COMPARISON OF NATIVE GRASSLAND MANAGEMENT TREATMENTS TO SPRING PRESCRIBED BURNS, A PILOT STUDY

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SUMMARY OF FINDINGS

We conducted a pilot study to measure the response of restored native grasslands to: (1) grazing; (2) fall biomass harvest; and (3) spring prescribed burning. Some fields included both a fall biomass harvest and a spring controlled burn within different portions of the same field. Fields were located on Wildlife Management Areas (WMAs) or Waterfowl Production Areas (WPAs) in Working Land Initiative Focus Areas of Grant, Kandiyohi, Pope, and Stevens 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. Vegetation characteristics varied considerably from field to field, and treatments (grazing, biomass harvest, burning) were not randomly assigned. Thus, comparisons involving fields undergoing a single treatment (e.g., grazing or fall biomass harvest only) were less informative than within-field comparisons involving biomass harvest and burn treatments. In these latter fields, biomass harvested and burned subplots appeared similar in most vegetative characteristics. We intend to survey vegetation in additional biomass harvest/burn field combinations in 2009, and will continue to survey all fields through summer 2011.

INTRODUCTION

Minnesota's Draft Grassland Biomass/Bioenergy Harvest on WMAs and Aquatic Management Areas (AMAs) 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 a pilot study during the summer of 2008 with the following objectives:

- to establish and test protocols for sampling vegetation; and
- to determine sample sizes necessary to have a high probability of detecting meaningful differences in vegetative characteristics among treatment groups in a subsequent 3-year study.

STUDY AREA

The pilot study was primarily conducted in Working Land Initiative (MNDNR unpublished brochure;http://files.dnr.state.mn.us/assistance/backyard/privatelandsprogram/working-lands-ini.pdf) Focus Areas within Grant, Stevens, and Pope Counties. Following the pilot year, the proposed study area will include Working Land Focus Areas in much of the prairie portion of Minnesota (Figure 1). Fields sampled during the pilot study were all located on state managed Wildlife Management Areas (WMAs) or federally managed Waterfowl Production Areas (WPAs). Study sites consisted of 4 with grazed, 5 with bioharvested, and 3 with both bioharvested and burned fields.

METHODS

We compared the response of restored native grasslands to (1) grazing, (2) fall biomass harvest (hayed) and (3) spring prescribed burning (control) Where possible, the fall biomass harvest versus spring prescribed burn comparison was made within different portions (subplots) of the same field. Visual obstruction measurements (VOMs, Robel et al. 1970) were taken every 2 weeks from early June through mid-August in grazed, hayed, and spring burned portions 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 diagonal across each field. GIS locations were permanently marked with stakes to define starting and sampling points. 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 from grazed, hayed and spring burned portions of each field on a given date.

A Daubenmire square (Daubenmire 1959) was used to determine coverage by various species across grazed, hayed, and burned fields. To sample, one corner of each field was randomly selected and, using a compass, a transect was walked across the field. The $1m^2$ Daubenmire frame was placed on the ground every 25 paces, and each plant species (and % coverage within the frame) that comprised $\geq$ 10% of the total number of individual plants within the frame was recorded. This procedure was repeated 10 times in each treated field every 2 weeks.

Litter depth (nearest 1mm) and vegetation height (nearest 0.5 dm) were also measured at 10 locations along the VOM transect in grazed, hayed, and burned portions of each field every 2 weeks. While walking the VOM transect, all exotic and woody species present were recorded, and 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 cover type (Stewart and Kantrud 1971), waterfowl numbers, and waterfowl pair status.

RESULTS

During the pilot study year, we measured vegetation on 4 sites with grazed fields, 5 sites with hayed fields, and 3 sites with both hayed and burned fields. Vegetative characteristics varied considerably among fields in the same treatment group (e.g., see Figures 2 and 3 for grazed fields), but were largely similar in hayed and burned subplots within the Eldorado and Grace Marsh fields (Figures 4-8). 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 at Klason.

We examined 12 seasonal and temporary wetlands in mid-April that had been at least partially harvested during the biomass treatment in fall 2007. One wetland remained closed (cover type 1), 4 had open scattered vegetation (cover type 2), 3 had an open central expanse of water (cover type 3), and 4 were completely open (cover type 4). Six wetlands had dabbling duck pairs present when visited in spring 2008.

DISCUSSION

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 3 management options for disturbing native grass stands.

Historically, the major factors influencing grassland ecosystems were fire, grazing by herbivores, and climatic variations (Kirsch et al. 1978). Grazing and mowing (Kirsch et al. 1978) and prescribed burning (Kirsch and Kruse 1972) are used to set back succession on managed areas. The suppression of these types of disturbances in prairie grasslands results in the invasion of woody species (Sauer 1950; Stewart 1956). Kirsch and Kruse (1972) found that species diversity of bird and vegetative species, as well as nest success, increased in burned versus non-burned grass fields in North Dakota.

Many species of upland nesting birds utilize residual vegetation as nest sites. Leopold (1933) noted that most waterfowl and gallinaceous birds depend upon residual vegetation for initial nesting attempts. Further, Bue et al. (1952) determined that ducks nesting in western South Dakota chose the tallest, most dense nesting cover available. Many studies have indicated the positive relationship between upland nesting birds and grassland disturbance from grazing (Mundinger 1976, Brown 1978, Duebbert et al. 1986), prescribed burning (Kirsch and Kruse 1972, Tucker et al. 2004, Thatcher et al. 2006), and both treatments in combination (Fuhlendorf and Engle 2004, Trager et al. 2004,Powell 2006). Evaluating treatment response over time is also important as Powell (2006) found that the effects of prescribed burning and grazing on habitats of grassland nesting birds benefited different species depending upon number of years post treatment.

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.

Management of native grass stands has become an important component of wildlife management in prairie portions of Minnesota (Kotts, personal communication). Historically, spring prescribed burning has been the management option most often used to create disturbances in these fields. However, the amount of habitat manipulated by spring burns is often dictated by spring weather conditions. Knowledge of the response of native grasses to management treatments other than spring burning may allow managers to treat additional acres, or manage grasslands in a more efficient manner. Further, determining alternate management scenarios for grasslands, particularly those that may have a financial incentive for the landowner (e.g. biofuel harvest, haying, or grazing), may entice some landowners to maintain their land in a grassland program such as Conservation Reserve Program or Wetland Reserve Program, rather than convert the land into cropland. This would have landscape wide benefits for wildlife, erosion control, and clean water.

Standardizing the many variables associated with grazing (e.g. stocking rates, grazing period, soil type) among grazing treatment fields proved challenging. Vegetative characteristics varied considerably among fields, treatments were not randomly assigned, and there was no within field comparison in grazed fields. Based on results from this pilot, it is unlikely we can learn much about the effects of grazing on vegetation. Therefore, we will likely drop the grazing component of this study in future years, and concentrate our efforts on the difference between bioharvested fields and spring controlled burn fields.

Differences in vegetation characteristics between bioharvested and burned fields were greatest at Klason WMA, possibly as a result of fields being burned at a later date at Klason (May 18, 2008) than at Grace Marsh WMA (April 25, 2008), or Eldorado WMA (May 9, 2008). The biomass harvest treatments on all 3 WMAs occurred in the fall of 2007, after vegetative growth had finished. However, vegetation began growing in the spring shortly after snow melt. Burning removes all old and new vegetation from the burned field. Therefore, the later in the spring the treatment field is burned, the larger the difference in vegetative growth between the hayed and burned portions of the same field would be expected.

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. We will continue to monitor the basins that were harvested in 2007 to document the duration of benefit from fall biomass harvest.

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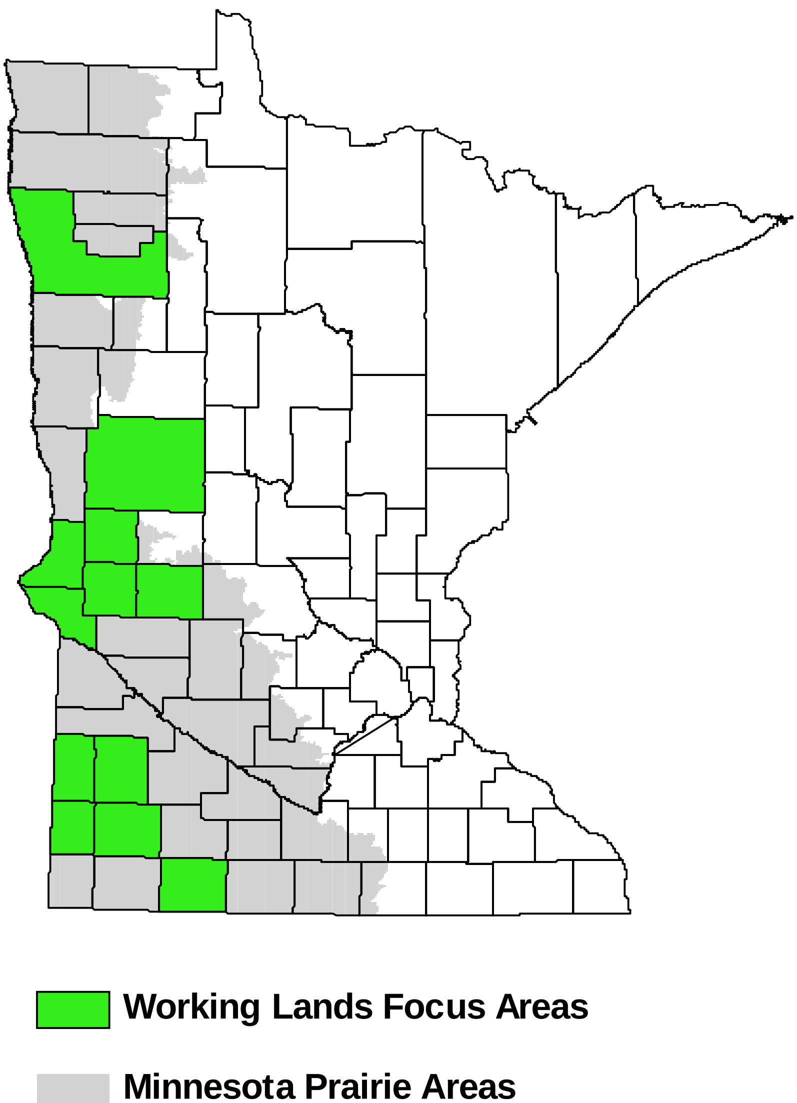

Figure 1. Minnesota Counties showing prairie areas and Working Lands Initiative focus areas, 2008.

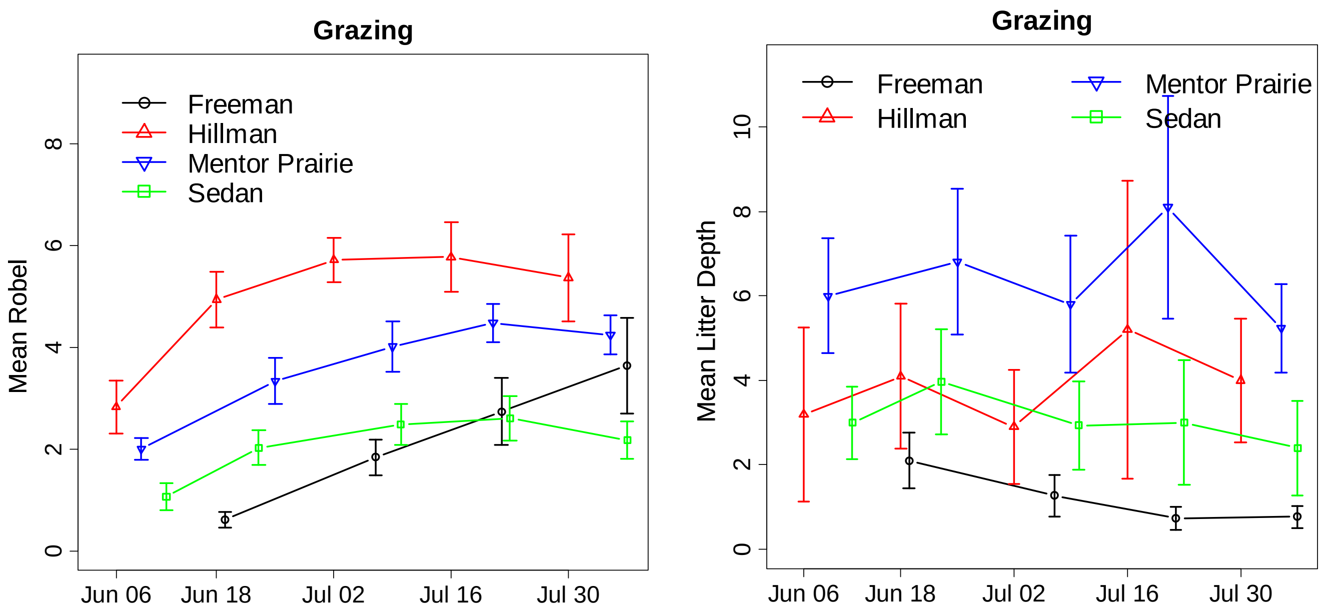

Figure 2. Comparison of mean Robel measurements (dm) and mean litter depth measurements (cm) across 4 units (Federal Waterfowl Production Areas and State Wildlife Management Areas) grazed in summer 2007 in west-central Minnesota.

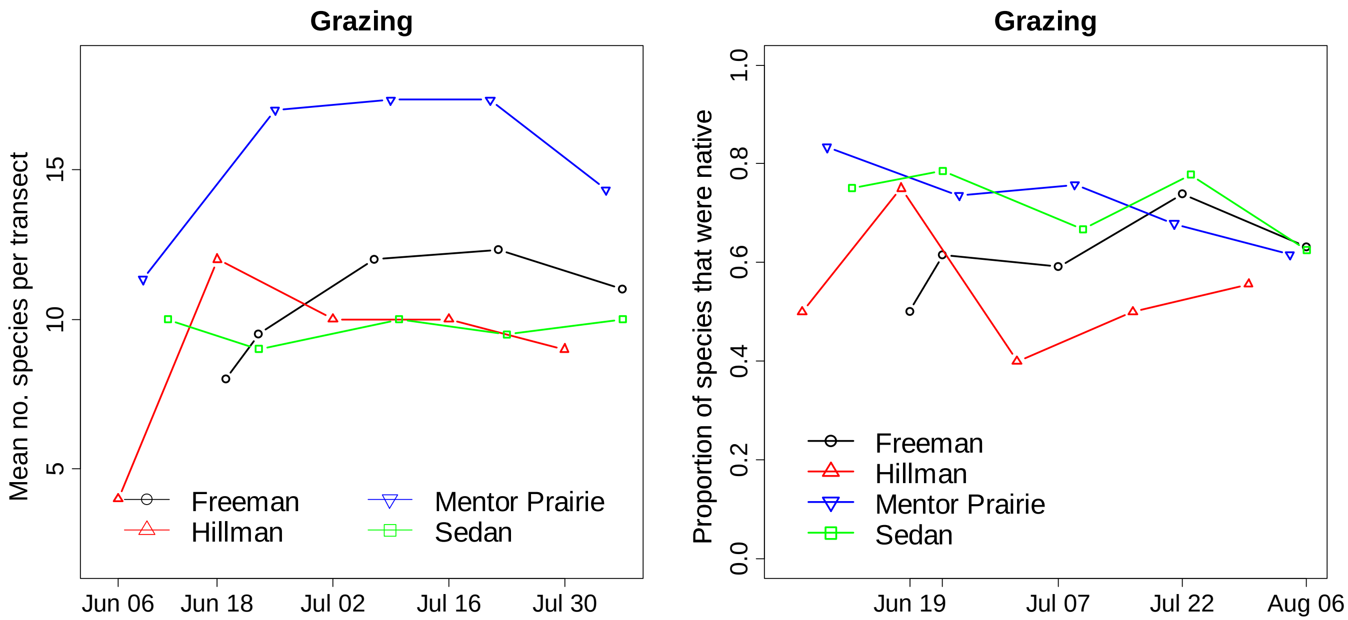

Figure 3. Comparison of mean number of species per transect and proportion of species that were native across 4 units grazed in summer 2007 (Federal Waterfowl Production Areas and State Wildlife Management Areas) in west-central Minnesota, summer 2008.

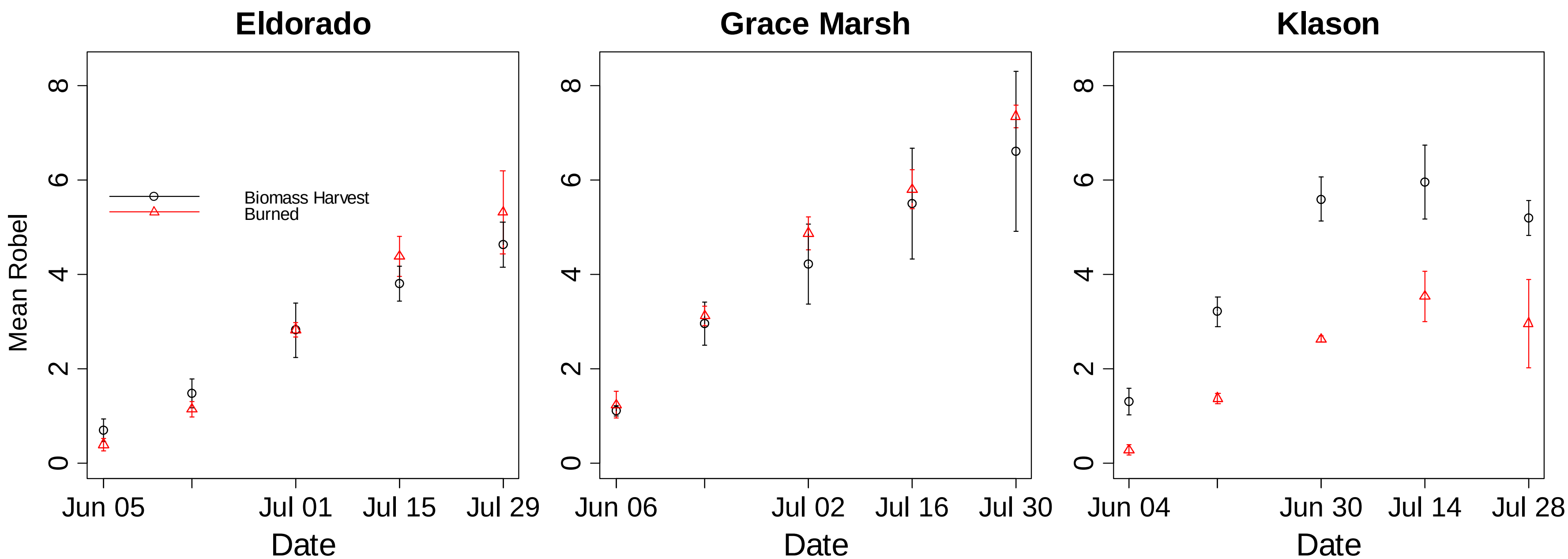

Figure 4. Comparison of mean Robel measurements (dm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

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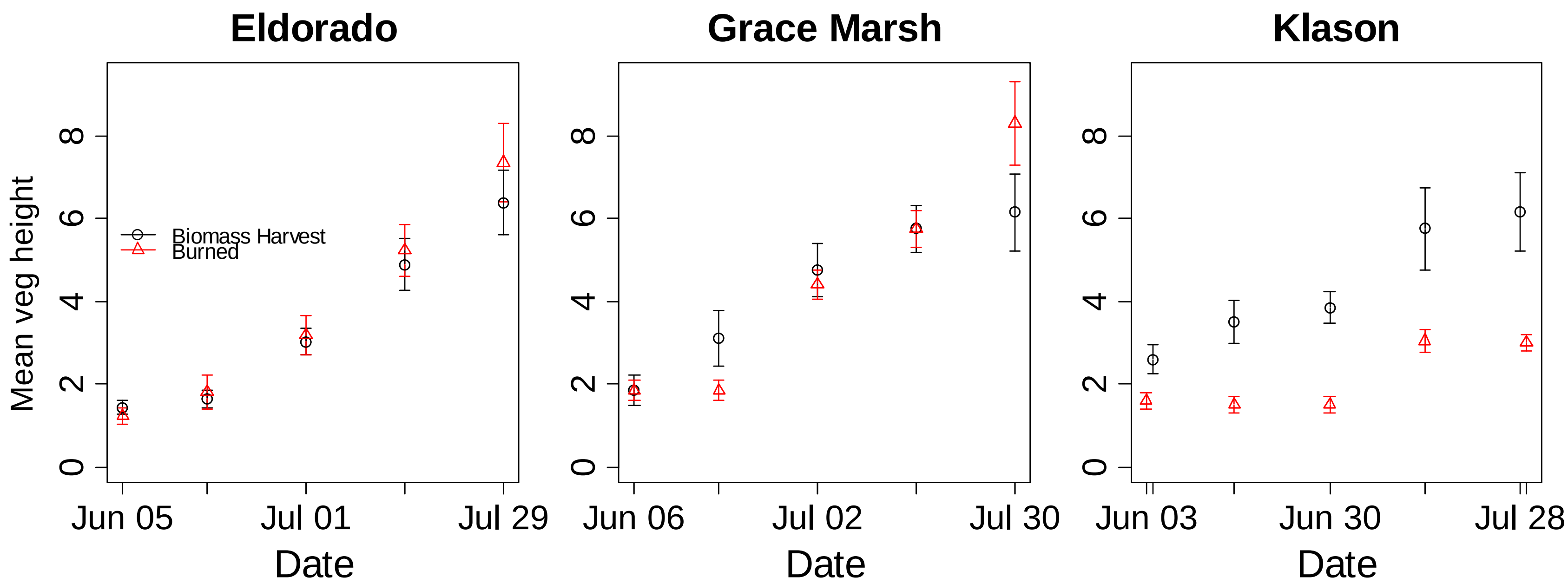

Figure 5. Comparison of mean vegetation height (dm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

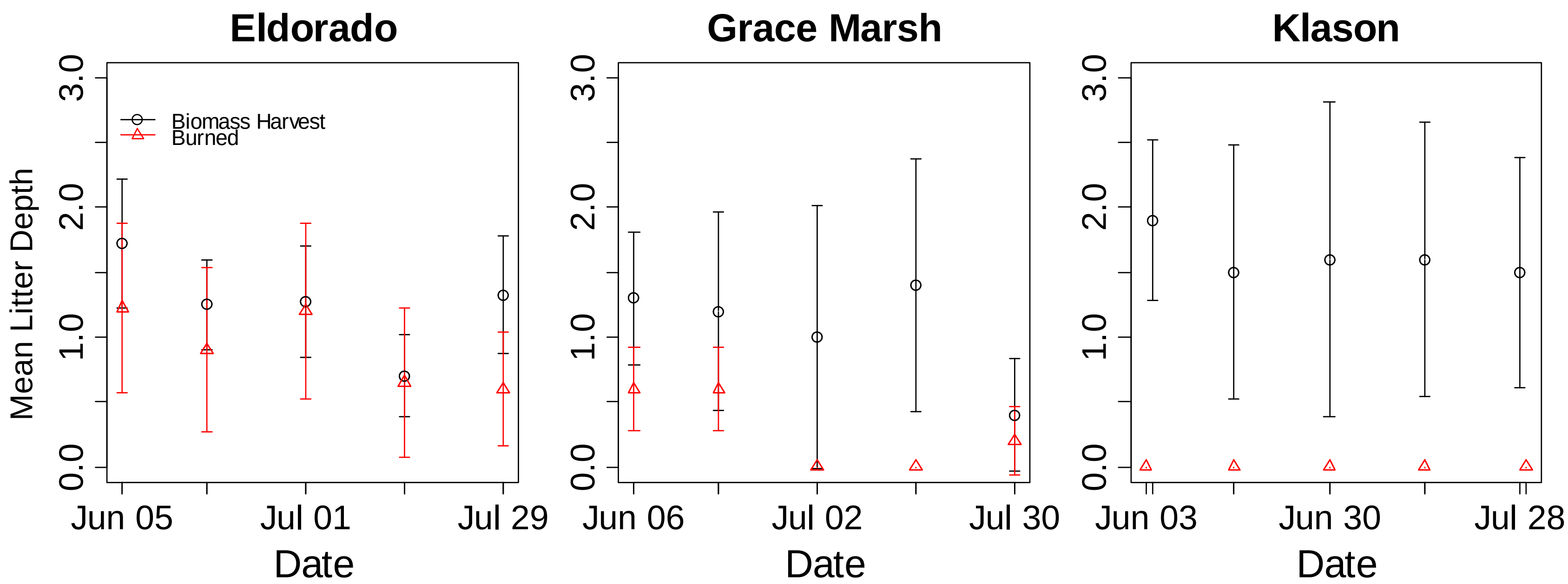

Figure 6. Comparison of mean litter depth (cm) between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

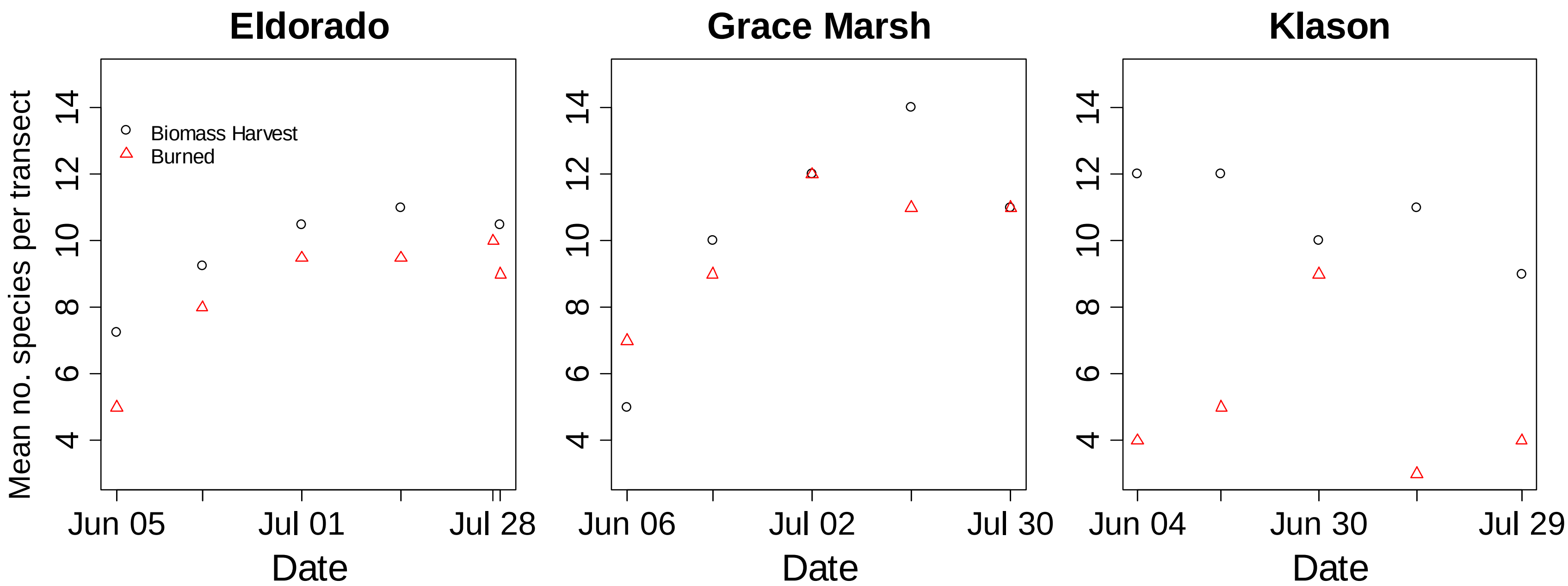

Figure 7. Comparison of mean number of plant species per transect between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

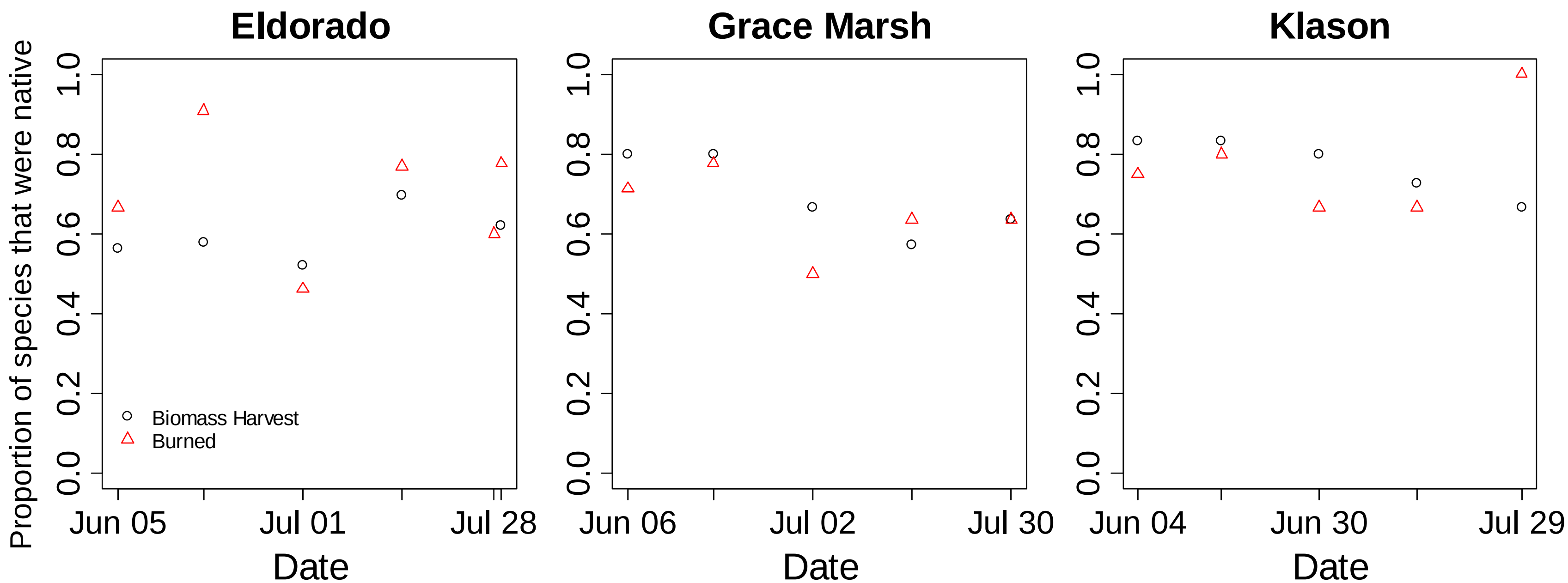

Figure 8. Comparison of the proportion of native plant species between 2 treatments (a fall 2007 Biomass harvest and a spring 2008 prescribed burn) in the same restored native grass field on 3 State Wildlife Management Areas in west-central Minnesota, summer 2008.

2008 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*) breeding population, because little was known about its distribution and relative abundance. We employed the survey design and methods developed during the pilot study (Zicus et al. 2006) to estimate the size of the population in 2007. In 2008, the survey was conducted again but we surveyed only 3 of 6 geographic strata due to budget limitations. The helicopter-based counts entailed 6 survey-crew days from 9-17 June totaling ~35 hrs of flight time. The survey included the portion of Minnesota considered primary breeding range. The 2008 breeding population was estimated to be ~9,500 indicated breeding pairs and ~19,500 birds, which are similar estimates to 2006 and 2007 for the 3 geographic strata.

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. At the time this survey was developed, little was known about the breeding distribution and abundance of resident ring-necked ducks. 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 2006a). 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).

A 3-year pilot study was initiated to develop a ring-necked duck breeding pair survey (Zicus et al. 2006), and 2007 represented the first year of an operational survey. In 2008, the survey was conducted again but was reduced in scope due to budget limitations. The primary objectives of this survey are to estimate breeding pair numbers and monitor population trends.

METHODS

Similar to the pilot study, we estimated number of breeding pairs and population size within a stratified random sample of survey plots 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. 2006). Surveys were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2005). 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. From 2004-2007, 6 ECS sections were surveyed (Western and Southern Superior Uplands [sections combined for the survey]; Northern Superior Uplands; Northern Minnesota and Ontario Peatlands; Northern Minnesota Drift and Lake Plains; Minnesota and Northeast Iowa Morainal; and Lake Agassiz, Aspen Parklands). In 2008, 3 of the ECS sections were dropped from the survey; ECS sections sampled for the survey included:

(1) Northern Minnesota Drift and Lake Plains; (2) Minnesota and Northeast Iowa Morainal; and (3) Lake Agassiz, Aspen Parklands, (Figure 1).

ArcInfo and ArcView software (Environmental Systems Research Institute, Inc., Redlands, California) were used to assign each PLS section to 1 of 4 model-based habitat classes (Zicus et al. 2006). We used the same habitat class definitions that were used for stratification in the last pilot year (i.e., 2006, Table 1). Plots with at least the median amount of nesting cover were assumed to have high potential (habitat class 1) for ring-necked ducks with plots having some nesting cover but less than the median amount assumed to have moderate potential (habitat class 2). Plots with no nesting cover but with near-shore water were assumed to have low potential (habitat class 3), and those without breeding habitat were assumed to have no potential (habitat class 4). In 2008, a stratified sampling design was used to estimate breeding ducks in the best ring-necked duck habitat (habitat class 1 and 2 plots), and the sampling frame consisted of 6 strata (i.e., 3 ECS sections x 2 habitat classes). We proportionally allocated 174 plots to the 6 strata (Zicus et al. 2005). 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). However, these areas were dropped from the survey due to limited funds.

As in previous years, we used a helicopter for the survey because visibility of ring-necked ducks from a fixed-wing airplane is poor in most ring-neck breeding habitats. For each plot, location, date, and time were recorded on data sheets as were all ring-necked ducks observed on study plots from the helicopter and their sex and social status (lone, paired, single-sex flock, mixed groups). Locations of these birds were also plotted on aerial photos. We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, MNDNR, personal communication). The total breeding ground 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 R programming language (R Development Core Team 2007) to estimate IBP and breeding population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area.

RESULTS

In 2008, plots were well distributed throughout the study area (Figure 1). Most plots (108 plots) were located in the Northern Minnesota Drift and Lake Plains section, while the fewest plots (13 plots) 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 (3.8% versus 1.5%; Table 2).

The survey was conducted 9–17 June and entailed 6 survey-crew days totaling ~35 hrs of flight time. A total of 296 ring-necked ducks were observed in 58 of 174 plots (Table 3). Overall, counts on occupied plots ranged from 1 to 25 birds (median = 3 birds/plot). Numbers of IBP on occupied plots ranged from 0 to 10 (median = 2 IBP/plot). Numbers of birds on occupied plots ranged from 0 to 28 ducks (median = 4 breeding birds/plot). Of the birds observed, 65% were classified as pairs, 14% flocked males, 12% lone males, 6% groups, 2% lone females, and <1% flocked females. Of IBP, 56% were classified as pairs, 24% flocked males, and 20% lone males. These IBP ratios suggest that survey timing was reasonably good for estimating the local breeding population. Observed pairs represented 56% of the IBP tallied during the 2008 survey, which was similar to the 2004 and 2007 surveys and slightly higher than the 2005 and 2006 surveys (Figure 2).

Estimated IBP in the survey area was 9,439 pairs (SE = 1,582 pairs; Table 4, Figure 3A). The estimated breeding ground population of ring-necked ducks in the survey area was 19,488 birds (SE = 3,240 birds; Table 4, Figure 3B). Because of sampling frame changes in 2008, 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 2008 were slightly higher than 2007 and slightly lower than 2006 but was within the error of both prior surveys. The breeding population ranged from a high of 4,948 pairs and 10,264 breeding birds in the Northern Minnesota Drift and Lake Plains section to a low of 803 pairs and 1,846 breeding birds and in the Lake Agassiz, Aspen Parklands section (Table 5).

In 2008, pair densities on habitat classes 1 and 2 ranged from 0.04 pairs/km^2 in the Lake Agassiz, Aspen Parklands section to 0.23 pairs/km^2 in the Northern Minnesota Drift and Lake Plains section. Densities of breeding birds ranged from 0.09 birds/km^2 in the Lake Agassiz, Aspen Parklands section to 0.48 birds/km^2 in the Northern Minnesota Drift and Lake Plains section. Compared to previous years, densities of birds and IBP were slightly lower in the Northern Minnesota Drift and Lake Plains section, higher in the Minnesota and Northeast Iowa Morainal section, and within the range of past years in the Lake Agassiz, Aspen Parklands section (Figure 4).

The survey was not designed explicitly to describe the distribution of breeding ring-necked ducks, but observations accumulated thus far have improved our knowledge of ring-necked duck distribution in the survey area (Figures 5 and 6). 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. 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.

DISCUSSION

Survey timing is considered optimal when most birds are counted as pairs and not in flocks (Smith 1995). Survey dates in 2008 appeared appropriate, because 56% of the indicated pairs were counted as paired birds. The breeding population appears to be relatively stable in the few years that this population has been surveyed, remaining between 18,000 and 22,000 breeding birds in the 3 ECS sampling frame. The Northern Minnesota Drift and Lake Plains section continues to have the majority of the ring-necked duck breeding population. These surveys are planned to continue in 2009.

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Table 1. Habitat classes assigned to Public Land Survey section plots in the Minnesota ring-necked duck breeding pair survey area, June 2004 – 2008.

Habitat class	Definition[a] 2004	Definition[a] 2005 - 2008[b]	Percent of survey area 2004	2005	2006-2007	2008
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, 2006, 2007, and 2008 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, 2007, and 2008.

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 – 2008.

ECS section	No. of plots[a]				No. of plots surveyed (Sampling rate [%])			
	2004	2005	2006-2007	2008	2004	2005	2006-2007	2008
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)
Minnesota & NE Iowa Morainal	3,510	4,033	3,561	3,561	50 (1.4)	35 (0.9)	32 (0.9)	53 (1.5)
Lake Agassiz, Aspen Parklands	316	363	340	340	15 (4.7)	8 (2.2)	8 (2.4)	13 (3.8)

[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, 2004-2008.

Year	No. of plots	No. plots with birds (%)	Birds			IBP			Breeding birds		
			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	0.58	3.20	353	1.77	7.06
2005	230	37 (16)	147	0.64	3.97	92	0.63	2.49	218	0.95	5.89
2006	200	50 (25)	279	1.40	5.58	167	0.60	3.34	375	1.88	7.50
2007	200	52 (26)	152	0.76	2.92	137	0.90	2.63	296	1.48	5.69
2008	174	58 (33)	296	1.70	5.10	173	0.58	2.98	364	2.09	6.28

Table 4. Estimated indicated breeding pairs (IBP) and breeding ground population size in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, 2004-2008.

	IBP (CV[%])		Breeding ground 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)

[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 breeding ground 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 2005-2008.

	IBP (CV [%])				Breeding ground population (CV [%])			
ECS section	2005	2006	2007	2008	2005	2006	2007	2008
W & S Superior Uplands[b]	444 (99.5[c])	669 (59.1)	671 (99.6)	-	889 (99.5[c])	1,338 (59.1)	1,342 (99.6)	-
Northern Superior Uplands	1,169 (46.8)	2,679 (33.7)	2,694 (46.5)	-	2,339 (46.8)	5,357 (33.7)	5,388 (46.5)	-
N Minnesota & Ontario Peatlands	239 (54.1[c])	1,572 (34.7)	717 (46.5)	-	477 (54.1[c])	4,076 (42.3)	1,434 (46.5)	-
N Minnesota Drift & Lake Plains	3,490 (33.0)	6,334 (31.5)	5,686 (26.0)	4,948 (24.6)	6,981 (33.0)	14,816 (29.6)	11,651 (25.4)	10,264 (24.3)
Minnesota & NE Iowa Morainal	918 (43.6)	2,102 (53.9)	2,118 (38.8)	3,689 (26.0)	4,122 (56.4)	4,204 (53.9)	4,236 (38.8)	7,377 (26.0)
Lake Agassiz, Aspen Parklands	1,235 (40.1[c])	1,414 (35.2)	902 (40.9)	803 (38.4)	2,471 (40.1[c])	2,829 (35.2)	1,976 (42.3)	1,846 (41.4)

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

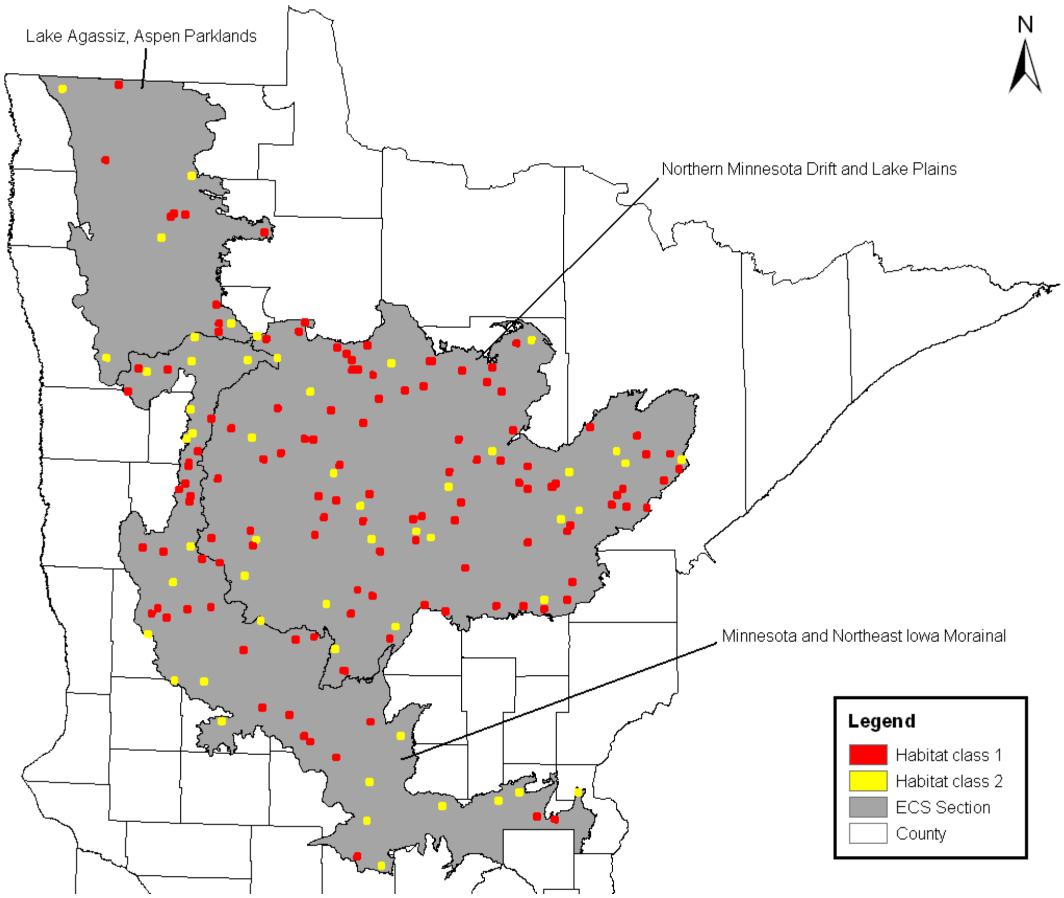

Figure 1. Study area with survey plots indicated by habitat class for Minnesota's 2008 ring-necked duck breeding pair survey.

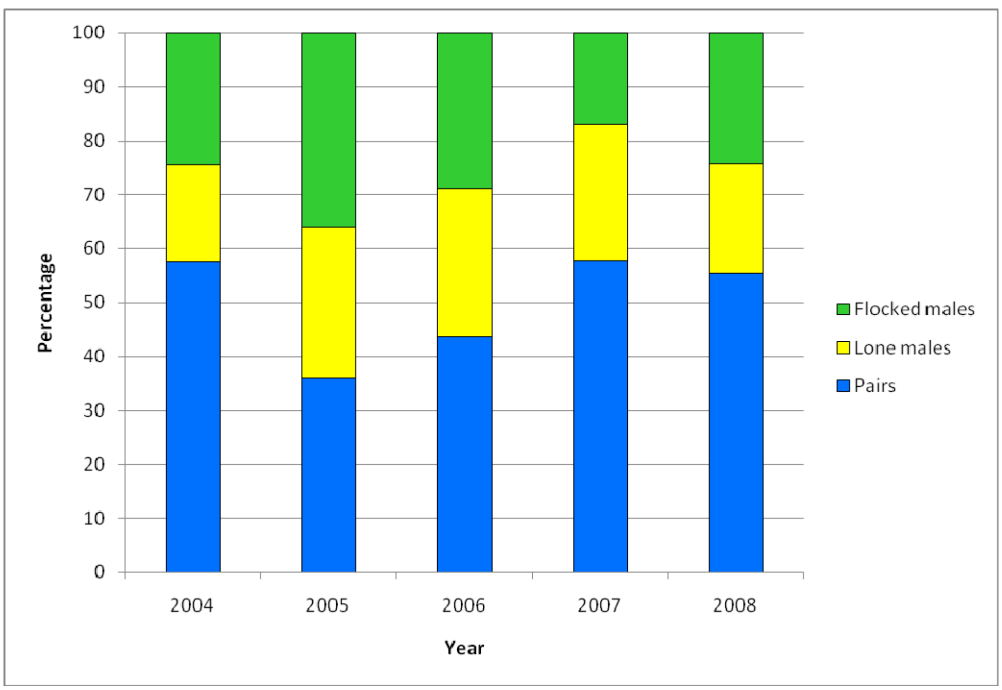

Figure 2. Social status of the indicated breeding pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004-2008. Surveys were conducted 6 – 17 June 2004, 12 – 24 June 2005, 6 – 16 June 2006, 5 – 13 June 2006, and 9 - 17 June 2008.

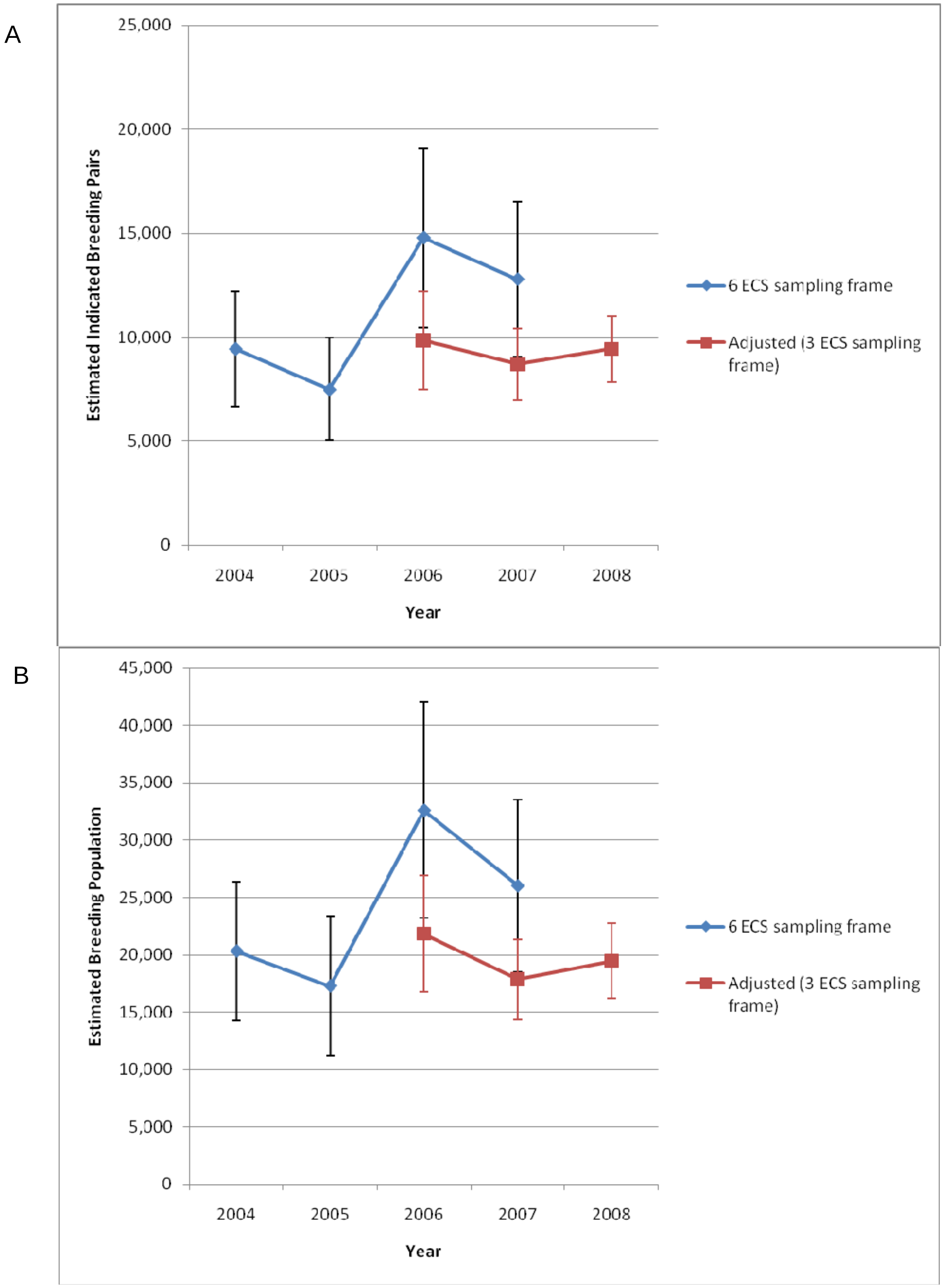

Figure 3. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs with SE bars and (B) estimated ring-necked duck breeding ground population with SE bars in the Minnesota ring-necked duck breeding pair survey area, June 2004-2008. Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 and 6 Ecological Classification System (ECS) sections in 2004-2007 and 3 ECS sections in 2008. Estimates from 2006 and 2007 were recalculated using the same sampling frame as 2008 (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.

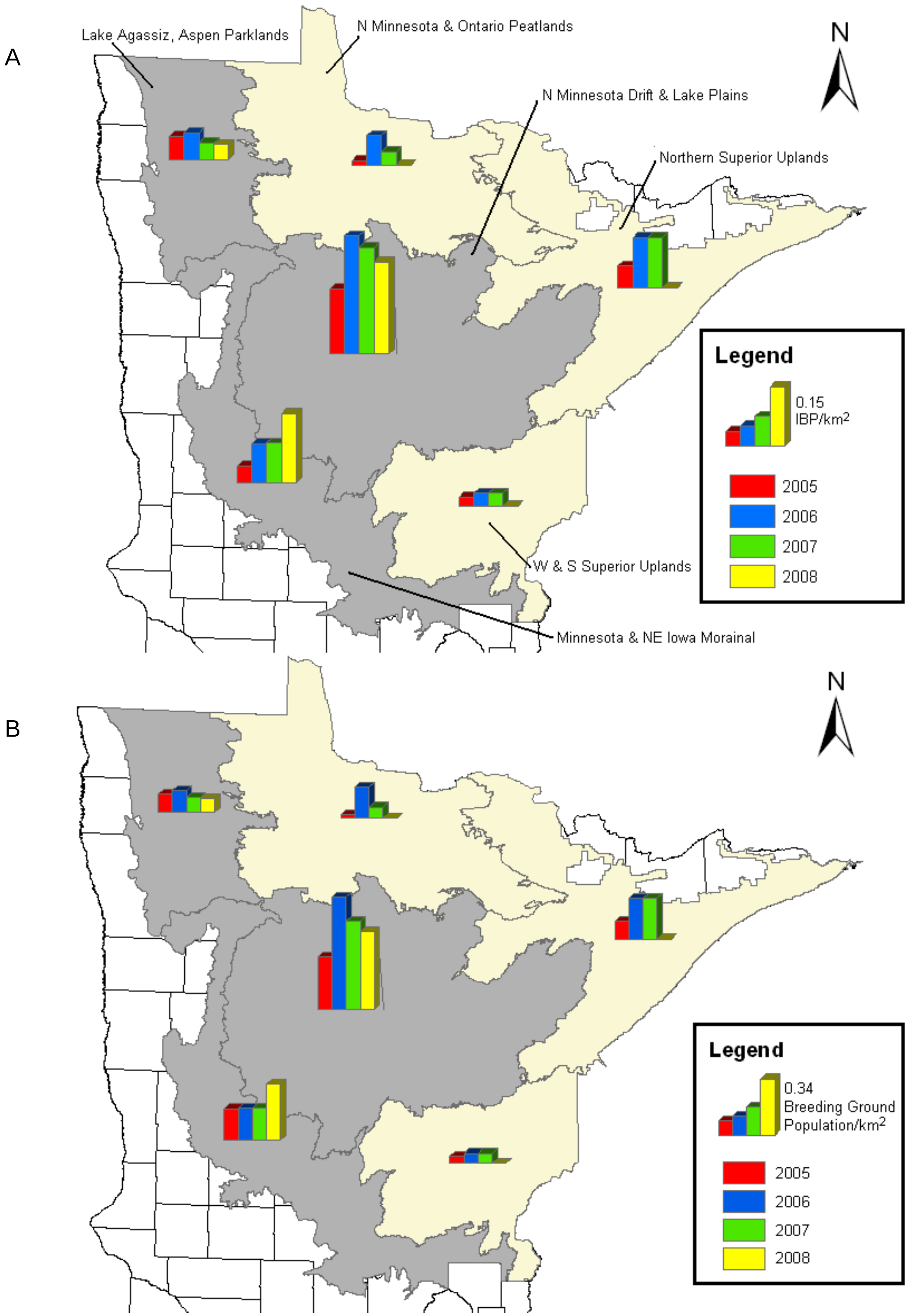

Figure 4. For the habitat class 1 and 2 strata (A) estimated indicated breeding pairs (IBP) per km^2 and (B) estimated breeding ground population per km^2 in the Minnesota ring-necked duck breeding pair survey area, June 2005-2008. Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

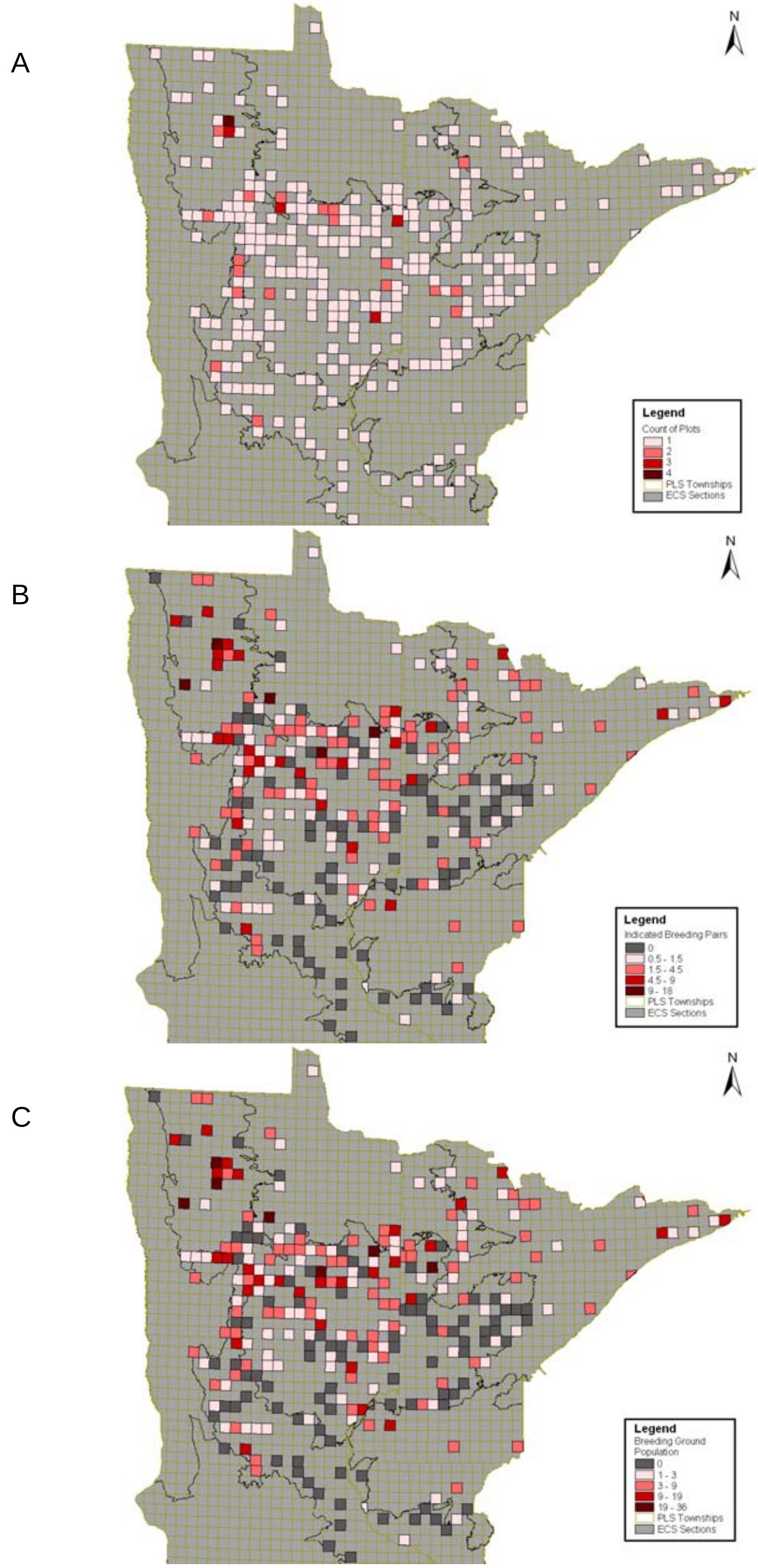

Figure 5. Maps are colored to indicate (A) total number of plots surveyed, (B) average number of indicated breeding pairs (IBP), and (C) average breeding ground population in each Public Land Survey (PLS) Township within the Minnesota ring-necked duck breeding pair survey area during 2004-2008. The Ecological Classification System (ECS) sections are also shown.

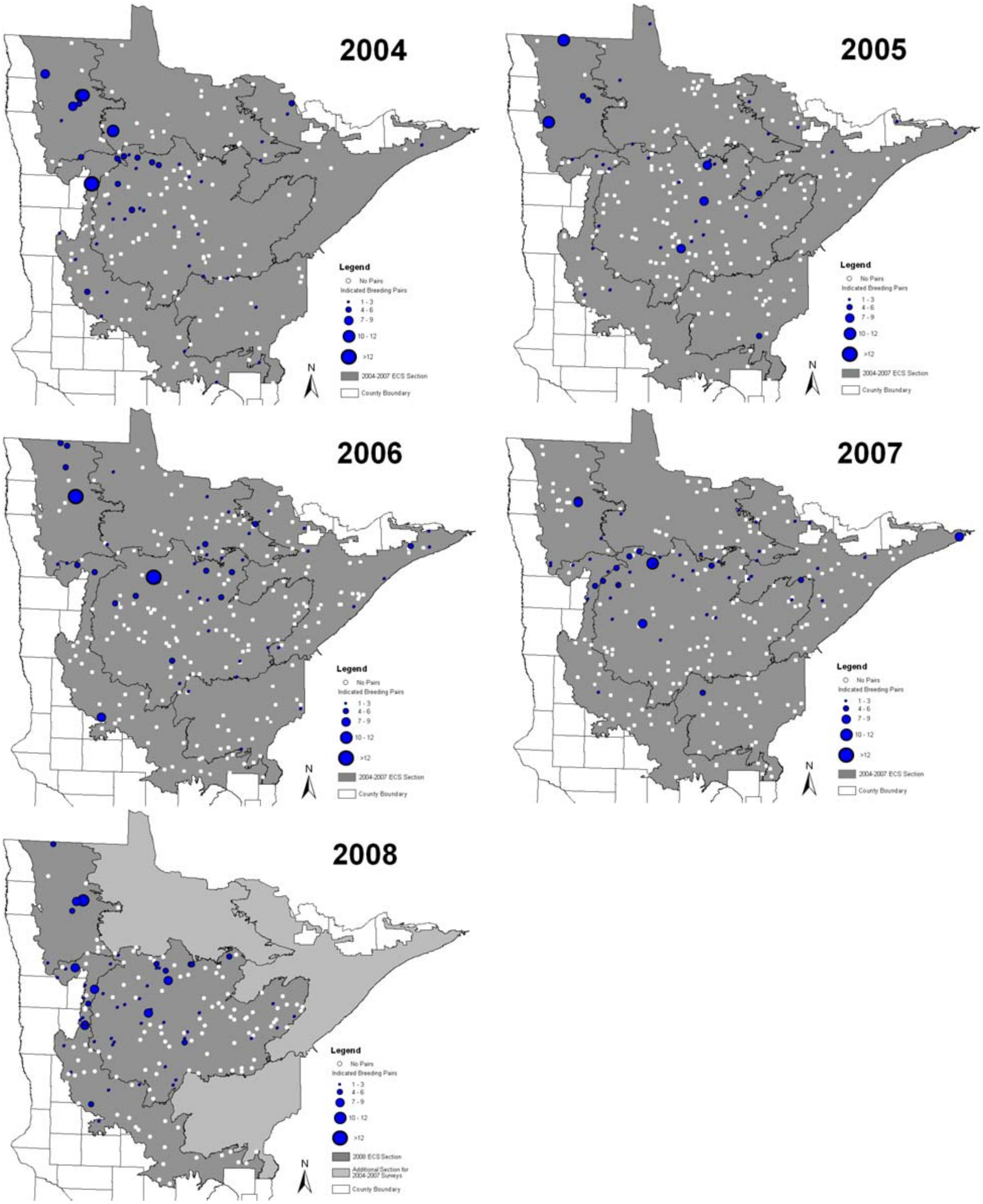

Figure 6. Plot locations and numbers of indicated breeding pairs observed on survey plots in the Minnesota ring-necked duck breeding pair survey area in June - 2008 (bottom left). White circles indicate plots where no indicated pairs were seen. Maximum number of indicated breeding pairs per plot was 10 pairs in 2008 (13 in 2004; 11 in 2005; 16 in 2006; 11 in 2007). The Ecological Classification System (ECS) sections are also shown.

NESTING ECOLOGY OF RING-NECKED DUCKS IN NORTHERN MINNESOTA

Charlotte Roy, Christine Sousa, Jody Kennedy[1], Elizabeth Rave[1]

SUMMARY OF FINDINGS

The first field season focused on data collection and refining methods to increase the efficiency of nest searches. We relied on 2 datasets to identify ring-necked duck (*Aythya collaris*) nesting habitat: 1) ring-necked duck breeding survey data from 2004-2008 (Zicus et al. 2006); and 2) GAP data layers indicating the juxtaposition of emergent herbaceous vegetation or low woody plants with water and <10% tree crown cover. After lakes were identified, we flushed ring-necked duck hens off nests using multiple methods, including nest dragging, disturbing vegetation with bamboo poles on foot and from canoes, and with low flights in a helicopter (Heyland and Munro 1967, Johnson 1977, Kaminski 1979). We found 18 ring-necked duck nests in this first year. We will improve effectiveness of finding nests next year by scouting for pairs and lone males on wetlands before conducting nest searches.

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood and duckling survival before fledging in the boreal forest.
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 pocked with lakes. Wetlands in the area commonly have wild rice or other emergent vegetation, sedges (*Carex* spp.), and floating bog mats along the margins.

METHODS

We identified lakes with suitable habitat using data on ring-necked duck use in the spring. These data were obtained from a helicopter survey conducted in 2004-2008 and from ground surveys conducted on 14 lakes in the Bemidji area beginning in 1969. We then used these data to identify lakes with similar land cover attributes (GAP types 12 and 13 surrounded by GAP types 10, 14, and 15). We also attempted to search lakes in our study area with pairs or lone males in the 2008 ring-necked duck survey.

We searched emergent vegetation along wetland margins using bamboo poles and nest drags to locate nests. We also searched with a helicopter to determine whether search efficiency could be improved from that of efforts on the ground (Heyland and Munro 1967, Johnson 1977, Kaminski 1979). This method involved flying low in a helicopter to blow vegetation and flush hens off nests. We documented lakes in which hens, lone males, and pairs were observed. We also recorded nests of other species encountered during nest searches from the ground.

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. Nests were monitored weekly to determine fate (abandoned, depredated, or successful). At each nest, and at a random point located 25 m from each nest, we determined water depth, concealment using a Daubenmire frame and Robel pole (Daubenmire 1959, Robel et al. 1970), predominant vegetation (e.g.,

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cattail, sedge), distance to dry land, and distance to open water. Wetland size, distance to roads and dwellings, wetland class, and disturbance variables will be determined in GIS for use in models of nest survival.

Late in incubation, we trapped ring-necked duck females 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 with which we had had previous success in wood ducks (*Aix sponsa*, Montgomery 1985). This attachment method is faster and less invasive than surgical methods. Briefly, hens received a transmitter fastened to a Herculite® fabric bib with dental floss and superglue (total weight of 11 g). We modified the method used unsuccessfully 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.

After the nest hatched, we monitored broods every 3-4 days. We conducted behavioral observations of hens with transmitters and hens without transmitters to determine whether behavior was altered by our methods. We continued to monitor hens after the brood-rearing period to examine hen survival until migration.

RESULTS

Nest Survival

We searched for nests from the ground on 39 wetlands a total of 73 times between 22 May-22 July 2008. We located 18 ring-necked duck nests on 10 of these wetlands. We also located nests of 6 mallards (*Anas platyrhynchos*), 4 American coots (*Fulica americana*), 2 blue-winged teals (*Anas discors*), a common loon (*Gavia immer*), a pied-billed grebe (*Podilymbus podiceps*), a red-necked grebe (*Podiceps grisegena*), a sora rail (*Porzana carolina*), and a Canada goose (*Branta canadensis*). Two black tern (*Chlidonias niger*) colonies were located, and many nests were associated with each colony. All nests were located on foot or from canoes. Helicopters flushed 16 hens in 4 hours, but many of these areas could not be searched due to difficulties obtaining landowner permission or because the bog mats could not withstand our weight without being submerged.

Of the 18 ring-necked duck nests, 8 hatched, 4 were depredated when found, 3 were depredated after they were found, and 3 nests were flooded by rising lake levels following rain events. Average clutch size was 9.1 $\pm$ 0.6 (range: 7-15) and 87.1 $\pm$ 0.1% of eggs hatched. We put transmitters on 8 hens late in incubation. Apparent nest success was 8/18, or 44%. Mayfield nest success for a 26-day period (Mendall 1958) was 37.3%. More thorough nest survival analyses will be conducted at the conclusion of the study, when sample sizes are larger. We will also analyze characteristics at each nest in subsequent reports.

Hen Survival

Of 8 birds radiomarked, 2 birds died during the brood-rearing period; one had a brood and the other lost her nest late in incubation. Both of these birds had been observed preening more than other birds with transmitters, although this behavior occurred during the first 2 weeks after marking and then subsided. Both deaths occurred after this period, one 3 weeks post-marking and the other 4 weeks post-marking. All birds continued to nest and rear broods after transmitter attachment, with the exception of one bird that lost her nest to flooding. Two more birds died during hunting season, but unfortunately we could not determine whether these mortalities were associated with hunting.

Brood and Duckling Survival

Of the 8 broods monitored (7 radio-marked, n = 66 ducklings), 1 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. The fate of one brood could not be determined because the hen died when the brood was at the IIA stage. 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. One hen with a transmitter lost her nest to flooding just before the expected hatch date, and thus did not have a brood.

DISCUSSION

Thus far, our results have been similar to findings by R. T. Eberhardt in northern Minnesota during 1978-1984 (Hohman and Eberhardt 1998). Our nest survival rates are comparable to his estimates of 44% based on 188 nests. The causes of nest failure in our study (30% flooding and 70% depredation) were also similar to those of other studies (flooding 16-24%, depredation 67-80%, and desertion 5%, Mendall 1958, McAuley and Longcore 1989). Early 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 spring of 2008 was very cool and rainy, which may have chilled eggs and flooded nests.

We have identified ways to improve our nest searching methods and success rate next field season. In 2009, we will scout lakes with ring-necked duck habitat for pairs and lone males before conducting nest searches. This modification will focus our nest searching efforts on lakes with nesting pairs rather than on lakes with nesting habitat. A small percentage of the lakes with nesting habitat were actually used by ring-necked ducks; our success rate finding nests at lakes with good nesting habitat was 25%. However, when pairs or lone males were observed on lakes, our success finding nests was 90%. Although the number of nests we found was 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), we expect to have even greater success in 2009 with these modifications to our methods.

ACKNOWLEDGMENTS

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

Charlotte Roy, Christine Sousa, David Rave, Wayne Bruninger[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 and 2008, we captured and implanted 108 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 both 2007 and 2008. Birds departing from their natal lakes did not exhibit strong directionality in movements. Young ring-necked ducks used state and federal waterfowl refuges, but this use was not evenly distributed among refuges; 2 refuges received the majority of use and 4 refuges have yet to be used by marked birds. Refuge use also increased markedly during hunting season. Additional data collection in 2009 will be aimed at increasing sample sizes to address survival benefits of refuge use to young birds.

INTRODUCTION

The MNDNR Fall Use Plan recognized sizable populations of resident breeding ducks as a cornerstone to improving fall duck use. 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 20 years, even as numbers of these birds staging on most traditional ring-necked duck refuges in the fall have increased in the state (Wetland Wildlife Populations and Research Group, unpublished data). Efforts to better understand population status began in 2003 with development of a ring-necked duck breeding-pair survey.

The Fall Use Plan identified the need to better understand the role of refuges in duck management. The influence of north-central Minnesota refuges on the distribution and welfare of resident ring-necked ducks is unknown. Factors influencing resident populations of ring-necked ducks are also poorly understood, as is the influence that the distribution of resident ring-necked ducks might have on that of migrant ring-necked ducks staging in the fall.

The intent of this project was to determine whether refuges benefit locally produced ring-necked ducks and increase survival. 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. Post-fledging ecology of many waterfowl species has not been documented, and this study provides information for an important Minnesota species.

OBJECTIVES

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

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

STUDY AREA

The study area lies in the heart of the Laurentian mixed forest province of Minnesota. This area is characterized by mixed coniferous and hardwood forest pocked 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, as well as 14 important ring-necked duck refuges which are not open to public hunting (Figure 1, Table 1).

METHODS

Night-lighting techniques were employed to capture flightless ring-necked ducks during July and August. Duckling age and sex was determined at capture (Gollop and Marshall 1954). We implanted radiotransmitters dorsally and subcutaneously on class IIb and IIc ring-necked ducklings following techniques developed by Korschgen et al. (1996), with one modification; we attached mesh to the back of transmitters to increase retention rates (D. Mulcahy, US Geological Survey (USGS), Alaska Science Center, personal communication). 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.

By early September, radiotelemetry stations were established at each refuge as a means of quantifying refuge use. These stations consisted of a tower with a four-element yagi antenna pointed toward the primary waterfowl use areas within the refuges. In some cases, more than 1 antenna was used so that a greater area could be covered. The receivers were programmed to scan all transmitter frequencies each hour and were equipped with data loggers to store the data (Advanced Telemetry Systems, Incorporated DCC II Model # D5041 and Model # R4500). Data were downloaded weekly from data-loggers from mid-September through early November, and examined to determine presence/absence of radiomarked birds. Reference radiotransmitters were stationed permanently at each refuge to ensure that receivers and data loggers functioned properly. Flights 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

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. Capture locations were distributed throughout the study area (Table 2 and Figure 2).

Birds moved in all directions from their natal lake. All but 1 bird left its natal lake before hunting opened over the 2 years. Success locating birds from aerial flights was higher before hunting season than the week hunting opened in both years (87% before and 66% after in 2007, 95% before and 83% after in 2008). Success locating birds also declined as birds began moving more in preparation for migration. For the tracking period, average weekly movements were 8.5 $\pm$ 1.9 km in 2007 and 8.3 $\pm$ 2.1 km in 2008. Average weekly movements tended to increase as the season progressed until mid to late October when birds started leaving the study area. Average weekly movements prior to the start of hunting (6.4 $\pm$ 1.1 km and 6.8 $\pm$ 1.6 km, in 2007 and 2008 respectively) were shorter than after hunting season opened (14.5 $\pm$ 3.0 km and 16.6 $\pm$ 3.5 km) in both years.

At the conclusion of the 2007 tracking season, 24 radiomarked birds were known to have died, of which 8 were harvested by hunters. Four of the 8 hunter harvested birds were shot during the first 2 days of the season (29 and 30 September). Two were harvested in Louisiana and 1 in Illinois. Natural sources of mortality based on evidence at the site where the transmitter was found included predation by mink (*Mustela vison*) and other mammals (7) and birds including great-horned owls (*Bubo virginianus*) or other raptors (3). Six radios were thought to have dehisced because they were retrieved from open water. Losses to predation prior to hunting season (7) were similar or slightly higher than those during hunting (3). Formal survival analyses have yet to be performed.

At the conclusion of the 2008 tracking season, 33 radiomarked birds were known to have died, of which 11 were harvested by hunters. Four (36%) of these birds were shot during opening weekend. Birds were shot predominantly in Minnesota (8), but losses also occurred in Louisiana (2) and South Carolina (1). We attributed mortality to predation by mammals (5), raptors (1), and unknown sources (5). Four radios were thought to have dehisced, because they were retrieved from open water, and a definitive cause of mortality could not be determined for 7 birds. Five of these cases might have been ducklings that were crippled and later scavenged. Before hunting season, predators took 10 marked birds. After hunting opened, 1 bird was lost to predation and 5 others were lost to unknown causes.

Refuges were rarely used before hunting season, but use increased markedly with the onset of hunting (Figure 3). Refuge use was documented for 18 radiomarked birds on 8 refuges during the fall of 2007 (Table 1), and 12 radiomarked birds used 8 refuges in 2008. The most heavily used refuges in 2007 were Mud-Goose (7 birds), Tamarac National Wildlife Refuge (6 birds), and Fiske and Blue Rock Lakes (5 birds). In 2008, Mud-Goose (6 birds) and Gimmer Lake (3 birds) were used the most by radiomarked birds. Rice Lake NWR was not used in either year, but we expected use of this refuge by radiomarked birds to be less than that of refuges located within the capture area. Nevertheless, Rice Lake NWR is an important staging area for ring-necked ducks in the fall, so we will continue to monitor this refuge next year.

DISCUSSION

One more field season is anticipated. Methods in 2009 will be similar to those of 2008. 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.

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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 2007-2008.

Refuge	Location	~Peak numbers	Stations	Refuge use 2007	Refuge use 2008
National Wildlife Refuge					
Rice Lake	5 mi SSW of McGregor	120,000	4	No	No
Tamarac	16 mi NE Detroit Lakes	10,000	3	Yes	Yes
State Waterfowl Refuge/State Game Refuge					
Donkey Lake	6 mi SW Longville	350	1	Yes	No
Drumbeater Lake	2 mi N of Federal Dam	160,000	1	Yes	Yes
Fiske and Blue Rock Lakes	8 mi SE Northhome	40,000	1	Yes	No
Gimmer Lake	10 mi SE Blackduck	200	1	No	Yes
Hatties and Jim Lakes	13 mi SE Blackduck	0	1	No	No
Hole-in-Bog Lake	2 mi SW Bena	4,000	1	No	No
Mud-Goose	4 mi SSW of Ballclub	2,100	1	Yes	Yes
Lower Pigeon Lake	4 mi S Squaw Lake	700	1	Yes	Yes
Pigeon River Flowage	6 mi S Squaw Lake	700	1	No	Yes
Preston Lakes	22 mi ENE of Bemidji	535	1	No	Yes
Round Lake	8 mi N Deer River	7,000	1	No	No
Rice Pond	9 mi E of Turtle River	15	1	Yes	Yes

Table 2. Ring-necked duckling captures per county in Minnesota during 2007 and 2008.

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

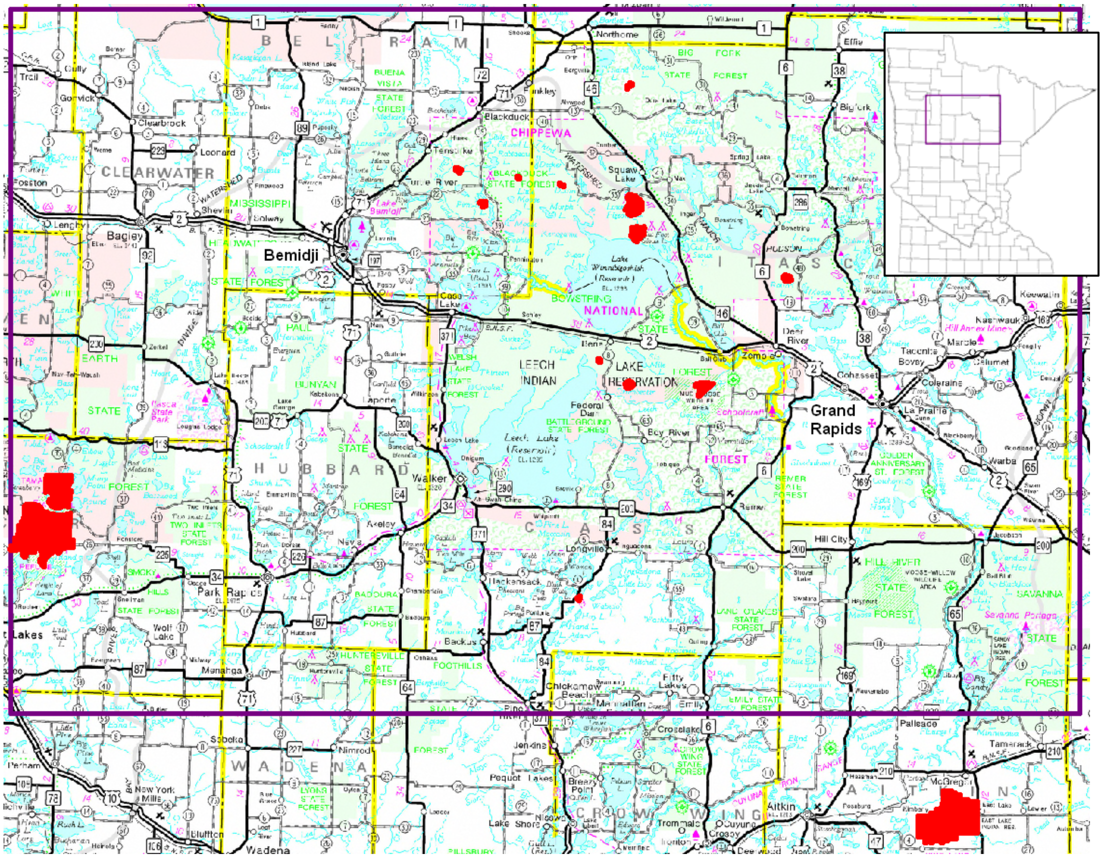

Figure 1. Ring-necked duck study area in Minnesota during 2007-2008 with 12 state waterfowl refuges/state game refuges and 2 National Wildlife Refuges depicted in red.

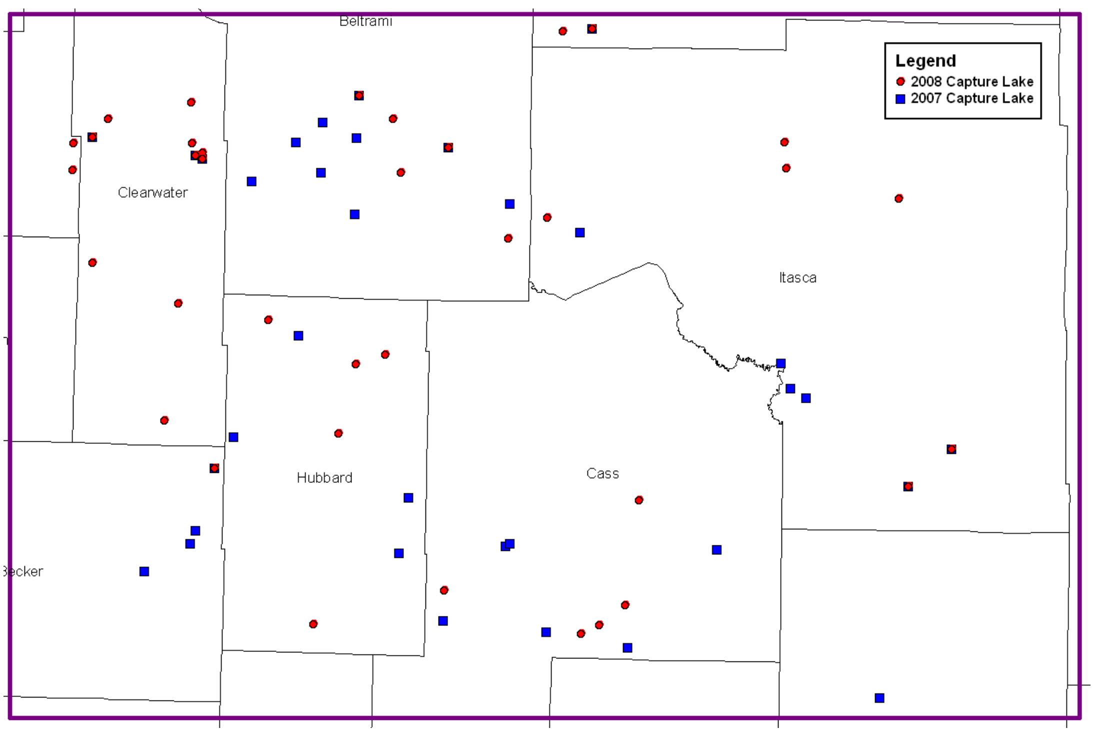

Figure 2. Capture locations for ring-necked duck ducklings in Minnesota during 2007 and 2008.

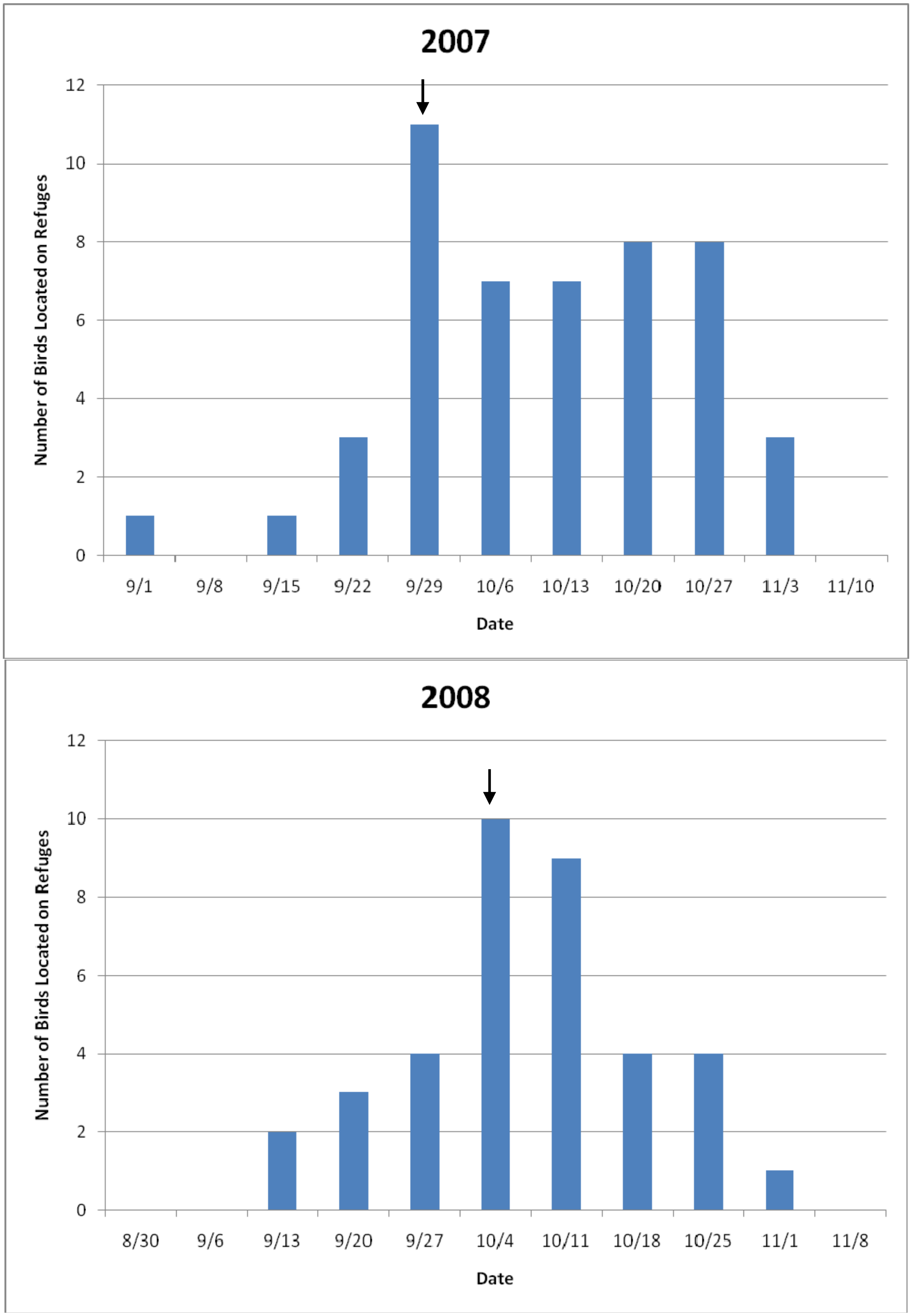

Figure 3. Weekly use of refuges by post-fledging ring-necked ducks before and during hunting season in 2007 and 2008 in Minnesota. Weeks are from Saturday through Friday with the Saturday date shown. Arrows indicate the week hunting opened.

ASSESSING CHARACTERISTICS OF KENOGAMA LAKE, A SHALLOW WATERFOWL LAKE IN NORTHERN MINNESOTA: TWO YEAR SUMMARY

Mark A. Hanson, Andrew Folkers[1], Neil Rude[1], and Donald Cloutman[1]

SUMMARY OF FINDINGS

Kenogama Lake (Kenogama) is a shallow lake in western Itasca County, MN, contained within the boundaries of the Laurentian Mixed Forest. The lake is believed to be of considerable importance to migrating diving ducks, especially Lesser Scaup (*Aythya affinis*). During the past 15 years, anecdotal evidence indicates that fall use of Kenogama by diving ducks has diminished. Mechanisms responsible for these declines are unknown but may include changes in duck migration patterns, weather and precipitation dynamics, or changing availability of aquatic invertebrates or other food resources important in diets of migrating Lesser Scaup and other ducks. Of particular interest is whether historical use of Kenogama as a site for rearing of walleye (*Sander vitreus*) fry is related to changes in lake characteristics and habitat suitability for migrating ducks. During 2007 and 2008, we monitored relative abundance of fish and aquatic invertebrates, water transparency, phytoplankton abundance, major nutrients, submerged macrophytes, and other characteristics of Kenogama. Fish were abundant, with golden shiners (*Notemigonus crysoleucas*) and walleyes comprising most biomass in our samples in 2007; however, adult walleyes appeared to be absent from the lake during 2008. We observed sparse populations of macroinvertebrates such as aquatic insects and amphipods. Zooplankton were abundant, but only small taxa were numerous, probably reflecting high predation by zooplanktivorous fish. Water quality data and relative abundance of submerged aquatic plants were indicative of a shallow lake in a "clear-water state" with a lighted substrate and rooted aquatic plants present in most areas throughout the lake. Adult walleye stomach contents indicated considerable consumption of aquatic invertebrates. It is not known whether this consumption is responsible for a low density of macroinvertebrates throughout the lake, but planktivorous fish may have been equally important. During 2008, plants and nutrients showed some evidence of a trend toward turbid conditions. We plan additional monitoring efforts at Kenogama and other forest lakes during 2009-2011. Current and future data should help clarify influences of various fishes and other factors on shallow lake characteristics and suitability for waterfowl.

INTRODUCTION

Kenogama Lake holds considerable interest to wildlife managers in north central Minnesota due to its history of fall use by migrating diving ducks. Located in the Laurentian Mixed Forest, Kenogama also represents a type of shallow lake that has received little study in North America. In Minnesota and elsewhere, shallow lakes are believed to exhibit a bimodal distribution of characteristics, tending toward opposite regime conditions along a continuum of water clarity and extent of submerged macrophyte development (Scheffer 2004). These "alternative states" are typically characterized by clear-water lakes containing abundant submerged macrophytes, and alternatively, by lakes with turbid water and sparse submerged macrophytes. In each alternative regime, shallow lakes are believed to exhibit stability and resist changes toward the opposite extreme, especially at either very high or very low levels of background nutrients. However, at intermediate nutrients, either regime is possible and lakes can quickly shift in response to water level changes, winter hypoxia and resulting fish "winterkill", chemical fish kills, introduction of fish, and other perturbations. For example, turbid shifts sometimes follow increased density of planktivorous/benthivorous fish populations, prolonged increases in water depth, or increased nutrient loading (although examples of the latter are rare). Complete removal of fish from shallow Minnesota lakes has been shown to

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induce transitions toward clear-water states (Hanson and Butler 1994a, Zimmer et al. 2001), but in such cases, regime shifts may be temporary.

Mechanisms influencing characteristics of shallow lakes in forested regions of Minnesota and elsewhere are poorly documented. At least some north temperate shallow lakes seem to follow a pattern of alternative regimes (Bayley and Prather 2003, Zimmer et al. in press). Minnesota's shallow lakes program has compiled data from 375 shallow lakes statewide, yet these efforts target relatively few lakes in the Laurentian forest. Data from Minnesota also indicate that patterns of shallow lake characteristics differ dramatically between prairie and transition ecoregions, perhaps indicating importance of different structuring mechanisms across regional gradients (Herwig et al. 2006).

Previous studies demonstrated that shallow lakes in the Minnesota parkland often support diverse fish communities (Herwig et al. 2006). Thus, although Kenogama is in the forest, we expected that it might also contain a rich fish community. This seemed especially likely given the lake's size, history of angler interest, and the recent pattern of mild winters. Limited reports from Kenogama indicated that water clarity was good, that abundance of submerged aquatic plants was relatively high, and that plants were not limited by poor water clarity (Hansel-Welch et al., unpublished data). Kenogama has been used to rear walleye since 1983 (MNDNR, unpublished data). Walleye fry are stocked in spring; juveniles (age-0) are removed during fall. Some unharvested walleye are known to survive over-winter because, summer and winter angling was sometimes popular, at least during the past decade.

Recent research evaluating stocking of walleye fry in shallow prairie lakes indicated that adding juvenile (age-0) walleye to sites containing dense fathead minnow (*Pimephales promelas*) populations sometimes bolster abundance of macroinvertebrates and zooplankton, and favors clear water shifts (Potthoff et al. 2008). However, it is currently not possible to predict long-term consequences of walleye fry stocking in shallow lakes with unknown fish communities, or where adult fish are well established.

During May 2007-September 2008, we monitored ecological characteristics and habitat suitability for waterfowl at Kenogama. Our objectives were to: (1) document current ecological conditions within the lake; (2) assess characteristics of the lake's current fish community; (3) describe the invertebrate community, with special emphasis on selected taxa known to be important for water quality and as waterfowl food; and (4) draw broad comparisons between Kenogama and other shallow MN lakes recently studied. Here, we summarize results of our efforts during May-September 2007 and 2008; we also discuss implications of our findings and offer some hypotheses about current characteristics and possible changes in the lake.

METHODS

During May 2007, we chose 6 transects by randomly selecting 6 compass bearings from the approximate center of the lake. All sampling for aquatic invertebrates and fish was conducted along these transects during the study.

Fish Community

Relative abundance and species composition of Kenogama's fish population was assessed using 3 gill nets and 12 mini-fyke (small trap) nets during June and August 2007 and 2008. For each sampling, a single mini-fyke net was deployed along the shore, or at the deep margin of emergent vegetation along each of the 6 transects. Gill nets were set concurrently, 1 each at the deepest location along transects 2, 4, and 6. Sampling gear was deployed in the morning and checked approximately 24 hours later. Fish were identified to species, and wet weights (g) and total lengths (mm) were determined in the field. During 2007, random samples of stomachs and otoliths were taken from walleye. Because we were especially concerned with population characteristics and functional influences of walleye, we also examined walleye length at age distribution, length frequency, relative weights (Wr, Pope and Carter 2007), and stomach contents. During late May 2008, DNR Fisheries staff stocked approximately 7 million walleye fry into Kenogama. Adult walleye were absent in 2008, apparently due to winter hypoxia (winterkill), but stomach analysis was conducted on young of the year (age-0) walleyes, yellow perch (*Perca flavescens*), and golden shiners.

Aquatic Invertebrates

Two sampling stations were established along each of the 6 transects, one 20 meters from the edge of the emergent vegetation and a second at the midpoint between the shoreline and the lake center. Aquatic invertebrates were sampled at the 12 stations using column samples (CS, Swanson 1978) and vertical activity traps (AT, after the design of Muscha et al. 2001) at approximately 2-3 week intervals (6-8 times each summer). AT samplers were used at deep and shallow locations; CS were taken at deep sites only. AT were deployed for approximately 24 hrs, then samples were condensed by passage through an 80 μm-mesh funnel. CS were concentrated by passage through a 64 μm-mesh funnel. Both CS and AT samples were preserved in 70% ethanol.

Aquatic macroinvertebrates were sampled during August and September 2008 using 500 μm mesh D-frame sweep net. Sweep nets were gathered concurrently from deep and shallow locations from 3 of the 6 transects. Substrate samples were taken by skimming the net along the bottom sediment for a distance equal to the depth at the sampling station described by Anteau and Afton (2008). Contents of the sweep nets were then concentrated with a 243-μm mesh wash net to reduce the amount of decaying organic material. Contents were put in plastic bags, refrigerated and later identified.

Invertebrates from CS, AT, and sweeps were identified to the lowest feasible taxonomic group (mostly family, sometimes genus) and were counted in a lab at Bemidji State University. We pooled organisms into 11 groups: all insects, all Diptera (Chaoboridae, Chironomidae, Culicidae), Corixidae, Ephemeroptera, Amphipoda, large cladocera (mainly *Daphnia, Ceriodaphnia, Simocephalus,* and Sididae), small cladocera (Chydoridae, Bosminidae, *Diaphanosoma* and *Eurycercus*), cyclopoid copepods, calanoid copepods, and *Leptodora*. We combined results of all CS and AT samples on each sampling date to develop a relative abundance estimate for each of the 11 groups. Organisms captured using sweep nets (2008 only), were pooled to form the following groups: Chironomidae, Hydracarina, *Leptodora,* Oligochaeta, *Hyalella*, Physidae, other Insects (including Diptera, Ephemeroptera, Trichopetera, Anisoptera) and Hirundea. We assessed trends in major taxa graphically.

Plant Community

Relative abundance of submerged macrophytes was measured during August 2007 and 2008 using methods of Deppe and Lathrop (1992) (an approach generally similar to that used by MNDNR Section of Wildlife Shallow Lakes Program staff and by researchers from MNDNR Wetland Wildlife Group). For plant surveys, we selected 8 transects, with 5 sampling locations equidistant from one another and from shorelines. At each location, we collected plants using 2 casts of a weighted plant rake. We recorded presence/absence of individual submergent species retained on each cast. A maximum score of 40 would indicate that a particular species was collected on each cast at all sampling stations.

We compared Kenogama to other shallow lakes in Minnesota using 2 procedures. First, we constructed a plant relative-abundance matrix by combining Kenogama plant survey results with similar data from a recent (2006) study of 74 shallow lakes in MN (Herwig et al. 2006). We then used Non-metric Multidimensional Scaling (NMS) to assess similarity between plant communities of Kenogama and the other shallow lakes. Second, we plotted plant extent (% vegetated points) and water clarity (average Secchi/average lake depth) of Kenogama along with values from other shallow lakes in a data set supplied by the MNDNR Section of Wildlife Shallow Lakes Program (Nicole Hansel-Welch, unpublished data). These approaches should not be considered formal statistical tests, but they do allow visual contrasts of potential similarities or differences between Kenogama and other shallow MN lakes.

Chemical Properties and Water Quality Features

We also measured water clarity, phytoplankton abundance (indexed using chlorophyll *a* (Chl a)), and concentrations of major nutrients at approximately 2-3 week intervals during open water periods from 31 May-18 September 2007 and 15 May-17 September 2008. Chemical analyses were performed from surface-dip water samples collected at 3 central-lake locations. Secchi disk transparency was measured at these 3 sites using a standard (20-cm) circular disk. We measured turbidity directly using a LaMotte turbidity meter following transport of water samples back to the lab. Water samples collected for determination of total phosphorus (TP), total nitrogen (TN), and ammonium (NH_4) were frozen and transported to laboratory facilities. Water samples were also collected and later analyzed for total dissolved phosphorus (TDP) and phytoplankton abundance (Chl a). We also measured TDP because this is sometimes more useful than TP for evaluating ecological change in shallow lakes (Potthoff et al. 2008). TDP samples were prepared by filtering lake water through GF/F glass fiber filters (0.7 μm nominal pore size) and immediately freezing the filtered water. Chl a samples were prepared by filtering lake water through GF/F glass fiber filters which were wrapped in tin foil and immediately frozen. TDP concentrations were determined using high-temperature persulfate digestion followed by ascorbic-acid colorimetry. Chl a was measured via fluorometric analysis following a 24-h, alkaline-acetone extraction of photosynthetic pigments. All chemical procedures for analysis of Chl a, TP, TDP, TN, and NH_3 were performed using laboratory facilities at the University of St. Thomas (St. Paul, MN). Evaluation of data trends was done graphically.

Relative Water Depth

Relative water level readings were recorded approximately biweekly from May-September 2007 and 2008 by reading a depth gauge near the boat access. On 8 June 2007, using a Lowrance sonar unit, we also measured water depth at various locations around the lake.

RESULTS

Fish Community

Fourteen and 10 fish species were captured during 2007 and 2008, respectively (Tables 1, 2). In 2007, golden shiner, fathead minnow and walleye were the most abundant fishes (highest relative mass, Table 1). Golden shiners and brook stickleback (*Culaea inconstans*) were most abundant in 2008 (Table 2). Golden shiners from several size (year) classes were captured in mini-fyke nets during both years; gill nets also collected golden shiners, but only larger sizes (135-185 mm) representing older year classes. Gill net catches were generally lower during 2008, when very few golden shiners, yellow perch, and walleye were collected in gill net samples.

We observed 3 peaks in length frequency of walleye collected during 2007 and walleye length increased from June to August, reflecting summer growth (Figure 1). Age-assignment based on otoliths confirmed 3 (or more) year-classes indicated by length-frequencies (Figure 1). Randomly selected walleye ranging from 230–300 mm were age-1 (2006 year class), fish ranging from 360–430 mm were age-2 (2005 year class), and larger walleye ranged from ages 3-5. No adult walleye were collected during 2008, but young of the year walleye that were stocked in May were collected during August that year.

Walleye appeared to be in good condition during summer 2007, with an average relative weight (W_r) ranging between 0.8 and 1.0 (Figure 2). Smaller fish achieved highest W_r during June 2007, whereas larger fish appeared healthier during August. In general, W_r values were negatively associated with total length during June 2007, but an opposite (positive) correlation was evident by August.

Summer diet of adult walleye (2007) consisted primarily of aquatic invertebrates (Table 3). Amphipods comprised the major percentage of food found in walleye stomachs during June (32.7% wet mass, n = 8) and August (58.3% wet mass, n = 9). Minnows (cyprinids) were absent in walleye stomachs during June, but occurred in 16.7 % of stomachs examined in August (31.3% wet mass, n = 9). Other food items present in walleye stomachs were Decapoda (crayfish), Hirundea (leeches), and larval insects. Almost one-fourth of walleye stomachs were empty, and considerable proportions of stomach contents (19.4%) were decomposed, thus were unidentifiable. In August 2008, the walleye population consisted entirely of age-0 fish, and stomachs contained mostly of cyprinids (94.2% wet mass, n = 9) (Table 4). Forty percent of stomachs were found to be empty, and contents of 20% (n = 5) were unidentifiable. Stomach contents were also taken from age-0 yellow perch in August 2008. Yellow perch diets included zooplankton (such as cladocera, 25% wet mass, n = 4) and some vegetation (20.5% wet mass, n = 4), and a lesser quantity of minnows (cyprinids, 13.6% wet mass, n = 2). Approximately one-third of the stomachs examined contained unidentifiable material (31.8% wet mass, n = 5), and 28% were empty. In August 2008, golden shiner stomachs showed a high presence of zooplankton (38.5% wet mass, n = 5), along with some Diptera (15.4% wet mass, n = 1) and vegetation (15.4% wet mass, n = 2) present. Most of the golden shiner stomachs examined were empty (60%), and almost one-third of the total proportion of stomach contents was unidentifiable (30.8% wet mass, n = 2).

Aquatic Invertebrates

Zooplankton samples during late May–September 2007 were numerically dominated by small-bodied cladocerans and copepods, but large cladocerans were also abundant during early August 2007 and late May, June and September 2008 (Figure 3a,b). Small cladocerans were present during both years, but were especially abundant during early 2007 (Figure 3a). Density of small cladocerans was variable during 2008, but peaked in June and August (Figure 3b). Large cladocerans occurred in relatively low numbers during 2007 (except early August), but were much more abundant in 2008, with peaks in late May and mid-September (Figure 3a,b). *Leptodora* (large, predatory cladocera) were absent in samples from the first 3 dates in 2007, appeared in July, but returned to low densities by mid-September (Figure 3a). In 2008, *Leptodora* were present at low density, but peaked briefly during August (Figure 3b).

Calanoid copepods were abundant but variable throughout 2007–2008 (Figure 4a,b). Throughout most of 2007 and 2008, cyclopoid copepods were present and less abundant than calanoids except for a peak during May–June 2008 (Figure 4b). In general, calanoid copepods were more abundant than were cyclopoids in both study years (a pattern opposite what we usually observe in shallow lakes elsewhere; Figure 4a,b).

Amphipods (*Hyalella* only) were collected in low numbers throughout 2007, but increased briefly during early-summer 2008 (Figure 5a,b). Amphipods ranged from 0 (31 May and 1 August) to 4 individuals in late July 2007 and remained similarly low during August–September 2008. Diptera were the most abundant insects during all sampling periods (except late September 2007). Corixidae (water boatmen) were periodically collected in 2007, but densities remained very low (< 5 individuals, lake-wide) (Figure 6a). Corixidae densities were higher in 2008, but they were never abundant (Figure 6b). Ephemeroptera (mayflies) occurred periodically in 2007, ranging from 0 (7 June and 1 August 2007) to 18 individuals (20 June 2007), with a peak during June (Figure 6a). Ephemeroptera also occurred periodically in 2008, with numbers ranging from 0 (15 May and 19 August 2008) to 26 individuals (26 June 2008) (Figure 6b). Peak density was observed in early summer (late May to late June) and numbers decreased steadily (Figure 6b).

Some aquatic macroinvertebrates were captured in greater numbers in sweep nets than with column samplers and activity traps. For example, Chironomidae were predominant in the 2008 sweep net sampling, with a peak in August (394 individuals/net), then declining in

September (73 individuals/net). Mean *Hyalella* catches in sweeps ranged from 105 individuals in August to 6.83 in September (2008 only). Other taxa were present at very low densities in sweep net catches (e.g. Hydracarina, Oligochaetes, Figure 7) and these were not represented in AT or CS samples.

Water Clarity, Phytoplankton, and Major Nutrients

Water clarity, represented by the Secchi depth:mean depth ratios, followed typical summer patterns, reaching seasonal highs early, then decreasing to lower levels by fall (2007 and 2008; Figure 8a,b). Annual ratios of mean Secchi/depth remained well above values of 0.5 (a theoretical maximum for depth of the photic zone when light penetration equals depth, Figure 12b), indicating that most of the lake bed received sufficient sunlight to facilitate growth of submerged aquatic plants.

TP values in Kenogama remained relatively low and nearly constant throughout May–August 2007, ranging from slightly below, to slightly above, 25 *u*g L^{-1} (Figure 9a). We observed a very different pattern during 2008, when TP was increased by a factor of approximately 3 and fluctuated throughout the summer (Figure 9b). Dissolved phosphorus was present during both years, but with slightly higher values and greater variability during 2008. Throughout 2007, TDP comprised >50 % of the TP pool, but in 2008 TDP was only about 25-30% of the TP pool.

Phytoplankton abundance remained very low during May–August/September, with mean observed values ranging from 5.5–8.5 and 3.4–9.3 *u*g L^{-1} in 2007 and 2008, respectively (Figure 10a,b). Ratios of TP:Chl a (nutrients:phytoplankton abundance) were relatively high during both years, indicating that a considerable portion of the TP pool was probably not associated with phytoplankton (Figure 10a,b). During 2008, a considerable portion of the water-column phosphorus was unaccounted for by phytoplankton biomass (Figure 10b) or dissolved pools (Figure 9b).

Relative Water Depth

Relative water depth generally decreased as the sampling season progressed (by approximately 0.7 ft (0.21 m)) during May– mid-September 2007 (Figure 11a) and by approximately 0.8ft (0.24 m) during May–mid-September 2008 (Figure 11b). We assessed depths lake-wide on 8 June 2007, when depth ranged from 3.1 – 4.9 ft (0.94 – 1.49 m) in various locations.

Submerged Aquatic Plants

Submerged aquatic plants were collected from 100% of our sampling locations during 2007 and 2008, but subtle differences were evident between years. Seven species were present during 2007, but, only 3 were widespread. These included Robbins' pondweed (*Potamogeton Robbinsii*, collected at 100% of sites), bushy pondweed (*Najas flexilis*, collected at 65% of sites), and large-leaf pondweed (*Potamogeton amplifolius*, collected at 43% of sites). Flatstem pondweed (*Potamogeton zosterformis)*, whitestem pondweed (*Potamogeton praelongus*), 1 *Sagittaria* spp., and 1 unidentified pondweed (*Potamogeton spp.*) were also collected, but these were far less abundant.

In 2008, submerged aquatic plants were again present at 100% of the sampling sites, but only 5 species were found. Only Robbin's pondweed (collected at 98% of sites) and large-leaf pondweed (collected at 63% of sites) were widespread. Bushy pondweed (collected at 20% of sites), *Sagittaria* spp. (collected at 13% of sites), and variable pondweed (*Potamogeton gramineus*, collected at 8% of sites) were present, but far less abundant.

Kenogama Lake's plant community differed from that observed in other shallow lakes recently sampled in prairie and parkland areas of Minnesota (Figure 12). Most notable was the widespread occurrence of Robbin's pondweed, which occurs throughout Kenogama, but was not collected from other shallow lakes sampled by our research group during 2007-2008.

DISCUSSION

Shallow lakes in North America show dramatic contrasts in lake features, typically conforming to either clear- or turbid-water regimes (Hanson and Butler 1994a,b, Scheffer 2004, Zimmer et al. in press). During 2007-08, Kenogama showed evidence of transition between regimes, with characteristics of both clear- and turbid-water states.

Trends in water transparency were similar between the 2 years of study. Water transparency was initially high and then decreased during the warmer months, with relatively clear water returning by late August–September. Water levels decreased as the summer progressed in both 2007 and 2008. Water levels were consistently higher in 2008, probably due to heavy spring precipitation.

Kenogama supported a diverse fish community during both 2007 and 2008. Species richness was 14 and 10 during 2007 and 2008 respectively, with lower values in 2008 probably reflecting influence of winterkill. The fish community was comprised of mostly planktivorous species during both years, with walleye being a notable exception. Benthivorous fish were not abundant and were not sampled in 2008. In 2007 1 white sucker (*Catostomus commersoni*) and 1 yellow bullhead (*Ameiurus natailis*) were captured. We collected no adult walleyes (>age 0) during 2008, apparently due to hypoxia and winterkill. Winterkill and lack of adult walleyes may help explain other changes such as increased amphipods and large cladocera, along with increased relative abundance of yellow perch in 2008.

Kenogama's fish community continued to differ markedly from those observed in other Minnesota shallow lakes, mostly due to its high golden shiner population. This dense population of golden shiners may result, in part, from accidental loss and dumping of bait by anglers. Fish sampling during 2007 confirmed that the lake supported an extensive population of adult walleyes resulting from walleye rearing activities and the incomplete removal of these fish during fall netting. During 2008, walleye biomass was again high by August 2008, but only age-0 fish were sampled, almost certainly due to severe hypoxia in winter 2007-08.

The combination of dense golden shiners (planktivores) concurrent with an established walleye population (piscivores) seems to contradict recent evidence indicating that walleye stocking limits planktivore abundance (Potthoff et al. 2008). A partial explanation for coexistence of these species may be that the extensive stands of submerged macrophytes in Kenogama provide refuge areas for golden shiners and other planktivores, thus uncoupling predator and prey densities. Because golden shiners achieve larger body size (than fathead minnows, for e.g.), it is also plausible that walleye predation did little to limit abundance or recruitment of the shiners.

Walleye diets consisted of macroinvertebrates and fish, but differed between 2007 and 2008, perhaps due to our methods (we examined only adult walleyes in 2007, then only juveniles in 2008). During 2008, cyprinids comprised over 90% of age-0 walleye diet. In contrast during 2007, amphipods were important food items of adult walleyes (40.7% wet mass), along with Decapoda (crayfish), Hirudinea (leeches), and Odonata (dragonflies); minnows (9.7%) were less important than a combination of other macroinvertebrates. It seems counterintuitive that young walleye in 2008 were targeting cyprinids and the older, larger fish were consuming macroinvertebrates. During 2008, yellow perch stomachs from (age-0 fish) contained mostly Branchiopoda (zooplankton; 25% wet mass) along with a large amount of vegetation (20.5%). Vegetation was probably due to incidental digestion from feeding in and around aquatic plants.

We did not expect to find that minnows were a major food source for fingerling walleyes in Kenogama during 2008, especially after observing little predation on minnows by adult walleyes in 2007. Ward et al. (2008) observed a shift from zooplankton to fish to macroinvertebrates during seasonal walleye development. It is plausible that walleye predation constrains zooplankton and macroinvertebrates in Kenogama, similar to the influence of walleye reported for large prairie wetlands in west-central Minnesota (Reed and Parsons 1999). However, given abundance of golden shiners and yellow perch, their consumption of zooplankton and macroinvertebrates probably exceeded that of the walleye population during the 2 years of our study. It is also likely that the extreme annual variability (and perhaps prey consumption patterns) reflect our small sample sizes and examination of walleye fry and adults during separate years.

During 2007–2008, Kenogama exhibited characteristics of a clear-water regime with widespread (but not lush) submergent or emergent macrophytes. This is consistent with our observation that sunlight penetration exceeded average lake depth during both years. Kenogama supported a relatively sparse invertebrate community in 2007, with a high proportion of small-bodied cladocerans (inefficient filter-feeders on phytoplankton) and relatively sparse macroinvertebrates. This contrasts somewhat with 2008, when we observed increases in small amphipods (*Hyalella*), large-bodied cladocera, and some aquatic insects. These patterns may signal a transition to clear-water regime conditions, or they may simply reflect absence of adult walleye during 2008, which were evidently foraging on both amphipods and aquatic insects during 2007 (51% total stomach mass). During 2008, sweep nets were also used to assess organisms closely associated with the bottom substrate. These results suggest that some invertebrates including those important as waterfowl food items (*Hyalella*, Chironomidae) were substrate-associated, thus may be poorly represented in data from column samples and activity traps. During both 2007 and 2008, density of most organisms drastically decreased between August and September, which may reflect reduced activity of invertebrates as waters cool, high consumption by fish (following a season's growth), or even migrating waterfowl foraging on the lake.

Presently, some patterns at Kenogama reflect characteristics of both clear- and turbid-water regimes. For example, submerged macrophytes are widespread throughout the lake and phytoplankton abundance remains low. At the same time, planktivorous fishes are abundant, richness of the submerged plant community is quite low, and TP:Chl a ratios are high. These findings may indicate that the lake is balancing near a threshold between clear and turbid regimes, perhaps due to sustained high water levels and a persistent, high-density population of planktivorous fishes.

Presently in Minnesota, there is a need for better understanding of basic ecological characteristics of shallow lakes in forested regions and elsewhere. Managers need to know typical ranges of conditions for shallow lakes in forested landscapes, and whether these lakes always provide good habitat for invertebrates and wetland wildlife simply because they exhibit clear water and support abundant submerged plants. Staff from the MNDNR Wetland Wildlife Populations & Research Group, MNDNR Fisheries, and the University of St. Thomas (St. Paul, MN) plan studies at Kenogama, along with approximately 15 other Laurentian Forest lakes and 135 additional shallow lakes statewide during 2009–2011. We expect that this work will increase basic knowledge of ecological characteristics of shallow lakes throughout Minnesota. More specifically, we hope that resulting data will improve understanding of regional patterns of variability in shallow lakes and identify mechanisms responsible for triggering shifts to turbid regimes (with poor water quality and little wildlife use). We are hopeful that additional data from Kenogama and similar forest lakes will provide practical guidance to resource managers in the Laurentian Forest. This is especially important because the mechanisms controlling shallow lake characteristics and water quality remain poorly understood, limiting our ability to properly manage these areas for wetland wildlife.

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Table 1. Relative abundance (mean weight (g) of catch per overnight set, standard error in parentheses) of fishes in Lake Kenogama, Minnesota, during summer 2007.

		14-Jun-07			9-Aug-07		
		Trap net	Gill net	Activity trap	Trap net	Gill net	Activity trap
Species	Common name	*N* = 12	*N* = 3	*N* = 12	*N* = 12	*N* = 3	*N* = 12
Cyprinidae							
Hybognathus hankinsoni	Brassy minnow	0.4(0.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Notemigonus crysoleucas	Golden shiner	3759.8(791.6)	277.3(17.3)	0.0(0.0)	26.5(9.7)	150.2(93.0)	0.0(0.0)
Notropis heterolepis	Blacknose shiner	23.5(5.1)	0.0(0.0)	0.0(0.0)	4.4(2.5)	0.0(0.0)	0.0(0.0)
Phoxinus eos	Northern redbelly dace	68.5(27.8)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus neogaeus	Finescale dace	11.3(5.4)	0.0(0.0)	0.0(0.0)	0.4(0.4)	0.0(0.0)	0.0(0.0)
Pimephales promelas	Fathead minnow	292.6(129.6)	0.0(0.0)	0.3(0.3)	6.9(2.7)	0.0(0.0)	0.0(0.0)
Catostomidae							
Catostomus commersoni	White sucker	0.0(0.0)	720.0(361.2)	0.0(0.0)	0.0(0.0)	412.3(296.7)	0.0(0.0)
Ictaluridae							
Ameiurus natalis	Yellow bullhead	48.3(48.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Umbridae							
Umbra limi	Central mudminnow	3.0(1.6)	0.0(0.0)	1.0(0.8)	17.9(13.5)	0.0(0.0)	0.5(0.5)
Gasterosteidae							
Culaea inconstans	Brook stickleback	38.3(16.6)	0.0(0.0)	3.4(2.2)	0.2(0.2)	0.0(0.0)	0.2(0.1)
Percidae							
Etheostoma exile	Iowa darter	1.7(1.1)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Etheostoma nigrum	Johnny darter	0.3(0.2)	0.0(0.0)	0.0(0.0)	0.0(0.2)	0.0(0.0)	0.0(0.0)
Perca flavescens	Yellow perch	0.0(0.0)	0.0(0.0)	0.0(0.0)	3.3(3.3)	303.7(303.7)	0.0(0.0)
Sander vitreus	Walleye	260.5(180.5)	9445.7(1304.6)	0.0(0.0)	214.2(94.2)	11447.7(2868.0)	0.0(0.0)

Table 2. Relative abundance (mean weight (g) of catch per overnight set, standard error in parentheses) of fishes in Lake Kenogama, Minnesota, during summer 2008.

		18-Jun-08		20-Aug-08	
		Trap net	Gill net	Trap net	Gill net
Species	Common name	*N* = 12	*N* = 3	*N* = 12	*N* = 3
Cyprinidae					
Notemigonus crysoleucas	Golden shiner	3270.8(761.3)	29.7(29.67)	312.1(66.3)	53.3(53.3)
Notropis heterolepis	Blacknose shiner	48.8(21.9)	0.0(0.0)	12.0(7.6)	0.0(0.0)
Notropos hudsonius	Spottail Shiner	6.0(2.3)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus eos	Northern redbelly dace	353.8(70.4)	0.0(0.0)	0.0(0.0)	0.0(0.0)
Phoxinus neogaeus	Finescale dace	243.3(93.0)	0.0(0.0)	185.4(20.7)	0.0(0.0)
Pimephales promelas	Fathead minnow	510.0(117.2)	0.0(0.0)	14.6(10.9)	0.0(0.0)
Umbridae					
Umbra limi	Central mudminnow	103.3(41.5)	0.0(0.0)	355.8(57.3)	0.0(0.0)
Gasterosteidae					
Culaea inconstans	Brook stickleback	2352.1(667.0)	0.0(0.0)	2781.3(535.0)	0.0(0.0)
Percidae					
Perca flavescens	Yellow perch	10.8(6.0)	450.3(391.87)	1290.0(442.2)	123.7(123.7)
Sander vitreus	Walleye	0.0(0.0)	0.0(0.0)	1095.0(191.7)	7.7(7.7)

Table 3. Percent by weight and prevalence (percent of stomachs containing a food item) of stomach contents of walleyes in Lake Kenogama, Minnesota, summer 2007.

	6/14/2007 *N* = 8 stomachs examined Empty stomachs = 1 (12.5%)		8/9/2007 *N* = 9 stomachs examined Empty stomachs = 3 (33.3%)		Overall *N* = 17 stomachs examined Empty stomachs = 4 (23.5%)	
Food	Percent by weight	Prevalence	Percent by weight	Prevalence	Percent by weight	Prevalence
Hirudinea	28.2	25			9.4	11.8
Crustacea						
Amphipoda	32.7	25	58.3	66.7	40.7	35.3
Decapoda	15.5	12.5			10.7	5.9
Insecta						
Odonata	9.3	12.5	2.1	16.7	7.1	11.8
Ephemeroptera			8.3	33.3	2.6	11.8
Diptera	0.1	12.5			0.3	5.9
Pisces						
Cyprinidae			31.3	16.7	9.7	5.9
Unidentified	28.2	37.5			19.4	17.6

Table 4. Percent by weight and prevalence (percent of stomachs containing a food item) of stomach contents of fish in Lake Kenogama, Minnesota, summer 2008.

	Walleye *N* = 25 stomachs examined Empty stomachs = 10 (40%)		Yellow Perch *N* = 25 stomachs examined Empty stomachs = 7 (28%)		Golden Shiner *N* = 25 stomachs examined Empty stomachs = 15 (60%)	
Food	Percent by weight	Prevalence	Percent by weight	Prevalence	Percent by weight	Prevalence
Hirudinea						
Crustacea						
Amphipoda			4.5	4		
Branchiopoda			25	16	38.5	25
Decapoda						
Insecta						
Odonata						
Ephemeroptera						
Diptera	0.9	4	4.5	4	15.4	4
Pisces						
Cyprinidae	94.2	36	13.6	8		
Unidentified	4.8	25	31.8	25	30.8	8
Vegetation			20.5	16	15.4	8

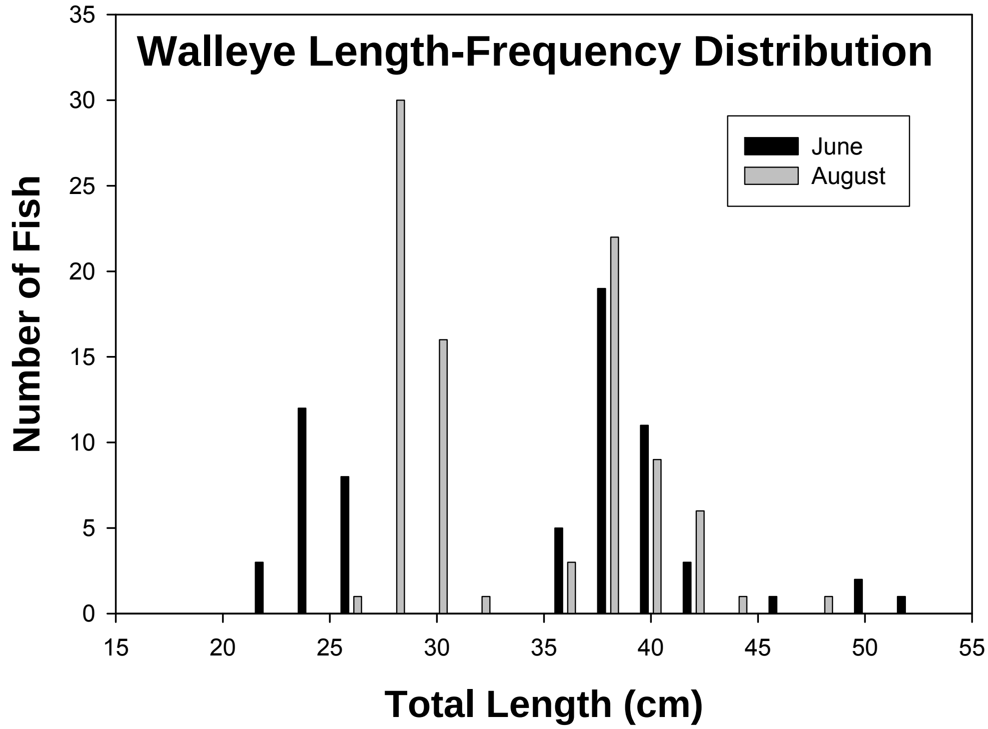


Figure 1. Length-frequency distribution of walleye captured in gill and mini-fyke nets during June and August 2007 in Kenogama Lake.

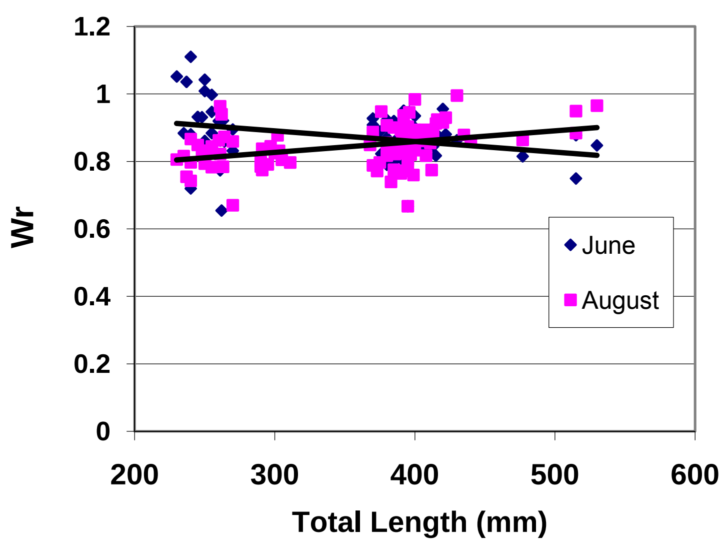
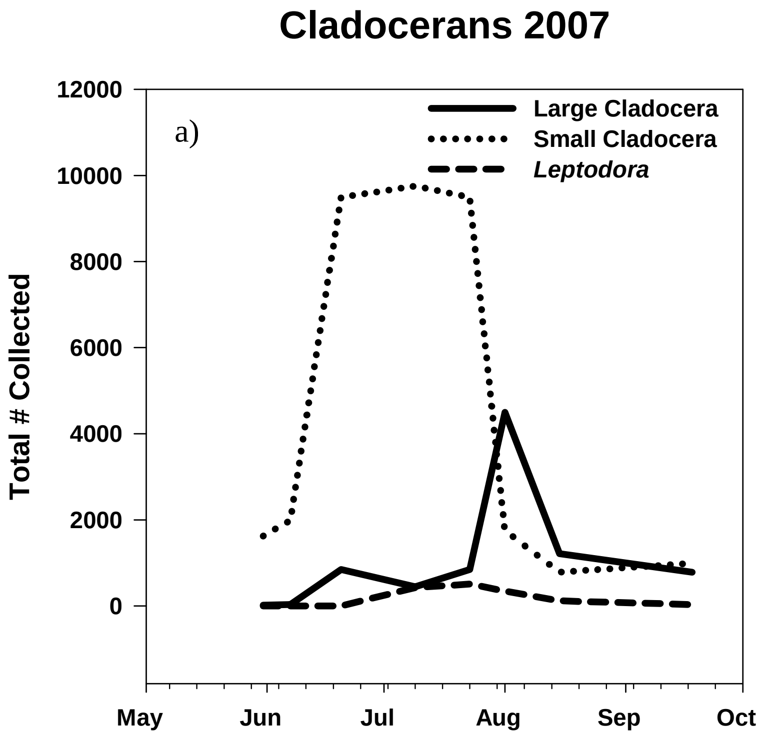


Figure 2. Relative weights (W_r) of walleyes sampled in Kenogama Lake, during June and August 2007 (June Wr = -0.0003x + 0.9853, R^2 = 0.098; August Wr = 0.0003x + 0.7308, R^2 = 0.137).



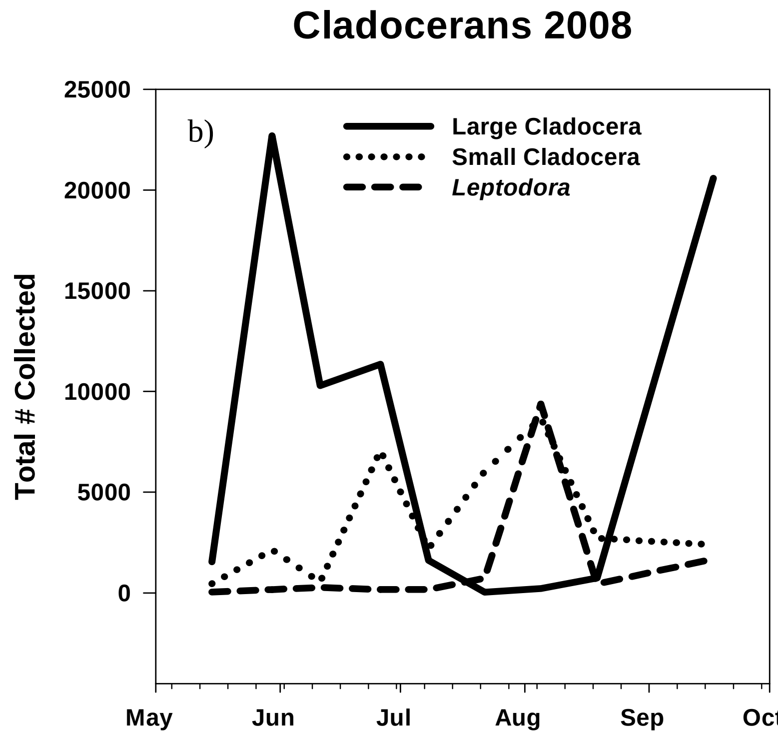


Figure 3. Seasonal patterns in total numbers of cladocerans captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Large cladocerans include *Daphnia, Ceriodaphnia,* Sididae, and *Simocephalus*; small cladocerans include Bosminidae, Chydoridae, *Diaphanosoma,* and *Eurycercus*.

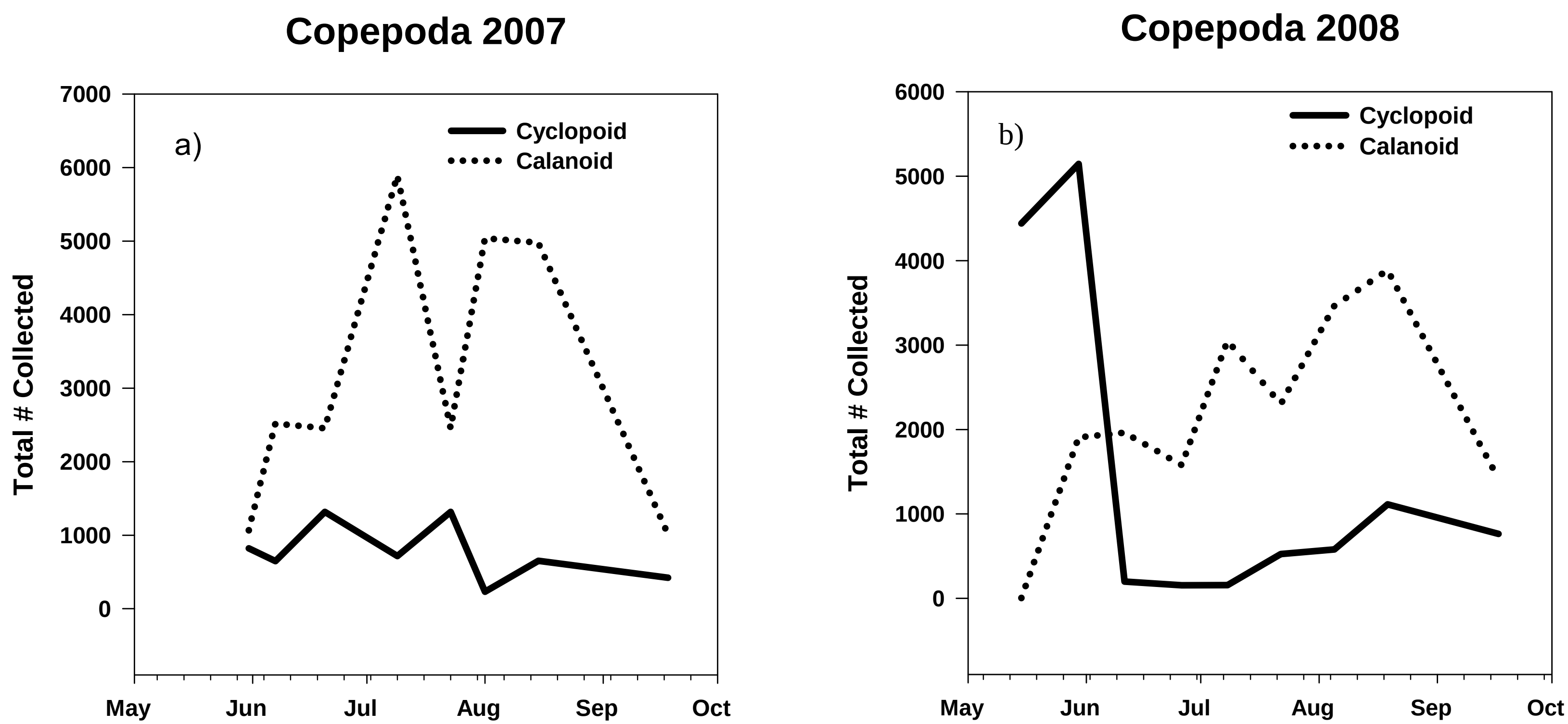


Figure 4. Seasonal patterns in total numbers of copepods captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Copepoda were classified to only suborder.

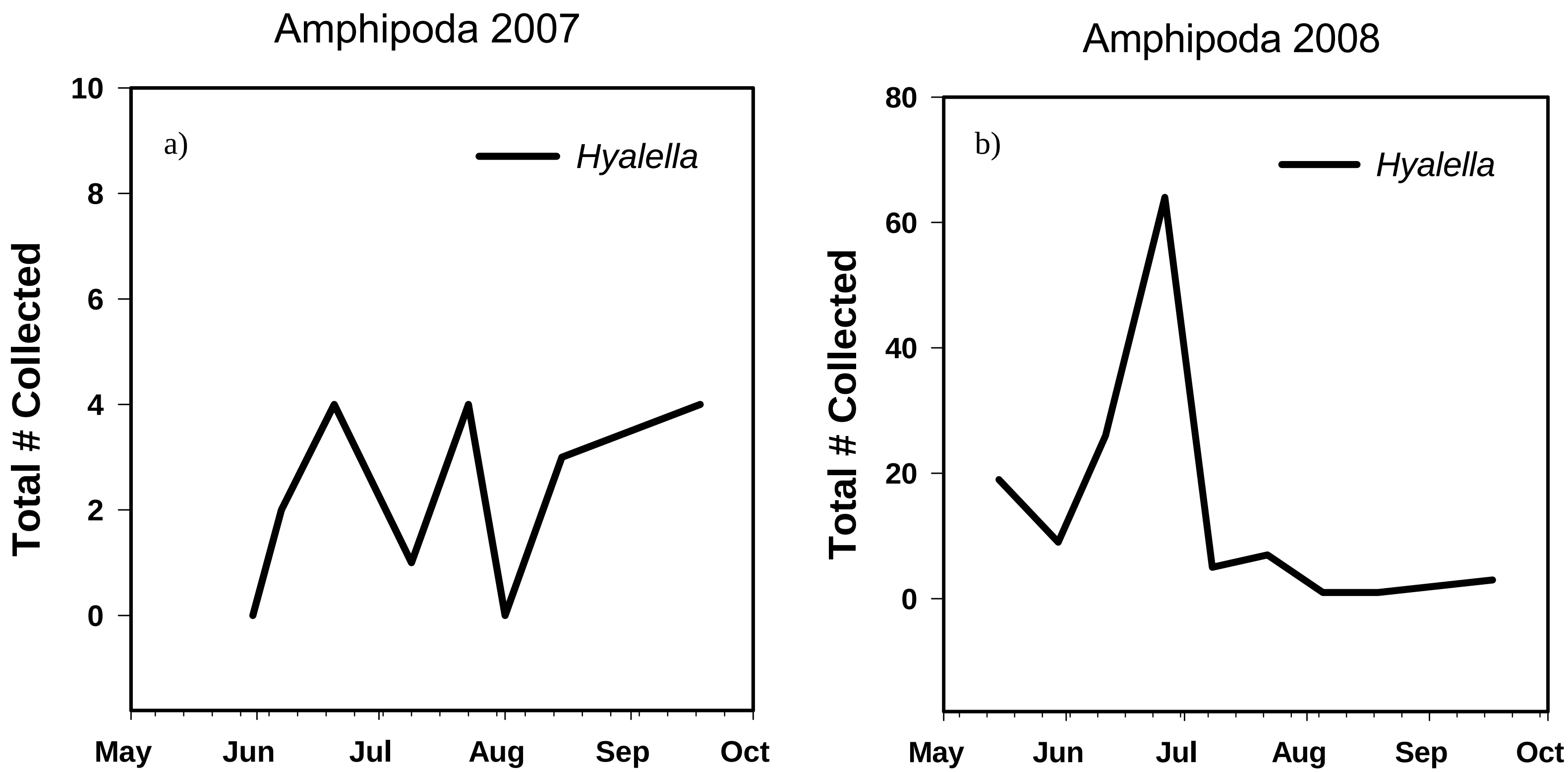


Figure 5. Seasonal patterns in total numbers of amphipods captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years.

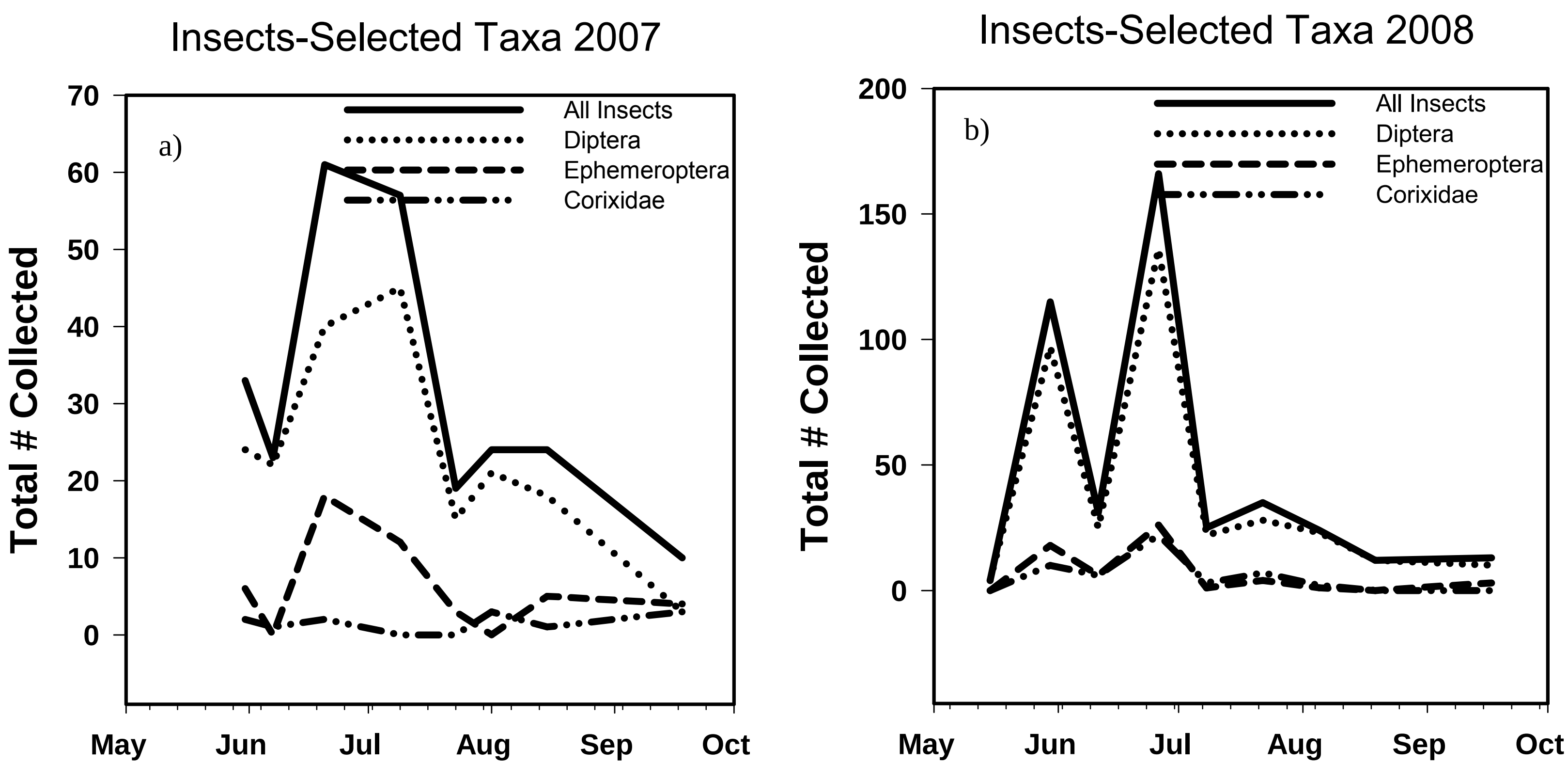


Figure 6. Seasonal patterns in total numbers of insects captured in activity traps (n = 12) and vertical column samplers (n = 12) on each sampling date of 2007 (panel a) and 2008 (panel b). Note the scales differ by years. Selected insect taxa groups of All Insects, Diptera, Ephemeroptera and Corixidae are shown.

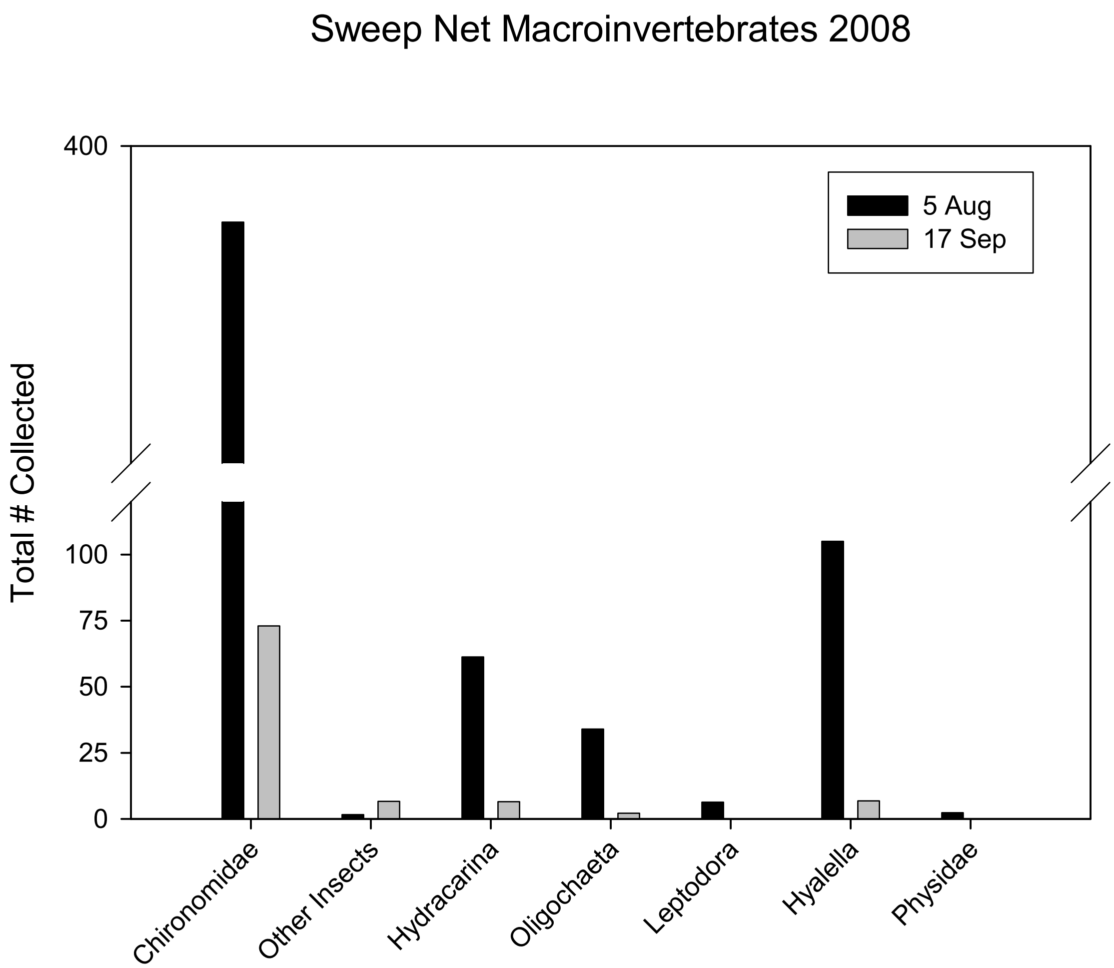


Figure 7. Mean density of aquatic macroinvertebrates collected in sweep nets in August and September 2008.

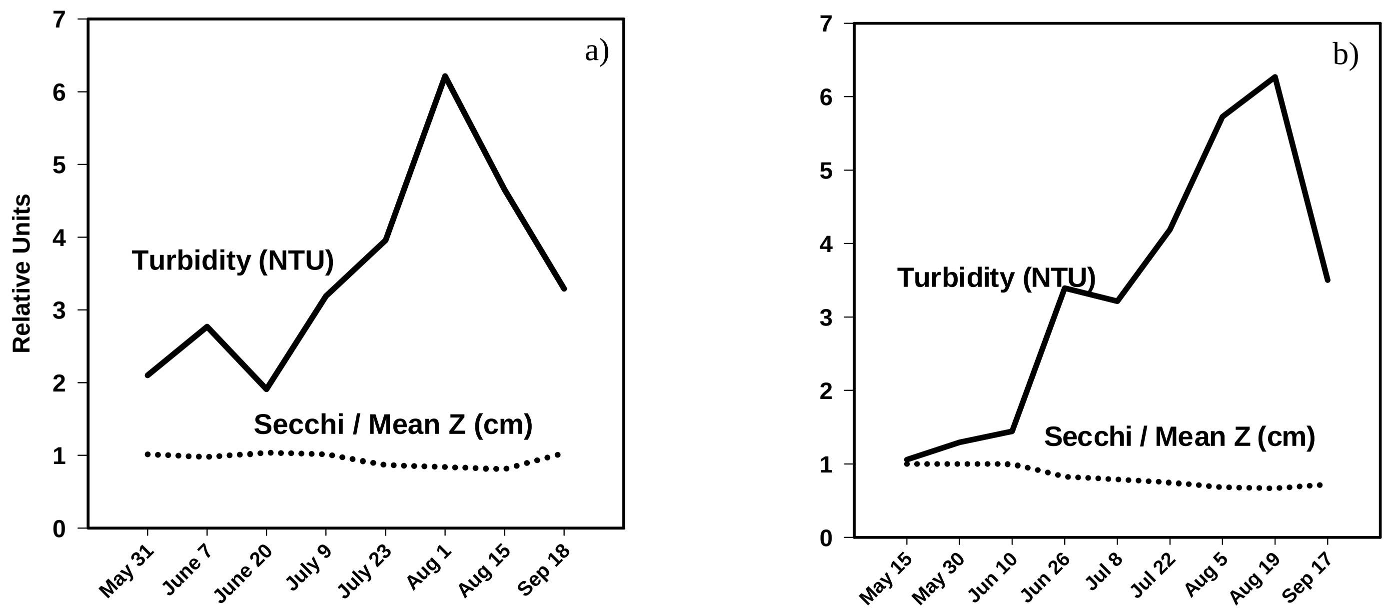


Figure 8. Water transparency patterns at Kenogama Lake during 2007 (panel a) and 2008 (panel b) depicted by ratio of Secchi disk transparency: mean lake depth, and turbidity measured using a nephelometer.

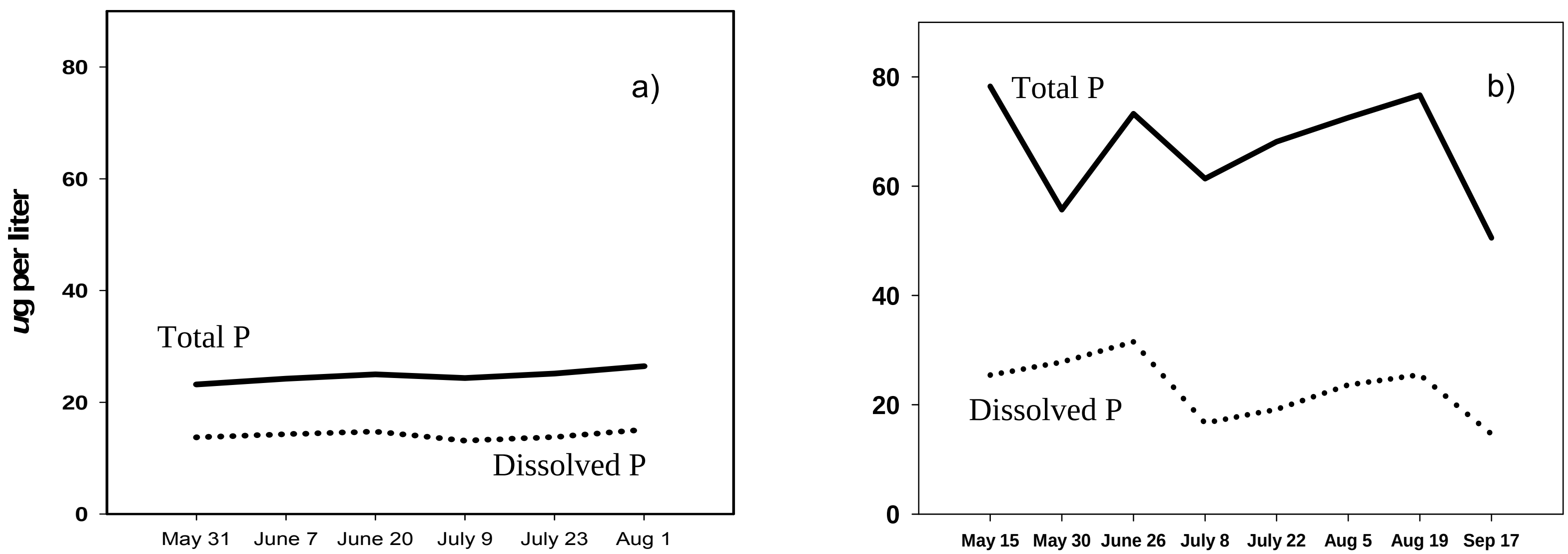


Figure 9. Patterns in mean phosphorus concentrations (total and dissolved forms, solid and dashed lines, respectively) at Kenogama Lake during 2007 (panel a) and 2008 (panel b).

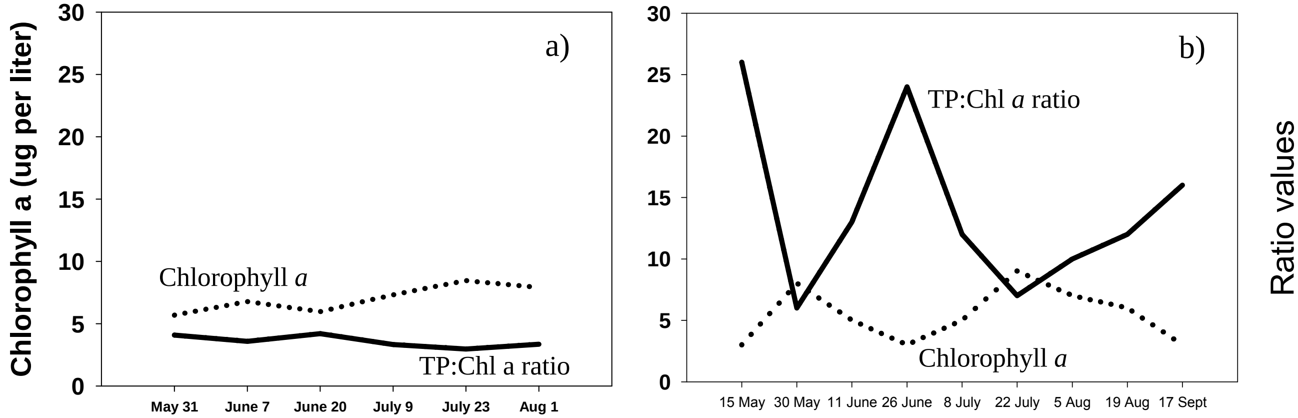


Figure 10. Phytoplankton abundance as indicated by water-column Cholorphyll a concentrations (dashed line) at Kenogama Lake during 2007 (panel a) and 2008 (panel b). Solid line depicts ratios of total phosphorus:chlorophyll *a* (same scale); ratio values above approximately 3 (as are all shown here) are often characteristic of lakes in a turbid-water regime (Dokulil and Teubner 2003).

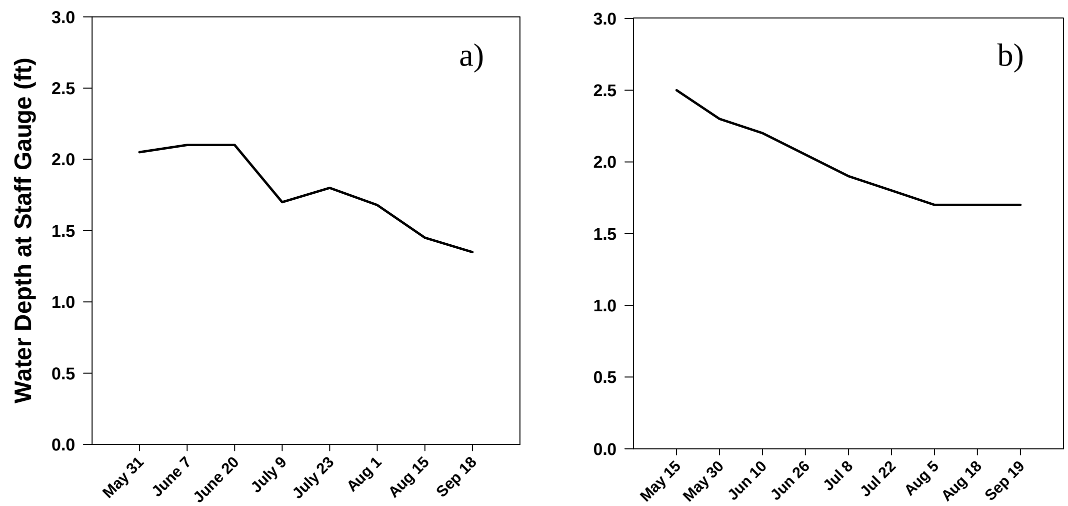


Figure 11. Water levels at the Kenogama Lake staff gauge during 2007 (panel a) and 2008 (panel b).

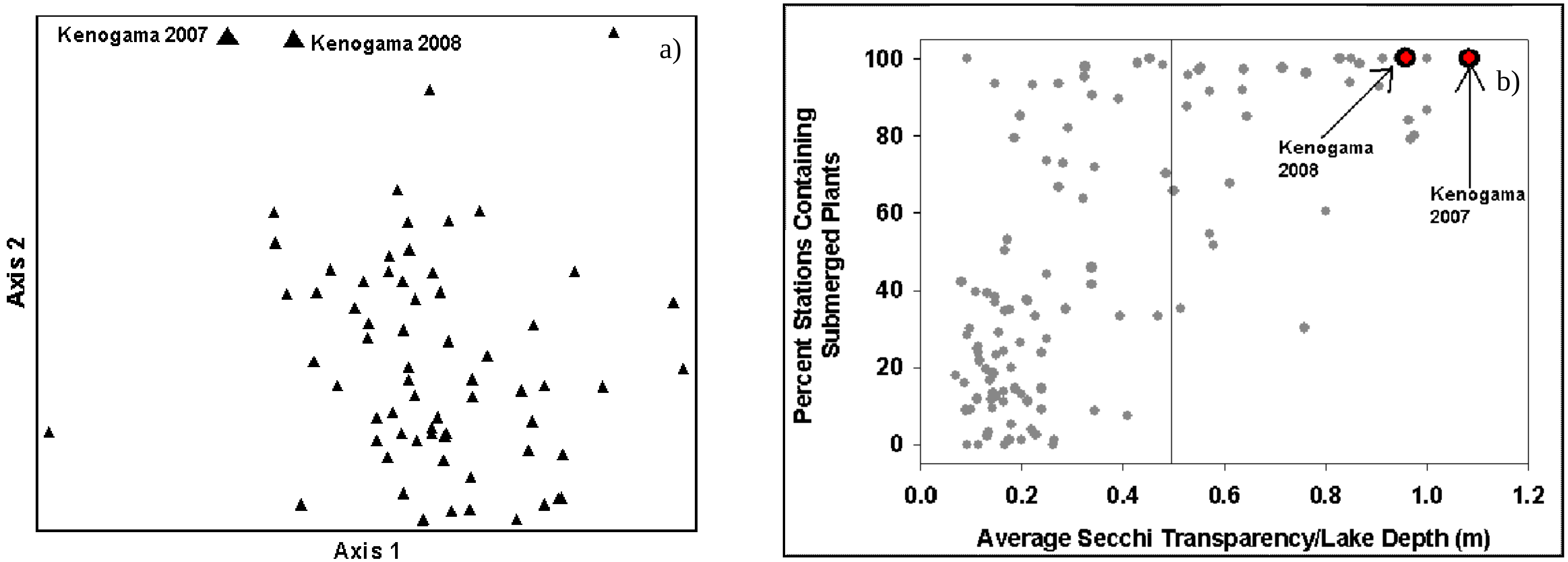


Figure 12. Plant community characteristics depicted by Non-metric Multidimensional Scaling (panel a) based on scores from a combined species matrix containing 71 shallow lakes in western and central MN, 2005 and 2006, and Lake Kenogama, 2007 and 2008. Panel b compares water clarity and macrophyte relationships of Kenogama and other shallow lakes surveyed by MNDNR Shallow Lakes Program (data provided by Nicole Hansel-Welch et al.). Vertical line (Secchi/depth value = 0.5) indicates approximate threshold depth where light penetration is sufficient to support rooted plants at mean lake depth. Separation in NMS space and water clarity/plant relationships indicate extent of similarity in abundance and species composition of Kenogama and water transparency relative to other shallow lakes recently studied in MN.