LOCAL RING-NECKED DUCK POST-FLEDGING MOVEMENT, SURVIVAL, AND REFUGE USE: A PILOT STUDY

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SUMMARY OF FINDINGS

Breeding ring-necked duck (*Aythya collaris*) populations have been increasing continentally, but appear to be declining in Minnesota. We initiated a pilot study in August 2006 to investigate post-fledging movement, survival, and the use of established refuges by locally produced ring-necked ducks. Between August 14 and 19, 2006, we captured and implanted radio transmitters subcutaneously in 25 locally produced, hatch year (HY) ring-necked ducks. We followed birds from the ground for 2 weeks, and then from the air until we lost contact with the last birds on October 19. We also set up 4 remote receiving stations on established waterfowl refuges. All birds survived the first 2 weeks following surgery. Retention of radios was a problem in the pilot study with at least 10 of 25 birds shedding transmitters prior to the end of the study. A different transmitter attachment strategy will be required in future years. The remote receiving stations worked well and we will set up receiving stations at 14 refuges in future years of the study.

INTRODUCTION

Minnesota's 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 appear to be declining in Minnesota. Further, hunter harvest of ring-necked ducks has declined markedly in the last 20 years even as numbers of these birds staging in fall on most traditional ring-neck refuges have increased (Wetland Wildlife Populations and Research Group, unpublished data). Factors influencing resident populations are poorly understood, and efforts to better understand their status began in 2003 with the development of a Minnesota ring-necked duck breeding-pair survey. Minnesota's Fall Use Plan also 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 as is the influence that the distribution of the resident population might have on that of migrant ring-necks arriving in the fall.

In response to these information needs, a pilot study of post-fledging resident ring-neck ducks was initiated during the 2006 summer and fall field season. This study was used to develop and test methods of capturing and monitoring birds and to gain a preliminary understanding of the post-fledging movements and fall distribution of local ring-necked ducks. The ability to assess the influence of refuges on survival will largely depend our ability to mark and follow an adequate sample of ducks. Therefore, information from this first year of data was used to plan a more expanded study to be completed over the next 3 years.

The objective of this research is to gain an understanding of the influence of north central Minnesota refuges on the distribution and welfare of resident ring-necked ducks. Specifically, we will employ radio telemetry to: 1) characterize the post-fledging movements of local ring-necked ducks prior to their fall departure, particularly as a function of distance from natal marshes and distance from waterfowl refuges; 2) quantify use of refuges and relate refuge use to refuge level characteristics (size, number of birds on refuge, vegetation characteristics) as well as individual level covariates (gender, proximity of natal marsh to refuge); and 3) estimate survival of locally raised birds during this period, and relate the survival of locally raised birds to their relative use of established refuges.

Study Area

The proposed study area (Figures 1 and 2) encompasses a significant portion of the core of the ring-necked duck breeding range in Minnesota (Zicus et al. 2005) and includes all important ring-necked duck refuges in this part of Minnesota (Figure 3). Presently, banding locations for resident ring-necked ducks are concentrated in the NW portion of the area.

METHODS

We decided to use 2006 as a pilot year to test equipment and methodology. We elected to utilize subcutaneous radio transmitters for our pilot study because this type of radio had been used successfully on hatch year mallards (*Anas platyrhynchos*), canvasbacks (*Aythya valisineria*) (Korschgen et al. 1996a, 1996b), and common loons (*Gavia immer*) (Kenow et al. 2003). Subcutaneous transmitters require a surgical technique that is less invasive to the birds than transmitters implanted in the body cavity and can be done without the need to hire veterinary assistance (R. Gatti Wisconsin DNR, K. Kenow, US Geological Survey, Upper Midwest Environmental Sciences Center, and J. Berdeen, Minnesota DNR Wetland Wildlife Populations and Research Group, personal communication). Surgical techniques followed those of Korschgen et al. (1996a). Transmitters were equipped with mortality switches.

We captured hatch year ring-necked ducks using night-lighting techniques (Lindmeier and Jessen 1961). The following morning, birds were weighed (g), tarsus and culmen lengths measured (mm), and surgery performed in our lab. Birds were then held in a darkened room throughout the day and released in the evening at the lake from which they had been captured. Radio-marked birds were relocated from the ground for the first 2 weeks post marking. We then attempted to locate birds weekly using aerial surveys.

Survival was estimated using the generalized Kaplan Meier estimator (Kaplan and Meier 1958). Birds that were located during one search, but were not located in any further searches, were censored the day after the last location. Birds that died or dehisced their transmitter between 2 searches (i.e. transmitter went into mortality mode) were censored on the day closest to half the length of the period between the last location and the location when the transmitter had gone into mortality mode. Birds that were not located for >14 days, but were located again, were censored for the period between the two location events and were treated as a new bird in the population following the period of absence. Birds that were killed on a known day were assigned that day as the end of their survival period.

We erected 4 remote receiving stations on refuges within the study area. Stations were located on Drumbeater Lake, Fiske Blue Rocks Lakes, Gimmer Lake, and Preston Lakes Refuges (Figure 4). Stations consisted of a 6-meter mast with 1-3 yagi antennas, depending on the size and shape of the refuge. At Preston Lakes and Gimmer Lake Refuges, we used Advanced Telemetry Systems (ATS) R4000 scanning receivers that we had on hand, coupled with ATS DCC (Data Collection Computer) standard data loggers. These receivers continuously scanned through all radio frequencies we used and stored any frequencies detected on the refuge to the data logger. These stations were visited weekly throughout the study to download data from the data logger to a portable computer. At Drumbeater Lake and Fiske Blue Rocks Lakes, we used ATS R4500S receivers that had integrated data loggers and were equipped with DSP (Digital Signal Processor) technology. These new receivers were equipped with a cell phone download unit to test remote downloading from the data loggers. Twice weekly, we called these stations using a modem and downloaded data directly to an office computer without the need to visit the station. All receiver-data logging systems were powered with 12-volt marine batteries recharged daily with solar panels to minimize the need to periodically change batteries. Reference radio-transmitters were stationed permanently on each refuge to assure that receivers and data loggers were functioning properly.

RESULTS

In 2006 we focused our capture efforts on lakes that have traditionally been used to leg band ring-necked duck ducklings. We marked 25 class II and class III ducklings (Gollop, J. B. and W. H. Marshall. 1954. A guide to aging duck broods in the field. Unpublished report. Mississippi Flyway Council). Marking occurred from 14–19 August, and 1–2 ducklings were marked from each banding location. Each surgery took about 15 minutes. Mean mass of radio-marked birds was 574.8 g (range 515 – 660 g). Mean tarsus and culmen lengths were 42.9 and 44.3 mm respectively (see Appendix 1).

Radio-marked birds were relocated from the ground for the first 2 weeks post surgery, then were relocated weekly from the air starting on 9 September. All birds survived at least 10 days post surgery. Two birds were still on the study area and alive as of 19 October, but no birds were relocated after that date. Most smaller wetlands and refuges were frozen by 24 October and we did not fly after that date.

We used a 50-day survival period to look at survival in 2006. The survival rate for ring-necked ducks during the pilot year was 0.750 (95% CI 0.505 – 0.995) between 17 August and 5 October. Between 14 August and 19 October, 5 radio-marked birds are known to have died, however, 2 birds had left the study area and had been right censored prior to their deaths. Hunters shot all 5 birds. Two birds were also reported shot by hunters after they left Minnesota, 1 in Texas and 1 in Illinois.

The remote receiving stations operated well. We were able to download data from each, either by visiting the station or via cell phone technology. The receivers worked well and continuously recorded the presence of reference radio-transmitters. One radio-marked ring-neck used a refuge for several days. This was verified both by the remote receiving station and by aerial flights over the refuge.

We had problems during the pilot year that will need to be resolved in future years. Radio transmitters were incorrectly assembled by ATS, leading to very poor signal strength. We were unable to receive transmitter signals at distances >1 mile, even from the air, during the pilot study. This led to difficulty finding birds after they began to disperse from their natal marshes. Further, we are unsure whether birds may have used portions of refuges that were beyond the range of transmitters. ATS has assured us that this problem will be resolved, however, other options will be explored. We also had problems with transmitter retention. Over the course of the study, 10 of 25 birds shed their transmitters and were right censored. Mass of birds that lost transmitters averaged slightly less than birds that retained them (Fig. 5). Transmitter retention will be a major focus in future years of the study.

DISCUSSION

Treating the first year of our radio-telemetry study as a pilot year proved invaluable. We used subcutaneous transmitters because they had been used successfully on mallards, canvasbacks, and loons. However, the subcutaneous transmitters we used did not work well on HY ring-necked ducks, as retention rates were poor. This is likely because body size of HY ring-necks is small, and there was little room under the skin in these birds for the transmitter. We will try a different attachment technique for the transmitters in 2007. Subcutaneous transmitters with minnow seine material glued to the back have worked to greatly increase retention rates in eiders and shorebirds (D. Mulcahey, U.S. Geological Survey Alaska Science Center, personal communication). However, it is possible that we will still have retention problems this year, and may be forced to try yet another attachment technique such as abdominally implanted transmitters in future years of the project.

The direct recovery rate of radioed birds in 2006 (28 %) was higher than recent recovery rates of ring-necks banded in the same area (J. Berdeen, unpublished data). This high recovery rate may be the actual recovery rate of locally produced ring-necks, an anomaly, or a transmitter effect. We plan to put $100 reward bands on all radio-marked birds and on a sample of normally banded birds to determine whether the recovery rate is different between these groups.

During 2006, we deployed 4 remote receiving stations. Two of the stations used ATS R4000 receivers as well as ATS data loggers. We visited these stations weekly to download data. At the other 2 stations, we used ATS R4500 receivers, with built in data loggers and an attached cellular phone so data could be downloaded via modem directly to a computer. This system seemed to work flawlessly and data could be downloaded without the need to travel to the remote station locations. In 2007 we will erect remote stations at 14 waterfowl refuges (Table 1). We will use 4 R4000 receivers with data loggers, and 8 R4500 receivers with cell technology on state designated refuges within the study area. Further, in 2007–2009, remote receiving stations will be located at Rice Lake and Tamarac National Wildlife Refuges, as these refuges have agreed to be cooperators in this project.

In future years of the study, we will need to radio-mark birds from additional locations to better represent the birds residing within the study area. We will locate additional banding lakes throughout the study area in early-mid August 2006. We may be able to use the same ring-necked duck habitat models that we currently use for the Ring-necked Duck Breeding Pair Survey to help locate lakes throughout the study area for ring-neck capture. Finding and capturing birds from lakes throughout the study area will be imperative to meet study assumptions. Wetland conditions may also be a determinant as to whether we can capture an adequate sample of birds in 2007. Low wetland conditions during summer 2006 made capturing ring-necks on many wetlands difficult, and if drought conditions persist into summer 2007, capturing ring-necked ducklings may be even more difficult.

Management Implications

Post-fledging ecology of most waterfowl has received relatively little study, and refuge management has been identified as an important element for duck management in the fall (Minnesota Department of Natural Resources. 2001. Restoring Minnesota's Wetland and Waterfowl Hunting Heritage, Minnesota Department of Natural Resources, St. Paul, Minnesota, USA). This study will attempt to relate the distribution and welfare of a local population of ducks to the pattern of refuges existing in north central Minnesota. The study also will provide information for a resident waterfowl species that has received little attention and which appears to be declining. Understanding factors influencing the distribution of locally raised ring-necked ducks in the fall also might be a key to understanding the distribution of migrant ring-necks in the fall. This understanding may provide valuable insights regarding the distribution of refuges required to meet management objectives for local ring-necked ducks.

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and radio track birds. John Finn, Leech Lake DNR, guided us into Drumbeater Lake. R. Lego allowed us access to the Leech River, and D. Barrett sterilized the transmitters for us.

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Table 1. Refuges to be included in the study and number of recording telemetry stations needed on each refuge.

Refuge	Location	~Peak numbers	Receivers
Donkey Lake	6 mi. SW Longville	350	1
Drumbeater Lake	2 mi. N of Federal Dam	160,000	1
Fiske and Blue Rock Lakes	8 mi. SE Northhome	40,000	1
Gimmer Lake	10 mi. SE Blackduck	200	1
Hatties and Jim Lakes	13 mi. SE Blackduck	0	1
Hole-in-the-Bog Lake	2 mi. SW Bena	4,000	1
Mud and Goose Lakes	4mi. SSW of Ballclub	2,100	1
Lower Pigeon Refuge	4 mi. S Squaw Lake	700	1
Pigeon River	6 mi. S Squaw Lake	700	1
Preston Lakes	22 mi. ENE of Bemidji	535	1
Rice Lake Waterfowl Refuge	8 mi. N Deer River	7,000	1
Rice Pond	9 mi. E of Turtle River	15	1
Tamarac National Wildlife Refuge	16 mi. NE Detroit Lakes	10,000	4
Rice Lake National Refuge	5 mi SSW of McGregor	120,000	4

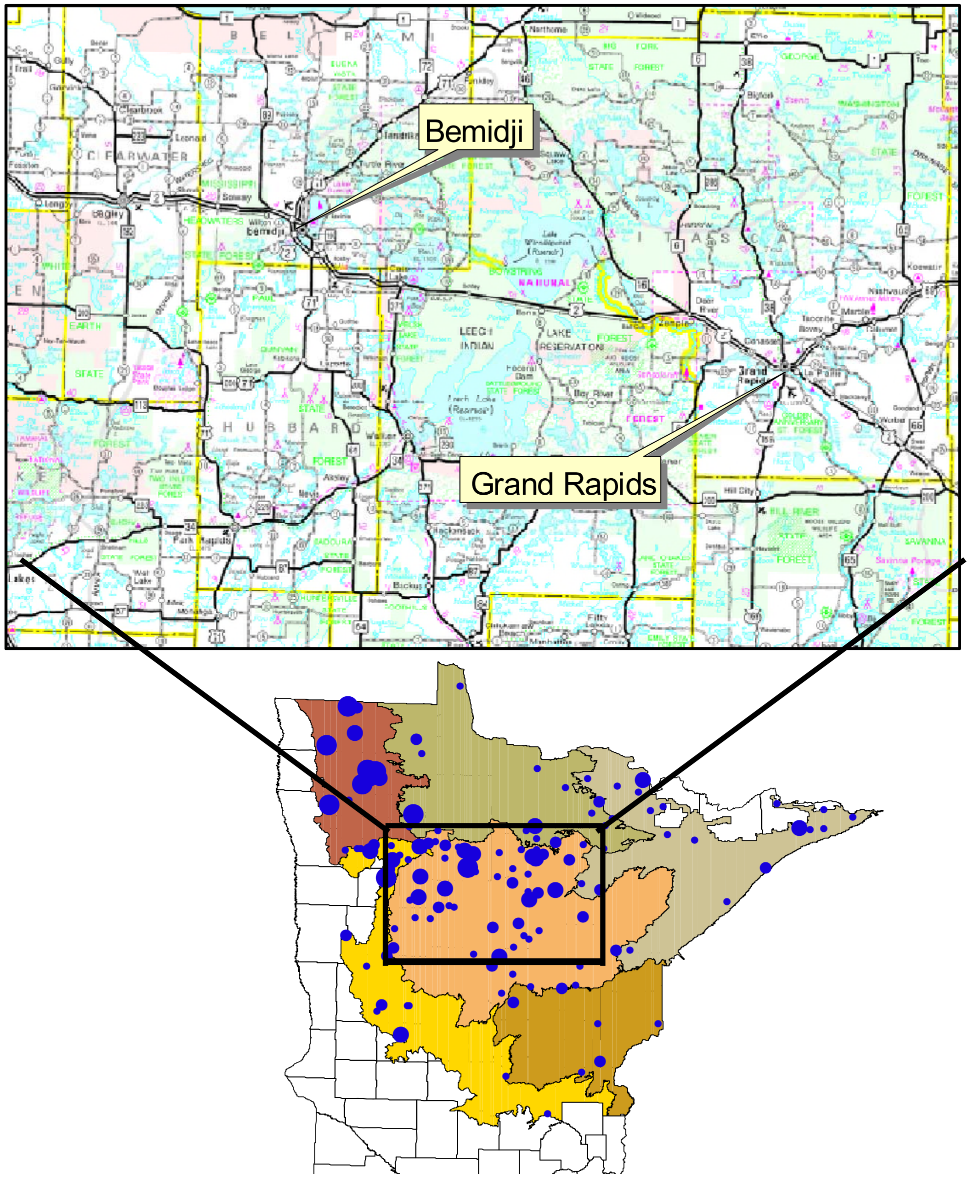


Figure 1. Proposed study area, 2006 – 2009. State map reflects results from 2004 – 2006 helicopter survey (see Figure 2 for details).

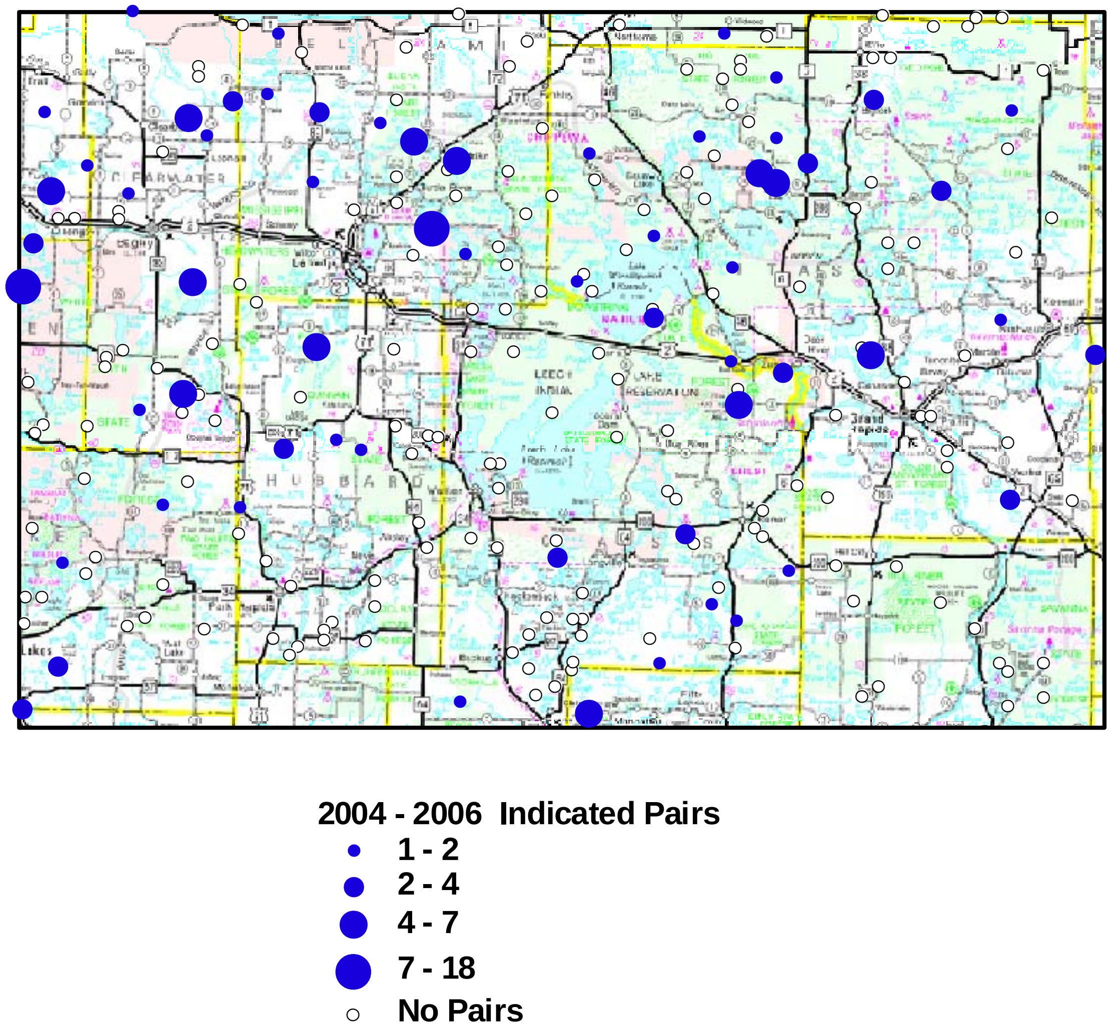
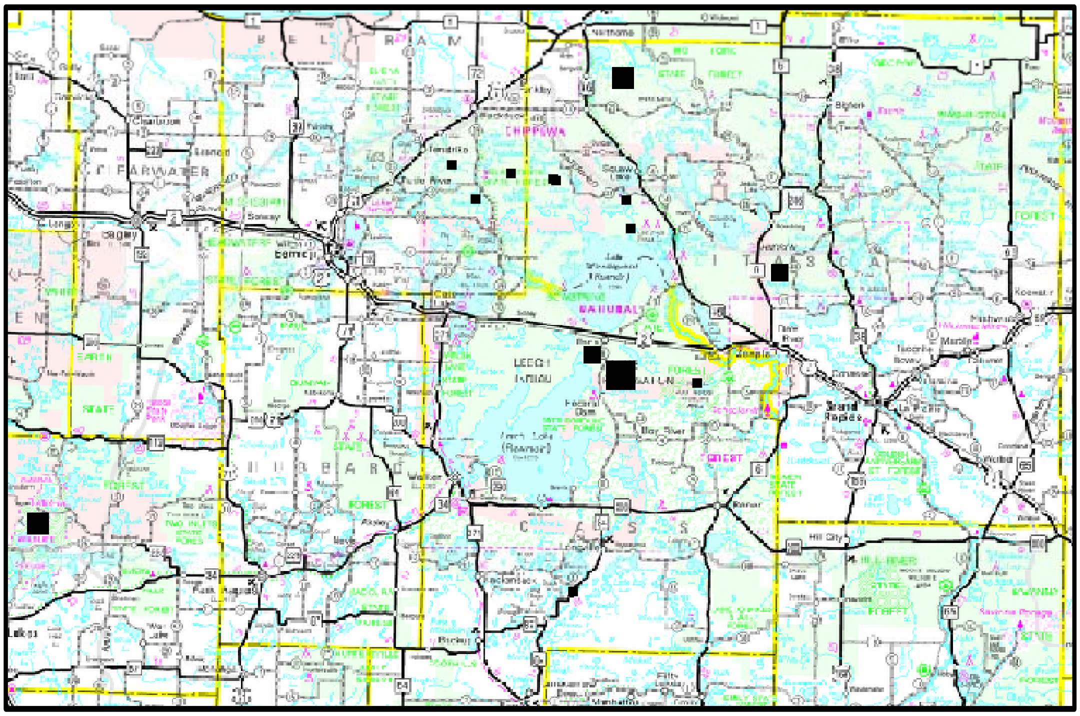


Figure 2. Distribution of indicated breeding pairs of ring-necked ducks based on survey plots in the 2004 – 2006 helicopter survey.



Peak numbers

- 0 - 2100
- 2101 - 7000
- 7001 - 40000
- 40001 - 160000

Figure 3. Approximate peak numbers of ring-necked ducks in fall on designated refuges, 2005.

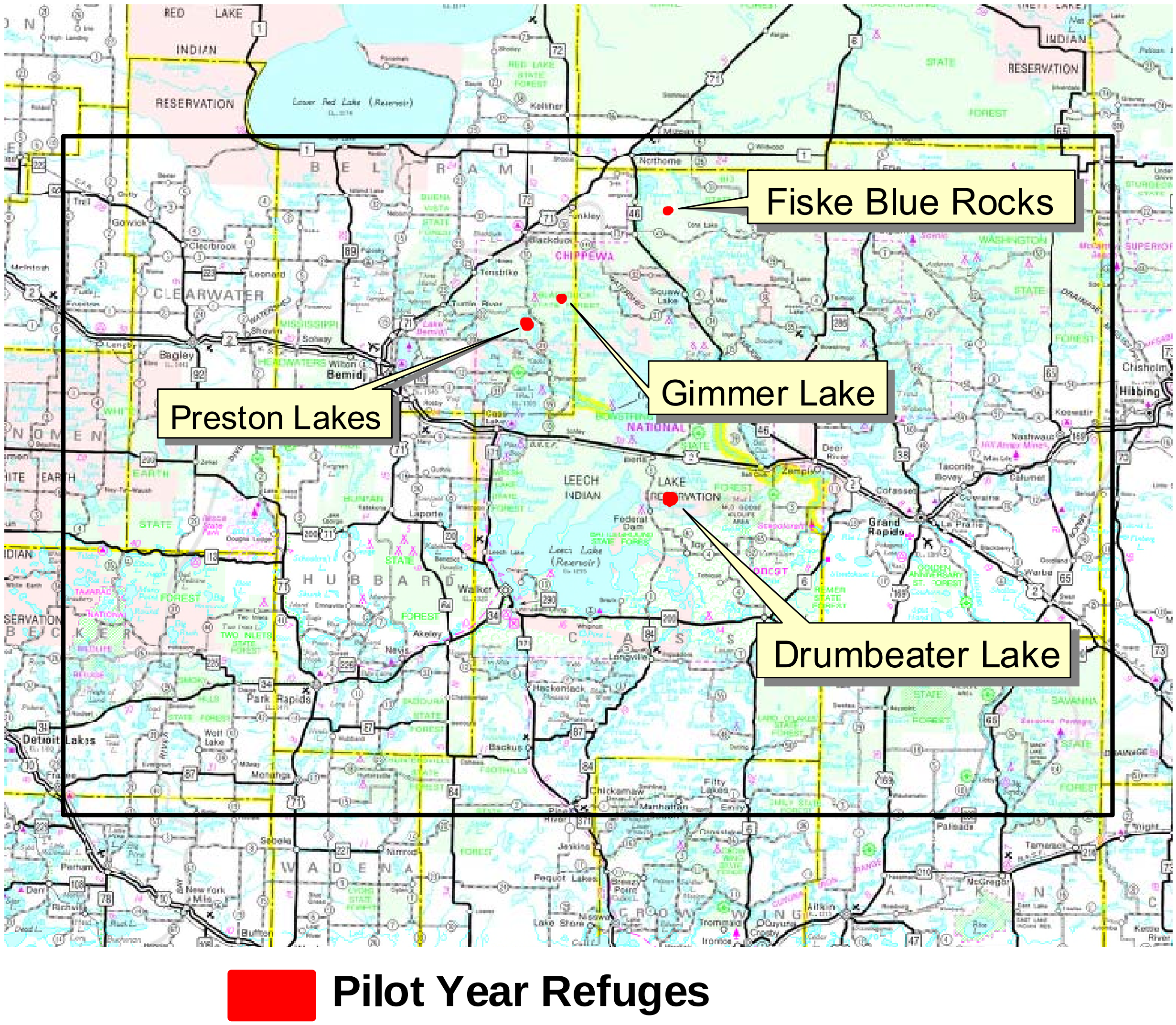


Figure 4. Refuges where remote receiving stations were located during the 2006 pilot year of the ring-necked duck telemetry study.

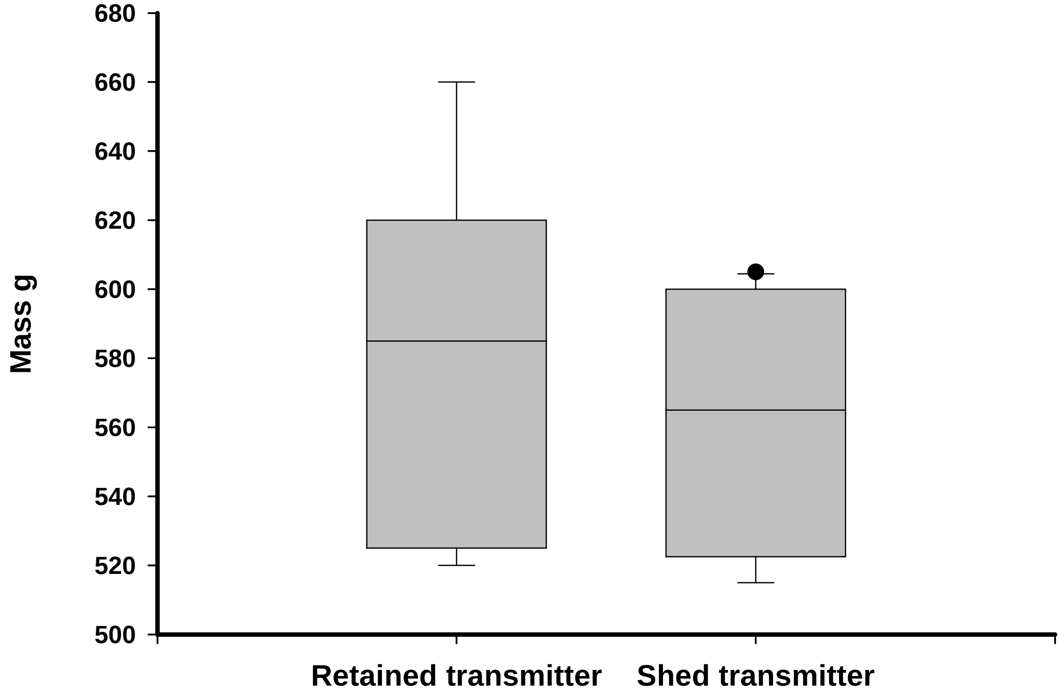


Figure 5. Box and whisker plot of the mass of ring-necked ducks at capture for those that retained their transmitters and those that shed them, August – October 2006.

Appendix 1. Data for hatch year ring-necked ducks captured between 14 and 19 August, 2006.

Year	Capt. Date	FWS Band Number	Mass (g)	Sex	Tarsus (mm)	Culmen (mm)	Capture Lake	Frequency	Term date	Term status
2006	8142006	104680754	600	F	44.4	46.8	East Four-legged	151.013	10022006	Dehisced
2006	8162006	108678227	650	M	43.2	44.1	White Oak	151.023	9302006	Killed
2006	8162006	104680790	565	M	43.4	40.2	Little Pine WMA	151.045	10092006	Not Found
2006	8182006	108679503	550	F	42.3	45	Little Moose	151.053	10092006	Dehisced
2006	8142006	104680744	565	M	38.4	43.1	W. Four-legged	151.065	9122006	Dehisced
2006	8192006	108679535	615	M	45.7	46	Little Puposky	151.075	9072006	Not Found
2006	8142006	104680751	585	M	41.2	42.2	E. Four-legged	151.084	9212006	Not Found
2006	8162006	104680788	620	M	40	45.6	Big Pine	151.104	10022006	Not Found
2006	8152006	108678220	520	F	42.6	42.2	Upper Rice	151.205	10092006	Not Found
2006	8182006	108679516	565	F	45.4	47.1	Rabideau	151.223	10022006	Dehisced
2006	8192006	108678234	605	M	45	46.8	Whitefish	151.245	10192006	Alive
2006	8142006	108678208	520	F	42.4	43.5	Muskrat	151.265	10192006	Alive
2006	8192006	108679533	660	M	44.9	46	Little Puposky	151.284	10092006	Not Found
2006	8162006	108678226	515	F	42.2	42.6	White Oak	151.324	10092006	Dehisced
2006	8162006	104680792	525	F	42.3	44.1	Little Pine WMA	151.344	10012006	Killed
2006	8182006	108679517	540	F	42.5	44.8	Rabideau	151.363	9212006	Dehisced
2006	8142006	104680743	585	M	41.3	45.9	W. Four-legged	151.383	9212006	Dehisced
2006	8152006	108678217	525	F	41.7	40.5	Upper Rice	151.402	9252006	Dehisced
2006	8162006	104680787	520	F	41.3	43.1	Big Pine	151.425	10022006	Not Found
2006	8152006	104680774	610	M	46.4	45.8	Little Pine	151.444	10192006	Not Found
2006	8152006	104680676	600	M	45.3	47.4	Dutchman	151.565	10182006	Dehisced
2006	8142006	108678210	660	F	43	45.6	Muskrat	151.584	10182006	Not Found
2006	8182006	108679504	550	F	41.9	44.6	Little Moose	151.603	9302006	Killed
2006	8162006	104680677	515	F	41.3	43.2	Damon	151.663	9262006	Dehisced
2006	8152006	104680775	605	F	43.2	41.5	Little Pine	151.685	9262006	Dehisced

MINNESOTA'S RING-NECKED DUCKS: A PILOT BREEDING PAIR SURVEY

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SUMMARY OF FINDINGS

Little is known about the distribution and relative abundance of Minnesota's ring-necked duck (*Aythya collaris*) breeding population. We conducted the third year of a pilot survey to better understand the issues involved in monitoring these important but poorly studied ducks. The helicopter-based counts (06–16 June 2006) entailed 10 flight days and included a portion of Minnesota that is considered primary breeding range. Minnesota Department of Natural Resources' MN-GAP land cover data again were used to quantify presumed ring-necked duck nesting cover in Public Land Survey (PLS) section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. Similarly to 2005, we combined results from 2 separate surveys to estimate population size. We apportioned 200 plots among 12 strata (i.e., 6 Minnesota Department of Natural Resources' Ecological Classification System sections x 2 habitat classes) using a stratified random sampling design to estimate population size in the best habitat. We used a simple random sample of 50 plots to estimate population size in the remaining habitat. The combined population was estimated to be ~15,600 indicated breeding pairs (~31,000 birds). Numbers of ducks counted from the air and the ground on 14 lakes differed less in 2006 than in 2005, and the difference was likely due to less time elapsed between the air and ground surveys. The stratification we used continued to account for geographical- and habitat-based differences in ring-necked duck abundance, whereas, we would have needed approximately 1.2 times as many plots to achieve the same precision under a simple random sampling design.

INTRODUCTION

Staff in the Minnesota Department of Natural Resources (DNR) Wetland Wildlife Populations and Research Group have been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks has been among the topics considered because the species has been identified as an indicator species for the Forest Province (Minnesota Department of Natural Resources. 2003. A Vision for Wildlife and its Use – Goals and Outcomes 2003 – 2013 (draft). Minnesota Department of Natural Resources, unpublished report, St. Paul), but little is known about the current distribution and abundance of breeding ring-necked ducks in Minnesota.

In 2004, a pilot survey was conducted in a portion of Minnesota that is considered primary breeding range (Zicus et al. 2005). Minnesota Department of Natural Resources' MN-GAP land cover data were used to quantify presumed ring-necked duck nesting cover in PLS section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. Plots in 2 habitat classes were not sampled because few ring-neck pairs were believed to occupy these plots. The resulting population estimate (~9,000 indicated pairs) was almost certainly biased low because >69% of the survey area was not sampled, and some survey plots in the habitat classes, that were not surveyed, were misclassified.

Our objectives were to: 1) conduct the third year of a pilot study to determine the most appropriate sampling design and allocation for an operational breeding-pair survey of ring-necked ducks in Minnesota; and 2) make recommendations for future operational surveys.

METHODS

Two separate surveys were again conducted in 2006 to reduce the bias associated with the 2004 estimate. We apportioned 200 plots among 12 strata (i.e., 6 Minnesota Department of

Natural Resources' Ecological Classification System (ECS) sections x 2 habitat classes) using a stratified random sampling design to estimate population size in the best habitat. We used a simple random sample of 50 plots to estimate population size in the remaining habitat. We continued to use a stratified random sampling design with 2 stratification variables: ECS sections and presumed nesting-cover availability (i.e., a surrogate for predicted breeding ring-necked duck density) to estimate population size in the best ring-necked duck habitat. We used a 2-stage simple random sampling design to estimate population size in the remainder of the survey area. 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. We considered pairs, lone males, and males in flocks of 2–5 to indicate breeding pairs (IBP; J. Lawrence, Minnesota Department of Natural Resources, personal communication). The total breeding population in the survey area was considered to be twice the IBP plus the number of birds in mixed sex groups and lone or flocked females.

Statistical Population, Sampling Frame, and Sample Allocation

The surveys were restricted to an area believed to be primary breeding range of ring-necked ducks for logistical efficiency (Zicus et al. 2005). However, we again used habitat class definitions modified from those used for stratification in 2004 (Table 1). Based on 2004 results, we added MN-GAP Level 4 cover class 10 (lowlands deciduous shrub) as presumed nesting cover. Furthermore, we reduced the maximum distance that we believed ring-necked ducks were likely to be from a shoreline from 250 to 100 m. We also corrected a GIS processing error that we made in 2004. Habitat class 1 and 2 plots were presumed to represent the best habitat, whereas, habitat class 3 and 4 plots represented the remainder of the survey area. As in 2004 and 2005, 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. Finally, we determined from the 2004 and 2005 survey that breeding ring-necked ducks did not use large fish lakes, therefore, for the 2006 survey we removed all "nesting cover" associated with lakes having a General or Recreational Development shoreline classification.

A stratified sampling design was used to estimate breeding ducks in habitat class 1 and 2 plots, and the sampling frame consisted of 12 strata (i.e., 6 ECS sections x 2 habitat classes). We proportionally allocated 250 plots to the 12 strata using the same approach as in 2004 (Zicus et al. 2005). We used a 2-phase sampling process to sample plots in habitat classes 3 and 4. The phase-1 sample consisted of 1,000 habitat class 3 and 4 plots, disregarding ECS sections. These plots were visually inspected using 2003 Farm Services Agency (FSA) true color aerial photography and classified as to their ring-necked duck potential (i.e., possible breeding pairs vs. no pairs). PLS sections containing open water except for small streams were considered potential ring-necked duck plots. The proportion of plots classified as potentially having pairs was used as an estimate of the proportion of all class 3 and 4 plots that had potential for breeding pairs. We then randomly selected 50 plots (phase-2 sample) from those having the potential for ring-necked duck pairs in order to estimate the mean number of breeding pairs in these plots.

Data Analyses

Estimated Population Size. – We used SAS PROC SURVEYMEANS (SAS 1999) to estimate population totals for habitat class 1 and 2 plots in each ECS section and the entire survey area. In this analysis, PLS sections were the primary sampling unit in a stratified random sampling design. For the second survey, we estimated population size (τ) for habitat class 3 and 4 plots in the entire survey area as follows:

$$\hat{\tau} = \hat{P} * \bar{x} * N \, ,$$

where $\hat{P}$ = proportion of phase-1 plots classified as habitat-class 3,
$\bar{x}$ = mean breeding ducks detected on phase-2 sample plots, and
N = total habitat-class 3 and 4 plots in sampling frame.

The variance of $\hat{\tau}$ was estimated using the delta method as:

$$\text{var}(\hat{\tau}) = N^2 ((\hat{P}^2 * \text{var}[\bar{x}]) + (\bar{x}^2 * \text{var}(\hat{P})).$$

Estimates from the 2 surveys were combined to produce an overall population estimate for the survey area.

Aerial Visibility

An implicit assumption in aerial waterfowl surveys is that the proportion of the population of interest that is observed from the air is known or can be estimated (Smith 1995). Surveys using helicopters usually rely on the assumption that virtually all individuals are seen (Ross 1985, Cordts 2002). In fact, counts of ring-necked duck pairs in boreal wetlands that were made from helicopters were similar to those made when walking around wetlands or by traversing wetlands in a canoe (Ross 1985). We again examined this assumption by comparing aerial counts of indicated ring-necked duck pairs on the 14 lakes included in the Bemidji Area Ring-Necked Duck Pair Survey (Zicus et al. 2004) with pair counts from these lakes that were made from boats. Only 13 lakes were compared in 2006 because of weather and timing constraints.

Stratification Evaluation

We estimated the relative efficiency (RE) of the stratified sampling design by dividing the estimated variance for a simple random sample [var(SRS)] by the variance of the stratified random sample [var(StRS)] (Schaefer et al. 1996, Cochran 1997) where:

var(SRS) = estimated variance of $\bar{x}$ if we treated the observations as having been drawn using a simple random sample (i.e., based on a weighted sum of sample variances in each stratum), and

var(StRS) = estimated variance of the stratified mean.

If stratification performed well, it would account for differences in indicated ring-necked duck pairs seen on plots among the strata in the survey. As a result, the population variance would be smaller than that obtained by a comparable simple random sample (Cochran 1997). If each estimator is unbiased, then RE will describe the relative gain in precision by using ECS and habitat classes as stratification variables. We also evaluated the stratification by comparing the mean number of indicated pairs seen among ECS sections, habitat classes, and the interaction between ECS sections and habitat classes using SAS Proc GLM (SAS 1999).

Data Acquisition

The 2006 survey utilized an ArcView 3.x extension (DNRSurvey) in conjunction with a Global Positioning System receiver and DNR Garmin program (real time survey technique) to collect the survey data. This approach allowed us to display the aircraft's flight path over a background of aerial photography and the survey plots. The flight path and ring-necked duck observations were recorded directly to ArcView shapefiles, all in real time (R. Wright, Minnesota Department of Natural Resources, personal communication).

RESULTS

More PLS sections in the northeast were classified as habitat classes 1 and 2 in 2005 and 2006 versus 2004 because we included MN-GAP cover class 10 as potential nesting cover. As a result, survey plots were distributed somewhat more to the northeastern portion of the survey area than they were in 2004 (Figure 1). Most plots (77) were located in the Northern Minnesota Drift and Lake Plains section. The fewest plots (8) were located in the Lake Agassiz, Aspen Parklands section this year, similar to 2005, rather than the Northern Superior Uplands section as in 2004 (Table 2). The highest and lowest sampling rate again occurred in the Lake Agassiz, Aspen Parklands section and Northern Superior Uplands section, respectively. The survey was conducted 06–16 June and entailed 10 survey-crew days. Observed pairs represented 44% of the indicated pairs tallied during the survey compared to 36% in 2005 and 57% in 2004 (Table 3).

Estimated Pair Density

Mean pair density on habitat class 1 and 2 plots ranged from a high of 4.16 pairs/plot in the Lake Agassiz, Aspen Parklands section to a low of 0.30 pairs/plot in the Western and Southern Superior Uplands section (Table 4). Mean pair densities were higher in all of the 6 ECS sections compared to 2005. Considering both years, pair densities were greatest in the Lake Agassiz, Aspen Parklands section with lowest pair densities in the Western and Southern Superior Uplands and the Northern Minnesota and Ontario Peatlands sections.

Estimated Population Size

Estimated indicated breeding pairs on habitat class 1 and 2 plots ranged from a high of 6,334 in the Northern Minnesota Drift and Lake Plains section to a low of 669 in the Western and Southern Superior Uplands section (Table 5). More breeding pairs were estimated in 2006 in all 6 ECS sections than in 2005. Pair numbers were greatest in the Northern Minnesota Drift and Lake Plains section and fewest in the Western and Southern Superior Uplands section.

The estimated population of ring-necked ducks on habitat class 1 and 2 plots ranged from a high of 14,816 in the Northern Minnesota Drift and Lake Plains section to a low of 1,338 in the Western and Southern Superior Uplands section (Table 6). As with indicated breeding pairs, more ducks were estimated in 2006 in all 6 ECS sections than in 2005. Considering both years, the most birds occurred in the Northern Minnesota Drift and Lake Plains section and the fewest in the Western and Southern Superior Uplands section.

In 2006, we estimated indicated breeding pairs and total birds for the entire survey area (Table 7). The estimated number of indicated breeding pairs for the survey area was 15,631 (90% confidence interval = 11,221 – 20,042), and the estimated ring-necked duck population was 34,342 (90% confidence interval = 24,766 – 43,918).

Observed Distribution

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. Indicated pair observations in 2005 and 2006 shifted somewhat to the east compared to 2004 (Figure 1). Estimates from 2004–2006 suggest that some ECS subsections or portions of a section might have substantial numbers of breeding ring-necked ducks even though few birds were observed in the ECS section (Figure 2). For example, pairs/plot and total estimated pairs were relatively high in the Northern Superior Uplands, yet few plots in the section had indicated breeding pairs (Table 5 and 6).

Aerial Visibility

Counts from boats generally agreed with aerial counts of IBPs on the individual lakes included in the 14-lake survey (Figure 3). Boat counts in 2004 were conducted 14–18 June in 2004 with the aerial survey of the 14 lakes done on 17 June. In contrast, boat counts were conducted 15–21 June with the aerial survey done on 24 June in 2005. In 2006, boat counts were conducted 8–13 June with the aerial survey flown on 12 June. Poorer agreement between the 2 surveys in 2005 than in either 2004 or 2006 was likely due to the greater time that elapsed between the boat counts and aerial surveys.

Stratification Evaluation

Analysis of variance indicated that the strata identified using the MN-GAP models were reasonable. For the most part, IBPs were related significantly to ECS sections and to habitat classes within the ECS sections (Table 8). Results from 2004 might have been an exception as IBPs were related to an interaction between ECS section and habitat class. Stratification by ECS section resulted in a thorough distribution of sample plots throughout the survey area (Fig 1). However, lack of IBP observations in some strata suggested we might have over-stratified relative to the number of plots we surveyed. Estimated relative efficiency suggested that a modest increase in plots would have been needed to achieve the same precision under a simple random design as we did using a stratified design. However, estimated relative efficiency should be interpreted cautiously because we lacked variance estimates for 1 and 3 strata in 2004 and 2005, respectively.

Data Acquisition

Generally less time was required to survey a plot in 2006 than in 2005 or 2004 (Table 9). Survey time ranged from 1–13 minutes (mean=4.5) compared to 1–22 minutes (mean=5.2) in 2005 and 1–29 minutes (mean=7.2) in 2004 (Figure 4). Use of the real time survey technique accounted for the reduction in plot survey time in 2005 (Fieberg et al. 2006), and it reduced the total airtime required to survey the plots by >8 hours.

DISCUSSION

We further improved our understanding of the issues involved in designing and conducting a survey to estimate the abundance and describe the distribution of breeding ring-necked ducks in Minnesota. Survey dates in 2004–2006 appeared appropriate because 36–57% of the indicated pairs were counted as paired birds, and survey timing is considered optimal when most birds are counted as pairs and not in flocks (Smith 1995). The stratified random sampling design that we employed was adequate for plots in habitat classes 1 and 2, while the second survey based on a simple random sample of plots in habitat classes 3 and 4 again provided an estimate for the survey area that was unbiased (i.e., included all potential breeding habitat). Detection rates appeared to be relatively high in all habitats, suggesting that any bias probably would be minor.

MN-GAP land cover data provided a convenient way to stratify the survey area, but they have shortcomings as well as strong points. They provided a consistent statewide source of land use/cover data that was available in an easy to use, raster format. However, the data are derived from 1991 and 1992 satellite imagery, which makes them dated. Further, the data exist at 4 levels of resolution, and classification accuracy of cover types is diminished at the level that we used. Nearly 50% (487 of 1,000) of habitat class 3 and 4 plots were incorrectly classified when compared to conditions that existed in 2003 (based on FSA photography).

Misclassifications resulted from MN-GAP data missing small wetland areas capable of supporting ring-necked duck pairs or from wetland conditions that had changed between 1991 and 2003. We improved the stratification in 2006 by eliminating emergent shoreline-vegetation associated with larger lakes containing fish from our definition of potential ring-necked duck nesting cover. Ring-necked ducks do not occupy these types of lakes during the breeding season.

The stratification approach that we used worked relatively well and assured a reasonable geographical distribution of survey plots throughout the survey area. However, failure to observe birds in 3 strata in 2005 indicated that we might have over-stratified given the sample size of 230 habitat class 1 and 2 plots. As a result, our variance estimates were biased low because the estimated sample variance in some strata was 0 and these strata contributed nothing to the overall variance. Likewise, the design effect (i.e., RE) becomes difficult to estimate when some strata have no observations; therefore, our estimate of relative efficiency should be viewed cautiously.

Survey costs are an important consideration with any wildlife survey, and survey efficiency is the product of optimal plot size as well as appropriate stratification and efficient data acquisition. A complete examination of plot size efficiency will require consideration of the time required to fly to and among plots in the sample as well as the number of refueling stops required. We began modeling to evaluate various plot sizes after the 2006 field season.

Recommendations

- Conduct the 2007 survey using the same proportional allocation of 200 habitat class 1 and 2 plots among the 6 ECS sections. Conduct a second survey choosing a simple random sample of 50 habitat class 3 and 4 plots. Rationale: An operational survey might need to focus on a core area within the primary ring-necked duck breeding range to reduce costs and improve the precision of the estimate. The 2005 and 2006 data contained a better geographical distribution of plots than 2004, and have helped define a core area for indicated breeding pairs. Another year with a similar sample distribution will continue to define the core area for breeding ring-necked ducks in Minnesota.
- Begin the survey as soon after 5 June as possible. Rationale: A set starting date will assure the needed flight time can be scheduled. Although phenology will vary from year to year, this date should result in the survey being done while most ring-necked ducks are still paired.
- Pending further discussions within the DNR Wetland Group and the Waterfowl Committee, conduct future operational surveys in enough of the primary breeding range to provide the desired population information in the most cost-effective manner. Rationale: Obtaining population estimates for the entire primary breeding range would be ideal. However, the information gained by surveying some areas that are logistically difficult to reach or that have few ring-necked ducks might not be worth the added cost.
- Continue using PLS sections as sampling units unless future modeling indicates some other unit is more efficient. Rationale: Preliminary modeling in 2004 suggested that quarter-sections might be a more efficient plot size. However, this modeling did not account for the time required to fly to and among plots in the sample as well as the number of refueling stops required. Consequently, we have no basis for recommending a different size plot at this time.

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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 – 2006.

Habitat class	Definition[a] 2004	Definition[a] 2005 and 2006[c]	%[b] 2004	%[b] 2005	%[b] 2006
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
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
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
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

[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]Percent of the survey area.
[c]Habitat class definitions in 2005 and 2006 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.

Table 2. Sampling rates by Ecological Classification System section for Minnesota's ring-necked duck breeding- pair survey, June 2004 – 2006

Ecological Classification System sections	Habitat Classes	~Area[a] 2004	~Area[a] 2005	~Area[a] 2006	Sampling rate (%) 2004	Sampling rate (%) 2005	Sampling rate (%) 2006
W & S Superior Uplands[b]	1	1,638	2,461	2,218	1.1	0.9	0.9
Northern Superior Uplands	2	1,810	4,648	4,209	0.7	0.8	0.8
N Minnesota & Ontario Peatlands	3	1,817	2,737	2,389	1.4	1.3	1.3
N Minnesota Drift & Lake Plains	4	5,048	8,383	7,145	1.5	1.1	1.1
Minnesota & NE Iowa Morainal	5	3,510	4,033	3,561	1.4	0.9	0.9
Lake Agassiz, Aspen Parklands	6	316	363	340	4.7	2.2	2.4

[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. Social status of the indicated pairs observed in the Minnesota ring-necked duck breeding pair survey area, June 2004-2006.

				Indicated pairs			
Year	Habitat class	No. of plots	Total ducks	n	% pairs	% Lone males	% Flocked males
2004[a]	1,2	200	278	160	57.5	18.1	24.4
2005[b]	1,2	230	147	92	35.9	28.2	35.9
2005	3,4	21	11	7	57.1	0.0	42.9
2006[c]	1,2	200	279	167	43.7	27.6	28.7
2006	3,4	50	4	3	33.3	66.7	0.00

[a]Survey conducted 6 – 17 June.
[b]Survey conducted 12 – 24 June.
[c]Survey conducted 6 – 16 June.

Table 4. Estimated indicated breeding pairs per plot in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005			2006		
Ecological Classification System sections	Plots	Mean pairs/plot	SE	Plots	Mean pairs/plot	SE
W & S Superior Uplands[a]	22	0.181	0.179[b]	20	0.302	0.178
Northern Superior Uplands	36	0.252	0.118	33	0.636	0.215
N Minnesota & Ontario Peatlands	35	0.087	0.045[b]	30	0.658	0.228
N Minnesota Drift & Lake Plains	94	0.416	0.138	77	0.887	0.279
Minnesota & NE Iowa Morainal	35	0.228	0.010	32	0.590	0.318
Lake Agassiz, Aspen Parklands	8	3.403	1.365[b]	8	4.160	1.463

[a]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[b]Standard error estimate is biased low because no birds were observed in one of the Ecological Classification System section's strata.

Table 5. Estimated indicated breeding pairs in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005				2006			
Ecological Classification System section	Pairs	LCL[a]	UCL[a]	CV(%)	Pairs	LCL	UCL	CV(%)
W & S Superior Uplands[b]	444	0	1,207	99.5[c]	669	0	1,355	59.1
Northern Superior Uplands	1,169	244	2,095	46.8	2,679	1,148	4,210	33.7
N Minnesota & Ontario Peatlands	239	20	457	54.1[c]	1,572	644	2,499	34.7
N Minnesota Drift & Lake Plains	3,490	1,577	5,404	33.0	6,334	3,011	9,657	31.5
Minnesota & NE Iowa Morainal	918	241	1,595	43.6	2,102	178	4,026	53.9
Lake Agassiz, Aspen Parklands	1,235	273	2,198	40.1[c]	1,414	448	2,381	35.2

[a]Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 and 6 ECS sections. LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[c]Variance estimate for the Ecological Classification System section is biased low because no birds were observed in one of the section's strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 6. Estimated ring-necked ducks in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2005-2006.

	2005				2006			
Ecological Classification System section	Birds	LCL[a]	UCL[a]	CV(%)	Birds	LCL	UCL	CV(%)
W & S Superior Uplands[b]	889	0	2,415	99.5[c]	1,338	0	2,710	59.1
Northern Superior Uplands	2,339	488	4,190	46.8	5,357	2,295	8,419	33.7
N Minnesota & Ontario Peatlands	477	40	915	54.1[c]	4,076	1,141	7,012	42.3
N Minnesota Drift & Lake Plains	6,981	3,154	10,808	33.0	14,816	7,504	22,127	29.6
Minnesota & NE Iowa Morainal	4,122	187	8,057	56.4	4,204	375	8,052	53.9
Lake Agassiz, Aspen Parklands	2,471	545	4,396	40.1[c]	2,829	896	4,762	35.2

[a]Estimates were based on a stratified random sample of Public Land Survey (PLS) sections in habitat classes 1 and 2 and 6 ECS sections. LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]Western and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.
[c]Variance estimate for the ECS section is biased low because no birds were observed in one of the ECS section's strata. As a result, the confidence interval is too narrow and the CV is optimistic.

Table 7. Estimated indicated breeding pairs and breeding population size in the Minnesota ring-necked duck breeding pair survey area, 2004-2006.

Year	Habitat classes	Indicated breeding pairs				Breeding population			
		Pairs	LCL[a]	UCL[a]	CV(%)	Birds	LCL[a]	UCL[a]	CV(%)
2004	1,2[b]	9,443	6,667	12,220	17.8[d]	20,321	14,248	26,395	18.1[d]
2005	1,2[b]	7,496	5,022	9,971	20.0[d]	17,279	11,156	23,402	21.5[d]
2005	3,4[c]	3,832	0	9,269	86.3	7,664	0	18,539	86.3
2005	All	11,328	5,359	17,298	32.0[d]	24,943	12,476	37,411	30.4[d]
2006	1,2[b]	14,770	10,465	19,075	17.6[d]	32,621	23,231	42,010	17.4[d]
2006	3,4[c]	861	0	1,908	74.0	1,721	0	3,816	74.0
2006	All	15,631	11,221	20,041	17.2[d]	34,342	24,766	43,918	17.0[d]

[a]LCL = lower 90% confidence level. UCL = upper 90% confidence level.
[b]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 ECS sections).
[c]Population estimates were based on a simple random sample of Public Land Survey (PLS) sections in habitat classes 3 and 4.
[d]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 8. General linear model evaluation of the Minnesota ring-necked duck breeding pair survey stratification developed using 2004 – 2006 MNGAP habitat models and the estimated relative efficiency of the resulting stratified random design.

MNGAP model	Stratification variable	df	F	P	RE[a]
2004	ECS section	5, 188	2.17	0.059	1.02
	Habitat class	1, 188	9.08	0.003	
	ECS section* habitat class	5, 188	0.93	0.462	
2005	ECS section	5, 218	7.17	<0.001	1.17
	Habitat class	1, 218	28.70	<0.001	
	ECS section* habitat class	5, 218	7.94	<0.001	
2006	ECS section	5, 188	3.51	0.005	1.06
	Habitat class	1, 188	7.25	0.008	
	ECS section* habitat class	5, 188	1.03	0.403	

[a]Relative efficiency of stratified random design compared to a simple random sample.

Table 9. Time required to complete the Minnesota ring-necked duck breeding pair survey, June 2004-2006.

Year	No. of plots	Flight days	Time (min)[a]		Min/plot	% Survey time
			Operation[b]	Survey[c]		
2004	200	13	4,686	1,441	7.2	30.8
2005	251	10	4,868	1,307	5.2	26.8
2006	250	10	4,399	1,126	4.5	25.6

[a]Includes all observers.
[b]Time between the initial start of the helicopter each morning and final shutdown of the helicopter each afternoon.
[c]Air time spent surveying the individual plots.

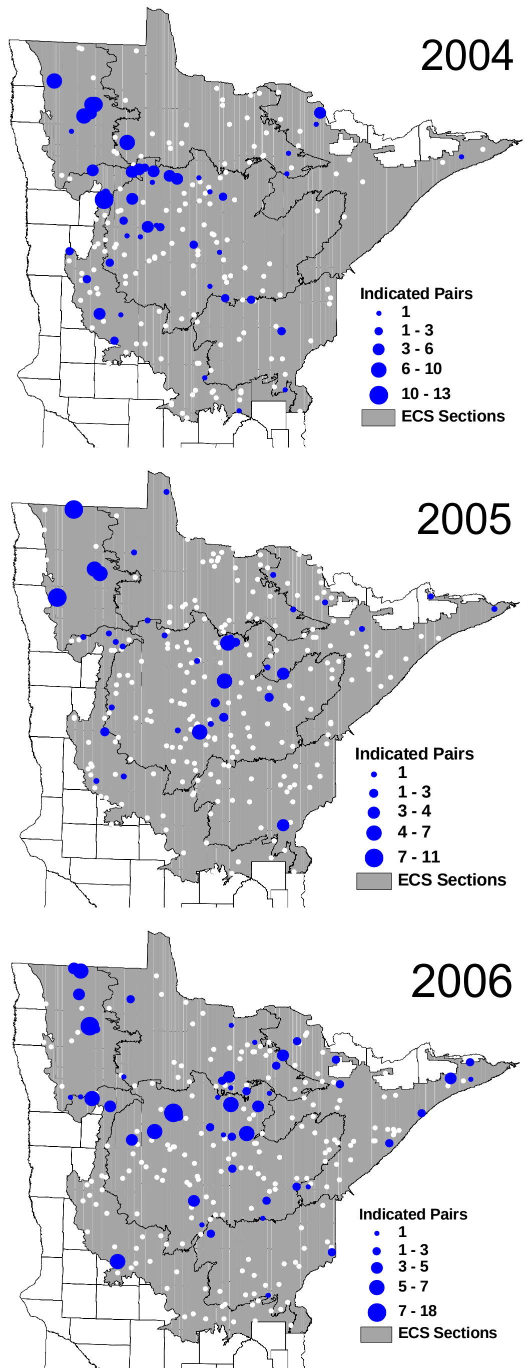

Figure 1. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area in June 2004 (top), 2005 (middle), and 2006 (bottom). White circles indicate plots where no indicated pairs were seen.

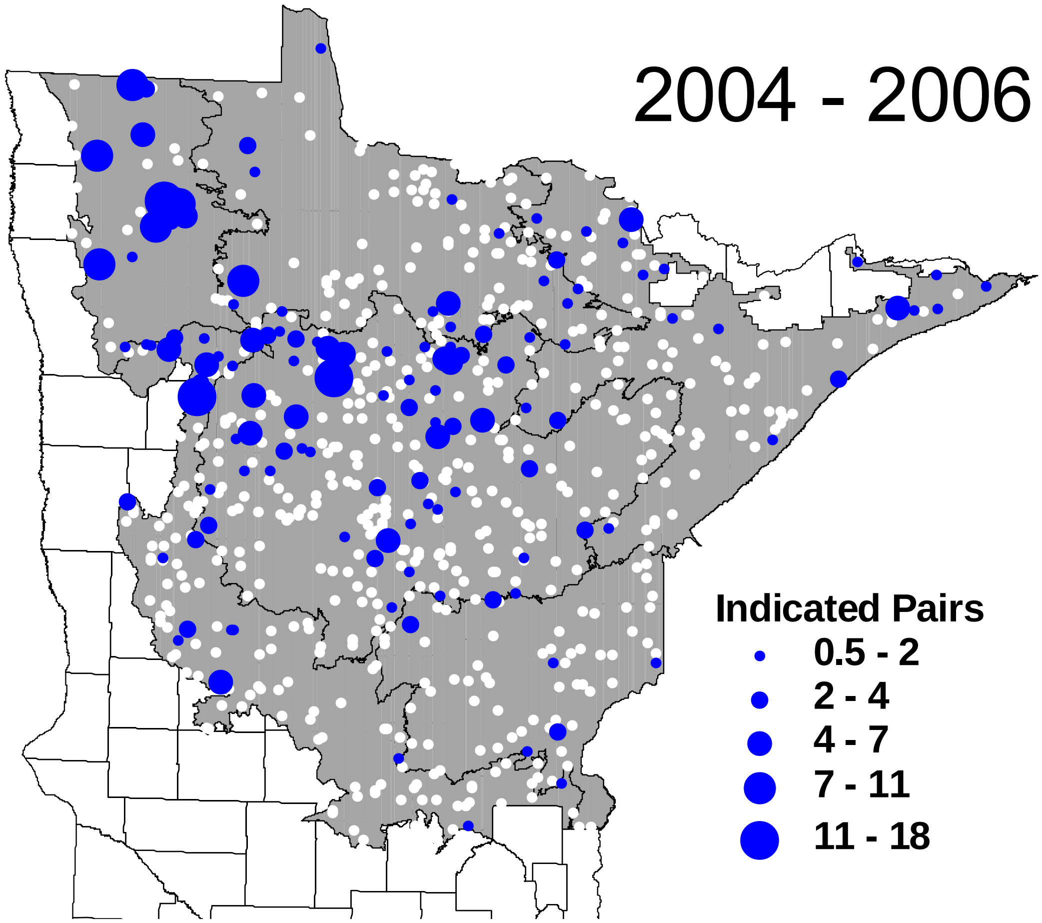

Figure 2. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area, June 2004-2006. White dot indicates a plot where no birds were seen.

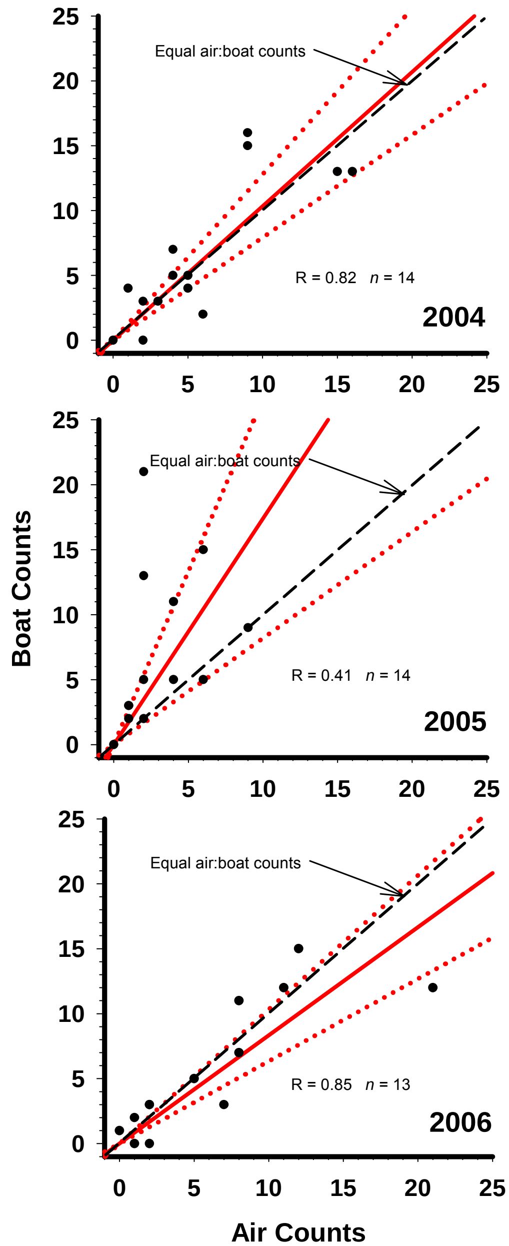

Figure 3. Regression lines and 95% confidence intervals comparing the numbers of indicated ring-necked duck breeding pairs counted from a boat and from the air on 14 lakes comprising the *Bemidji Area Ring-necked Duck Survey*, June 2004 (top), 2005 (middle), and 2006 (bottom).

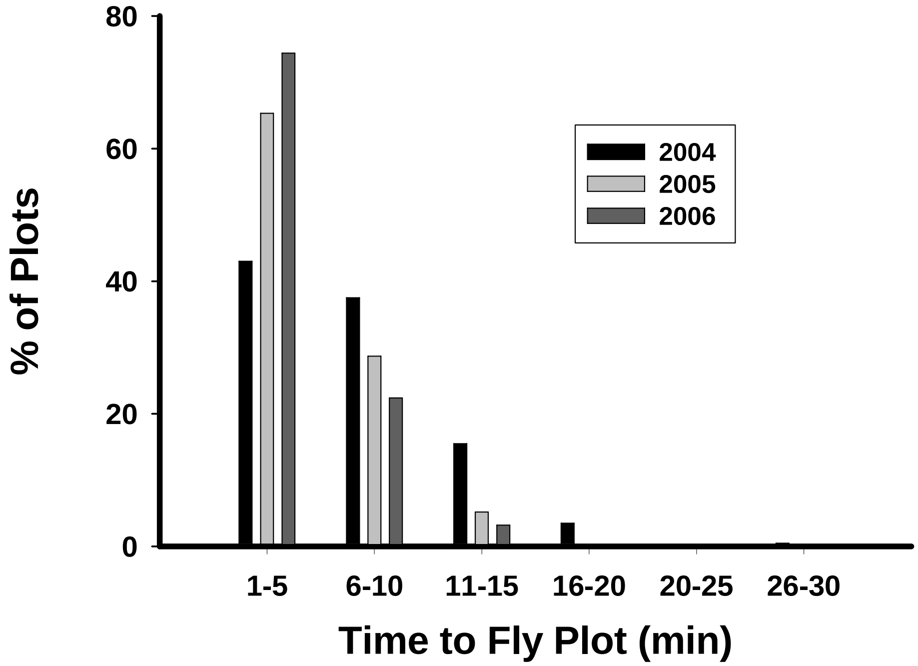

Figure 4. Time required (all observers) to survey individual ring-necked duck breeding pair plots in the Minnesota survey, June 2004 – 2006.

EVALUATING FUNCTIONAL LINKAGES AMONG LANDSCAPES AND WETLAND ATTRIBUTES: ASSESSING THE ROLES OF GEOMORPHIC SETTING, LAND USE, AND FISH ON WETLAND COMMUNITY CHARACTERISTICS

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SUMMARY

During 2005-06, we assessed fish community patterns and influences of site- and landscape-level variables on fish communities and ecological features of prairie wetlands in two areas in western Minnesota (generally Polk and Grant County areas). Fish populations were found to occur in nearly all wetlands. Diverse, multi-species fish communities were common, and often contained combinations of planktivorous, benthivorous, and piscivorous species. Preliminary analyses indicated that landscape-scale variables were poor predictors of fish populations in study wetlands. However, fish communities did reflect wetland size and site-level influences of piscivores. Here, we summarize procedures used in development and analyses of spatial (landscape) and site-level wetland data, and discuss preliminary trends in major variables including fish communities, aquatic invertebrates, limnological characteristics, submerged macrophytes, waterfowl use (breeding pairs and broods), amphibians, and periphyton. Future publications will more thoroughly describe relationships among these variables and landscape characteristics at spatially-explicit scales, and will clarify site-level influences of fish on wetland invertebrates, submerged macrophytes, and characteristics of clear – vs. turbid – water states in shallow Minnesota lakes.

INTRODUCTION

Installation of drainage tile and ditches, consolidation of wetlands, and other anthropogenic activities (e.g., agricultural land uses, road construction, nonnative invasive flora and fauna, intentional fish stocking, water control structures) are widespread in prairie regions of Minnesota. It is plausible that these landscape modifications have increased ecological influences of wetland fishes (reviewed by Bouffard and Hanson 1997), favoring preponderance of turbid, phytoplankton-dominated wetlands with low abundances of invertebrates and submerged aquatic vegetation. Furthermore, a prolonged period of above-average precipitation in Minnesota has increased depth of many prairie wetlands, increased surface connectivity among wetlands, and favored lower frequency of winter anoxia. These interacting influences contribute to development of permanent populations of fathead minnows (*Pimephales promelas)* and other fish species in a large proportion of wetlands remaining in Minnesota's prairie region (Hanson et al. 2005).

In shallow lakes and wetlands, reductions in herbivorous zooplankton due to predation by planktivorous fish are thought to reduce water transparency, favoring shifts towards increased turbidity and loss of submerged vegetation (Scheffer et al. 1993; Scheffer 1998). Across western and southern Minnesota, landscape modifications, along with resulting changes in fish distribution and population persistence, may have favored shifts toward a large proportion of degraded prairie marshes. Presently, many such sites are characterized by high turbidity, sparse communities of submerged aquatic plants and invertebrates, and limited suitability for waterfowl.

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Aquatic food web characteristics reflect density and community structure of associated fish populations. Fish-mediated influences on invertebrate community structure and water transparency are often pronounced (Bendell and McNicol 1987; Wellborn et al. 1996; Zimmer et al. 2000, 2001). Scheffer proposed that shallow-water ecosystems exist in 1 of 2 alternative conditions, either a clear-water, macrophyte-dominated state, or a turbid-water, phytoplankton-dominated state (Scheffer et al. 1993). Recent studies in Minnesota's Prairie Pothole Region (PPR) documented the strong negative influences of fathead minnows on invertebrate populations (Zimmer et al. 2000, 2001, 2002). Reductions in herbivorous zooplankton resulting from fish predation have been shown to increase phytoplankton biomass and turbidity consistent with predictions of models by Scheffer et al. (1993) and Scheffer (1998). Minnesota PPR wetlands largely conform to a binomial distribution (clear or turbid), rather than a normal distribution of features along a theoretical continuum (Zimmer et al. 2001; Herwig et al. 2004).

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

Fish community composition in lakes reflects interplay of isolation and extinction, but Magnuson et al. (1998) suggested that extinction is a more important influence. Extinction factors generally include environmental features of lakes such as surface area, habitat heterogeneity, depth and depth-related factors, winter oxygen concentrations, pH, presence of piscivores, watershed position, and local water chemistry (Tonn and Magnuson 1982; Rahel 1984; Marshall and Ryan 1987; Robinson and Tonn 1989; Keller and Crisman 1990, Magnuson et al. 1998). Isolation features are also important in structuring fish assemblages in lakes, and reflect differences in geomorphic setting (Magnuson et al. 1998; Hershey et al. 1999). Landscape features important in structuring fish assemblages in arctic lakes are primarily attributes of the lake/stream drainage network (Hershey et al. 1999). Alternatively, in small northern lakes, a combination of factors including presence of connecting streams, barriers, and characteristics of nearby species source pools have been identified as important predictors of fish community characteristics (Magnuson et al. 1998).

Fish community composition has been successfully predicted from only a few landscape and environmental variables, likely indicating that structuring mechanisms are robust (Tonn and Magnuson 1982, Rahel 1984, Robinson and Tonn 1989; Magnuson et al. 1998; Hershey et al. 1999). Fish assemblages in lakes also reflect regional and geographic patterns of fish distributions (i.e., reflecting local species pools) when larger spatial scales are considered (lakes - Jackson and Harvey 1989; wetlands - Snodgrass et al. 1996). In isolated wetlands in the southeastern US, disturbance frequency (drying) and connectivity determined the presence or absence of fish (Snodgrass et al. 1996). In contrast to prairie wetlands where low winter oxygen concentrations and sometimes drought influence fish distributions (Peterka 1989), drying and colonization rates were more important in determining the distribution of fish in coastal-plain wetlands (Snodgrass et al. 1996). Along the eastern part of the PPR (e.g., Minnesota), there is a propensity for intermittent surface water connections, and frequent fish invasions, as a result of natural east-west gradients in precipitation and topography (Leibowitz and Vining 2003; Hanson et al. 2005). In Minnesota's PPR wetlands, isolation (or connectivity) and extinction (environmental) characteristics are likely both important in structuring fish assemblages, but relative magnitude of influences are unknown.

Hershey et al. (1999) suggested a "geomorphic trophic" model to illustrate how stream drainage networks influenced the dispersal and subsequent distribution of native fishes in arctic lakes. There, fish controlled lake trophic structure (invertebrates, prey fish, etc.), but influences of fish also reflected extinction and isolation of fish populations due to constraints of landscape features. Thus, landscape configuration indirectly controlled trophic structure and expression of specific biological attributes within these lakes (Hershey et al. 1999). This model forms the basis of our overall working hypothesis of landscape control of PPR wetland food webs, where:

Geomorphic setting → Fish distributions → Ecosystem characteristics

We hypothesize that landscape and environmental features constrain fish communities, and interactively regulate the distribution of wetland fish throughout PPR regions of Minnesota. Fish, in turn, influence ecological characteristics of semi permanent and permanent wetlands in Minnesota's prairie landscape. Hence, by extension, landscape setting indirectly influences wetland ecosystem characteristics.

Landscape setting, including site-level wetland characteristics, may also directly influence water body features. For example, watershed position of a lake or wetland determines a variety of physical, chemical, and biological attributes of lakes (Kratz et al. 1997; Riera et al. 2000) and wetlands (Euliss et al. 2004). These properties include potential responses to drought, predominant groundwater interactions, and concentrations of dissolved constituents including organic carbon in lakes (Kratz et al. 1997). Other landscape features that have been found to influence water quality in lakes include percentage wetland extent in the watershed (Detenbeck et al. 1993; Prepas et al. 2001) and land use, where agricultural land was associated with a higher trophic state index (Detenbeck et al. 1993). Site-level wetland characteristics and processes that may also influence community characteristics include nutrient status (Scheffer et al. 1993; Bayley and Prather 2003; Jackson 2003), lake surface area (Hobæk et al. 2002; Wellborn et al. 1996), wetland depth (Scheffer et al. 1993), and macrophytes (Scheffer et. al. 1993; Paukert and Willis 2003; Zimmer et al. 2003).

The goal of our study was to develop conceptual and empirical models linking landscape features, site-level environmental influences, and wetland fish assemblages, and to assess the influences of these factors on characteristics of semi-permanent and permanent prairie wetlands. Our overall working hypothesis was that landscape setting indirectly influences wetland characteristics through structuring influences on fish communities.

METHODS

Study Area, Site Selection, Development of Landscape Predictor Variables

Our proposed study areas ("study landscapes") were selected to reflect a range of human-induced modifications. This gradient of anthropogenic influence results largely from a north to southwest transition toward increasing agricultural land use within Minnesota's PPR. Thus, our study focuses on 2 landscapes, 1 high-impact (HI) and 1 low-impact (LO) landscape. The HI landscape is located primarily in the southern portions of Grant County, and extends into northern Stevens County and western Douglas County, and includes 1 site in Ottertail County (Figure 1). The LO landscape is located primarily in eastern half of Polk County, with 1 study site located in northern Mahnomen County (Figure 1). In addition to differences in extent of human influence between our study landscapes, the HI and LO landscapes also fall into different ECS classifications due to variation in geomorphic features, climate, and vegetation patterns (Almendinger et al. 2000). Our study landscapes are also positioned in different major river drainages. The HI landscape lies between the Red River and Minnesota River drainages, while the LO landscape is located entirely within the Red River drainage. The LO and HI landscapes encompass approximately 1,292 km^2 and 1,435 km^2 respectively.

Within each study landscape, wetlands were selected for measurement of fish assemblages, wetland characteristics, and surrounding landscape attributes. For each landscape study area, we identified all candidate Type IV and V wetlands using the Minnesota Department of Natural Resource's (MN DNR) National Wetlands Inventory (NWI) Quick Theme layer. From this layer, we selected all "MnWet 4s and 5s", which best match the Circular 39 Types IV and V (Shaw and Fredine 1956; Stewart and Kantrud 1971). We imposed additional requirements such that all resulting candidate wetlands were between 2-40.5 ha, and were not licensed for aquaculture activities. Determining aquaculture status was accomplished by referencing the population of candidate study sites against the Division of Waters (DOW) numbers corresponding to basins licensed for aquaculture (either white sucker or walleye [*Sander vitreus]*) (Roy Johannes, MN DNR Fisheries, Aquaculture Program Coordinator). The remaining population of study sites within each study area was then stratified among 27 different bins based on the following criteria: 1) wetland size; 2) distance to nearest permanent stream, wetland, or lake; and 3) proportion agriculture within a 500 m buffer surrounding the wetland. We then randomly selected 1 study site from each of the resulting categories (for a total of 27 sites), and 9 (LO landscape) or 10 (HI landscape) additional sites across the 27 categories, with a maximum of 2 study sites per category imposed. If we were unable to obtain permission for a wetland in private ownership, or if some other conflict was identified (e.g., inaccessible), we then randomly selected a new site within that category, and repeated this process until a suitable study site was identified. In 2006, we selected 2 additional sites that were known to be fishless from previous studies within the HI landscape to facilitate statistical comparisons between fish and fishless sites.

A total of 36 study sites were selected for study in the LO landscape (35 sites in Polk County, 1 site in Mahnomen County) and a total of 39 study sites were selected for study in the HI landscape (31 sites in Grant County, 4 sites in Stevens County, 3 sites in Douglas County, 1 site in Ottertail County). In the LO landscape, 22 of the 36 study sites are either partially (6 sites), or completely (16 sites) within public ownership (i.e., Waterfowl Production Area (WPA), Wildlife Management Area (WMA), or National Wildlife Refuge (NWR)), with the remaining 14 study sites completely within private ownership. In the HI landscape, 21 of the 39 study sites are either partially or completely within public ownership (i.e., WPA, WMA, or county-owned), with the remaining 18 study sites completely within private ownership.

Existing GIS layers will be used to derive metrics that characterize features of the landscape surrounding each study wetland. Data layers not currently available will be developed as needed. Landscape attribute summaries might include, but are not limited to, the following: 1) distance to permanent and ephemeral water bodies; 2) distance to roads and driveways; 3) distance to (or presence of) drainage ditches and culverts; 4) latitude, elevation, and position of the study sites within the watershed; 5) surrounding land use assessed at multiple spatial scales; and 6) watershed ratios: direct contributing area (DCA) and total watershed areas (TWA) to the wetland surface area will be calculated. Watersheds for the DCAs and TWAs for each wetland will be manually delineated in Arc View using standard heads-up digitizing techniques. The on screen digitizing environment will incorporate hydrologically-corrected digital elevation models (DEM), digital raster graphics (DRGs), and digital orthoquads (DOQs), digitized hydrological connections and directional flow captured from DOQs, DEMs and/or corroborated from field inspections, as well as evidence from several other data sources (e.g., NWI, DNR streams and rivers coverage with proper connectivity and directionality, DOW Protected Wetlands Inventory (PWI) lake coverage, Department of Transportation (DOT) culvert point coverage, and existing digital major and minor watershed coverages). From this information, we plan to extract watershed areas at several spatially-explicit scales, summarize land cover types within watersheds, develop variables that capture influence of hydrological connectivity and geomorphic setting, and calculate average watershed slope. Within the DCA of each wetland, surrounding land cover types and connectivity features (streams, ditches) will be captured and categorized as outlined in Table 1. Our primary references for delineating land cover features were 2003 FSA Color DOQs, and 1 square mile land use maps obtained from county Farm Service Agency (FSA) offices. We will apply existing GAP data layers (or 2000 land cover data layers furnished by the USFWS, HAPET Office; Fergus Falls, MN), and existing flow network layers (MN DNR-Division of Waters) to characterize cover types and hydrological features within the watershed areas extending beyond the DCAs. We will use ArcView to summarize land cover types at various distances from the study basin, up to and including the DCA, and TWA scales. Existing aerial photographs (2003 FSA Color DOQs) and Global Positioning System (GPS) mapping were used to develop updated estimates of wetland size in 2005. Maximum depth of the study wetlands was also determined during the 2005 field season by measuring depths along parallel transects throughout the open water zone of each wetland.

Fish Community Assessments

Fish species composition was determined from July surveys using a combination of gear deployed overnight. Three mini-fyke nets (6.5 mm bar mesh with 4 hoops, 1 throat, 7.62 m lead, and a 0.69 m X 0.99 m rectangular frame opening into the trap) were set overnight in the littoral zone of each wetland. One experimental gill net (76.2 m multifilament net with 19, 25, 32, 38, and 51-mm bar meshes) was set along the deepest depth contour available in wetlands less than 2 m deep or along a 2 m contour in wetlands with sufficient depth. Preliminary results from the LO landscape indicated that results from minnow traps were redundant with the other types of gear, so fish sampling was restricted to 3 mini-fyke nets and 1 gill net per wetland in the HI landscape.

The protocol outlined above has been shown to be effective in sampling fish assemblages in small lakes from other regions (Tonn and Magnuson 1982; Rahel 1984; Jackson and Harvey 1989; Robinson and Tonn 1989), and enabled us to capture fish of different sizes, species, and from all major trophic guilds (e.g., planktivores, benthivores, piscivores) in the study wetlands. Number of individuals and total biomass of each species collected were determined for each type of gear in each site.

Aquatic Invertebrates

Zooplankton were sampled twice each year, once in late May or early June, and again in late July or early August by collecting 2 replicate vertical column samples (Swanson 1978a) at 6 locations in each wetland. Resulting data were used to estimate density of major invertebrate groups and taxon richness of these communities. Relative abundance of free-swimming macroinvertebrates was estimated using submerged activity traps (ATs) (Swanson 1978b; Murkin et al. 1983; Ross and Murkin 1989) placed in each wetland for 24-hours. Six ATs were deployed at the interface of open water and emergent macrophytes because this area often concentrates organisms. Estimates of relative abundance and taxon richness were developed for each study site. We collected aquatic invertebrate samples from 73 wetlands during 2005 and from 75 sites during 2006.

Limnological and Phytoplankton Sampling

Surface (dip) water samples were taken from the center of each wetland once during late May or early June, and again in late July or early August each year. Samples were acidified to a pH of 2 using concentrated sulfuric acid, then frozen. Samples were analyzed by the Minnesota Department of Agriculture chemistry lab (St. Paul, MN) for total Kjeldahl, nitrate-nitrite, and ammonia nitrogen, as well as total phosphorus. Additional water was collected at the same time as the surface samples for total dissolved phosphorus (TDP) and phytoplankton abundance, measured as chlorophyll *a* (Chla). TDP samples were collected by filtering lake water through GF/F glass fiber filters (0.7 µm nominal pore size) and immediately freezing filtered water. Chla samples were collected in the field by filtering lake water through a GF/F glass fiber filter. The filters were then wrapped in tin foil and immediately frozen. In the lab TDP was determined using high-temperature persulfate digestion followed by ascorbic-acid colorimetry (APHA 1989). Chla was measured in the lab using a 24 h, alkaline-acetone extraction, followed by fluorometric analysis (APHA 1989). Turbidity and specific conductance were measured in the field with a portable nephelometer and a conductivity meter.

Submerged Macrophyte Surveys

Species richness, frequency of occurrence, and community-scale biomass of submerged macrophytes were assessed using techniques of Jessen and Lound (1962), and Deppe and Lathrop (1992). In each wetland, submerged macrophytes were sampled in early August at 20 stations along 4 transects. Two throws were made at each station using a weighted plant rake, and frequency of occurrence was recorded for each plant species. We then calculated a whole-wetland score (number of times each species was collected in 40 rake throws; resulting species scores were then summed across all submerged plant taxa); values hereafter referred to as “submerged plant score”. We also measured the total plant biomass (all species combined) for the first rake throw at each station. Metaphyton (e.g. *Cladophora* spp.) and macroalgae (e.g. *Chara* spp.) were assessed along with vascular plant species during these surveys.

Waterfowl Surveys

Waterfowl numbers were assessed during the breeding season and brood-rearing period using helicopter survey techniques (Cordts 2002). Indicated breeding pairs (lone male, pairs, and flocked males < 6) were tallied by species on each wetland during early May 2005 and 2006. Groups by species were also recorded. We assumed that all individuals were seen using the helicopter survey technique (Ross 1985, Cordts 2002). We also counted the number of waterfowl broods on each wetland during late-June or July 2005 and 2006. Broods were recorded by species where possible and number of ducklings was estimated.

Amphibian Surveys

We sampled larval amphibians concurrently with fish (using the same gear used to sample fish, described above). In each wetland, we determined the total number of larval frogs, larval salamanders, and painted turtles captured with the 3 trap nets and 1 gill net set in each wetland during late July. Results are expressed as the total number of individuals captured in each study site.

Periphyton Measurements

Periphyton biomass (Chla) was determined by deploying artificial substrates for 5 weeks in 2005 and 4 weeks in 2006 (4 weeks was found to be sufficient to get maximum growth without sloughing of periphyton). Sampling devices were set out in mid-June and collected in late-July each year. These devices were constructed out of a polyester braided rope (6.35 mm thick, 1.5 m long) with a brick anchor attached to one end and a float on the other. Along the rope, 3 vinyl microscope slides were attached using zip-ties at 10, 50, and 90 cm below the surface because we hoped to assess whether periphyton abundance differed with depth. Total height of the sampling device was 1.5 m when placed vertically in the water column. Three devices were deployed in each wetland, near locations where invertebrates were sampled. Each sample was carefully removed from the water column to limit disturbance to the periphyton, placed in a container of well water and stored in a dark cooler until taken to the lab and processed within 12 hours. Chla was measured in the lab using a 24 h, alkaline-acetone extraction, followed by fluorometric analysis (APHA 1989).

RESULTS

Fish Communities

Sites within the HI landscape generally contained fewer fish species on average than sites within the LO landscape during both 2005 (4.00 vs. 5.61 species) and 2006 (3.82 vs. 5.57 species; Figure 2). The HI landscape had 19 sites and 21 sites with 3 or fewer fish species in 2005 and 2006, respectively, while the LO landscape had just 6 sites with 3 or less species of fish in both years. Maximum number of fish species sampled in sites within the HI landscape was 10, while the LO landscape had 3 sites with 11 or more fish species in 2005. In 2006, the maximum number of species in the HI landscape increased to 11, but dropped to 9 in the LO landscape. Twenty-seven different fish species were sampled across both the HI and LO landscapes. We sampled 23 species of fish across the HI landscape sites, and 21 species of fish across the LO landscape sites in 2005. In 2006, we sampled 2 fewer species in both the HI (21 species) and LO (19 species) landscape sites.

Study areas (HI and LO) shared 14 species of fish across years, including black bullhead, black crappie (*Pomoxis nigromaculatus)*, bluegill (*Lepomis macrochirus)*, brown bullhead (*Ameiurus nebulosos)*, brook stickleback (*Culaea inconstans)*, central mudminnow (*Umbra limi)*, fathead minnow, golden shiner (*Notemigonus crysoleucas)*, largemouth bass, northern pike, pumpkinseed (*Lepomis gibbosus)*, tadpole madtom (*Noturus gyrinus)*, white sucker, and yellow perch (*Perca flavescens)*. Species common to both study areas in 2005, but not in 2006, included common carp, Iowa darter (*Etheostoma exile)*, and walleye. In 2006, common carp and walleye were sampled in the HI landscape but not in the LO landscape, while Iowa darter was found in the LO but not in the HI landscape. Green sunfish (*Lepomis cyanellus)* was common to both landscapes in 2006. However, in 2005, green sunfish was sampled only in the LO landscape. Species that were present in the HI landscape, but never present in the LO landscape, included bigmouth buffalo (*Ictiobus cyprinellus)* (2005 only), channel catfish (*Ictalurus punctatus)* (2005 only), freshwater drum (*Aplodinotus grunniens)* (both years), orangespotted sunfish (*Lepomis humilis)* (2005 only), shorthead redhorse (*Moxostoma macrolepidotum)* (both years), and yellow bullhead (*Ameiurus natalis)* (both years). Species present in the LO landscape, but never present in the HI landscape, included blacknose shiner *Notropis heterolepis* (2006 only), brassy minnow (*Hybognathus hankinsoni)* (both years), and northern redbelly dace (*Phoxinus eos)* (both years). Common shiner (*Luxilus cornutus)* was found in the LO but not in the HI landscape in 2005, and vice versa in 2006.

There were also some interesting differences between study landscapes with respect to proportions of the 4 major fish community types. The HI landscape contained 3-4 fishless sites, while only 1 site was fishless in LO landscape (Table 2). The number of planktivore-only (P) sites was similar in both areas, but sites with planktivores, benthivores, and piscivores (PBP) were approximately twice as numerous (n=15-16) as the number of sites containing planktivores and benthivores (PB) (n=8) in the HI Landscape (Table 2). Proportion of sites with common carp also differed between the 2 study landscapes; we sampled 12-14 sites with carp in the HI landscape, but only 0-2 sites in the LO region. Common carp were sampled in 2 additional sites in the HI landscape in 2006. Carp were absent in 1 site in the LO landscape where they were captured in 2005; also, during 2006, we were unable to sample the only other LO landscape site that contained carp in 2005.

Within each of the major fish community types, the biomass of each guild (i.e., planktivore, benthivore, piscivore) was higher in the HI landscape sites than in the LO landscape sites in both 2005 and 2006 (Figure 3). Most striking was the pattern for benthivores, which were 2.9-5.3 times higher in the HI than LO sites in both the PB, and PBP community types. Planktivores in the P sites (HI and LO landscapes), and benthivores in the PBP sites (HI landscape only) also were lower in 2006 compared to 2005. This inter-annual variation may reflect differential over-winter mortality or summerkill between years, less basin interconnectivity in spring 2006 (drier, less snowpack), or spatial-temporal sampling effects (much hotter during 2006 fish sampling—less active, restricted to deep-water refuges). It is also interesting to note that planktivore biomass was always low in the presence of piscivores (i.e., predators), suggesting that piscivory may be an important structuring mechanism in these wetlands, as it is in larger lakes (Tonn and Magnuson 1982, Robinson and Tonn 1989).

Future analyses will focus on predicting fish community characteristics from basin characteristics and geomorphic setting, including the study site's relationship to characteristics of the surface water drainage network, surrounding land uses, as well as anthropogenic factors and features. Other focus areas will include: 1) exploring relationships between fish community "types" and wetland characteristics (invertebrate communities, submerged plant vs. algal dominance, etc.); 2) better understanding inter-specific interactions among wetland fish species (i.e., roles of predation and competition); and 3) understanding the relative roles of watershed features (e.g., agricultural land use) and fish in determining alternative equilibria in wetland ecosystems (clear vs. turbid, etc.).

Aquatic Invertebrates

We assessed potential relationships among fish communities, characteristics of wetland study sites, and wetland invertebrates graphically and using model selection procedures (Anderson and Burnham 2002). In our case, "best" models were selected from combinations of fish community characteristics, wetland features, study site location (focus area), and other site attributes that have been previously shown to influence aquatic invertebrates. In all cases, model fit was assessed using AICc (Anderson and Burnham 2002). To reduce the number of comparisons, we combined taxa, creating 3 aggregate variables including "macroinvertebrates" (common aquatic insects including Diptera, Coleoptera, Odonata, selected Hemiptera, and others), "amphipods" (Gammarus lacustris and Hyallela azteca), and "zooplankton" (primarily Daphnia spp.). Here, we describe preliminary results of analyses using relative abundance of these 3 aggregate taxa from July 2005 (lab processing of 2006 samples has only recently been completed).

Crustacea, aquatic insects, water mites, and snails were the most common aquatic invertebrates collected from study wetlands during 2005. In general, invertebrate taxa in these wetland communities were similar to those reported from recent work in wetlands in Minnesota (Hanson and Riggs 1995, Zimmer et al. 2000, Zimmer et al. 2002).

Relative abundance of macroinvertebrates in our study wetlands was best predicted by models including only mass of submerged aquatic macrophytes (plants) (R^2=0.28); however, a model including mass of planktivorous fish and plants performed nearly as well (R^2=0.30) (Table 3). By a wide margin, our best zooplankton model also included mass of plants and planktivorous fishes (R^2=0.36; Table 4). All of our amphipod models explained < 5 percent of observed variance; here, our best model included only mass of planktivorous fishes (R^2=0.03; Table 5). Alternative amphipod models, including total mass of planktivorous and benthivorous fishes (R^2=0.02) or a single variable depicting water clarity status (turbid or clear), achieved similar fits to observed data (Table 5).

Results indicated that mass of plants and planktivorous fish were important determinants of macroinvertebrates and zooplankton in our study wetlands during 2005, but these influences may interact in complex ways. For example, macroinvertebrate abundance was most influenced by plant mass, but also reflected abundance (mass) of planktivorous fish (Figure 4). All of our preliminary models explained < 5 percent of observed variability in amphipods. In no cases did our other invertebrate models explain more than approximately 30 percent of observed variability in 2005 data. Along with results of amphipod models, this probably indicates that important determinants of invertebrate community structure were not accounted for in our preliminary analyses.

We will continue to develop data from vertical column and activity trap samples collected during 2006. We have nearly completed enumeration of the samples collected in 2006. We expect that aquatic invertebrates will constitute an important response variable in our analyses of fish and land use effects. We will also explore potential influences of spatial and hydrological variables (such as distance to other water bodies and position in a watershed) on wetland invertebrate characteristics in these study sites.

Limnological Characteristics

In 2005, average total phosphorus (TP) concentrations (µg/L) increased between June and July in both the HI and LO landscapes (HI: 20% increase, LO: 55% increase), and were 7.2 and 5.6 times higher in the HI than LO landscape sites in June and July, respectively (Figure 5). In 2006, average total phosphorus (TP) concentrations (µg/L) again increased between June and July in the HI and LO landscapes (HI: 18% increase,
LO: 60% increase). The HI landscape sites had 4.9 and 3.6 times higher TP level than the LO landscape site in June and July, respectively (Figure 5).

TP averaged 22.2-34.5 μg/L across months in the LO landscape in 2005, sometimes falling within the range of TP values favoring persistent, clear-water, macrophyte-dominated conditions (Moss et al. 1996). In contrast, TP in the HI landscape averaged 161-193 μg/L across months in 2005, levels where basins can exhibit either clear-water, macrophyte-dominance or turbid, phytoplankton dominance (Moss et al. 1996). In 2006, TP exhibited a greater range in values, averaging 12.5-240 μg/L across months in the LO landscape and 24.5-609 μg/L across months in the HI landscape. Despite this greater range and higher TP values in 2006, TP exceeded 50 μg/L in 6 of 36 sites and 150 μg/L in just 1 site in the LO landscape. This contrasts strongly with the HI landscape, where TP exceeded 50 μg/L in 34 of 39 sites, and 150 μg/L in 15 of 39 sites. Data from the UK and elsewhere suggests that TP <150 μg/L is required for dominance by a diverse macrophyte community, and that macrophyte communities exhibit higher stability at TP <50 μg/L (reviewed in Madgwick 1999). Thus, many of the study sites (~56%) in the HI landscape have TP concentrations within a range where there is considerable potential for restoring macrophyte dominance through fish community manipulations, etc.

In 2005, mean turbidity was 2.4 times higher in the HI landscape sites than in the LO landscape sites in both June and July (Figure 6). Turbidity increased by approximately 50% between June and July in the HI landscape and by 12% in the LO landscape, reflecting a seasonal increase in phytoplankton abundance. In 2006, HI landscape sites had 2.7 and 2.8 times higher mean turbidity than the LO landscape sites in June and July, respectively. There was a 51% increase between June and July in the HI landscape and 44% increase in the LO landscape sites.

LO landscape sites were relatively clear (<5 NTUs), with just 9 sites having a turbidity >5 NTUs in both 2005 and 2006. In contrast, the HI landscape sites exhibited considerably more variability, with some sites characterized by clear water (10 sites had a turbidity < 5 NTUs in 2005, and 15 in 2006), but many sites characterized by very turbid water (14 sites had a turbidity >20 NTUs in 2005, and 11 in 2006).

Distributions of chlorophyll *a* (Chla) concentrations (μg/L) between the 2 landscapes were dramatically different in both 2005 and 2006 (Figure 7). In June 2005, there were only 9 sites in the LO landscape with Chla >15 μg/L, while there were 23 sites in the HI landscape with Chla >15 μg/L. In the HI landscape, this increased to 27 sites in July, with 10 sites exceeding 90 μg/L Chla. In June 2006, the LO landscape had only 5 sites with Chla >15 μg/L, while the HI landscape had 22 sites. In the HI landscape, this increased to 26 in July, with 9 sites exceeding 90 μg/L Chla (highest was 534.5 μg/L).

As for turbidity, there were several sites in the HI landscape with low levels of Chla (e.g., 10 sites in July 2005 with Chla <15 μg/L). These data suggest that alternative conditions of clear water, macrophyte dominance and turbid, phytoplankton dominance are represented in both landscapes, although the proportion of turbid sites is considerably higher in the HI landscape.

Submerged Macrophytes

Submerged plant biomass was similar between the HI and LO landscapes, as evidenced by overlapping standard errors (Figure 8). In contrast, the average submerged plant score showed that the coverage and diversity of macrophytes was higher in the LO landscape than in the HI landscape (68 vs. 32 in 2005, 74 vs. 40 in 2006). Although mean biomass of submerged plants was similar among study areas, the distribution of submerged plant biomass differed appreciably (only 2006 data summarized; Figure 9). In the HI landscape, there were many sites with low plant biomass (about half of the sites). The remainder of sites had intermediate to high plant biomass (long right tail). In contrast, in the LO landscape we observed only a few sites with low or high submerged plant biomass, with most sites having intermediate to high plant biomass.

Waterfowl Use

Despite high variability, more puddle and diving duck breeding pairs were observed within the HI landscape area, especially during 2005 (Figure 10). Overall, for both puddle and diving ducks combined, breeding pairs were observed on 92 and 77 percent of wetland study sites in HI and LO landscapes (respectively) during 2005 (Table 6). This trend continued during 2006 when breeding pairs were observed on 90 and 71 percent of wetlands in HI and LO landscapes. As with breeding pairs, more duck broods were also observed within the HI landscape area; this geographical contrast was much more obvious during the first year of our study (2005; Figure 10). Overall, duck broods were observed on 44 and 23 percent of wetland study sites in HI and LO landscapes (respectively) during 2005. This trend continued during 2006 when breeding pairs were observed on 45 and 36 percent of wetlands in HI and LO landscapes (Table 6).

Amphibian Populations

Our current analysis is limited to comparisons between landscapes and relative to fish community characteristics, as we do not yet have final data on the surrounding land cover types. Results from 2005 and 2006 indicated that highest abundances of tadpoles and larval salamanders were found in fishless wetlands in both HI and LO landscapes (Figures 11,12). Across all types of fish communities, the general abundance pattern for both tadpoles and salamanders in 2005 was fishless > planktivores (P) > planktivores/benthivores (PB) > planktivores/benthivores/piscivores (PBP) (Figure 11). A similar pattern was observed in 2006, but only in the HI landscape (Figure 12).

Few tadpoles or salamanders were captured in wetlands with piscivores (PBP sites) in either study landscape in both 2005 and 2006 (Figures 11,12). Finally, we observed no consistent relationships between relative abundance of painted turtles and fish community types. However, the lowest abundance of painted turtles occurred in the HI landscape PBP sites in 2005, and PB and PBP sites in 2006.

Periphyton Distribution and Dynamics

In 2005, sites in the HI landscape (534 µg/L) had a higher average periphyton biomass per wetland than those in the LO landscape (266 µg/L). HI landscape sites had a wider range of periphyton biomass than LO landscape sites (Figure 13). Periphyton biomass generally decreased with water depth (slide positions: top, middle, bottom) (Figure 14).

During 2006, lakes in the HI landscape (918 µg/L) again had a higher average periphyton biomass per wetland than those in the LO landscape (109 µg/L). Wetlands in the HI landscape also had a wider range of periphyton biomass values than those in the LO landscape (Figure 13). One wetland in Grant County (HI) had an average biomass of 5727 µg/L, whereas the highest value in Polk County (LO) was 411 µg/L. There were no consistent trends between periphyton biomass and water (slide) depth (Figure 14). Overall, results show that there is considerable variance within each of the landscapes, especially among sites in the HI landscape.

Model selection and model averaging showed that macrophyte biomass was the best single predictor of periphyton biomass at all depths in 2005 (data not shown). No fish parameters were present in any of the candidate models, thus fish were poor predictors of periphyton biomass. Nutrient (particularly phosphorus) and macroinvertebrate variables occurred repeatedly in candidate models predicting periphyton biomass at each depth. Therefore, preliminary results show that periphyton biomass is influenced by both top-down (macroinvertebrates) and bottom-up (macrophytes and nutrients) factors.

SUMMARY

Data gathered during 2005 and 2006 indicated that fish populations occurred in nearly all wetland study sites. Diverse fish communities were common and often contained combinations of planktivorous, benthivorous, and piscivorous species. As far as we know, our research is the first to simultaneously measure direct and indirect influences of fish on prairie wetland characteristics in the sense that, in addition to site-level effects, we are also assessing fish communities in response to landscape characteristics at several spatial scales.

Previously, we reported results of preliminary analyses indicating that attributes of adjacent landscapes were poor predictors of fish populations in study wetlands; however, fish communities did reflect wetland size and depth, along with site-level influences of piscivores (Hanson et al. 2006). Preliminary data summarized here suggests that wetland fish abundance and/or community type were associated with important components of wetland study sites including aquatic invertebrates, limnological characteristics (such as water clarity and phytoplankton abundance), and relative abundance of amphibians. Early results also indicated that fish influences differ among feeding guilds (planktivores – fathead minnows, benthivores – black bullheads). For example, increasing mass of benthivorous fishes was significantly associated with declining mass of submerged aquatic plants during 2005 (Hanson et al. 2006) and 2006 (data not shown here); in contrast, no similar relationships was observed between submerged plants and planktivorous fishes in either study year. It is also notable that fish influences are complex and often interact with other wetland characteristics. For example, during 2005, planktivore mass was negatively associated with relative abundance of macroinvertebrates in study wetland. However, our best invertebrate models also showed strong evidence of interactions between mass of planktivorous fish and submerged plant mass; this indicates that macroinvertebrates were most strongly suppressed in wetlands with high mass of planktivorous fish and low mass of submerged plants.

Finally, location was also an important determinant of wetland characteristics; study sites often differed dramatically in key ecological features between our HI and LO landscape study areas. For example, benthivorous fish mass, turbidity, total phosphorus, and phytoplankton concentrations tended toward higher values in our HI (Grant County) study sites. In part, this probably reflects an increasing gradient of nutrient availability along a statewide SW to NE trajectory, but may also relate to differences in proportions of agriculture or other anthropogenic differences between these regions. It may also indicate greater vulnerability of Grant County wetlands to shift to turbid-water states. Future analyses will focus on clarifying relationships among landscape cover types, surface-water connectivity, site-level geomorphic setting, and other spatial characteristics of wetland sites and associated fish communities, along with additional clarification of site-level influences of fish populations.

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Table 1. Landscape features captured using existing GIS layers or digitized using 2003 FSA air photos and 1 square mile land use maps as primary references.

Description	Definitions	Polygon source	Our label	FSA label
Grasslands: CRP, pasture, WPA and WMA uplands	· Grassy, does not include row crops or hay/alfalfa, but does include pasture · Established grassy uplands on WPAs and WMAs	Digitized	GRA	NC, FWS, DNR, CRP
	· Vegetated portions of right-of-ways associated with transportation	Buffered		
Woodlands	· Forested areas, with ground cover of greater than 75% mature trees	Digitized	WDL	NC, FWS, DNR
Shrubs	· Shrubby area mixed with grassy area, woodland area with ground cover of less than 75% mature trees	Digitized	SBL	NC, FWS, DNR
Row crops and hay	· Tilled crops, generally corn, soybeans, and small grains · Areas that are hayed annually including alfalfa and wild hay	Digitized	AGR	HEL, MHEL, NW
Non-study site lakes	· Entire area of lake or wetland including emergent vegetation (Lakes, Type IV, V wetlands, bogs with at least 10% open water and lakes)	Digitized	LKS	W
Non-study site wetlands	· All non-Type IV or V wetlands, and bogs with <10% open water; minimum size of 0.1 ha to be digitized	Digitized	WTL	W, CW, FW
Study sites	· Open water portion of the wetland	Digitized	OWT	W
	· Emergent vegetation along basin margins (use GPS reference points as guide). Includes cattails, sedges, Phragmites spp.	Digitized	EAV	
	· Islands with trees and shrubs	Digitized	ISL	
	· Emergent vegetation in the interior of basins (cattail islands)	Digitized	CTI	
Streams	· Continuously wetted and intermittent streams.	Quick Themes	CST	No label
Ditches	· Ditches containing water in fields (straight/linear, and contain water that you can see on an air photo) · Ditches associated with public roads and driveways	Digitized	DWT	No label
Farmsteads	· Active and abandoned farmsteads/homesteads and associated buildings and shelterbelts etc. regardless of size, but not the associated woodlands	Digitized	FST	NC
Roads	· Transportation surfaces	Quick Themes	RDS	No label
Other impervious surfaces	· Gravel pits and parking lots, towns	Digitized	OIS	No label
Driveways	· Driveways associated with farmsteads/homesteads	Digitized	DVW	No label

Table 2. Number of sites corresponding to each of the four major fish community types within the HI ("Grant County") and LO ("Polk County") landscapes in 2005 and 2006. Also tabulated is the number of sites falling within the P-Benthivores and P-B-Piscivores community types that also contain common carp.

	HI		LO	
Fish community "type"	No. of sites	No. of sites with common carp	No. of sites	No. of sites with common carp
2005				
Fishless	3	-	1	-
Planktivores	10	-	9	-
P-Benthivores	8	4	17	0
P-B-Piscivores	16	8	9	2
2006				
Fishless	4	-	1	-
Planktivores	12	-	10	-
P-Benthivores	8	5	16	0
P-B-Piscivores	15	9	8	0

Table 3. Results of model selection procedures using macroinvertebrate data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Plant Mass was the best indicated model (R^2=0.28; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	**3**	**73**	**26.27**	**1.0**
Planktivore Mass	3	73	45.09	12,206.0
Benthivore Mass	3	73	46.96	31,085.4
Planktivore+Benthivore Mass	3	73	39.79	866.9
Water clarity (turbid or clear)	3	73	45.57	15,537.2
July Chlorophyll a	3	73	43.08	4481.0
Plant Mass * Planktivore Mass	5	73	26.47	1.1
Plant Mass * Benthivore Mass	5	73	30.48	8.2
Plant Mass * Plank + Benth Mass	5	73	29.30	4.5

Table 4. Results of model selection procedures using zooplankton (*Daphnia*) data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Plant Mass* Planktivore Mass (interaction term) was the best indicated model (R^2=0.31; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	3	73	124.32	17.6
Planktivore Mass	3	73	142.88	187765.7
Benthivore Mass	3	73	131.67	695.5
Planktivore+Benthivore Mass	3	73	135.20	4057.0
Water clarity (turbid or clear)	3	73	136.62	8256.6
July Chlorophyll a	3	73	132.74	1188.7
Plant Mass * Planktivore Mass	**5**	**73**	**118.58**	**1.0**
Plant Mass * Benthivore Mass	5	73	125.77	36.3
Plant Mass * Plank + Benth Mass	5	73	128.40	135.6

Table 5. Results of model selection procedures using amphipod data gathered from study wetlands during 2005. Model fit was assessed using AICc values and resulting evidence ratios. Planktivore Mass was the best indicated model (R^2=0.03; indicated in bold).

Model terms	K	n	AICc	Evidence ratio
Plant Mass	3	73	51.46	3.3
Planktivore Mass	3	73	49.05	1.0
Benthivore Mass	3	73	51.29	3.1
Planktivore+Benthivore Mass	3	73	49.63	1.3
Water clarity (turbid or clear)	3	73	50.40	2.0
July Chlorophyll a	3	73	51.33	3.1
Plant Mass * Planktivore Mass	5	73	51.10	2.8
Plant Mass * Benthivore Mass	5	73	54.80	17.7
Plant Mass * Plank + Benth Mass	5	73	53.67	10.1

Table 6. Breeding pair and duck brood characteristics in the HI (Grant County) and LO (Polk County) study landscape during 2005-2006.

	Study Landscape			
	HI		LO	
Characteristic	2005	2006	2005	2006
Number of sites with at least one breeding pair	34	35	27	25
Total number of study sites	37	39	35	35
Percentage of sites with at least one breeding pair	92%	90%	77%	71%
Number of sites with at least one brood	16	17	8	13
Total number of study sites	36	38	35	36
Percentage of sites with broods	44%	45%	23%	36%

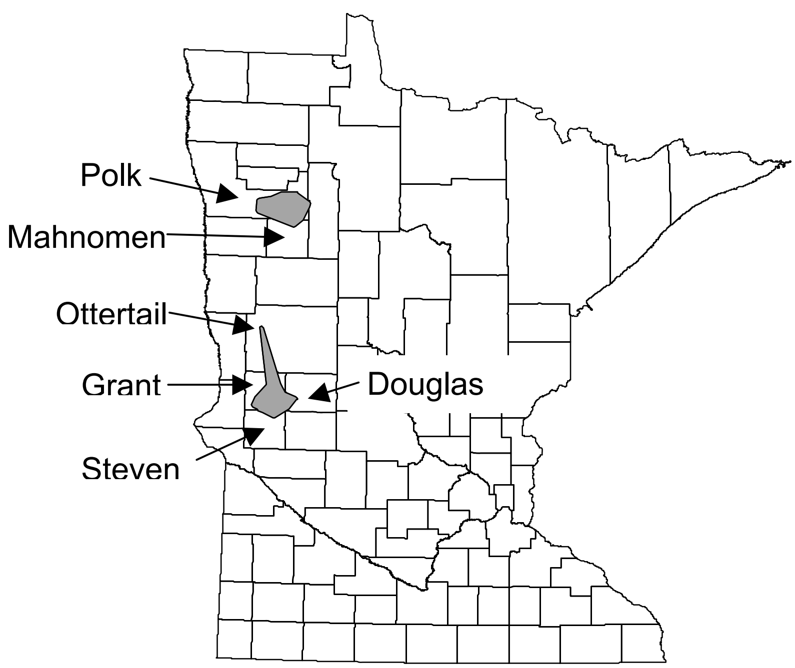

Figure 1. Map showing locations of study landscapes. Study areas are defined by a polygon drawn around the outermost 1-mile buffers surrounding each of the study sites.

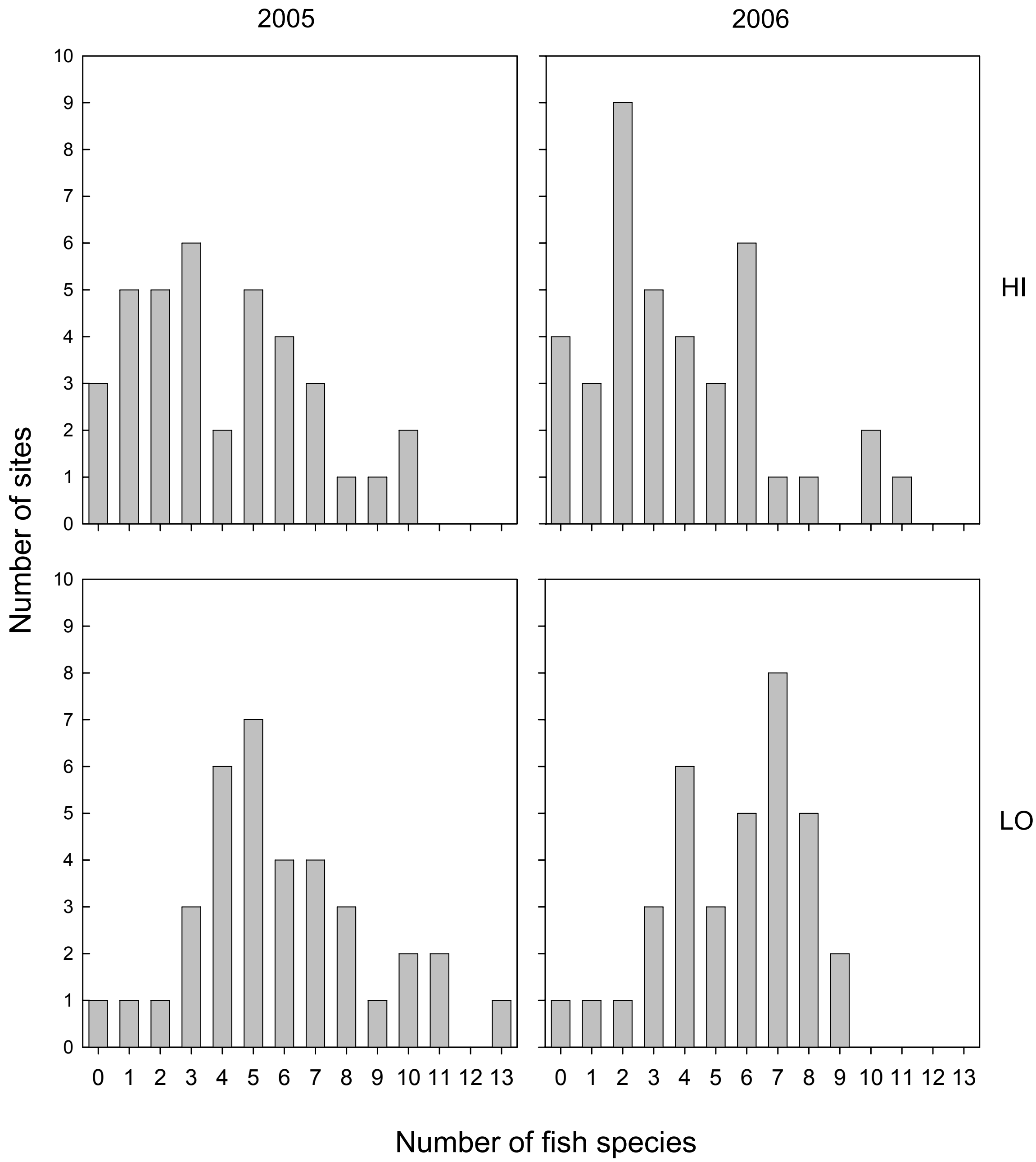

Figure 2. Frequency distributions showing fish species richness for study sites located in the HI landscape (“Grant Co” – top panels) and LO landscape (“Polk Co.” – bottom panels) in summer 2005 and 2006.

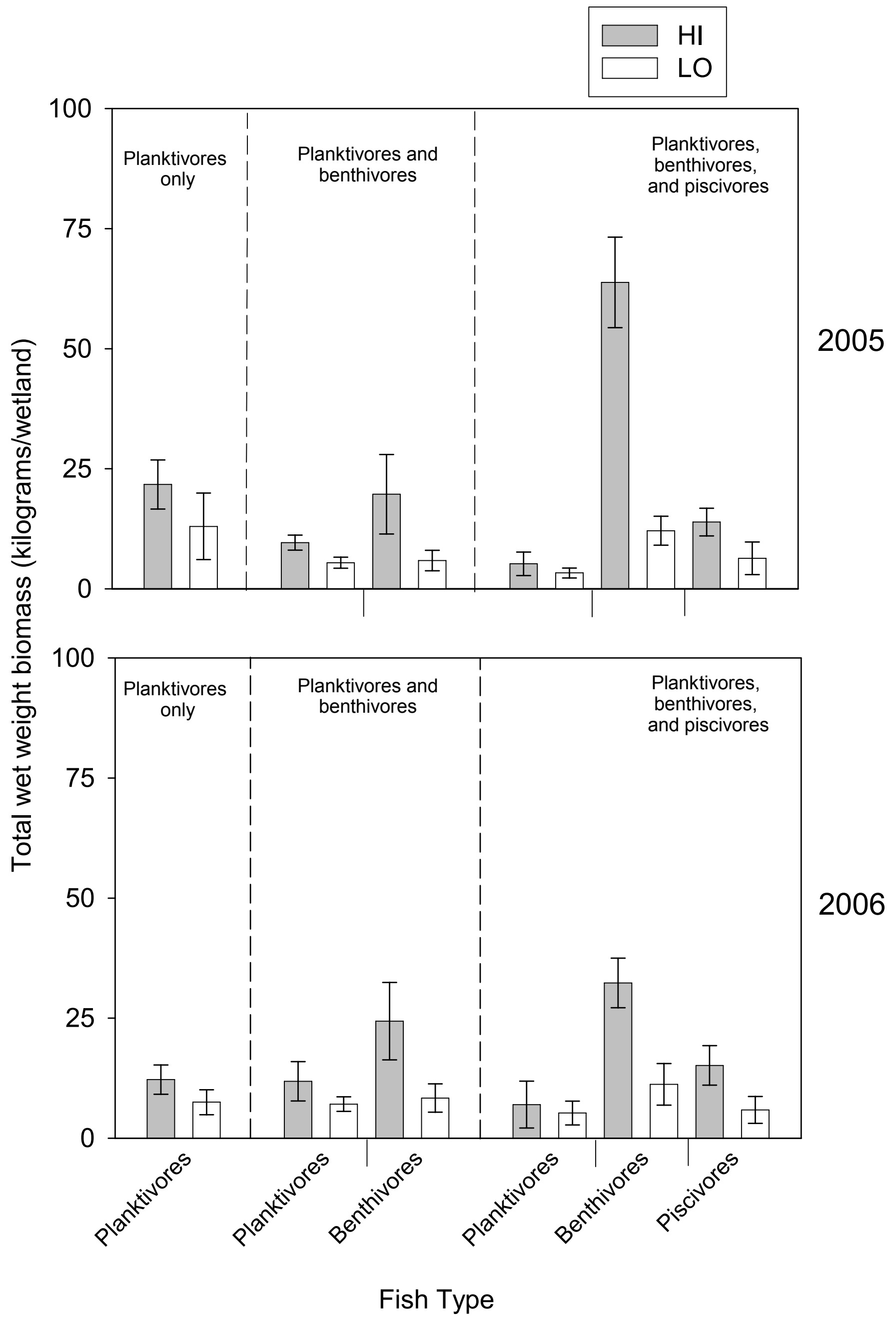

Figure 3. Average fish biomass (+/- 1 SE), summarized by guild (i.e., planktivores, benthivores, piscivores), for each of the major fish community types for the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co." – white bars) in 2005 (top panel) and 2006 (bottom panel).

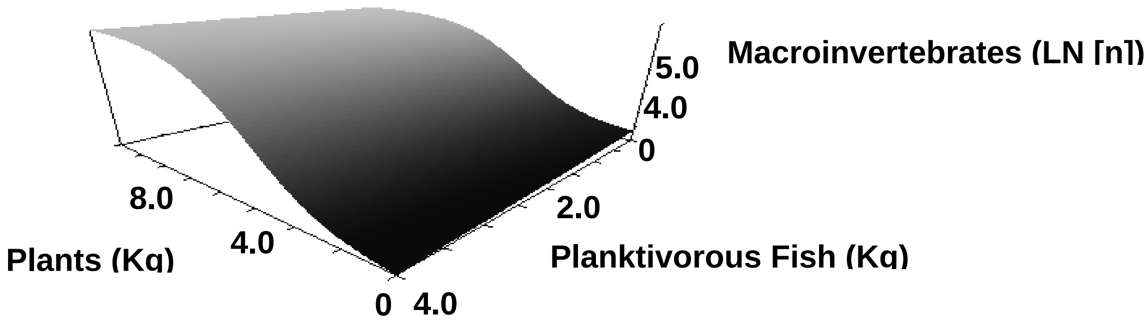

Figure 4. Relationship among macroinvertebrate abundance, mass of submerged aquatic plants (plants) and mass of planktivorous fish measured in 73 study wetlands during July 2005. Smoothed surface depicts predicted values derived using nonparametric multiplicative regression model (Hyperniche [McCune and Mefford 2004]).

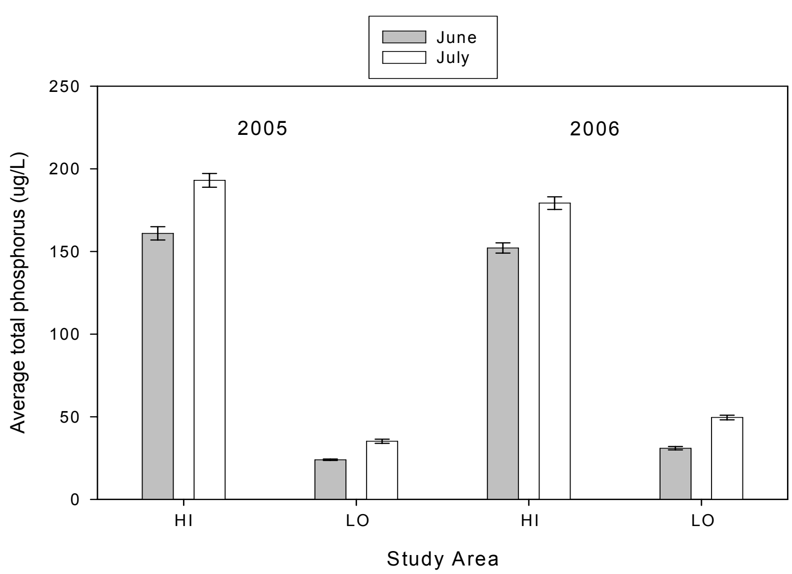

Figure 5. Average total phosphorus concentration (μg/L) for study sites in the HI landscape (“Grant Co”) and LO landscape (“Polk Co”) in June (black bars) and July (grey bars) 2005 and 2006. Error bars are +/- 1 SE.

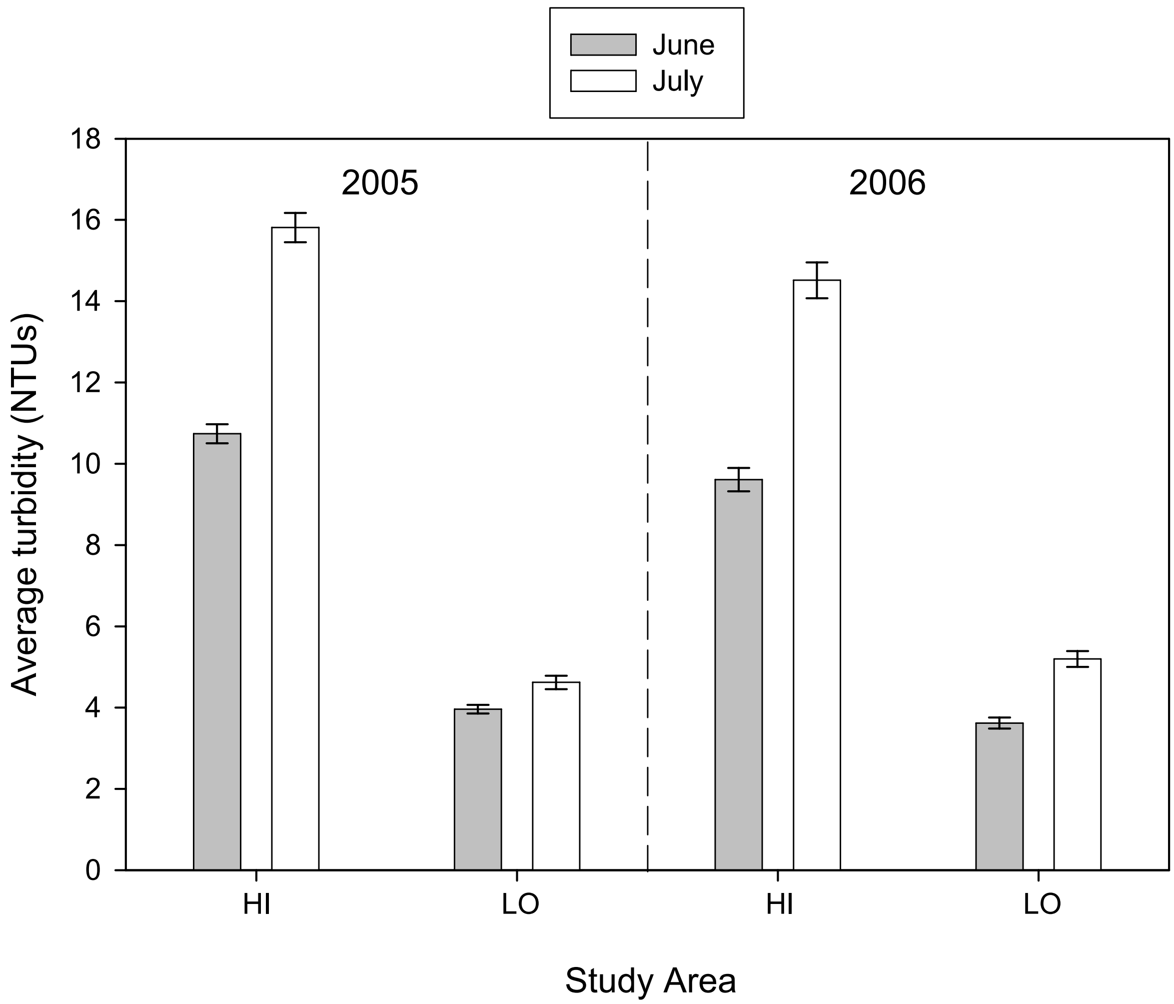

Figure 6. Average turbidity readings (NTUs) for study sites in the HI landscape (“Grant Co”) and LO landscape (“Polk Co”) in June (black bars) and July (grey bars) 2005 and 2006. Error bars are +/- 1 SE.

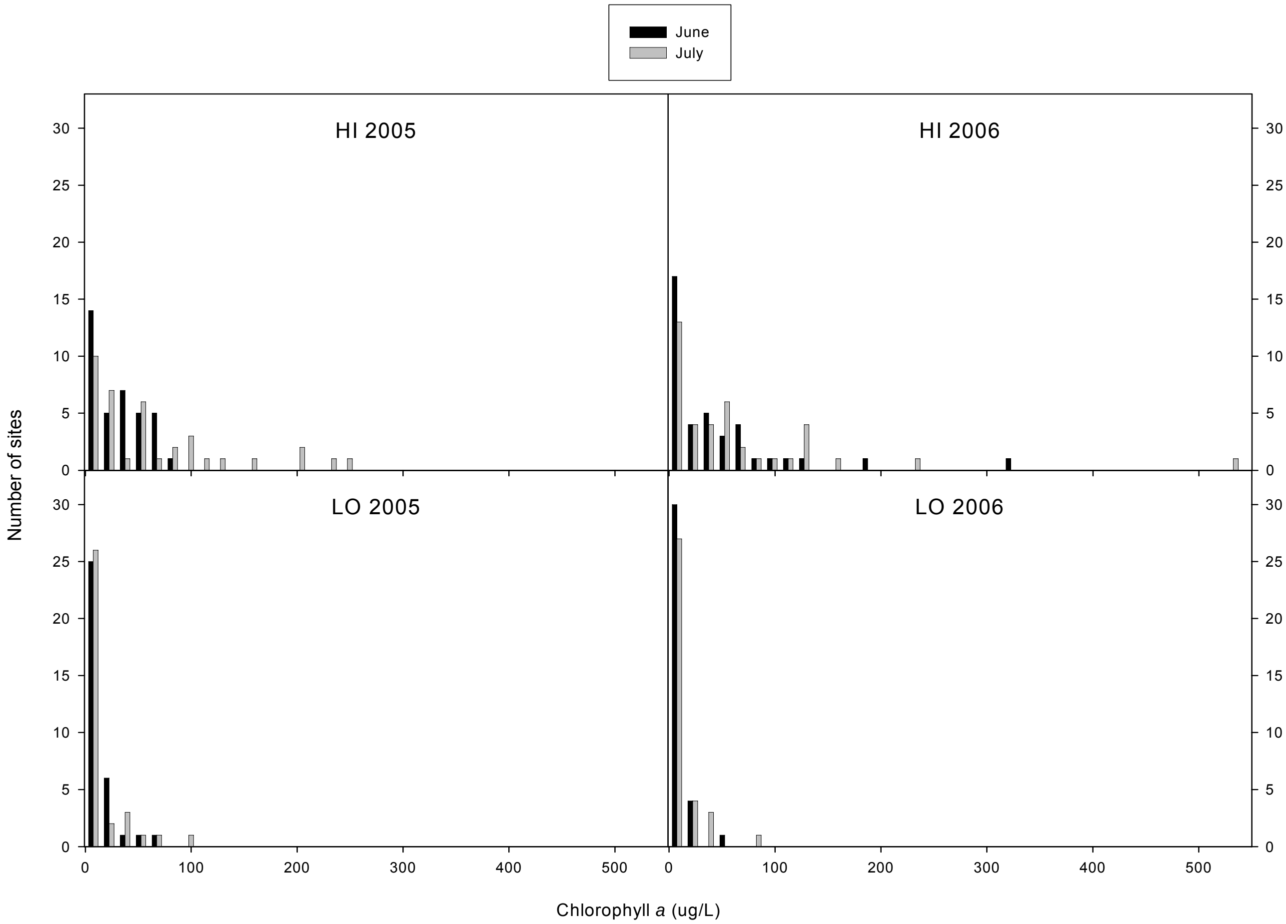

Figure 7. Histogram showing the distribution of average chlorophyll *a* concentrations (μg/L) observed in study sites located in the HI landscape (“Grant Co” – top panels) and LO landscape (“Polk Co” – bottom panels) in June (black bars) and July (gray bars) 2005 and 2006.

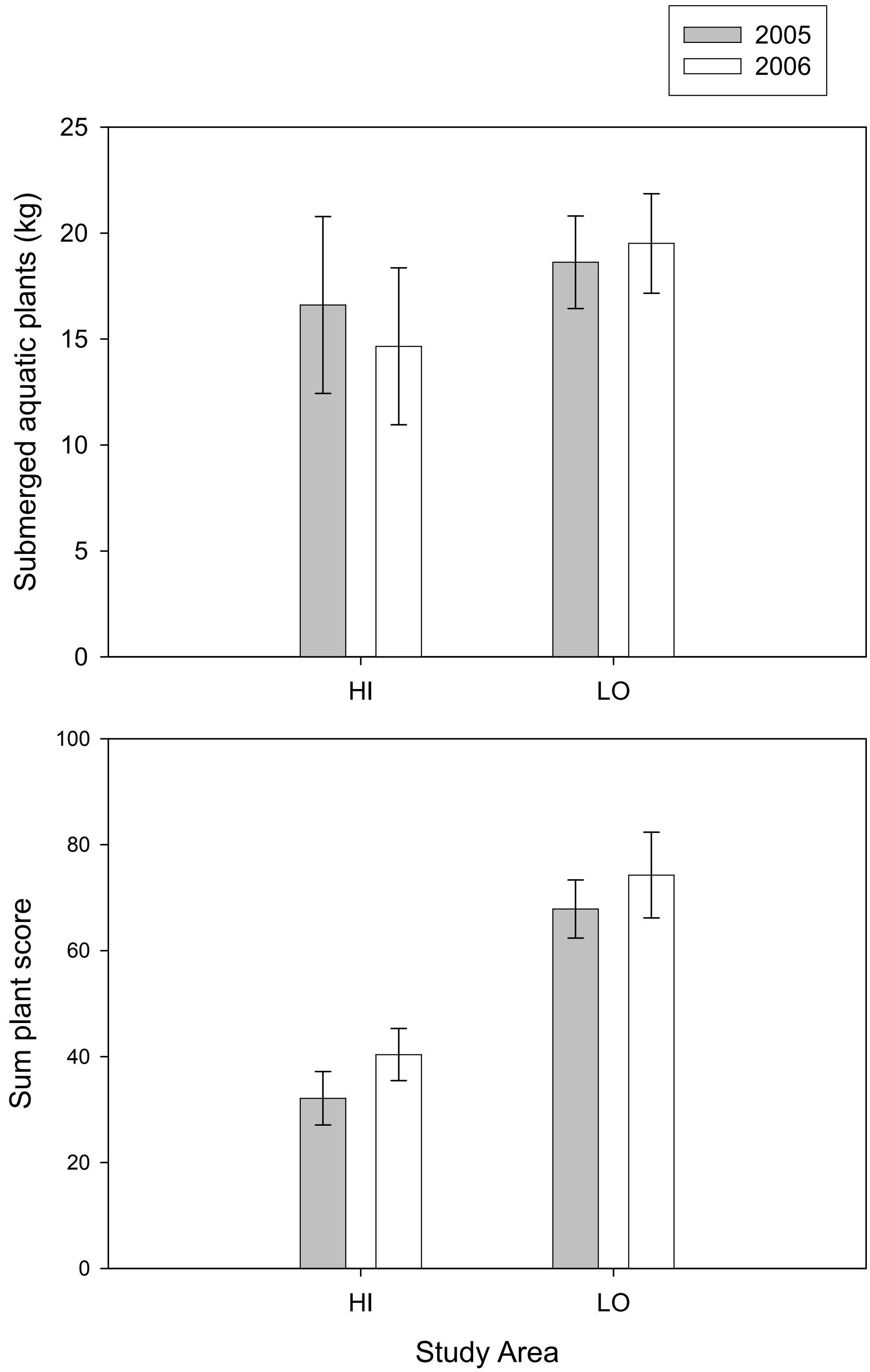

Figure 8. Average sum of submerged plant biomass (top panel) and average submerged plant score (bottom panel) for study sites in the HI landscape (“Grant Co”) and LO landscape (“Polk Co”) in 2005 and 2006. Error bars are +/- 1 SE.

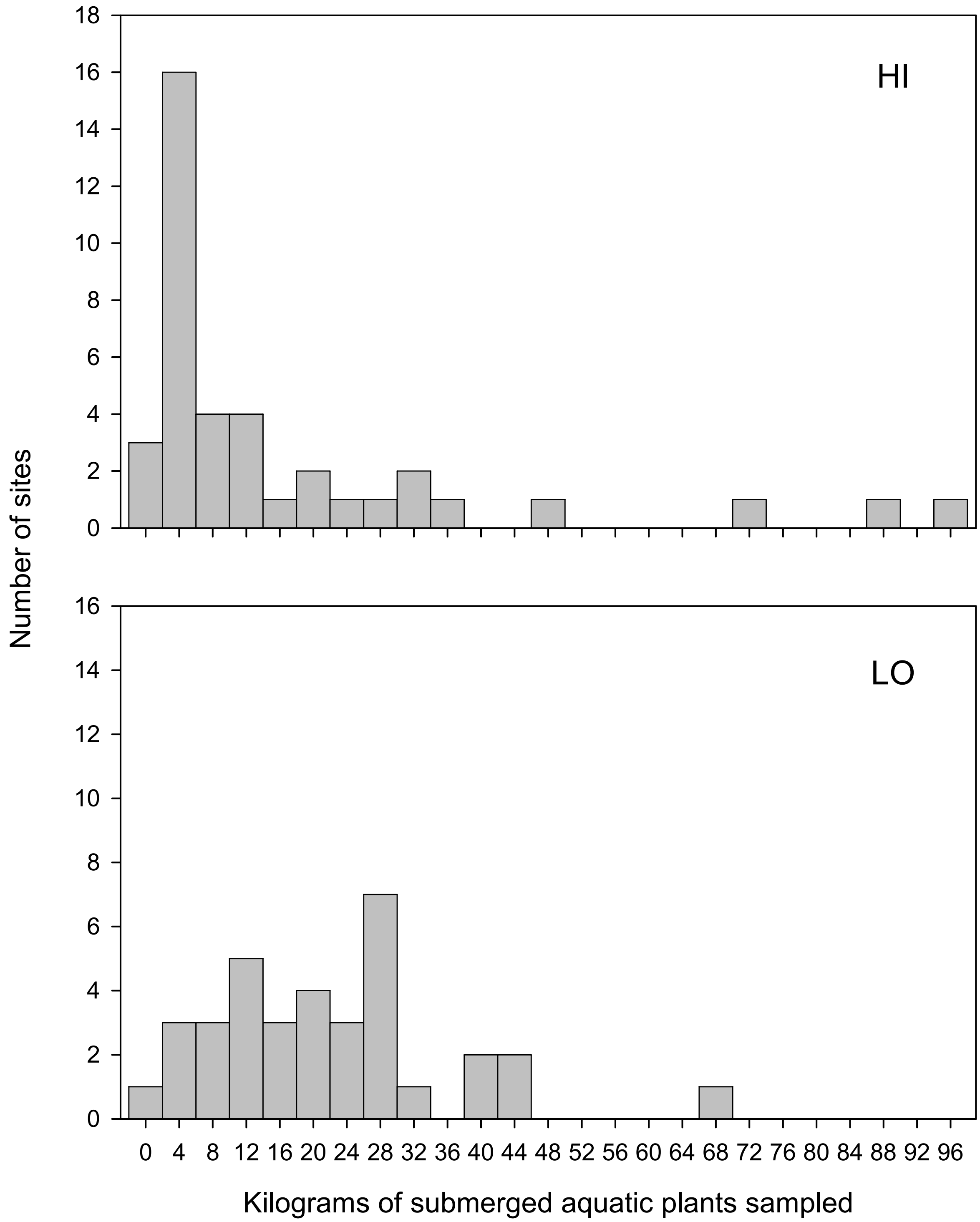

Figure 9. Histograms showing distribution of submerged plant biomass (summed across stations) within study sites in the HI landscape focus area ("Grant Co" - top panel) and LO landscape focus area ("Polk Co" – bottom panel) in summer 2006.

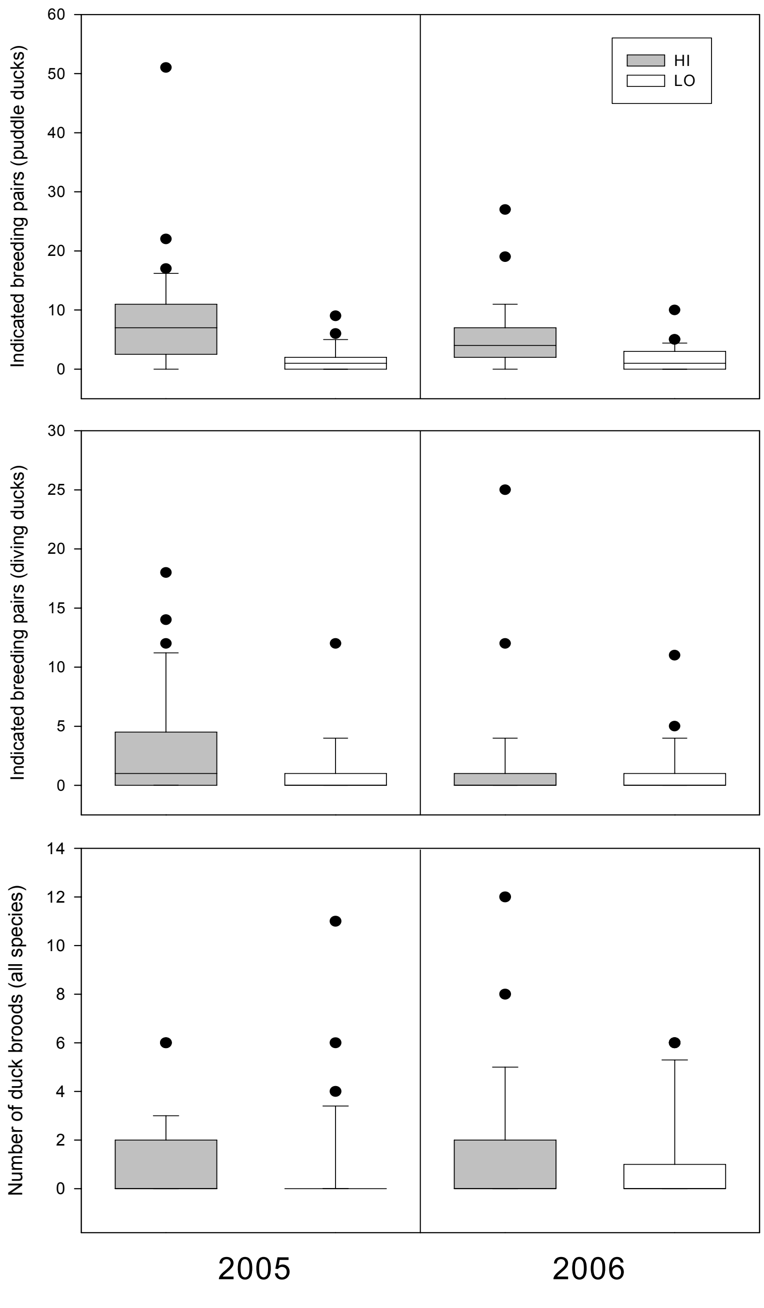

Figure 10. Number of indicated breeding duck pairs and duck broods observed on wetland study sites during 2005 and 2006. Box plots depict median values (central horizontal line), along with the 10th, 25th, 75th, and 90th percentiles and outliers beyond 10 and 90 percentiles (indicated by whiskers). Left-hand bars indicate ducks observed in HI study landscape; right-hand bars indicate ducks observed in LO study landscape.

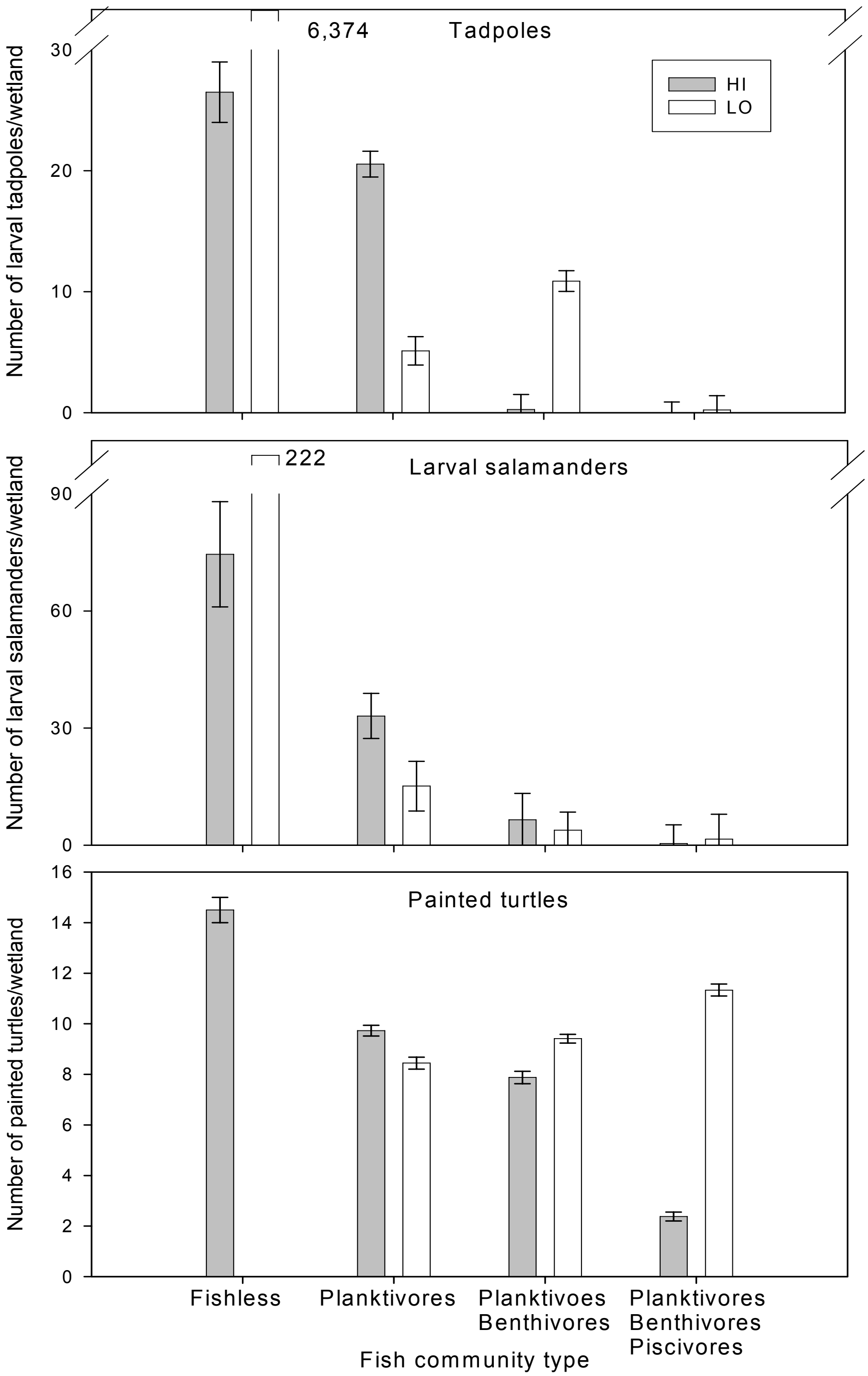

Figure 11. Average number of tadpoles (top panel), salamanders (middle panel), and painted turtles (bottom panel) per wetland in 2005, summarized by fish community types for the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co." – white bars). Error bars represent +/- 1 SE.

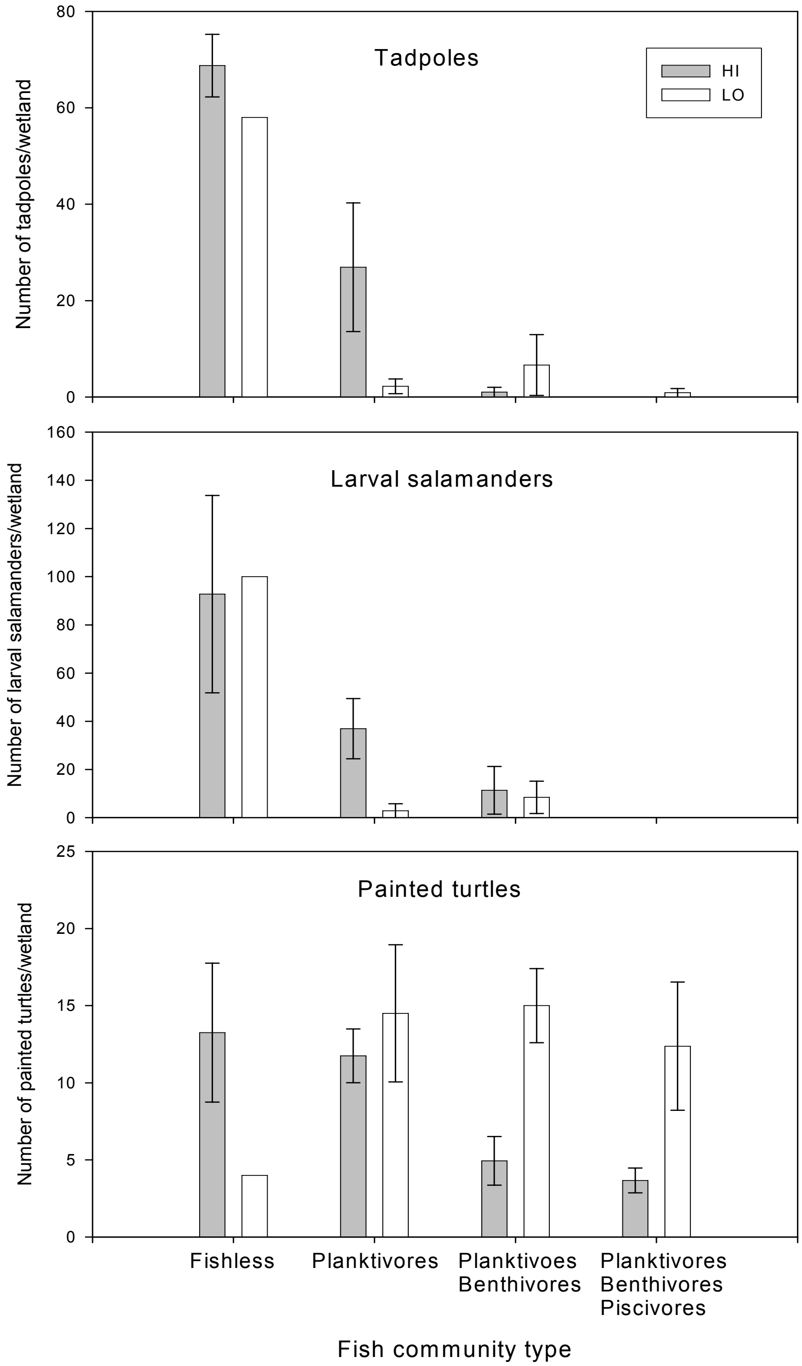

Figure 12. Average number of tadpoles (top panel), salamanders (middle panel), and painted turtles (bottom panel) per wetland in 2006, summarized by fish community types for the HI landscape ("Grant Co" – black bars) and LO landscape ("Polk Co." – grey bars). Error bars represent +/- 1 SE.

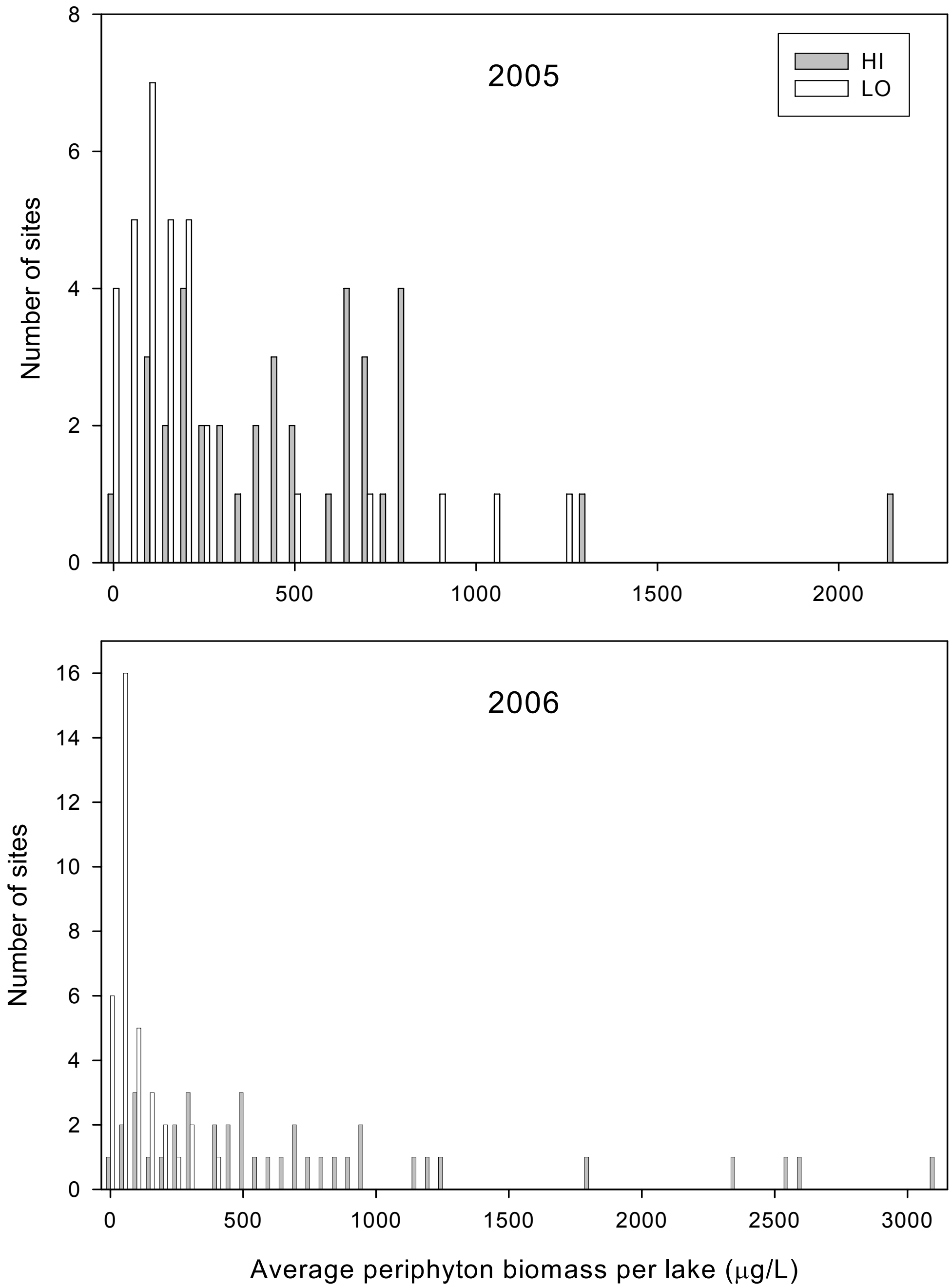

Figure 13. Histograms showing the distribution of chlorophyll *a* concentrations (ppb) of periphyton samples (average per lake) collected from study sites located in the HI landscape ("Grant Co" – gray bars) and LO landscape ("Polk Co" – white bars) in 2005 (top) and 2006 (bottom). There was a value of 5727 ug/L in the HI landscape in 2006 but it is not shown in the histogram.

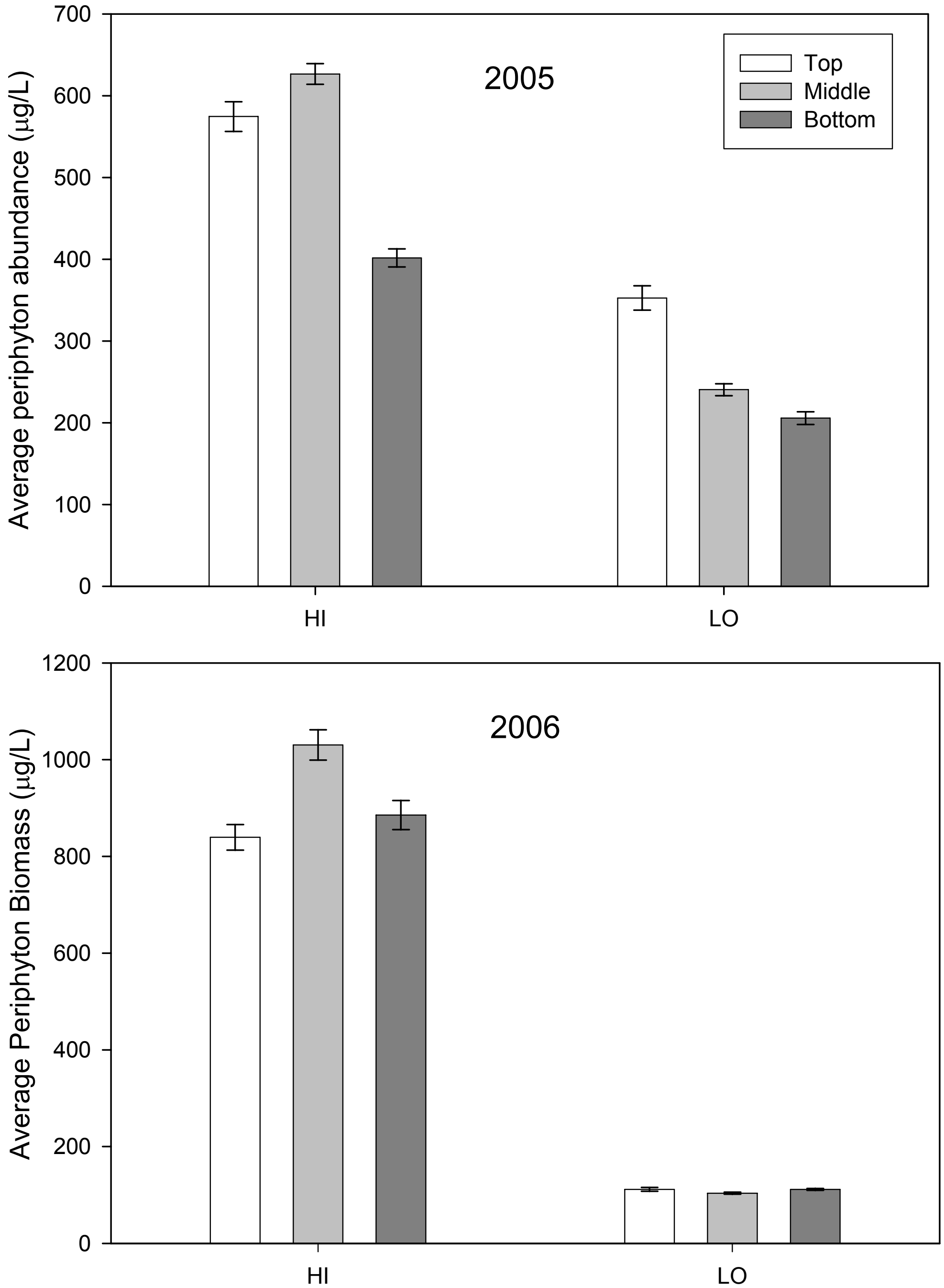

Figure 14. Average chlorophyll *a* concentration (ppb) of periphyton samples on the top slide (black), middle slide (light grey), and bottom slide (dark grey) from study sites located in the HI landscape (Grant Co.) and LO landscape (Polk Co.) in 2005 (top) and 2006 (bottom). Error bars represent +/- 1 sample SE.