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BLACK TERN NEST HABITAT SELECTION AND FACTORS AFFECTING NEST SUCCESS IN NORTHWESTERN MINNESOTA¹

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Abstract: We documented nest habitat selection, nest success, and factors affecting nest success of Black Terns (*Chlidonias niger*) at Agassiz National Wildlife Refuge in northwestern Minnesota. Over three years, 289 Black Tern nests and 400 random sites were sampled on search areas totaling 1,331 ha. Four habitat characteristics were measured at each nest and random site: (1) mean water depth, (2) distance to open water, (3) dominant vegetation within a 2-m radius, and (4) amount of open water within a 2-m radius. Habitat variables were highly correlated with each other, making it difficult to estimate independent effects of each habitat variable on nest-site selection. However, conditional logistic regression models indicated that locations closer to open water and in deeper water were more likely to be associated with nest sites. Locations in bulrush and sedge/grass were also more likely than those in cattails

to be associated with nest sites, although 68% of nests were in cattail reflecting the greater availability of that habitat in the study area. Nest success ranged from 48-69% (Apparent) and 33-62% (Mayfield) among years. Except for five nests that were abandoned or had infertile eggs, nests that failed to hatch appeared to have been depredated. Nest success was higher for nests with larger clutch sizes, nests located farther away from other nests, and for nests initiated earlier in the nesting season. Nests with 3-egg clutches were 2.8 times as likely to hatch as 2-egg nests. The odds of a nest being successful increased by 25% for each 5 m increase in distance to the nearest nest and decreased 7% for each additional day that passed before the nest was initiated. Nest success was not related to nest cluster size and was negatively related to the strength of nest site-selection (estimated from logistic regression models).

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INFLUENCE OF LAND USE ON MALLARD NEST STRUCTURE OCCUPANCY¹

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Abstract: We investigated the relationship between land use and mallard (*Anas platyrhynchos*) occupancy of single- and double-cylinder nest structures on a 658 km² (254 mi²) western Minnesota study area from 1997-1999. We used hierarchical logistic regression to spatio-temporally model structure occupancy as a function of land use, number of nearby structures, number of mallard pairs with access to the structure, size of the open-water area including the structure, and structure type. We fit models to data from 4 different sized buffers around each structure to investigate scale influences. Goodness of fit, predictive ability, and amount of reduced spatio-temporal correlation were similar for each buffer-size model. We made inferences using the 1.6 km radius buffer model because it produced the lowest deviance. The amount and attractiveness of nesting cover (i.e., as indexed by VOMs) within a buffer interacted with nest initiation period ($P = 0.003$). VOMs and nest occupancy were positively associated early in the nesting season, but the pattern reversed

later in the nesting season. Structure occupancy and area of open water around a structure were related quadratically ($P = 0.004$), with odds of a structure in median sized open-water areas being occupied increasing until the open-water area was ~16 ha. Year and nesting season period interacted ($P = 0.002$), reflecting different nest initiation phenology. Number of pairs with access to a structure had no effect on nest initiations ($P = 0.7$), perhaps due to our inability to account for within-season changes in pair numbers. Number of nearby structures ($P = 0.8$) was unrelated to initiation probability, but structure density was low (0.05/km²). We suspect that mallard settling patterns and an unmeasured temporal relationship between VOMs and numbers of pairs with access to structures produced the VOM by period interaction. Structures deployed in larger open water areas where surrounding residual upland cover is abundant can improve mallard nest success early in the nesting season when duckling survival is the greatest and can reduce hen mortality associated with nest destruction and re-nesting.

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USING GIS TO PREDICT MALLARD NEST STRUCTURE OCCUPANCY

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SUMMARY OF FINDINGS

We used the relationships described in a study of mallard nest structures to build a Geographic Information System (GIS) based model that would predict the probability of structure use by mallards. We assessed the model performance using data from a long-term study and used the assessment to illustrate a useful approach to predictive model building and validation. The model employed an existing GIS developed to aid in waterfowl management in western Minnesota. We used 3 predictors: 1) nest structure type, 2) 4 measures of the size of open water area containing the structure, and 3) a measure that described the mean aggregate visual obstruction of all residual cover during the early part of the nesting season (15 March – 20 April) in a buffer with a 1.6 km radius around each structure. We built the predictive model using the approach outlined by Harrell (2001), which is an alternative to data-based model selection methods (e.g., stepwise variable selection). We used a bootstrap procedure to obtain an unbiased measure of future predictive performance of the models that we fit. Unfortunately, we failed to produce a GIS model with much predictive power. Constantly changing features in the landscape were likely responsible for the difficulty in predicting biological outcomes. The process we employed forced us to think about the problem rather than using a data-based selection algorithm to determine the most important variables in the model.

INTRODUCTION

Knowing which type of nest structure to use and where to deploy them in a landscape should be important to waterfowl managers. Zicus et al. (2006a) studied mallard (*Anas platyrhynchos*) nest

structure occupancy in an attempt to understand how landscape features affected structure use. They were interested in the effect of 5 covariates, and their final fitted model was complex, including 3 interactions and 1 main effect. More nests were initiated as the size of the open water area where structures were deployed increased. Simultaneously, cover influence interacted with period of the nesting season such that nesting probability was positively associated with cover height and density early in the season, and negatively associated with cover height and density late in the season.

Nest success in structures is generally good (Eskowich et al. 1998) with early nests having higher nest success (M. Zicus, Minnesota Department of Natural Resources, unpublished data). Consequently, hen mortality associated with renesting (Sargeant et al. 1984) would be reduced for hens nesting in structures early in the year. Further, brood and duckling survival from early-hatched nests is believed to be greater than that of later-hatched nests (e.g., Rotella and Ratti 1992, Dzus and Clark 1998, Krapu et al. 2000). These understandings led Zicus et al. (2006a) to recommend that nest structures be deployed in larger wetlands where early-season residual cover in the surrounding uplands was most abundant within 1 km of the structure. Geographic Information System (GIS) models might provide powerful tools to help waterfowl managers decide where nest structure should be placed in complex landscapes.

OBJECTIVES

- Build a GIS-based model that wildlife managers can use to help determine best placement of mallard nest structures;

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- assess the model performance using data from a long-term study; and
- as a secondary objective, illustrate a useful approach to predictive model building and validation.

METHODS

We used the relationships described in a study of mallard nest structures (Zicus et al. 2006a) to build a GIS-based model that would predict the probability of structure use by mallards. The response that we were interested in modeling was the mean number of mallard ducklings (DUCKS) produced in each structure (Zicus et al. 2006b). We used 3 predictors: 1) nest structure type (TYPE), 2) 4 measures of the size of open water area containing the structure (NWI, GAP, FSA03, FSA97), and 3) a measure that described the mean aggregate visual obstruction (MVOM) of all residual cover during the early part of the nesting season in a buffer with a 1.6 km radius around each structure.

DATA USED TO BUILD THE MODEL

We began with a GIS developed to aid in waterfowl management in western Minnesota (D. Hertel, unpublished data). Classified Landsat Thematic Mapper data from 2000 and 2001 was used to estimate the area of each habitat class within buffers (1.6 km radius) around each nest structure.

The following variables were included in the model:

DUCKS. – We determined the mean number of ducklings from 110 nest structures across the entire nesting season from 1996 – 2003 (M. Zicus, unpublished data).

TYPE. – We considered 2 types of cylindrical nest structures, those having either a single or a double cylinder (Zicus et al. 2006a).

Open water area measures. – Different measures of the size of the open water area containing the structure were determined to compare model performance with different data sources. These measures were from: 1) open water polygons in National Wetland Inventory data (i.e., NWI; D. Hertel, unpublished data), 2) areas classified as open water in MN-GAP land cover data (i.e., GAP; Minnesota Department of Natural Resources 2004, U. S. Geological Survey 1989), 3) open water areas digitized from 2003 Farm Services Agency (FSA) aerial photography (i.e., FSA03; M. Zicus, unpublished data), and 4) open water areas digitized 1997 FSA aerial photography (i.e., FSA97; Zicus et al 2006a). The distribution of the NWI water data was highly skewed. As a result, we expected a few data points with extreme values (e.g., >100 ha) to have substantial influence on the model fit. Therefore, we also considered $\log(\text{NWI} + 0.1)$ which had a more bell-shaped distribution. Both NWI and GAP data are readily available for large areas of western Minnesota, whereas FSA97 and FSA03 data were included here to determine the potential gain in predictive power that might be obtained if efforts were made to obtain more up-to-date measures of open water.

MVOM. – We created a variable for the mean aggregate visual obstruction measurement (MVOM) for 15 March – 20 April for each buffer around each structure (D. Hertel, unpublished data). First, each 28 m x 28 m GIS cell within a particular habitat class in the buffer was assigned a habitat-specific VOM (Table 1). Next, a weighted VOM was calculated for each cell in a particular habitat class by multiplying the area of that habitat class in the buffer by the habitat-specific VOM. A mean aggregate visual obstruction measurement (MVOM) was then calculated for all cells in the buffer by summing the weighted VOMs across all habitat classes in the buffer and dividing by the total area of the buffer.

MODELING

We built predictive models using the approach outlined by Harrell (2001). We first determined a reasonable degree of model complexity using guidelines based on our sample size. This approach can be summarized as “determine the number of degrees of freedom (df) that can be spent, and then spend them without any further model simplification.” Harrell suggested a minimum of 10 – 20 observations per parameter considered, including those that account for potential non-linear effects. Burnham and Anderson (1998) suggested a similar liberal rule of 10 observations per predictor. Consequently, we believed 5-10 parameters to be a maximum for the 110 structures that we observed.

We used Spearman's ρ^2 (i.e., between response and predictors) to help determine how to apportion the df among the available predictors (e.g., to account for potential non-linearities) (Harrell 2001). Spearman's ρ^2 is a generalization of the rank correlation between two variables that can account for nonmonotonic relationships (e.g., using quadratic ranks) (Harrell 2001:127). We included all variables for which we examined ρ^2 in the model (i.e., ρ^2 was used only to determine the degree of non-linearity in the model). These steps defined an *a priori* full model from which we made our inferences; thereby avoiding problems associated with model selection algorithms (e.g., over fit models that predict new data poorly and biased p-values and confidence intervals arising from models selected using data-based selection procedures).

We used a bootstrap procedure to obtain an unbiased measure of future predictive performance of the models that we fit (Harrell 2001). We fit the model to 1,000 bootstrapped data sets, and the fitted parameters were used to calculate predicted values for all observations in the original dataset (as well as the bootstrap data set). We then calculated two R^2 values for each bootstrap replication: 1) using the original data and predicted

values from the bootstrap model fit, and 2) using the bootstrap data and the predicted values from the bootstrap model fit. The difference between these two values is an estimate of “optimism” (i.e., resulting from fitting and “testing” the model on the same dataset). A final adjusted R^2 value was then determined by subtracting the mean “optimism” from the R^2 obtained from the original fit of the model to the full dataset. Bootstrap calculations were carried out using functions in the Design library of the R computing package (Harrell 2001, R Core Development Team 2005). We also calculated the usual adjusted R^2 .

RESULTS

MODEL COMPLEXITY

Values of Spearman's ρ^2 indicated that both TYPE and MVOMs had less potential for explaining variation in DUCKS than open water area (Figure 1). Consequently, we assumed the MVOM effect was linear (i.e., a single df was used to model the relationship between MVOMs and DUCKS). The relatively greater values of Spearman's ρ^2 for open water area and previous work (Zicus et al. 2006a) suggested that more dfs should be spent to model the effect of open water area. Values of Spearman's ρ^2 were considerably higher for the digitized water measures (FSA03 and FSA97) than either NWI or GAP measures of open water.

Two models were fit using digitized water data (FSA03 and FSA97):

DUCKS = TYPE + MVOM + water
(using a linear spline with 2 df), and (1)

DUCKS = TYPE + MVOM + water
(using a restricted cubic spline with 2 df)
(2).

Model (1) used a single knot (i.e., the location where the slope was assumed to change), while model (2) used 3 knots (2 of these were located at the boundary of the data; the fit of a restricted cubic spline is constrained to be linear outside the range of the boundary knots). The medians of non-zero

observations (3.66 and 3.14 for FSA03 and the FSA97 data, respectively) were chosen as the knot location for the linear spline. Knots for the cubic spline used the 10th, 50th, and 90th percentiles of the data.

The GAP data only had 6 observations that were >0 and were not considered further. Given the low values of Spearman's ρ^2 for the NWI water data, we considered a model that assumed the effect of open water area was linear. In addition, we examined a model with a 2 dfs restricted cubic spline with knot locations again determined using the 10th, 50th, and 90th percentiles of the data.

ESTIMATES OF PREDICTIVE POWER

Models that used FSA03 and FSA97 water data performed considerably better than models using the NWI or GAP water data (Table 2). However, none of the models performed particularly well. The model using the FSA97 data had an R^2 of 0.14, suggesting that the open water area measured in Zicus et al. (2006a) along with structure type and MVOM values explained 14% of the variation in mean duckling production per structure. However, bootstrap validation suggested this model would perform considerably worse when applied to new data (i.e., it would explain only 6% of the variation). By comparison, R^2 measures for models using the NWI data were all less than 5% and their adjusted measures were negative, suggesting that the grand mean might predict new data better than the fitted model.

TYPE and MVOM values had p -values considerably >0.05 in all of the models, suggesting that they were not associated DUCKS (see also exploratory plots with smoothing lines; Figure 2). These results suggest that the MVOM values are not likely to be useful for predicting the mean duckling production (across all periods and years) in nesting structures, and that the available measures of open water area (NWI and GAP) are of questionable value for modeling duckling production.

DISCUSSION

Models having strong predictive ability are often difficult to construct (Steyerberg et al. 2001, Ambler et al. 2002, Steyerberg et al. 2003). There are a number of reasons why our efforts may have failed to produce a GIS model with much predictive power. First, mean visual obstruction measurements (MVOM) within 1 km of each structure may not accurately reflect the importance of surrounding cover. In particular, the height and density of cover in individual buffers having the same land use could actually differ markedly. Second, while Zicus et al. (2006a) recommended making structure placement decisions using early spring landscape conditions (as described by aggregate MVOMs in the buffer), their recommendations were intended to encourage production of young early in the season and not necessarily the maximum production of young across the entire nesting season. Zicus et al. (2006a) found that occupancy rates increased with VOM measurements early in the nesting season and decreased with VOMs later in the nesting season. Given the time-varying effect of VOM on occupancy rates, it was not surprising to discover that MVOM was unrelated to season-long duckling production. Lastly, although cover and water body size both vary temporally, we were forced to use measurements of these variables from a single year. The relationship between these habitat measurements and the average productivity of structures (across the 8 years of the study) may be much weaker than the relationship between habitat covariates and productivity in any given year.

The question as to how much predictive power a model would need to have in order to be useful is difficult to answer. Regardless, the models using either NWI or GAP measures of open water had essentially no predictive power, and a better measure of open water would be needed to produce a model with even

low predictive ability. FSA97 open water values produced the model with the most predictive ability, but even this was low, perhaps because water conditions had changed significantly between 1997 and 2003. Identifying specific locations for management actions such as nest structures will be difficult when the desired biological outcomes are determined by features in the landscape that are constantly changing. A sensible strategy for structure placement and management would be to place structures in larger wetlands (>4.0 ha) where early-season residual cover in the surrounding uplands is most abundant (Zicus 2006a; Minnesota Department of Natural Resources. 2006. Using cylindrical nest structures to increase mallard nest success. Unpublished pamphlet.). This should reduce the number of structures that never get used as 19 of 20 structures that were not used during the 8-year study were deployed in open water areas <0.8 ha in size (M. Zicus, unpublished data). In addition, we recommend that managers continue to collect data on structure use as well as habitat measurements surrounding the structure (e.g., cover types, wetland size) so that we might refine our models in the future.

Despite the poor predictability of the models considered, we believe the general modeling approach is a useful alternative to data-based model selection methods (e.g., stepwise variable selection). Harrell (2001:56-57) provides 7 disadvantages of stepwise selection methods (repeated verbatim below):

1. It yields R^2 values that are biased high.
2. The ordinary F and χ^2 test statistics do not have the claimed distribution. Variable selection is based on methods (e.g., F tests for nested models) that were intended to be used to test only prespecified hypotheses.
3. The method yields standard errors of regression coefficient estimates that are biased low and confidence

intervals for effects and predicted values that are falsely narrow.

4. It yields P-values that are too small (i.e., there are several multiple comparison problems) and that do not have the proper meaning, and the proper correction for them is a very difficult problem.
5. It provides regression coefficients that are biased high in absolute value and need shrinkage. Even if only a single predictor were being analyzed and one only reported the regression coefficient for that predictor if its association with Y were "statistically significant," the estimate of the regression coefficient $\hat{\beta}$ is biased (too large in absolute value). To put this in symbols for the case where we obtain a positive association ($\hat{\beta} > 0$), $E(\hat{\beta} | P < 0.05, \hat{\beta} > 0) > \beta$.
6. Rather than solving problems caused by collinearity, variable selection is made arbitrary by collinearity.
7. It allows us to not think about the problem.

Wildlife biologists have become familiar with problems associated with stepwise selection methods due to the popular book by Burnham and Anderson (2002) on model averaging and multi-model inference. As a result, model averaging and multi-model inference using AIC weights (Burnham and Anderson 2002) have become exceedingly prevalent in the wildlife literature. Unfortunately, few alternatives to AIC model averaging have been presented in applied ecology/wildlife journals (Guthery et al. 2005), and therefore model averaging is applied routinely without critical thinking. We would argue that approaches that utilize a full model with candidate predictors chosen based on subject matter considerations will often provide a viable alternative to model averaging/multi-model inference. The former approach offers several advantages over the AIC-

based model-averaging paradigm. For example, more time can be spent on diagnostics and model validation since a single model is considered rather than a suite of candidate models. In addition, if interest lies in estimation (rather than prediction), calculation of valid confidence intervals is straightforward (estimates of regression coefficients and σ^2 are not biased from considering multiple models or model reduction) (Harrell 2001, Ambler 2002).

The benefits of using a full model for inference are likely to be greatest when the effective sample size is $>10 - 20$ times the number of candidate predictors (Harrell 2001, Ambler 2002). For problems where the ratio of effective sample size to number of predictors is smaller, we recommend first trying to eliminate variables that do not have strong biological support (e.g., based on prior studies). This process is advantageous because it forces the researcher to think about the problem rather than using a data-based selection algorithm to determine the most important variables. In addition, it is generally beneficial to eliminate redundant variables, variables with lots of missing values, and variables that have very narrow distributions (Harrell 2001). If the number of remaining predictors is still $>10 - 20$ times the effective sample size, model averaging or other methods of shrinkage (e.g., penalized estimation or lasso) may offer improved predictions (Harrell 2001, Ambler 2002).

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Table 1. Land use cover types and source of visual obstruction measurements (VOM) used to estimate mean visual obstruction measurements (MVOMs) in the GIS model.

GIS model		Source data				
Cover type	VOM (dm) ^a	Cover type	VOM (dm)	Reference		
Grassland	1.16	CRP grass	1.30	Zicus ^d		
		WMA grass	1.02	Zicus		
		WPA grass	0.86	Zicus		
		Other grass	0.86	Zicus		
		Cropland ^b	0.001	Mack 1991		
Cropland	0.001					
Hayland	0.80	Hayland	0.80	Mack 1991		
Right-of-way	0.75	Gravel township road	0.71	Zicus		
		Gravel county road	0.40	Zicus		
		Gravel CSAH ^c	0.40	Zicus		
		Paved CSAH	0.65	Zicus		
		State highway	0.41	Zicus		
		Railroad	1.60	Zicus		
		Woodland	1.70	Woodland	1.70	Mack 1991
		Odd areas	1.70	Odd areas	1.70	Mack 1991
Vegetated wetlands	0.67	Seasonal	1.00	Mack 1991		
		Semi-permanent	2.00	Mack 1991		
		Temporary	0.50	Mack 1991		
		Permanent	1.00	Mack 1991		
		Open water/barren	0.00	Open water/barren	0.00	Mack 1991

^aVisual obstruction measurement corresponding to residual conditions in early spring (15 March – 20 April). Values are weighted by the area of the various source types occurring in western Minnesota.

^bMack (1991) presents values for many types of cropland. The value for fall-plowed cropland was used.

^cCASH = county state aid highway.

^dVOM is the mean value for 1997-1999 based on unpublished data collected as part of Zicus et al. (2006a).

Table 2. Measures of future predictive accuracy of GIS models predicting average duckling production from 110 nest structures in Grant County Minnesota, 1997 – 2003.

Model ^a	R ²		
	Original	Adjusted (from linear regression)	Adjusted (bootstrap)
FSA03, lsp	0.087	0.052	0.009
FSA03, rcs	0.084	0.050	0.009
FSA97, lsp	0.138	0.105	0.061
FSA97, rcs	0.134	0.102	0.056
NWI, linear	0.024	-0.013	-0.042
NWI, rcs	0.042	0.006	-0.045
Log(NWI), linear	0.027	0.000	-0.036
Log(NWI), rcs	0.053	0.017	-0.031

^alsp = linear spline model with 1 knot (2 dfs); rcs = restricted cubic spline model with 2 knots (3 dfs).

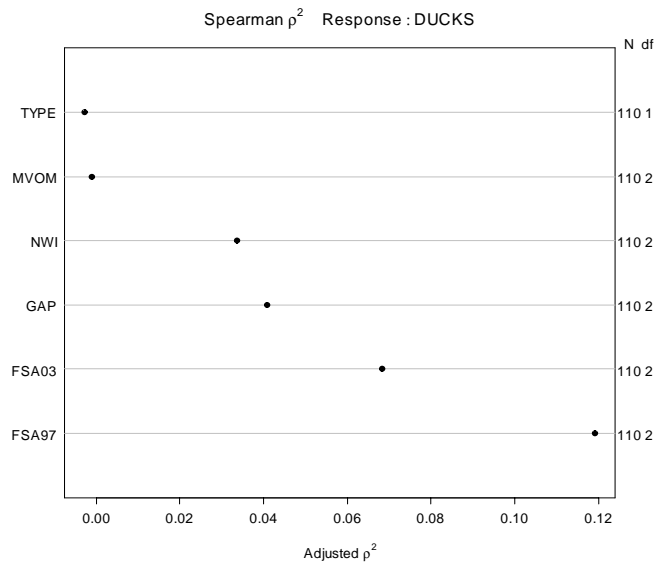


Figure 1. Spearman's ρ^2 indicating the strength of the relationship between mean duckling production (DUCKS) and each predictor variable (TYPE = indicator variable for structure type, MVOM measures, NWI open water measure, GAP open water measure, FSA03 open water measure, FSA97 open water measure).

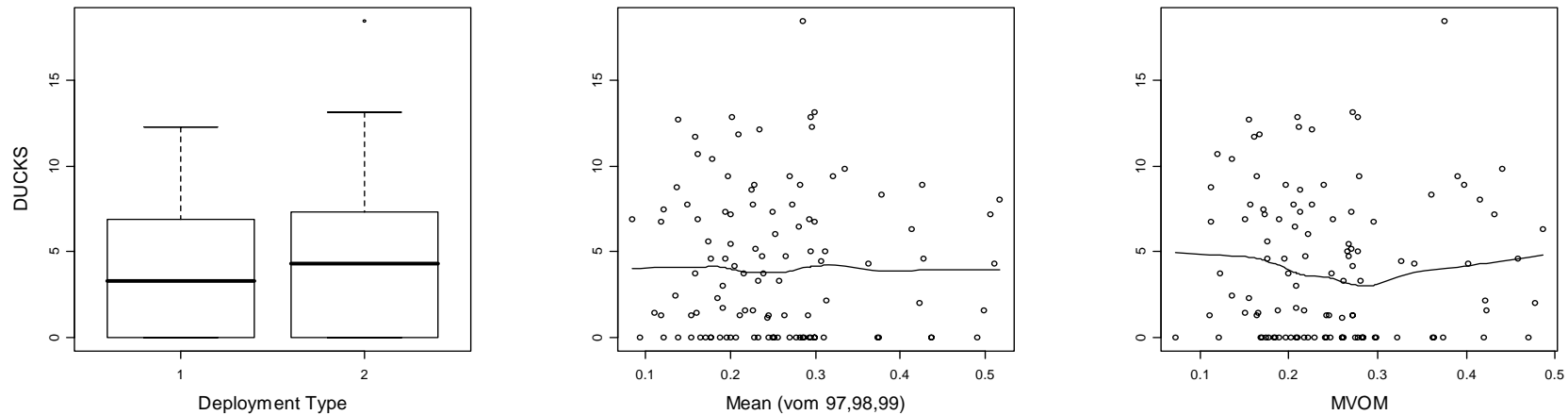


Figure 2. Exploratory plots of mean duckling production/year for each structure versus structure type, mean VOM measures across 1997-1999 (M. Zicus, unpublished data), and MVOM (D. Hertel, unpublished data). Lines represent smooth curves estimated using locally weighted regression via the `lowess` function in R.

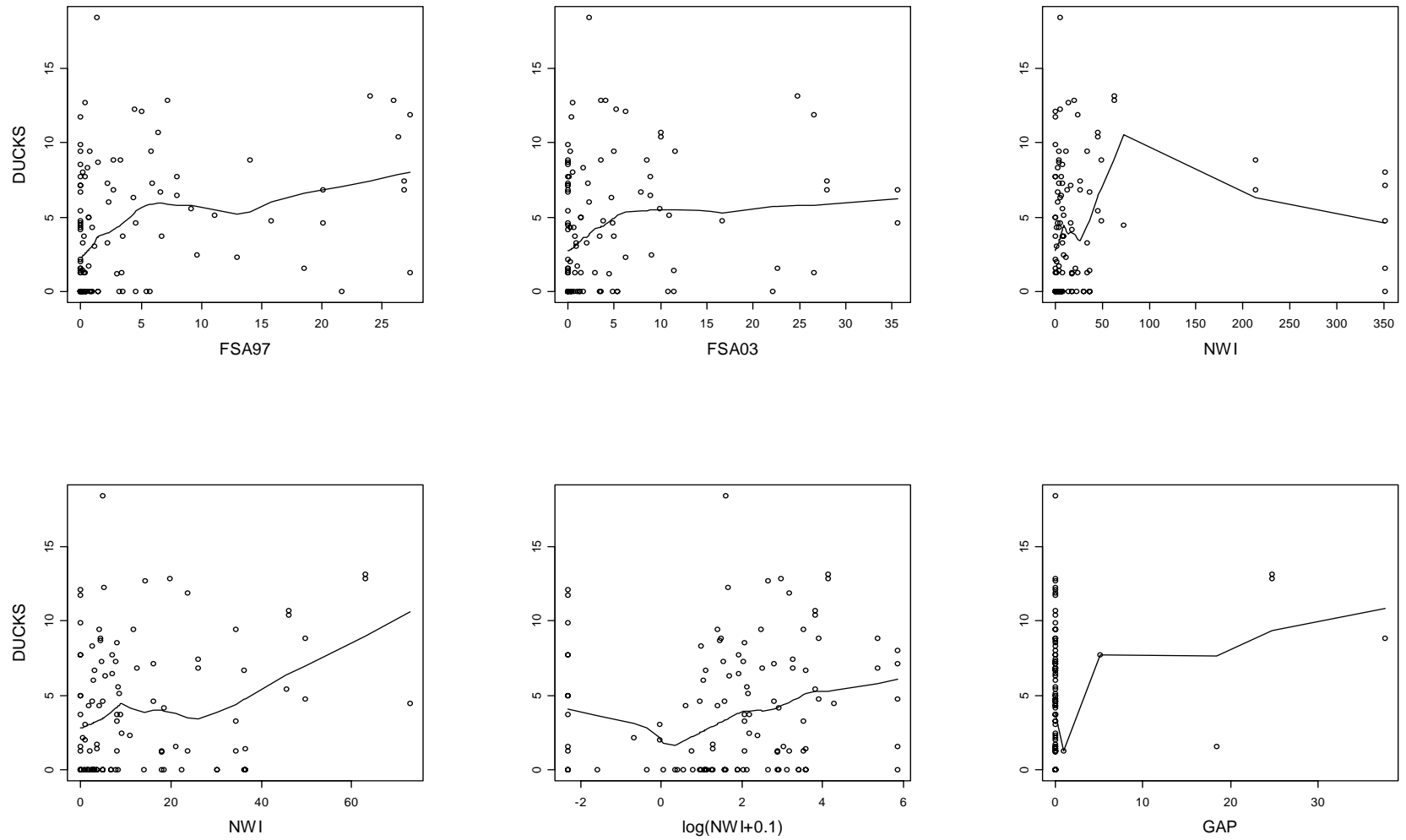


Figure 3. Exploratory plots of mean duckling production/year/structure versus FSA97 open water, FSA03 open water, NWI open water (all values), NWI open water (only values < 100), $\log(\text{NWI} + 0.1)$, and GAP open water. Lines represent smooth curves estimated using locally weighted regression via the `lowess` function in R.

EFFECTS OF SUBCUTANEOUS TRANSMITTER IMPLANTS ON MOURNING DOVES¹

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Abstract: An important assumption of telemetry studies is that radiomarking does not negatively affect study animals. To test this assumption for mourning doves (*Zenaida macroura*), we evaluated whether subcutaneous transmitter implants (STI) would affect bird weight in cage studies and hunting mortality in field studies. At three weeks post-implantation, caged adult birds in the sham surgery and control groups gained and STI birds lost weight. Males gained and females lost weight. When percent weight change (PWC) for caged adult and juveniles was pooled the trends were similar, suggesting a STI treatment effect. In the field study, 16.3% of observed mortalities of STI birds during July–November 1998–2000 occurred during the first 3 days post-

release. The overall 45-day summer period survival rate was relatively high, 0.9446 (95% CI = 0.8907–0.9986), when birds were entered into the population at-risk on the fourth day post-release. Although most observed mortalities were hunting-related (62.7%), similar direct recovery rates ($P = 0.186$) for STI (14.7%) and leg-banded birds (9.2%) suggests that implanted radios did not increase a bird's vulnerability to hunting mortality in the year of marking. However, the difference between the direct recovery rates of the 2 cohorts may be large enough to be biologically significant. Further research is needed to determine whether STI birds are especially susceptible to hunting mortality.

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COST SAVINGS FROM USING GIS-BASED “REAL TIME” IN A RING-NECKED DUCK SURVEY

John R. Fieberg, Robert G. Wright, and Michael C. Zicus

SUMMARY OF FINDINGS

Staff in the Minnesota Department of Natural Resources Wetland Wildlife Populations and Research Group recently began surveying Public Land Survey (PLS) sections with helicopters to estimate numbers of breeding ring-necked ducks (*Aythya collaris*). Data were recorded on paper tally sheets in 2004, while the 2005 survey utilized customized GPS and GIS software to record data directly to a tablet style computer. These customizations allowed the observers to display the aircraft's flight path over aerial photography or maps, and record both the flight path and animal observations directly to ArcView GIS shapefiles in real time. This provided an efficient means of data capture and may reduce the amount of flight time required to conduct the survey. We estimated that the approach saved ~1.75 minutes of flight time per sample plot using statistical methods developed for estimating causal effects in observational studies. As a result, survey cost was reduced by ~\$2,100 when aircraft and staff expenses were considered.

INTRODUCTION

Geographic Information System (GIS) staff at the Minnesota Department of Natural Resources (MDNR) recently developed an ArcView GIS 3.x (Environmental Systems Research Institute, Redlands, California USA) extension called DNR Survey (MDNR 2005a), which provides menu-driven data entry forms for recording animals observed during aerial surveys. DNR Survey was designed to be used with DNR Garmin software (MDNR 2005a,b), a Global Positioning System (GPS) receiver, and a tablet style computer. This configuration allows the observer to view

the aircraft's flight path over aerial photography, maps and survey boundaries, and record both the flight path and animal observations directly to ArcView GIS shapefiles, all in real time. This “real time” survey technique provides for efficient data capture and greatly enhances navigation between and within sample units. These efficiencies can reduce both the aircraft and staff costs associated with conducting aerial surveys. A ring-necked duck (*Aythya collaris*) breeding survey conducted in 2004 and 2005 is among several recent surveys conducted using the real time survey technique. In 2004, we surveyed 200 Public Land Survey (PLS) sections as survey plots, without the use of real time technology. The technology was employed in 2005 when 251 plots were surveyed.

Quantifying the amount of time saved by employing the real time survey technique can be difficult because other factors that may influence flight time are usually not held constant across years. For example, observers, number of ducks seen on a plot, and amount of potential nesting habitat in a plot differed between the 2 years in the ring-necked duck survey. These difficulties are common in observational studies, where covariates are not balanced between treatment and control groups. As a result, students in introductory statistics classes are often taught that observational studies can only provide evidence of correlation and not causation (Schield 1995).

We provide a brief introduction to statistical estimation of causal effects via counterfactuals. We then use data from the 2004 – 2005 ring-necked duck survey to illustrate the use of matching for estimation of a causal treatment effect in an observational study (i.e., where randomization to treatment group is not possible). Throughout, we will refer to

observations in 2005 as “treated” and observations in 2004 as “controls”, with the goal of estimating the causal effect (in terms of flight time savings) of using the real time survey technique.

OBJECTIVES

- To estimate the time and cost savings from employing the GIS-based real time survey technique in 2005, and
- to introduce a useful methodology for estimating causal effects in observational studies.

CAUSAL INFERENCE AND MATCHING COUNTERFACTUAL MODEL

Define two possible responses for each PLS survey plot:

$Y_i(t=1) = Y_i(1)$ = a continuous random variable representing the observation time for survey plot i flown under the “treatment condition” (i.e., using the real time technique in 2005)

$Y_i(t=0) = Y_i(0)$ = a continuous random variable representing the observation time for survey plot i flown under the “control” condition (i.e., no real time technique in 2004)

Similarly, we can define actual realizations of these random variables as $y_i(1)$ and $y_i(0)$ (typically, it is not possible to observe both random variables and therefore they are termed “counterfactuals” or potential outcomes). We then define a “realized causal effect” (Ho et al. 2005a) for sample plot i as: $y_i(1) - y_i(0)$. Realized causal effects are not observed and cannot be estimated because we record only $y_i(1)$ or $y_i(0)$, never both. Instead, we can attempt to estimate the expected causal effect for sample plot i , i.e., $E[Y_i(1) - Y_i(0)]$. Further, we can estimate the average treatment effect (ATE) overall and the average treatment effect for the treated (ATT) as:

$$ATE = \frac{1}{n} \sum_{i=1}^n E[Y_i(1) - Y_i(0)] \quad \text{and}$$

$$ATT = \frac{1}{n_T} \sum_{i=1}^{n_T} E[Y_i(1) - Y_i(0)],$$

where n_T = the number of treated observations and the second sum is only over treated subjects (Ho et al. 2005ab). The sampling frame for the ring-necked duck survey changed significantly between 2004 and 2005 (Zicus et al. 2005, 2006); therefore, these 2 effects may differ. The ATT provides a measure of the treatment effect that applies to a sampling frame similar to that used in 2005 (since it estimates causal effects only for treated observations), while the ATE provides an estimate of treatment effect that applies to the combined 2004 and 2005 sampling frames (since it estimates causal effects for control and treated observations).

$E[Y_i(0)]$ and $E[Y_i(1)]$ will usually depend on covariates (e.g., number of ducks observed on the survey plot, hectares of water/nesting cover in the survey plot) and are often estimated using regression models. In observational studies, the distribution of important covariates will often differ between treatment and control groups since the sampling units are not randomized to treatment group. This imbalance has important implications for model-based estimates of treatment effects as estimates will be biased if important confounders are not included in the model, or if the relationship between these confounders and the response is misspecified. For example, a linear relationship might be assumed when the true relationship is non-linear (Ho et al. 2005a). Similarly, model-based estimates of treatment effects may depend heavily on the assumed model (e.g., estimates may be influenced by inclusion/exclusion of various covariates, assumptions regarding the degree of non-linearities and extent of interactions, distributional assumptions, etc.).

Matching control and treated observations with respect to potential confounders can help minimize the bias (and improve the robustness) of model-based estimates of expected causal effects (Ho et al. 2005a). Matching can be done in a number of ways, including exact matching (i.e., matching based on exact values of covariates) and nearest neighbor methods. Matching serves as a “preprocessing” step that pairs treated and control observations with respect to important covariates, resulting in a data set that is more balanced between these two groups (Ho et al. 2005ab). This balance helps provide assurance that any observed differences between control and treatment groups is due to the treatment rather than inherent differences between the two groups.

METHODS

We limited our analysis to observations that were made by 2 observers (DR and JH). Observer effects were large (Figure 1a) and DR and JH were the only pair of observers that flew plots in both 2004 and 2005. Before formulating and fitting regression models relating plot survey time to covariates (hectares of nesting cover, hectares of water, total ring-necks observed), we used functions in the R package MatchIt (Ho et al. 2005b) to create a dataset where each control observation (2004) was matched to a single treated observation (2005) using nearest neighbor matching with the distance between observations measured using a propensity score (Ho et al. 2005a). The propensity score measures the probability of an observation belonging to the treatment group as a function of covariates (hectares of nesting cover, hectares of water, total ring-necks observed) and is typically estimated using logistic regression. Treatment observations that were outside the convex hull of the control data (and vice versa) were discarded (Ho et al. 2005b, Stoll et al. 2005, King and Zeng in press,), leaving 113 controls matched to 113 treated

observations. The convex hull is the smallest convex set containing all observations (in two dimensions, this is a polygon; e.g., “minimum convex polygon’s” are often considered in animal home range analyses). Observations outside the convex hull of the data are “far away” from the rest of the data and require extrapolation (rather than interpolation) to estimate their counterfactuals (King and Zeng in press).

The Models for Survey Time

We fit 4 models to the matched and original datasets. In each model, the response was the time required to fly each survey plot:

1. A least-squares regression model with linear effects of hectares of nesting cover, hectares of water, and total number of ring-neck ducks observed. In addition, an additive treatment effect was assumed.
2. A Poisson regression model that assumed the log (mean survey plot time) was linearly related to hectares of nesting cover, hectares of water, and total ring-necks observed. In addition, an additive treatment effect was assumed on the log scale.
3. A Poisson regression model that assumed the relationship between log(mean survey plot time) and hectares of nesting cover, hectares of water, total ring-necks observed were non-linear. We used orthogonal polynomials of degree 2 to account for the non-linearities. In addition, an additive treatment effect was assumed on the log scale.
4. A Poisson regression model that assumed the log(mean survey plot time) was linearly related to hectares of nesting cover and hectares of water. We used 2 degrees of freedom to model the effect of the number of observed

ducks. First, we included an indicator variable to reflect differences between plots that contained ducks and those plots that did not contain ducks. In addition, we included the number to reflect the assumption that the mean survey time increased of ducks observed as a covariate linearly (on the log scale) for each additional duck observed. Finally, an additive treatment effect was assumed on the log scale.

Letting X represent all covariates of interest, the expected survey time in each of the models is given by:

Model 1:

$$E[Y_i(1) | X] = \beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + ducks_i \beta_3 + \gamma$$

$$E[Y_i(0) | X] = \beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + ducks_i \beta_3$$

Model 2:

$$E[Y_i(1) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + ducks_i \beta_3 + \gamma)$$

$$E[Y_i(0) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + ducks_i \beta_3)$$

Model 3:

$$E[Y_i(1) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + nest_acres_i^2 \beta_2 + water_acres_i \beta_3 + water_acres_i^2 \beta_4 + ducks_i \beta_5 + ducks_i^2 \beta_6 + \gamma)$$

$$E[Y_i(0) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + nest_acres_i^2 \beta_2 + water_acres_i \beta_3 + water_acres_i^2 \beta_4 + ducks_i \beta_5 + ducks_i^2 \beta_6)$$

Model 4:

$$E[Y_i(1) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + I(ducks_i > 0) \beta_3 + ducks_i \beta_4 + \gamma)$$

$$E[Y_i(0) | X] = \exp(\beta_0 + nest_acres_i \beta_1 + water_acres_i \beta_2 + I(ducks_i > 0) \beta_3 + ducks_i \beta_4)$$

For model 1, the effect of treatment is estimated directly by γ (assuming the model is correct) since $\gamma = E[Y_i(1) - Y_i(0)]$ regardless of the value of X . For models 2-4, the effect of treatment is assumed to be multiplicative and therefore $E[Y_i(1) - Y_i(0)]$ will depend on X . In such cases, one may choose to estimate the causal effect of treatment for an observation with all covariates set to the mean values in the data, $E[Y_i(1) | X = \bar{x}] - E[Y_i(0) | X = \bar{x}]$.

However, this causal effect may not be very meaningful [e.g., this “subject” may be very different from any of the subjects in the study, particularly for model 4 where one of the covariates is an indicator variable that is always either 0 or 1]. Thus, for models 2 – 4 we report an estimate of the multiplicative effect of treatment on survey time [i.e., $\exp(\gamma)$].

For models 1, 2, and 4, we also estimated the ATE and ATT in the matched and full datasets (we did not estimate the ATE or ATT for model 3 because of minor complexities with applying the approach when using orthogonal polynomials and because models 2, 3, and 4 all gave similar estimates of γ). We followed the steps outlined in Ho et al. (2005b):

1. We fit the model (1, 2 or 4) first to the control observations (without γ in the model). We used the fitted model to estimate $E[Y_i(0)]$ for all of the treated observations in the dataset.

- We fit the model (1, 2 or 4) to the treated observations (again without γ in the model). We used the fitted model to estimate $E[Y_i(1)]$ for all of the control observations in the dataset.
- We estimated ATE using:

$$ATE = \frac{1}{n} \left(\sum_{i=1}^{n_t} \left\{ y_i(1) - E[Y_i(0)] \right\} + \sum_{i=1}^{n_c} \left\{ E[Y_i(1)] - y_i(0) \right\} \right)$$

where “ $\hat{\cdot}$ ” denotes estimated values determined using steps 1 and 2 and n_c = the number of “control” observations and n_t = the number of “treated” observations in the matched/full dataset.

- We estimated the ATT using:

$$ATT = \frac{1}{n_t} \sum_{i=1}^{n_t} \left\{ y_i(1) - E[Y_i(0)] \right\}.$$

Importantly, the estimation procedure fits separate models to the control and treatment observations (steps 1 and 2). These steps provide a means of essentially “imputing” values for $y_i(1)$ for control observations and $y_i(0)$ for treated observations. Using separate models in the two steps helps to reduce bias by eliminating the assumption of constant parameter values for treated and control observations (Ho et al. 2005b). Uncertainties in the estimates of ATE and ATT were determined by generating 1,000 random samples of all model parameters from their asymptotic sampling distributions (i.e., a multivariate normal distribution) using the R package, Zelig (Imai et al. 2005). For each set of sampled parameters, we estimated the ATE/ATT and then report the 0.025 and 0.975 percentiles across the set of 1,000 estimated ATEs/ATTs.

Lastly, we examined plots of flight time/plot versus date to determine if flight times decreased systematically as observers became more experienced with the survey. We also examined residuals plots to assess the fit of the regression models.

Cost Comparisons

Survey costs include airtime for the helicopter and the pilots, air and ground time for the observers, and lodging and meals for the pilots and observers. We determined the difference in the cost of the 2005 survey compared to the expected cost had we not used the real time approach. We determined the difference in airtime costs by multiplying the per-plot ATT by the number of plots surveyed in 2005 (251) and the helicopter/pilot rental rate (\$230/hr). The difference in ground time was calculated by assuming 40-minute refueling stops for every 2.67 hrs of flight time (D. Rave, unpublished data). We determined observer cost difference by multiplying the air and ground time by an average observer salary (plus fringe) of \$32/hour. Lodging and meal costs for the survey crew was assumed to be \$150/day.

RESULTS

Matching significantly improved the balance between treated and control units with respect to important covariates (Figures 2, and 3). Model-based estimates of γ were quite consistent across models (2 – 4) using either the full or matched data sets (Table 1). Estimates of γ were all statistically significant (all $p < 0.05$). The linear model estimated that the 2005 survey technique would save on average >2 minutes/plot, while the Poisson regression models estimated approximately a 30% time reduction per plot (Table 1).

Conclusions regarding the importance of hectares of water, hectares of nesting cover, and number of observed ducks were also similar across the fitted models. Survey time was estimated to increase with hectares of water and number of ducks observed ($p < 0.05$), but hectares of nesting cover was not significantly related to survey time ($p > 0.05$) (Figure 1).

Average Treatment Effect (Overall and for the Treated)

Estimates of the ATE and ATT from the matched dataset were more conservative and also had slightly wider confidence intervals (reflecting the smaller sample size) than estimates from the full dataset (Table 2). While estimates ATE were similar to estimates of the ATT for the full dataset, estimates of the ATT were consistently lower than the corresponding estimates of the ATE for the matched dataset (average time savings ~1.75 minutes/plot compared to ~2 minutes/plot) (Table 2). Survey time did not appear to systematically decrease in either year (Figure 4).

Cost Comparisons

Use of the real time survey approach resulted in an estimated savings of >\$2,100 over the expected cost of the 2005 survey if it had been conducted without using the real time approach (Table 3). Almost a full day was saved in airtime alone which resulted in further saving for on the ground refueling time and travel expenses for lodging and meals.

DISCUSSION

Estimating treatment effects from observational data can be problematic because treatment and control groups often differ with respect to important covariates that may also influence the response of interest. Regression models are frequently used to obtain adjusted estimates. Unfortunately, estimates of treatment effects will remain biased unless the relationships between confounders and the response are correctly specified in the model. In addition, model-based estimates of treatment effects may be highly sensitive to assumptions of the regression model (e.g., inclusion/exclusion of covariates, assumed non-linearities or interactions). Matching (treated and controlled units)

with respect to important covariates can reduce the sensitivity of estimated treatment effects to model assumptions and also reduce bias (Ho et al. 2005a,b).

Matching significantly improved the balance between treated and control units with respect to factors thought to influence flight time. Therefore, we expected estimates of causal effects to be more consistent across models. Somewhat surprisingly, we found that estimates of γ , ATE, and ATT, while slightly more consistent for the matched data, were quite robust using either the full and matched datasets.

The ATT provided an estimate of the treatment effect for plots surveyed in 2005, while the ATE provided an estimate of the treatment effect for the combined 2004 and 2005 plots. On average, plots sampled in 2004 contained more ducks, even in the matched dataset (Figure 3). Since flight time increased with numbers of ducks, it was not surprising that estimates of the ATT were lower than estimates of the ATE for the matched data. On the other hand, estimates of ATE and ATT were more similar for the full dataset because it consisted mainly of 2005 survey plots (the matched data included 113 plots from each of the 2004 and 2005 surveys, while the full dataset included 251 plots from 2005 and only 130 from 2004).

In calculating cost savings from use of the real time survey technique, we used estimates of the ATT from the matched dataset. As a result of matching and dropping observations outside of the convex hull of the data, this estimate did not consider all plots sampled in 2005. Therefore, our estimate of cost savings may be biased. Using the most conservative estimate of the ATT from the full dataset (~2 minutes/plot) resulted in an estimate of cost savings of ~\$2400. This estimate, while using all plots sampled in 2005, is likely to be more model-dependent as a result of imbalance with respect to important covariates between the 2004 and 2005 sample plots. Other methods exist for estimating causal

effects in observational studies (Lunceford and Davidian 2004), and these may be explored in the future.

We cannot rule out the possibility that systematic differences among plots flown in 2004 and 2005 were partially responsible for observed reduction in flight times. However, our estimate of time savings was robust to assumptions regarding the effects of numbers of ducks, hectares of nesting cover, and hectares of water. Further, we controlled for observer differences by only considering observations made by the same 2 observers in both years. Lastly, survey time did not appear to systematically decrease in either year, and observers believed that it took only 1 or 2 plots to “get up to speed on things” (D. Rave, personal communication) suggesting experience with the survey was not responsible for the reduced flight times. While exact cost savings are impossible to determine, we believe that actual flight time was reduced by ~7 – 8 hours through the use of the real time approach. This amounts to ~10% of the total survey flight time (Zicus et al. 2006). Although we did not attempt to estimate them, further savings were realized because the survey data were recorded directly in ArcView shapefile. Consequently, data entry from field sheets and the related data checking were eliminated.

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Table 1. Estimates of the treatment effect (i.e., time savings from sampling plots with the real time survey technique in 2005). For model 1, the effect of treatment is assumed to be additive: $\gamma = E[Y_i(1) | X = x] - E[Y_i(0) | X = x]$. For models 2 – 4, the effect of treatment is assumed to be multiplicative, $\gamma = E[Y_i(1) | X = x] / E[Y_i(0) | X = x]$. For models 2 – 4, we determined 95% confidence intervals using $\exp[\hat{\gamma} \pm 1.96 \cdot se(\hat{\gamma})]$.

Model	Before matching		After matching	
	Estimate	95% C.I.	Estimate	95% C.I.
1	-2.19	(-1.54, -2.84)	-2.05	(-1.18, -2.92)
2	0.71	(0.65, 0.77)	0.67	(0.59, 0.76)
3	0.73	(0.67, 0.80)	0.71	(0.62, 0.82)
4	0.73	(0.67, 0.79)	0.71	(0.62, 0.81)

Table 2. Estimates (minutes/plot) of the average treatment effect (ATE) and average treatment effect for the treated (ATT) using the full and matched datasets in the 2004 and 2005 ring-necked duck breeding pair survey.

Model	Average treatment effect (ATE)				Average treatment effect for the treated (ATT)			
	Before matching		After matching		Before matching		After matching	
	Estimate	95% C.I.	Estimate	95% C.I.	Estimate	95% C.I.	Estimate	95% C.I.
1	-2.18	(-1.61, -2.71)	-1.98	(-1.26, -2.64)	-2.18	(-1.61, -2.71)	-1.74	(-1.14, -2.40)
2	-2.24	(-1.81, -2.70)	-2.00	(-1.40, -2.54)	-2.24	(-1.79, -2.73)	-1.83	(-1.31, -2.34)
4	-2.11	(-1.65, -2.55)	-1.90	(-1.29, -2.43)	-2.02	(-1.58, -2.49)	-1.74	(-1.20, -2.25)

Table 3. Approximate cost savings realized by using the real time survey approach in the 2005 ring-necked duck breeding pair survey. Calculations were based on a reduction of ~1.75 minutes of survey time per plot (average causal treatment effect) for all plots.

Expense	Crew member	Hours or days	Cost/hr or day (\$)	Cost (\$)
Air time (hrs)	Pilot	7.3 ^a	230 ^b	1,684
	Observer	7.3 ^a	32 ^c	234
Ground time (hrs)	Observer	1.84 ^d	32 ^c	59
Travel (days)	Pilot and observer	1.0	150	150

^aAir time is equal to the per plot average causal treatment effect times 251 plots divided by 60 minutes.

^bAir time rate includes helicopter cost and pilot salary and fringe.

^cAverage observer salary and fringe.

^dGround time is equal to the hours of air time divided by 2.67 (hours between refueling) times 0.67 (hours to refuel).

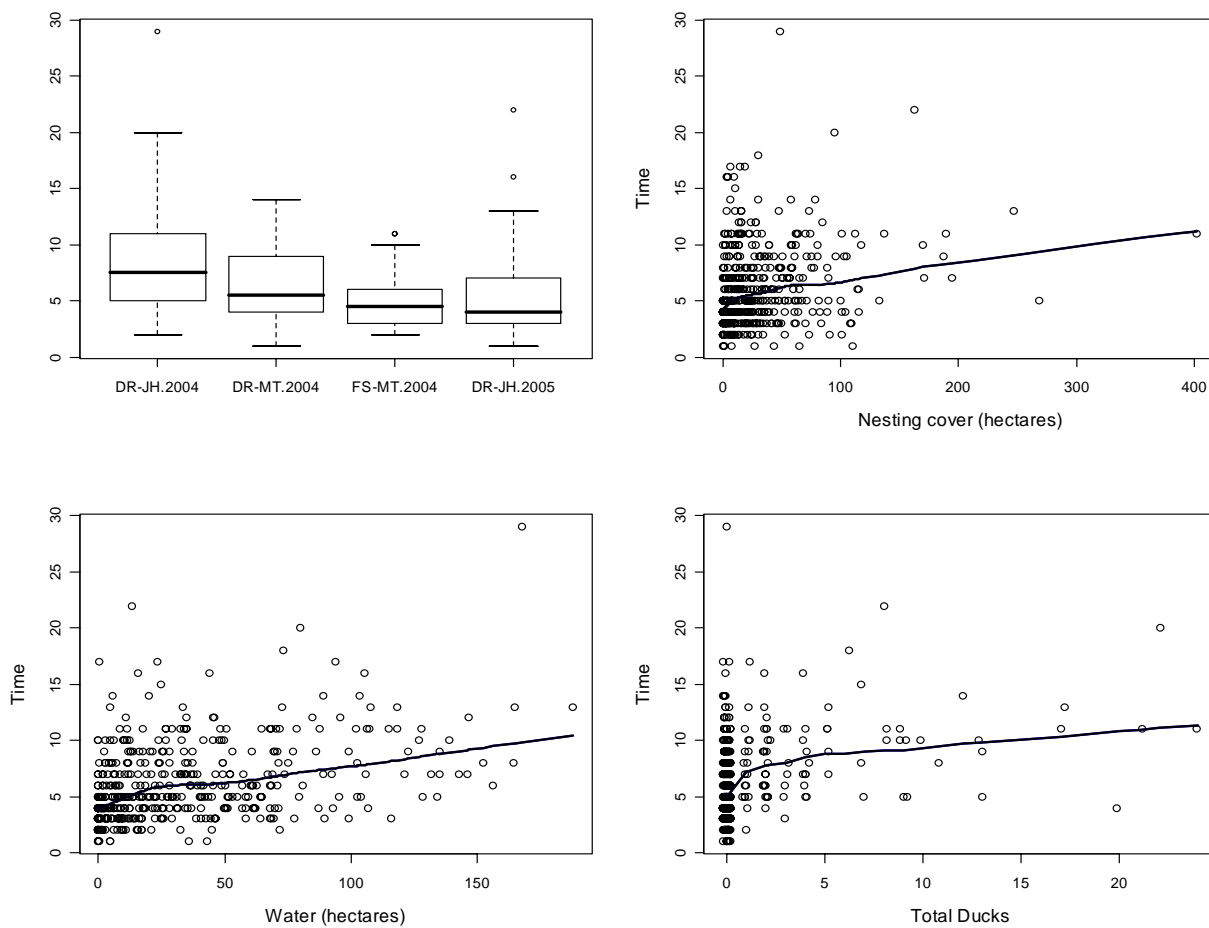


Figure 1. Time required to fly each plot versus covariates (observer/year, nest acres, water acres, ducks observed). Lines indicate loess smooths of the data using the `lowess` function in R (R Core Development Team 2005).

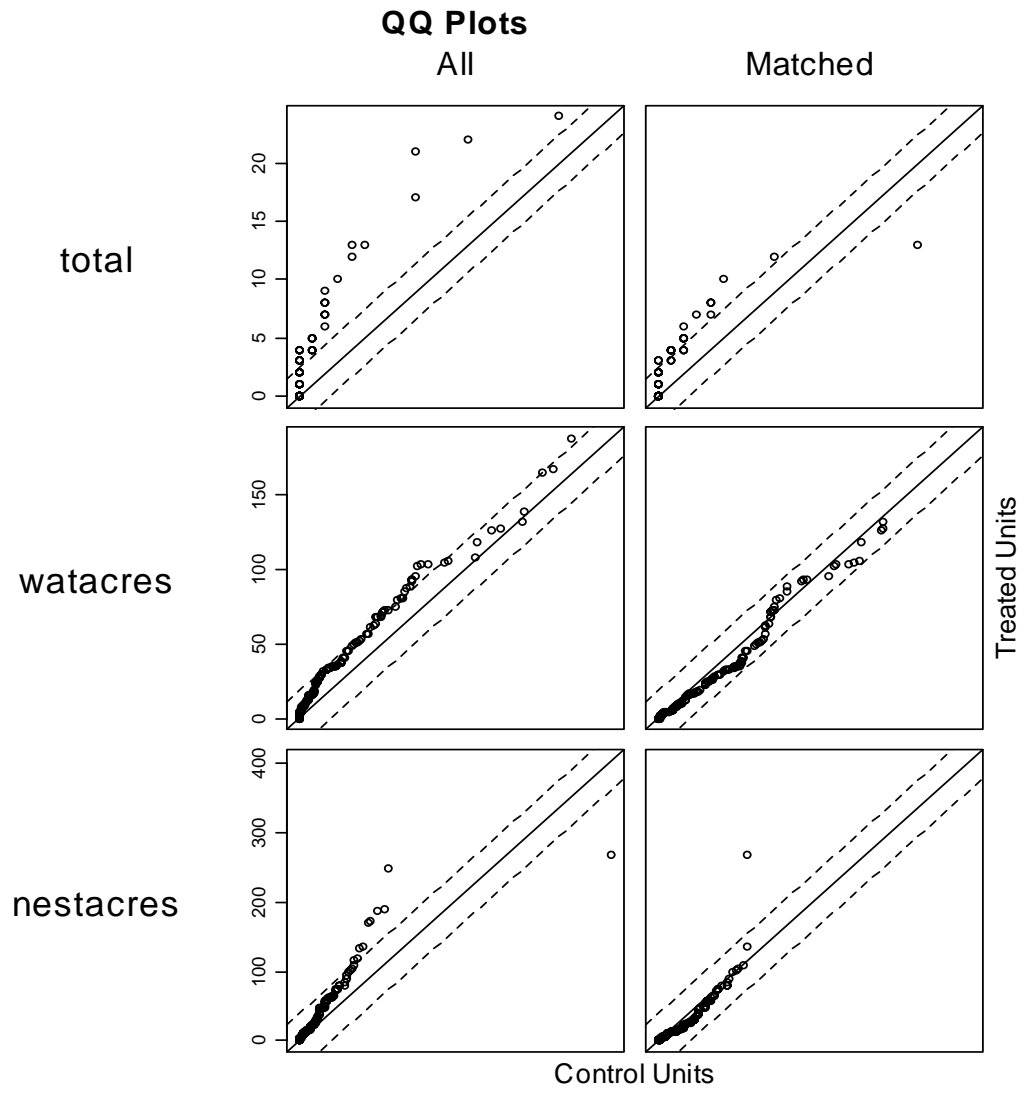


Figure 2. Quantile-quantile plots of the empirical distributions of each covariate before and after matching.

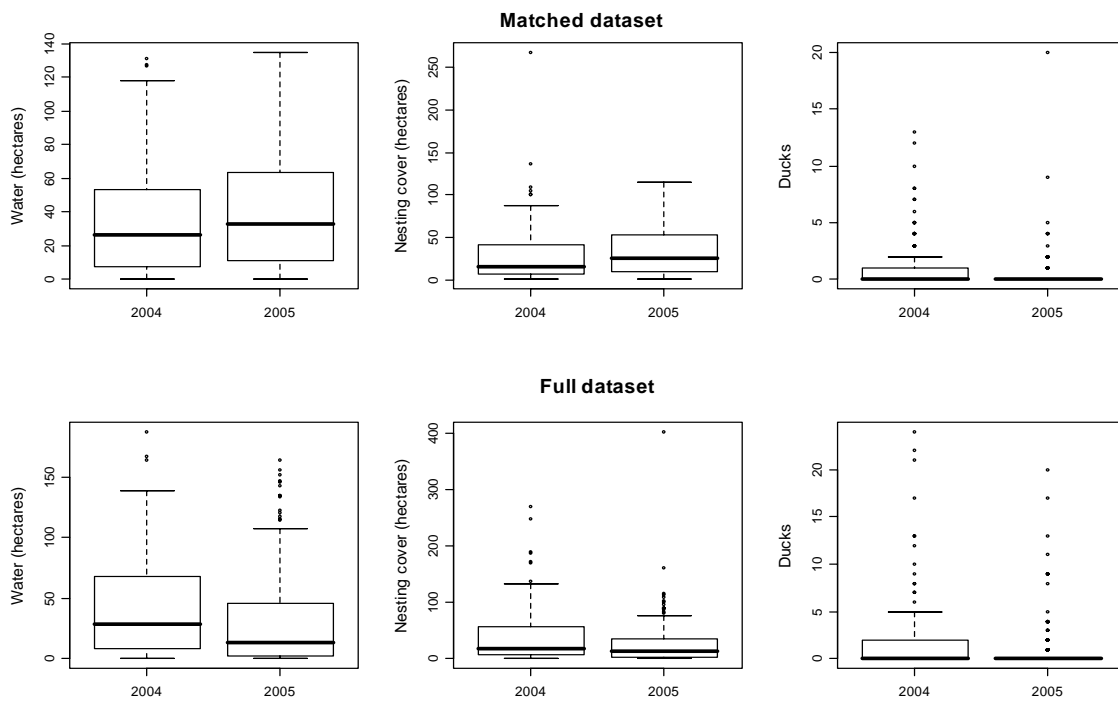


Figure 3. Boxplots of water (hectares), nesting cover (hectares), and ducks observed for the matched and full datasets.

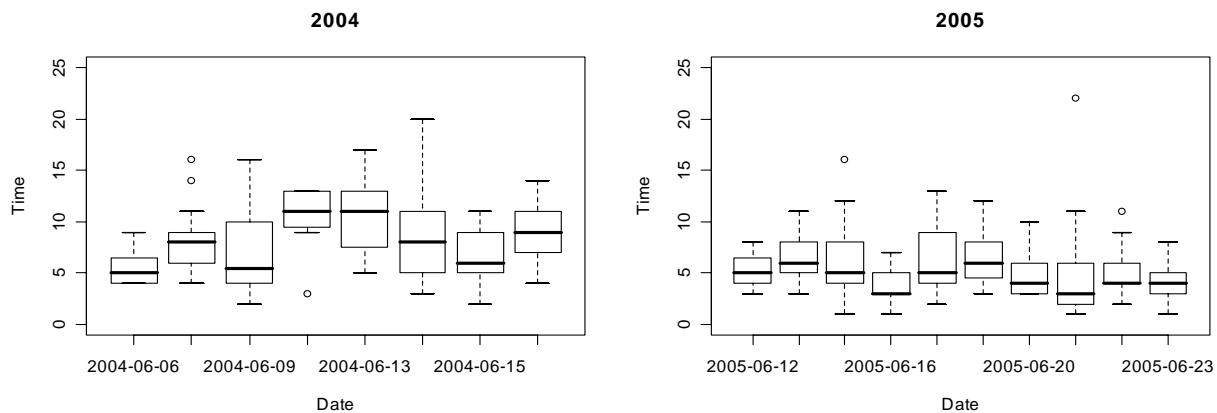


Figure 4. Boxplots of survey plot times versus date.

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 second year of a pilot survey to better understand the issues involved in monitoring these important but poorly studied ducks. The helicopter-based counts (12 – 23 June 2005) entailed 10 flight days and included a portion of Minnesota 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 section-sized survey plots, and 4 habitat classes were defined based on the amount of nesting cover in each plot. In contrast to 2004, we combined results from 2 separate surveys to estimate population size. We apportioned 230 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 21 plots to estimate population size in the remaining habitat. The combined population was estimated to be ~11,300 indicated breeding pairs (~25,000 birds). Numbers of ducks counted from the air and the ground on 14 lakes differed more in 2005 than in 2004, and the difference likely due to the 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, and 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 has been developing a forest wetlands and waterfowl initiative. The status of ring-necked ducks (*Aythya collaris*) 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 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 Public Land Survey (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 second 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 2006 and future operational surveys.

METHODS

Two separate surveys were conducted in 2005 to reduce the bias associated with the 2004 estimate. We continued to use a stratified random sampling design with 2 stratification variables: Ecological Classification System (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 modified the habitat class definitions used for stratification in 2004 (Table 1). Based on 2004 results, we also included 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, 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.

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 230 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 20 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.

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

$\text{var}(\text{SRS})$ = 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

$\text{var}(\text{StRS})$ = 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 2005 survey utilized an ArcView 3.x extension (DNRSurvey) in conjunction with a GPS 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 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 (94) were located in the Northern Minnesota Drift and Lake Plains Section. However, the fewest plots (8) were located in the Lake Agassiz, Aspen Parklands section this year rather than the Northern Superior Uplands

Section (Table 2). The highest and lowest sampling rate again occurred in the Lake Agassiz, Aspen Parklands Section and Northern Superior Uplands section, respectively. A total of 21 rather than 20 habitat class 3 and 4 plots was surveyed because a replacement plot was flown before permission to survey one of the originally selected plots was granted. The survey was conducted 12 – 24 June and entailed 11 survey-crew days. Observed pairs represented 36% of the indicated pairs tallied during the survey compared to 57% in 2004 (Table 3).

Estimated Pair Density

Mean pair density on habitat class 1 and 2 plots ranged from a high of 3.40 pairs/plot in the Lake Agassiz, Aspen Parklands Section to a low of 0.09 pairs/plot in the Northern Minnesota and Ontario Peatlands Section (Table 4). Mean pair densities were lower in 2005 in 4 of the 6 ECS sections compared to 2004. 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 3,490 in the Northern Minnesota Drift and Lake Plains Section to a low of 239 in the Northern Minnesota and Ontario Peatlands Section (Table 5). Fewer breeding pairs were estimated in 2005 in 3 of the 6 ECS sections than in 2004. Considering both years, pair numbers were greatest in the Northern Minnesota Drift and Lake Plains Section and fewest in the Western and Southern Superior Uplands and the Northern Minnesota and Ontario Peatlands sections.

The estimated population of ring-necked ducks on habitat class 1 and 2

plots ranged from a high of 6,981 in the Northern Minnesota Drift and Lake Plains Section to a low of 477 in the Northern Minnesota and Ontario Peatlands Section (Table 6). As with indicated breeding pairs, fewer ducks were estimated in 2005 in 3 of the 6 ECS sections than in 2004. 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 and the Northern Minnesota and Ontario Peatlands sections.

In 2005, 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 11,329 (90% confidence interval = 5,359 – 17,298), and the estimated ring-necked duck population was 24,943 (90% confidence interval = 12,476 – 37,411).

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 shifted somewhat to the east compared to 2004 (Figure 1). Estimates from 2004 and 2005 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

There was a greater discrepancy between boat counts and the aerial counts of indicated breeding pairs for the individual lakes included in the *Bemidji Area Ring-necked Duck Pair Survey* in

2005 than in 2004 (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. Poorer agreement between the 2 surveys in 2005 than in 2004 was likely due to the greater time that elapsed between the boat counts and the aerial surveys.

STRATIFICATION EVALUATION

Analysis of variance indicated that the stratification used in the 2005 survey performed well. Indicated pairs were related significantly to ECS sections ($F_{5,218} = 7.17$, $P < 0.001$) and to habitat classes within the ECS sections ($F_{1,218} = 28.7$, $P < 0.001$). The importance of habitat class varied among ECS sections ($F_{5,218} = 7.94$, $P < 0.001$), although more mean indicated pairs were seen in habitat class 1 plots than in class 2 plots in 5 of 6 ECS sections. Pair density was greatest in the Lake Agassiz, Aspen Parkland habitat class 1 stratum plots. In contrast, no indicated pairs were observed in habitat-class 2 plots in the Northern Minnesota and Ontario Peatlands ($n = 16$) or Lake Agassiz, Aspen Parkland sections ($n = 3$). However, indicated pairs also were not observed in high-density plots in the Western and Southern Superior Uplands ($n = 11$). Our best estimate of relative efficiency of the stratified design compared to a simple random sample suggested we would have needed approximately 1.2 times as many plots to achieve the same precision under a simple random sampling design. However, we lacked variance estimates for 3 strata because no birds were observed on sample plots in those strata. Thus, standard error estimates and design effects should be interpreted cautiously.

Data Acquisition

Generally less time was required to survey a plot in 2005 than in 2004 (Table 8). Survey time ranged from 1 –

22 minutes (mean = 5.2) compared to 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 and 2005 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, but a second survey based on a simple random sample of plots in habitat classes 3 and 4 was needed to provide 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. Furthermore, emergent shoreline-vegetation associated with larger lakes containing fish was defined as potential ring-necked duck nesting cover when stratification decisions were based on MN-GAP data alone. Ring-necked ducks do not occupy these types of lakes during the breeding season. Stratification would likely be improved somewhat by not including emergent shoreline-vegetation associated with these larger lakes when quantifying potential nesting cover in each PLS section. Additional GIS data would be required to identify this cover.

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 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 zero 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 intend to begin modeling to evaluate various plot sizes in 2006.

Recommendations

- Conduct the 2006 survey using the same proportional allocation of 230 habitat class 1 and 2 plots among the 6 ECS sections. Conduct the 2006 survey choosing a simple random

sample of 20 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 2004 and 2005 data alone suggest somewhat different geographical distributions for indicated breeding pairs, and a third year would help better define the core area.

- 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 2004-2005.

Habitat class	Definition ^a		% ^b	
	2004	2005	2004	2005
1	Plots with \geq the median amount of MN-GAP class 14 and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially high pairs).	Plots with \geq the median amount of MN-GAP class 10, 14, and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially high pairs).	15.3	24.5
2	Plots with $<$ the median amount of MN-GAP class 14 and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially moderate pairs).	Plots with $<$ the median amount of MN-GAP class 10, 14, and/or 15 cover within 250 m of and adjacent to open water (i.e., potentially moderate pairs).	15.3	24.5
3	Plots with no MN-GAP class 14 and/or 15 cover that include open water that is within 250 m of a shoreline (i.e., potentially low pairs).	Plots with no MN-GAP class 10, 14, and/or 15 cover that include open water that is within 100 m of a shoreline (i.e., potentially low pairs).	25.2	7.7
4	Plots with no MN-GAP class 14 or 15 cover and no open water within 250 m of a shoreline (i.e., potentially no pairs).	Plots with no MN-GAP class 10, 14, and/or 15 cover and no open water within 100 m of a shoreline (i.e., potentially no pairs).	44.2	43.3

^aPlots are Public Land Survey sections. MN-GAP cover class 10 is described as lowlands with $<10\%$ tree crown cover and $>33\%$ cover of low-growing deciduous woody plants such as alders and willows. MN-GAP cover class 14 is described as wetlands with $<10\%$ tree crown cover that is dominated by emergent herbaceous vegetation such as fine-leaf sedges. MN-GAP cover class 15 is described as wetlands with $<10\%$ tree crown cover that is dominated by emergent herbaceous vegetation such as broad-leaf sedges and/or cattails.

^bPercent of the survey area

Table 2. Sampling rates in the habitat class 1 and 2 strata in the Minnesota ring-necked duck breeding pair survey area, June 2004-2005.

Ecological Classification System section	~Area ^a		Sample plots		Sampling rate (%)	
	2004	2005	2004	2005	2004	2005
W & S Superior Uplands ^b	1,638	2,461	18	22	1.1	0.9
Northern Superior Uplands	1,810	4,648	13	36	0.7	0.8
N Minnesota & Ontario Peatlands	1,817	2,737	26	35	1.4	1.3
N Minnesota Drift & Lake Plains	5,048	8,383	78	94	1.5	1.1
Minnesota & NE Iowa Morainal	3,510	4,033	50	35	1.4	0.9
Lake Agassiz, Aspen Parklands	316	363	15	8	4.7	2.2

^aNumber of Public Land Survey sections in habitat classes 1 and 2.

^bWestern 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 2004-2005.

Year	Habitat class	No. of plots	Total ducks	Indicated Pairs			
				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

^aSurvey conducted 6 – 17 June.

^bSurvey conducted 12 – 24 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 2004-2005.

Ecological Classification System section	2004			2005		
	Plots	Mean pairs/plot	SE	Plots	Mean pairs/plot	SE
W & S Superior Uplands ^a	18	0.167	0.122	22	0.181	0.179 ^b
Northern Superior Uplands	13	0.566	0.396	36	0.252	0.118
N Minnesota & Ontario Peatlands	26	0.465	0.381 ^b	35	0.087	0.045 ^b
N Minnesota Drift & Lake Plains	78	0.707	0.155	94	0.416	0.138
Minnesota & NE Iowa Morainal	50	0.797	0.298	35	0.228	0.010
Lake Agassiz, Aspen Parklands	15	2.959	0.948	8	3.403	1.365 ^b

^aWestern and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

^bStandard 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 2004-2005.

Ecological Classification System section	2004				2005			
	Pairs	LCL ^a	UCL ^a	CV(%)	Pairs	LCL	UCL	CV(%)
W & S Superior Uplands ^b	273	0	626	74.1	444	0	1,207	99.5 ^c
Northern Superior Uplands	1,025	0	2,311	69.9	1,169	244	2,095	46.8
N Minnesota & Ontario Peatlands	845	0	2,030	82.0 ^c	239	20	457	54.1 ^c
N Minnesota Drift & Lake Plains	3,567	2,278	4,856	21.7	3,490	1,577	5,404	33.0
Minnesota & NE Iowa Morainal	2,799	1,041	4,556	37.4	918	241	1,595	43.6
Lake Agassiz, Aspen Parklands	935	405	1,465	32.0	1,235	273	2,198	40.1 ^c

^aEstimates 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.

^bWestern and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

^cVariance 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 2004-2005.

Ecological Classification System section	2004				2005			
	Birds	LCL ^a	UCL ^a	CV(%)	Birds	LCL	UCL	CV(%)
W & S Superior Uplands ^b	546	0	1,252	74.1	889	0	2,415	99.5 ^c
Northern Superior Uplands	2,049	0	4,622	69.9	2,339	488	4,190	46.8
N Minnesota & Ontario Peatlands	2,183	0	5,385	85.7 ^c	477	40	915	54.1 ^c
N Minnesota Drift & Lake Plains	7,849	5,015	10,682	21.7	6,981	3,154	10,808	33.0
Minnesota & NE Iowa Morainal	5,597	2,082	9,113	37.4	4,122	187	8,057	56.4
Lake Agassiz, Aspen Parklands	2,097	856	3,339	33.4	2,471	545	4,396	40.1 ^c

^aEstimates 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.

^bWestern and Southern Superior Uplands sections combined due to the small area of the Southern Superior Uplands occurring in the survey area.

^cVariance 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, 2004-2005.

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

^aLCL = lower 90% confidence level. UCL = upper 90% confidence level.

^bPopulation 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).

^cPopulation estimates were based on a simple random sample of Public Land Survey (PLS) sections in habitat classes 3 and 4.

^dVariance 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. Time required to complete the Minnesota ring-necked duck breeding pair survey, June 2004-2005.

Year	# 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

^aIncludes all observers.

^bTime between the initial start of the helicopter each morning and final shutdown of the helicopter each afternoon.

^cAir time spent surveying the individual plots.

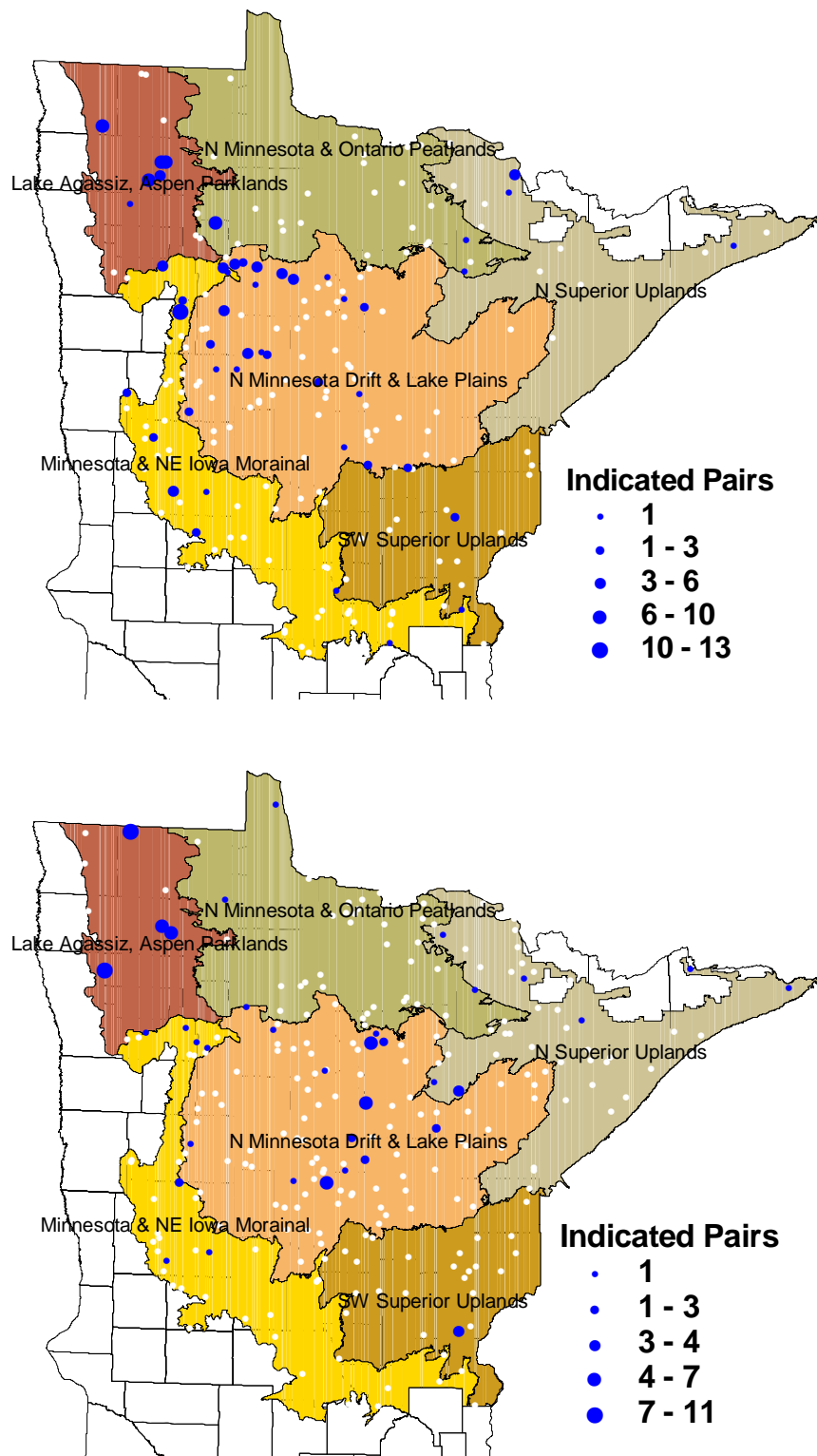


Figure 1. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area in June 2004 (top) and 2005 (bottom). White circles indicate plots where no indicated pairs were seen.

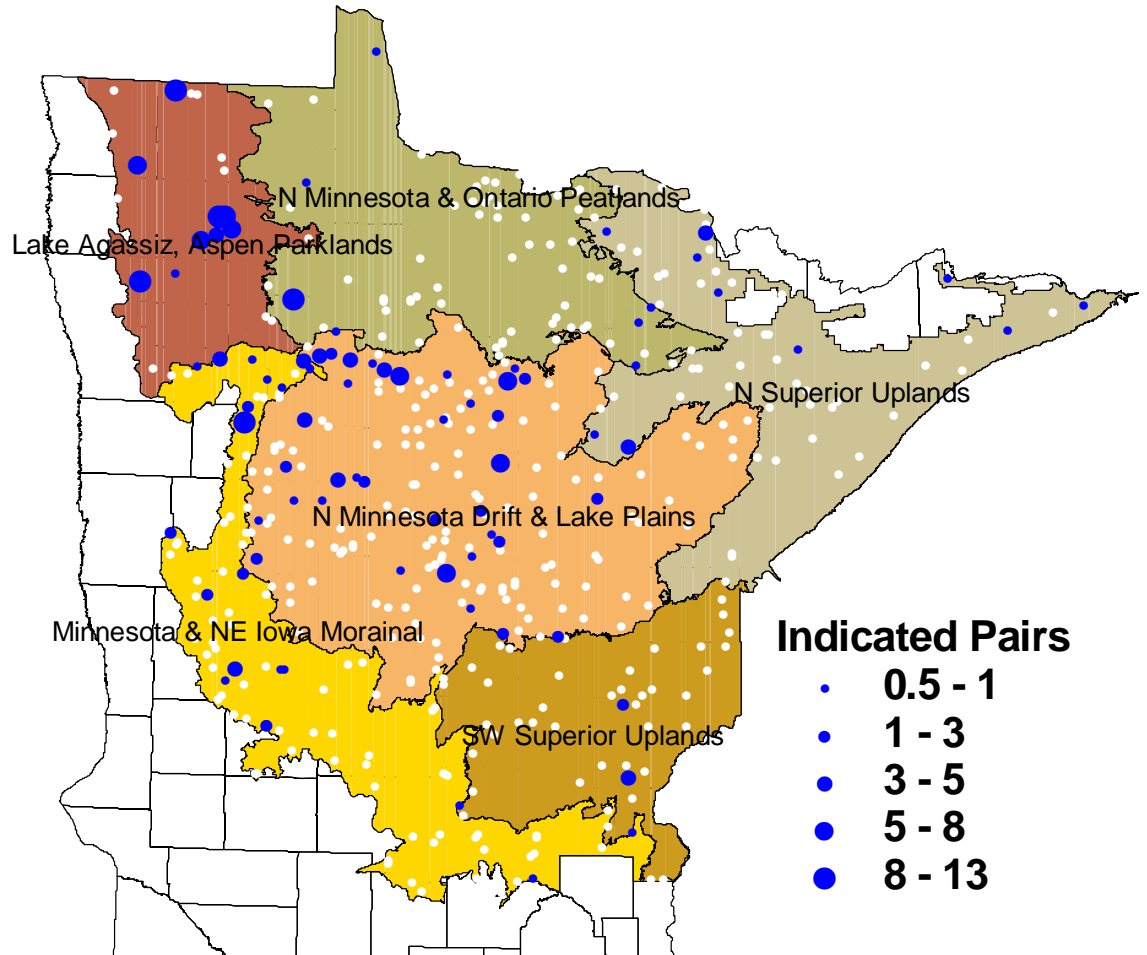


Figure 2. Plot locations and numbers of indicated breeding pairs of ring-necked ducks observed on survey plots in the Minnesota survey area, June 2004-2005.

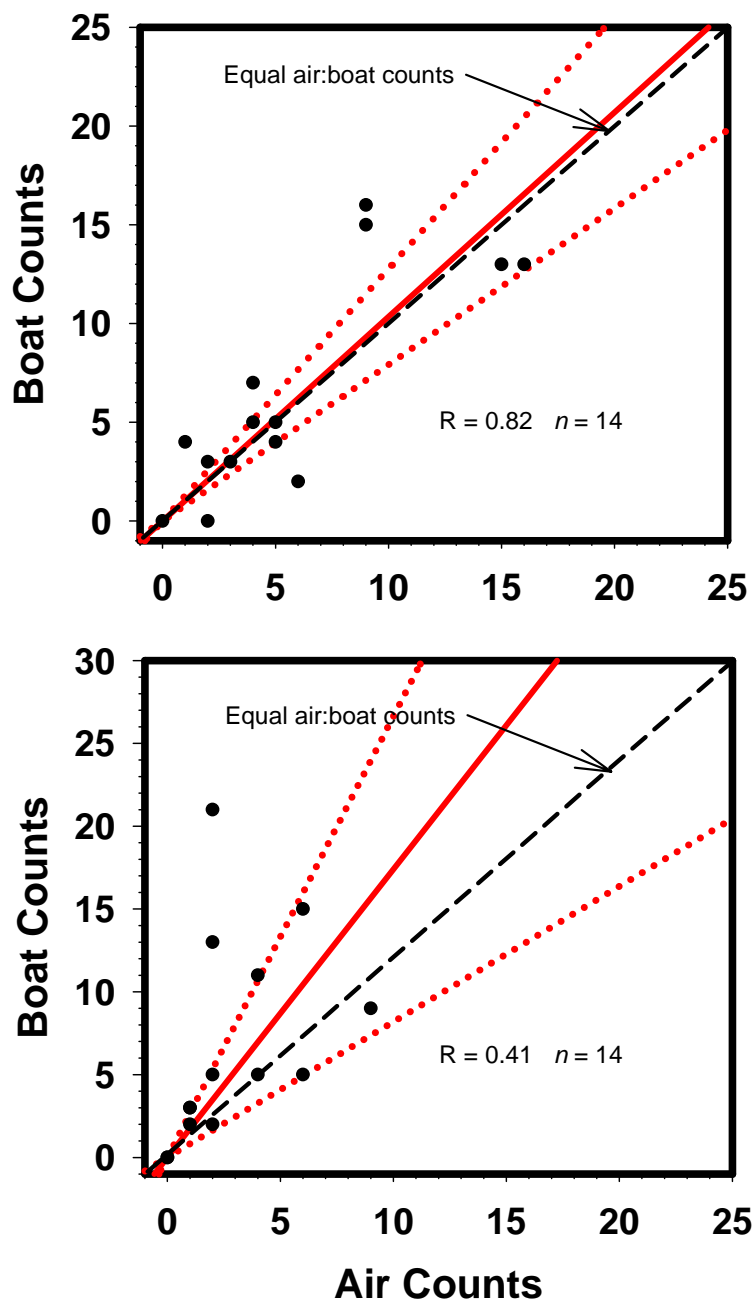


Figure 3. Regression lines and 95% confidence intervals comparing the numbers of indicated ring-necked duck breeding pairs counted from a boat and from the air on 14 lakes comprising the *Bemidji Area Ring-necked Duck Survey*, June 2004 (top) and 2005 (bottom).

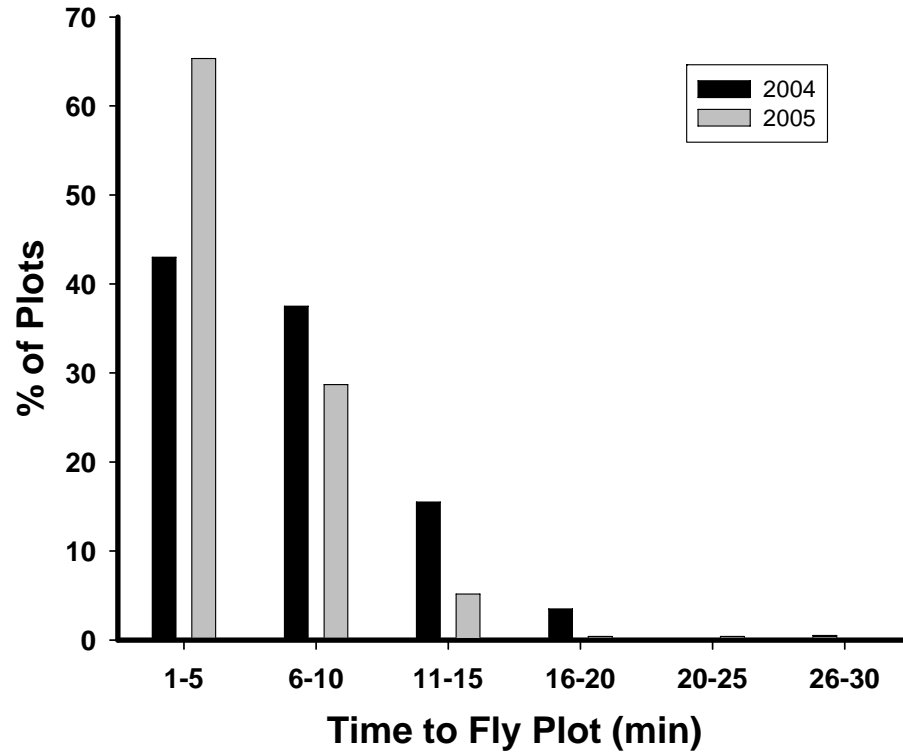


Figure 4. Time required (all observers) to survey individual ring-necked duck breeding pair plots in the Minnesota survey, June 2004 – 2005.

COST EFFECTIVENESS OF SINGLE- VS. DOUBLE-CYLINDER OVER-WATER NEST STRUCTURES¹

Michael C. Zicus, David P. Rave, and John R. Fieberg

Abstract: Minnesota waterfowl management plans prescribe widespread deployment of mallard (*Anas platyrhynchos*) nest structures. We compared 53 single- and 57 double-cylinder structures from 1996 – 2003 because managers used both structure types but were uncertain about their respective cost effectiveness. More nests occurred in doubles, but numbers of successful nests and hatched ducklings were comparable for both types. Nest success in singles and doubles was 92.8% and 79.4%, respectively, with nest abandonment being >4.5 times greater in doubles. Structure damage occurred only at ice out and was greater for doubles. However, relative risk of failure for double-vs. single-cylinder structures was similar (1.26; 95% confidence interval = 0.91 –

.75) and increased with size of the open-water area containing the structure. Modeling indicated ~95% of recruits from nest structures were additional recruits. A case history approach indicated doubles produced an additional recruit for \$23.11 vs. \$23.25 for singles. However, these estimates were sensitive to assumptions used to apportion costs between structure types and ignored structure placement influences. Placement affected cost effectiveness significantly with structures placed in open-water areas >10 ha being more cost-effective. Results also suggested singles might be more effective than doubles when placement is considered. Lower nest abandonment alone might make single-cylinder structures the better choice.

¹ Abstract of paper in press in the Wildlife Society Bulletin. 2006. Volume 34.

LAKE CHRISTINA RECLAMATION: ECOSYSTEM CONSEQUENCES OF BIOMANIPULATION

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SUMMARY OF FINDINGS

We assessed early patterns in fish community characteristics, limnological features, and wildlife use of Lake Christina following the 2003 rotenone treatment. Following rotenone application, fish were reduced, but not eliminated, and a diverse population comprised of planktivorous, benthivorous, and piscivorous species was again present by 2006. However, dramatic improvements in water transparency, increased density of large-bodied zooplankton (*Daphnia* spp.), and increases in occurrence of submerged aquatic plants (especially *Chara* spp.) were also evident and, collectively, these results seem to indicate the onset of a shift back to the clear-water state.

INTRODUCTION

Lake Christina, a 1,619-ha shallow lake in Douglas County in west-central Minnesota, is nationally recognized as a critical staging area for migrating canvasbacks, and also is a breeding location for a number of unique nongame bird species. Since the 1950s, the lake has alternated between ecological extremes, sometimes characterized by favorable conditions, and at other times of little use as waterfowl habitat. Sustained high water and dense populations of undesirable fish species are believed to be associated with shifts toward high turbidity and other unfavorable limnological characteristics, along with extreme habitat deterioration for waterfowl and other wildlife. Following obvious trends of habitat deterioration, the lake was "reclaimed" in 1965 and 1987 via chemical removal of fish. Extensive scientific monitoring was conducted in association with the 1987 treatment.

Limnological and waterfowl-use data were gathered before and after the 1987 fish kill to assess the nature and causal mechanisms associated with observed changes. Dramatic improvements in water quality features, extensive development of submerged aquatic plants, and increased fall use by migrating ducks followed the 1987 reclamation (Hanson and Butler 1994, Hansel-Welch et al. 2003). Research before and after treatment contributed to improved understanding of ecology and management potential of shallow lakes in North America. Unfortunately, data gathering efforts at Lake Christina subsequently dwindled, more or less at the same time as habitat quality and suitability for wildlife again declined. During 2000-2003, water clarity, distribution of submerged macrophytes, and fall use by migrating ducks all indicated that the lake had again stabilized in a deteriorated condition characterized by poor water quality, a sparse community of submerged macrophytes, and limited suitability for diving ducks and other wildlife species. Fish were removed from Lake Christina using rotenone during October 2003 to stimulate a limnological shift to more favorable habitat conditions. Here, we summarize responses of fishes, limnological features, and wildlife use during 2004-05, the first two years following the fish removal. Our objectives were to evaluate broad ecosystem-level responses of the lake to the 2003 fish removal, with special emphasis on patterns of recruitment by fish that either survived the rotenone treatment, or immigrated into the lake following the fish kill. Here, we report preliminary patterns in fish populations, seasonal water transparency, abundance of large filter-feeding zooplankton (*Daphnia* spp.),

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occurrence of submerged macrophytes, fall use by waterfowl, and nesting efforts by colonial waterbirds.

METHODS

We used a variety of techniques to collect data summarized in this report. Adult, juvenile, and larval fish were sampled from May – August using gill and trap nets, beach seines, minnow traps, ichthyoplankton push nets, and boom electrofishing. Water transparency was determined using a Secchi disk, and by measuring vertical light attenuation in the water column. Zooplankton were collected using a vertical column sampler. Submerged macrophytes were sampled using weighted plant rakes. Fall use by ducks and geese was assessed as numbers observed during aerial and ground counts during late September – mid-November. Western grebe (*Aechmophorus occidentalis*) nests were counted during weekly surveys using kayaks. Methods used in collecting these and other data are discussed in greater detail in Hanson et al. (2006).

RESULTS AND DISCUSSION

We believe conditions observed during 2004-05 indicated that Lake Christina has entered a period of transition, and is tending back toward the clear-water state. Our results indicated presence of a persistent fish community during spring 2004, approximately 6 months after the October 2003 rotenone treatment. Recruitment by remnant fishes was very strong (Figure 1) and, by 2005, a diverse fish community was again present and included benthivorous, planktivorous, and piscivorous species (Figure 2). Disappointing, but not unexpected, was evidence of rapid recovery by bullheads, carp, and fathead minnows during the 2 years immediately following the rotenone treatment. Data gathered during 2004-05 also contain strong signals indicating a shift towards more favorable ecological conditions (as described by Scheffer et al. 1998). While changes in abundance of

large-bodied herbivorous zooplankton (*Daphnia* spp., Figure 3) were equivocal, concomitant lake-wide trends toward higher water transparency during spring periods (Figure 4), and changes in abundance and composition of submerged aquatic plants (Figure 5) are consistent with outcomes lake managers had hoped to achieve, and with patterns observed following the 1987 rotenone treatment. One of the most encouraging signals observed following the 2003 rotenone treatment was the sharp increase in *Chara* spp. during 2004, the first post-treatment year (Figure 5B). Sharp increases in *Chara* spp. often portend major ecological shifts towards a clear-water state in shallow lakes and a similar trend was also observed within a year following the 1987 rotenone treatment at Lake Christina. Fall use by migrating ducks, coots, and Canada Geese (*Branta Canadensis*) also increased during 2004, a pattern also similar to that observed during 1988-1989 (Figure 6). Finally, we emphasize that even if the over-all lake response is ultimately similar to that observed following the 1987 treatment (and induces a transition to the clear-water state), more dramatic, sustained improvements in water transparency may not be evident until 2006, or even later. Non-target effects of rotenone in shallow lakes and wetlands may be considerable, but are rarely considered in lake rehabilitation studies. For example, Lake Christina has supported breeding western grebes since the late 1960s and a large population was observed using the lake during 2003. Availability of small prey fishes is considered crucial for successful recruitment of western grebes because adults fly infrequently other than during migration. During 2004, and following the 2003 rotenone treatment, adult western grebes returned to Lake Christina, but quickly abandoned traditional nesting areas and left the lake, presumably due to absence of suitable prey. By 2005, western grebes returned in large numbers and over 300 nests were identified and monitored. This may indicate that non-

target effects of rotenone on some colonial waterbirds should be expected, but are short-term in that breeding waterbird populations return in response to recruitment of young fishes.

Comparison among historical relationships has great potential to help researchers identify signals of transition, thus indicating if and when lake-wide changes are underway. Lake managers have continuing needs to identify limnological signals useful for anticipating periods of rapid change, especially when the lake is entering transition to the turbid-water state. This would facilitate better use of less drastic measures to maintain a clear-water state. For example, since 1999, environmental signs showed evidence that the lake was probably transitioning towards the turbid state. In retrospect, we know that this was true. For example, TP:chl *a* ratios may be important indicators of the ecological state of this and other shallow lakes (Dokulil and Teubner 2003), and researchers may benefit from monitoring trends relative to the 3:1 threshold (Figure 7). Alternatively, based on results of indicator species analyses, concern may be justified when high counts of small cladocerans such as *Bosmina* spp. consistently occur. Additionally, it may be possible to use the importance values of *Chara* to monitor whether the lake is stable or in transition. If *Chara* spp. shows sharp lake-wide declines, as it did during the period of 1999-2001, then perhaps the onset of a period of deterioration and a shift to the turbid state may be anticipated.

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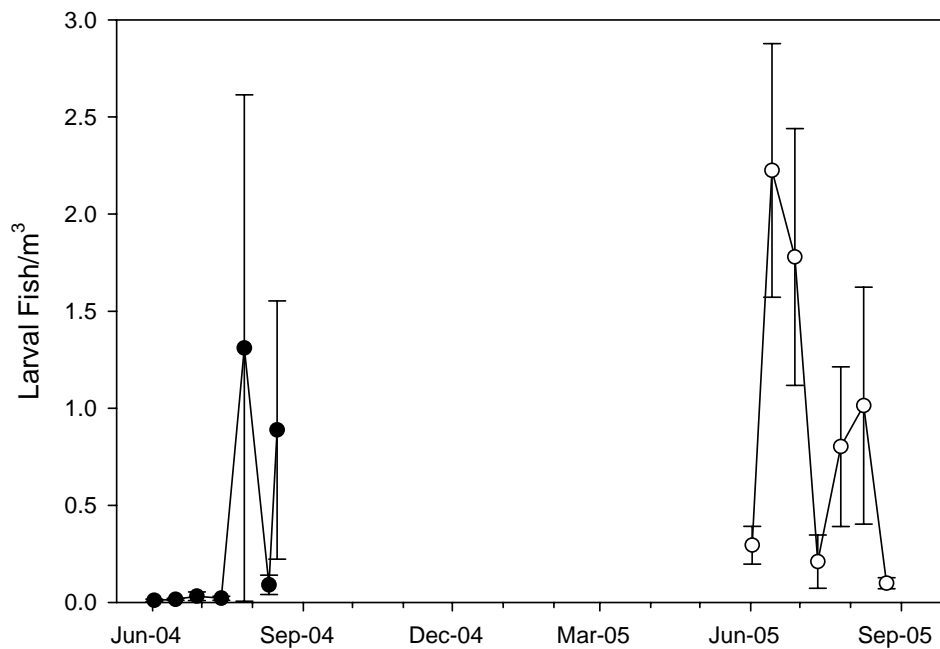


Figure 1. Larval fish tow results for 2004 and 2005 (average larval fish/m³). Vertical bars indicate +/- 1 standard error.

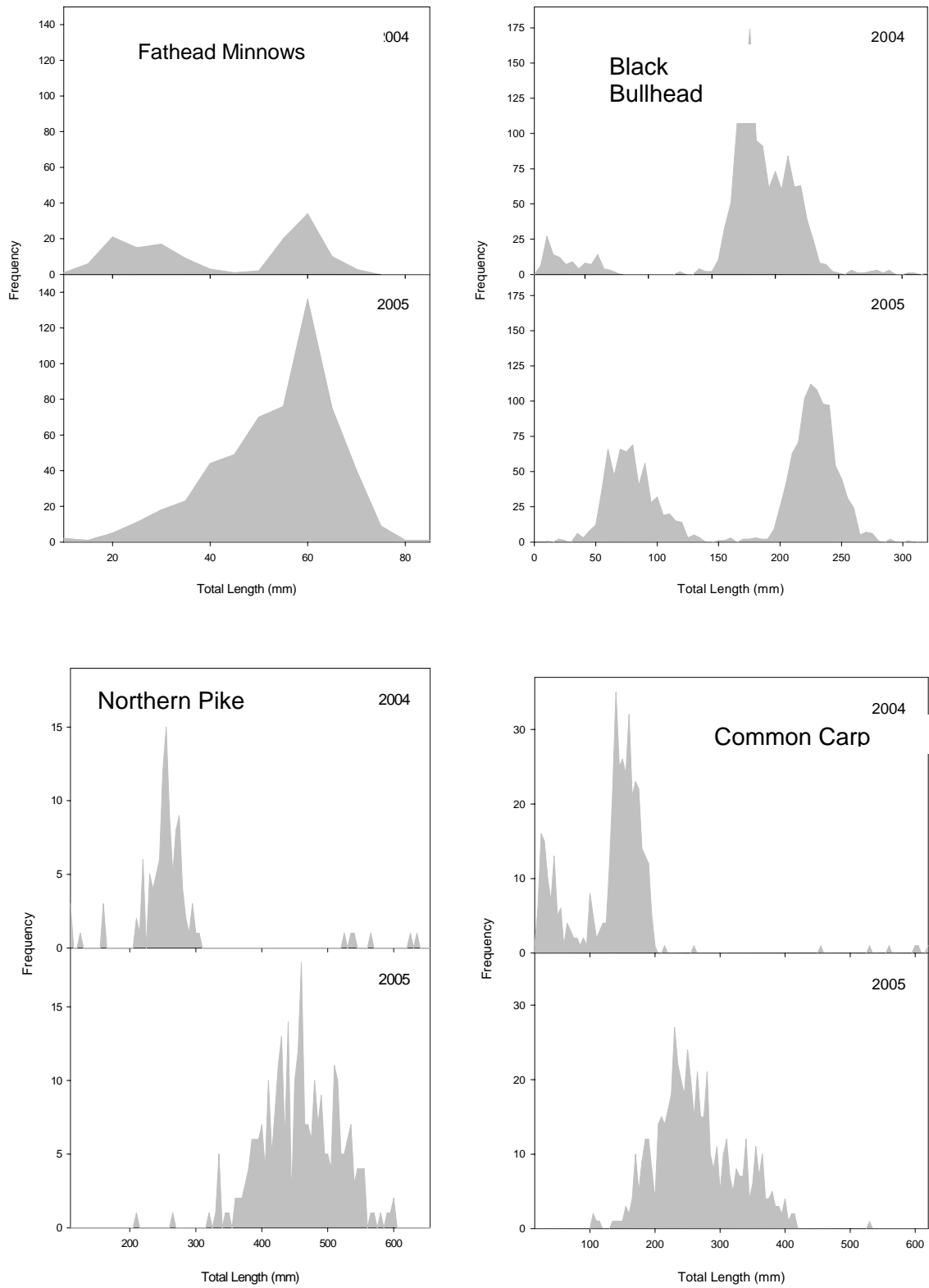


Figure 2. Length distribution of common fishes captured using beach seines at Lake Christina during 2004-05.

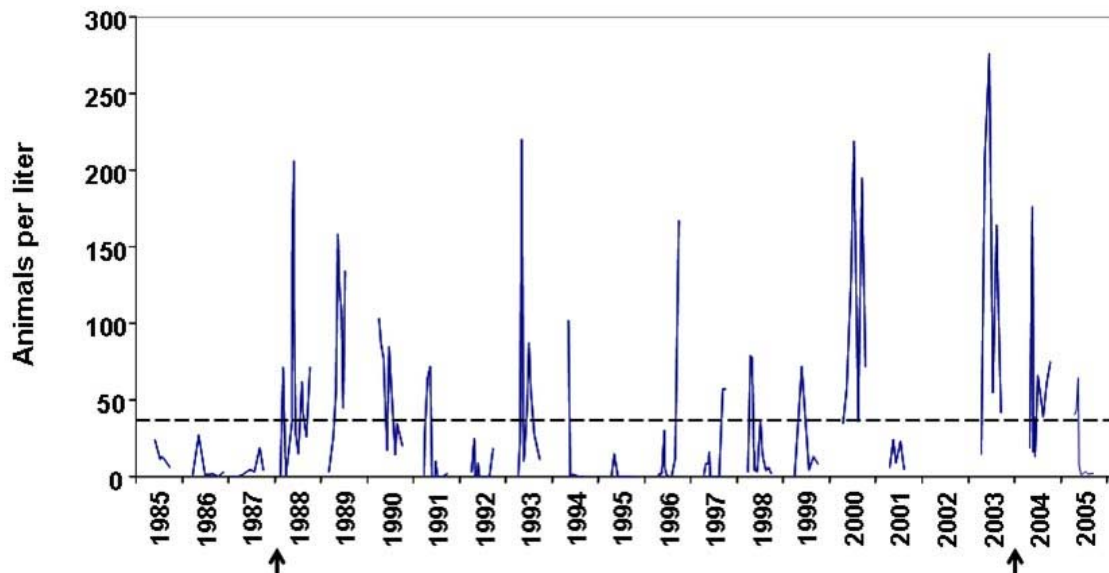


Figure 3. Mean lake-wide *Daphnia* spp. density (no./liter) in Lake Christina, 1985-2005. Arrows indicate rotenone treatments in October of 1987 and 2003. Dashed line indicates long-term mean over the 21-year record.

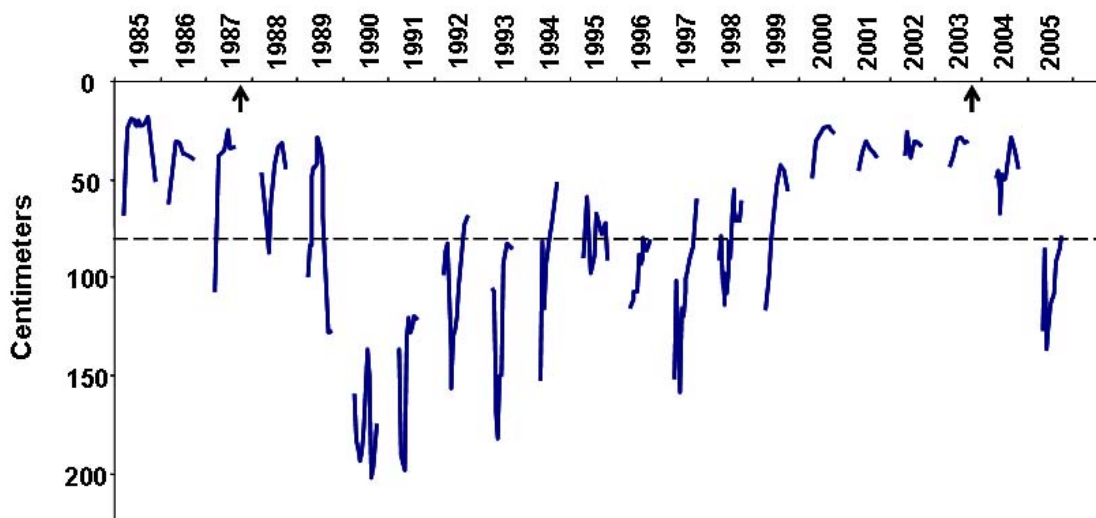


Figure 4. Mean lake-wide secchi depth (cm) in Lake Christina, 1985-2005. Arrows indicate rotenone treatments in October of 1987 and 2003. Dashed line indicates long-term mean over the 21-year record.

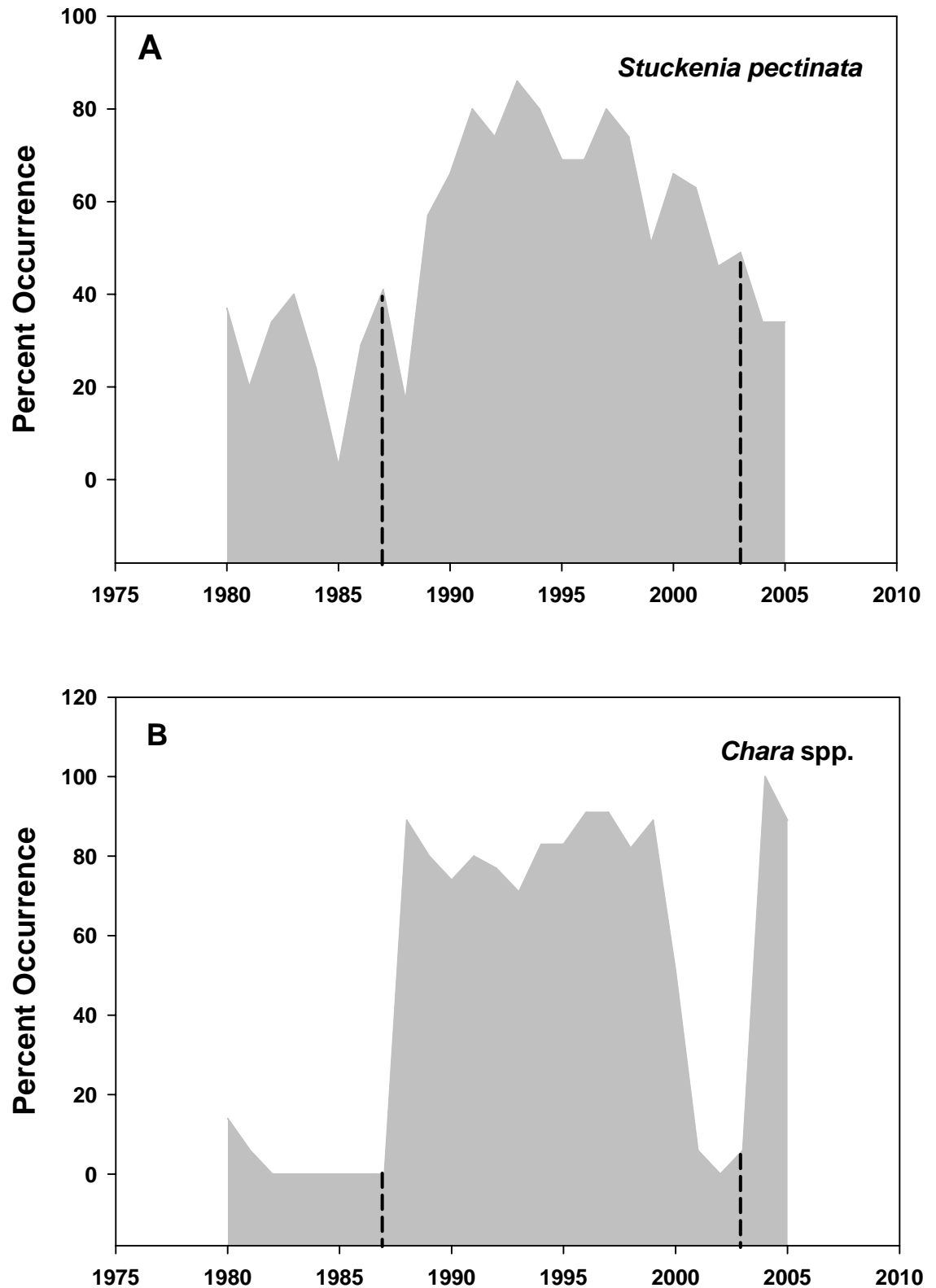


Figure 5. Results of submerged aquatic plant surveys at Lake Christina during 1980-2005. Plotted values indicate percent occurrence of 2 species (*Stuckenia pectinata* (A), and *Chara* spp. (B)), sampled at 35 locations around the lake. Hatched lines indicate timing of rotenone treatments (1987, 2003).

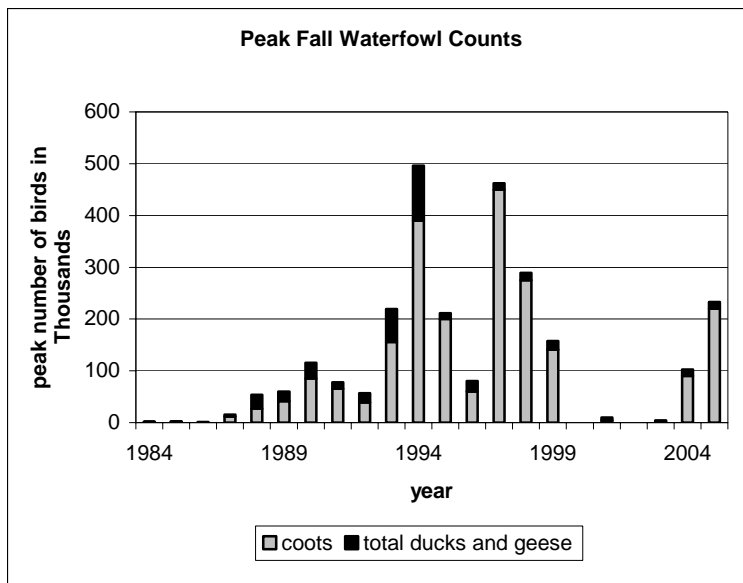


Figure 6. Annual peak waterfowl estimates for Lake Christina during 1984-2005.

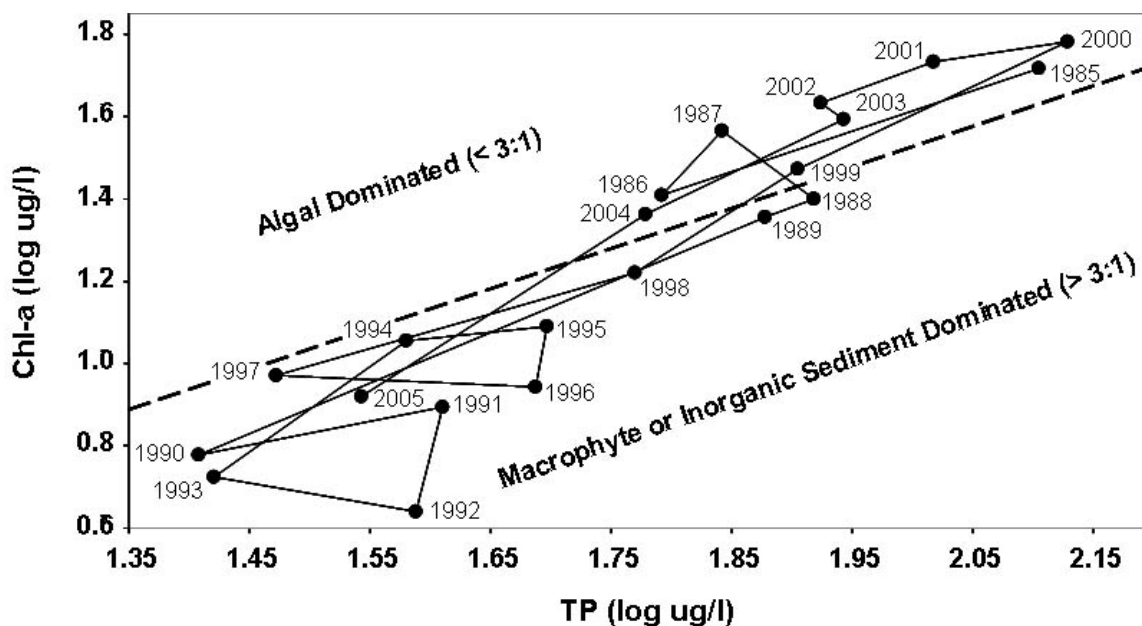


Figure 7. Chronological history of total phosphorus (TP):chlorophyll a (chl-a) ratios in Lake Christina, 1985-2005. Dashed line indicates 3:1 ratio of TP:chl-a; shallow lakes with values below this line often exhibit characteristics of the clear-water state.

RELATIONSHIPS AMONG LANDSCAPE FEATURES, FISH ASSEMBLAGES, AND SUBMERGED MACROPHYTE COMMUNITIES IN PRAIRIE WETLANDS

Mark A. Hanson, Brian R. Herwig, Kyle D. Zimmer¹, and Jerry A. Younk

SUMMARY OF FINDINGS

We are assessing fish community patterns and influences of site- and landscape-level variables on fish assemblages and various ecological features of prairie wetlands in two areas in western Minnesota (generally Polk and Grant County areas). Fish populations during the first year of the study (2005) 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. In general, landscape-scale variables were not useful in predicting presence of fish populations in study wetlands, but fish communities tended to reflect influences of wetland size, depth, and presence of piscivorous fish species. Biomass of planktivorous fish was not related to abundance (mass) of submerged macrophytes in our study wetlands. In contrast, biomass of benthivorous fish was negatively related to mass of submerged macrophytes in Grant, but not Polk County wetland sites. We believe this indicates presence of a strong interaction between benthivorous fish and ambient nutrient concentrations, perhaps indicating greater potential for macrophyte loss with introduction of benthivorous fish in Grant County wetlands. These results are preliminary and similar data will be gathered in 2006.

INTRODUCTION

Fish communities exert strong, but variable, influences on ecological properties of deep prairie wetlands and shallow lakes. For example, previous research has shown that dense populations of fathead minnows (*pimephales promelas*) have key structuring influences on invertebrate populations and wetland community

characteristics (Zimmer et al. 2002), although additions of piscivores (e.g., walleye fry) may negate those effects (Herwig et al. 2004). Less is known about ecological roles of benthivorous fishes, but their presence is often associated with turbid conditions.

Winter hypoxia and isolation are believed to be major constraints on wetland fish communities throughout the Prairie Pothole Region (PPR) of North America. Recently, some authors have suggested that distribution of fishes has increased among PPR wetlands due to anthropogenic activities and, perhaps, climate extremes. However, the distribution and community characteristics of wetland fishes across the PPR are poorly known. Past research has not assessed influences of both scale-dependent spatial factors and site-level environmental mechanisms that control distribution of fishes in prairie landscapes, while simultaneously evaluating influences of specific fish assemblages on wetland features.

During 2005-06, we were exploring patterns and assessing influences of spatial and site-level variables on fish communities in 73 deep wetlands and shallow lakes (wetlands) in west-central Minnesota, USA, an area along the eastern margin of the PPR (Figure 1). Two focus areas were chosen for study, with 36 and 37 sites along borders of Polk/Mahnomen (PM) and Grant/Stevens (GS) counties, respectively. Because it is widely believed that anthropogenic disturbance is greater in the GS area, including data from these regions provided a means of capturing influences of a potential land-use gradient in our spatial and environmental data. Here, we report results of preliminary analyses used to 1) identify patterns in wetland fish communities, 2) relate fish community assemblages to site- and landscape-level variables, and 3) assess potential

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relationships between biomass of planktivorous and benthivorous fish and submerged macrophytes in study wetlands.

METHODS

We estimated presence and abundance of fish in study wetlands using a combination of mini-fyke nets, gill nets, and minnow traps. Chlorophyll *a* was estimated according to procedures followed by the Minnesota Department Agriculture chemistry lab (St. Paul, MN) and was used as an index of phytoplankton biomass. Submerged macrophytes were sampled using a weighted plant rake. Samples of fish and submerged plants were weighed on site to provide indexes to abundance. We used Principle Components Analysis (PCA), to examine potential fish assemblage patterns in preliminary data collected during 2005. We used Canonical Correspondence Analysis (CCA) to relate site- and landscape-level variables to patterns in wetland fish communities. Lowess regression was used to evaluate the relationship between chlorophyll *a* (natural log) and biomass of submerged aquatic plants (natural log [$n+1$]) sampled in wetland study sites. Finally, we used analysis of covariance (ANCOVA) to relate biomass of planktivorous and benthivorous fish to mass of submerged plants.

RESULTS AND DISCUSSION

Fish were more widespread and fish communities were more complex than expected (Figure 2). PCA of fish abundance data indicated four distinct fish community types, including: 1) fishless, 2) minnow-only, 3) multi-species communities with black bullheads, and 4) multi-species communities including piscivores, where minnows were strongly suppressed. Observed fish community patterns reflected strong gradients of piscivory as well as wetland depth and size. For GS sites, CCA identified two

significant environmental variables ($p < 0.05$): maximum wetland depth and surface area. For PM sites, CCA identified only maximum depth as a significant source of variance. Our results indicated that piscivory is an important mechanism structuring fish communities in these wetlands, but also that smaller, shallower wetlands tended to have relatively simple fish communities and were often dominated by planktivorous species such as fathead minnows.

Because submerged macrophytes and planktonic algae reflect broad ecological properties of wetlands and shallow lakes (Scheffer 1998), we also assessed influences of fish communities and ambient nutrient levels on abundance of submerged macrophytes and algae during 2005. Nutrient levels were generally much higher in the GS wetland sites. Either submerged macrophytes or planktonic algae dominated wetlands in both study regions. As chlorophyll *a* increased from 5 to 50 ppb, submerged macrophytes declined 71-fold (Figure 3). Frequency of algal dominance (chlorophyll *a* > 19ppb) differed between areas, with 31 of 37 wetlands algal-dominated in the GS, compared to 8 of 35 sites in the PM region (Figure 4). Planktivore and macrophyte abundance were not related in either study area. However, benthivore and macrophyte abundance were negatively related in the GS, although no similar relationship was detected in the PM region (Figure 5). Our results indicated that macrophyte abundance was much more strongly influenced by benthivores than by planktivores, but the strength of benthivore influences depended upon ambient nutrient levels in this landscape.

Ducks depend upon quality wetland and shallow lake habitats throughout the PPR. Certain fish communities have the potential to reduce ecological integrity of wetlands, limiting suitability of these areas for breeding and migrating ducks. Wetland managers need tools useful for predicting ecological consequences of practices that increase

connectivity and permanence of wetlands and shallow lakes throughout the PPR. Our results should aid in development of models useful for predicting both fish presence and community types in PPR wetlands, and for assessing potential ecological implications of specific fish assemblages in wetland habitats.

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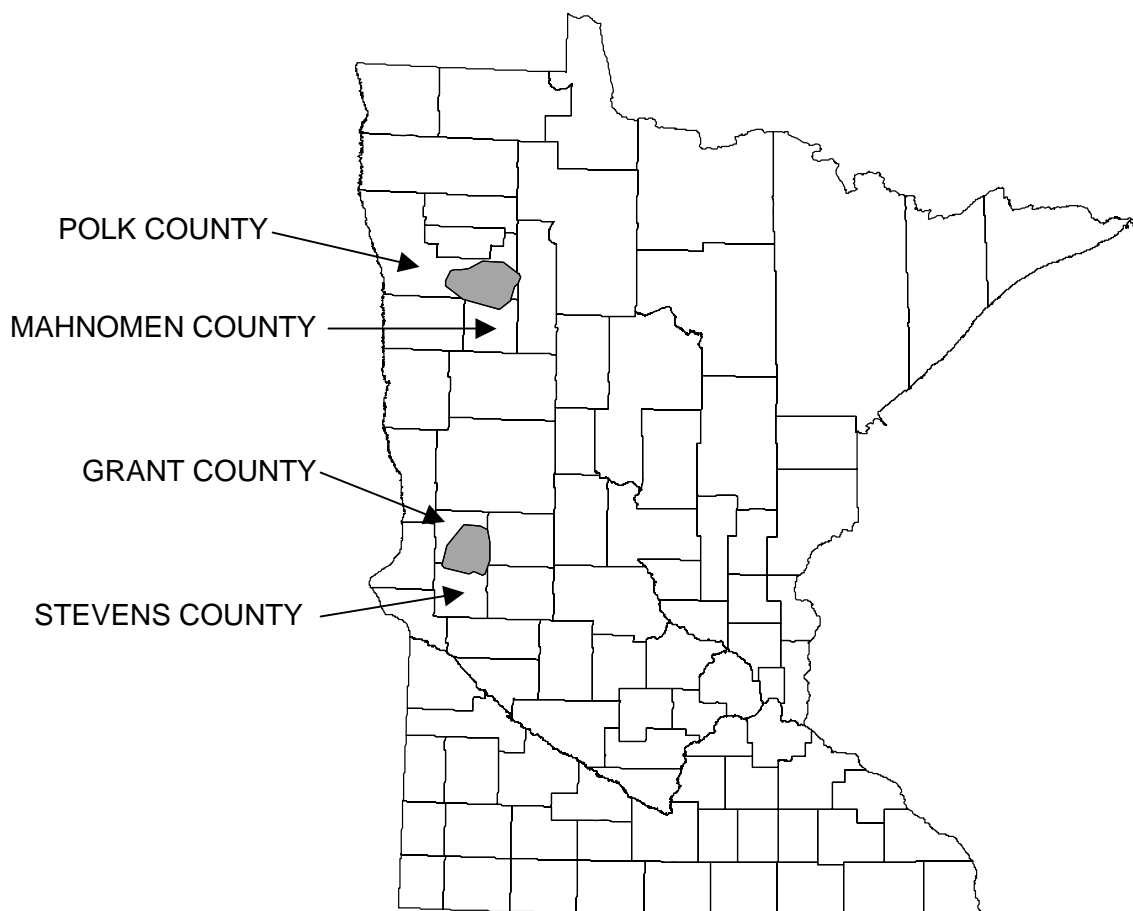


Figure 1. Locations of study focus areas, each defined by a polygon drawn around the outermost 1-mile buffers surrounding each of the study sites.

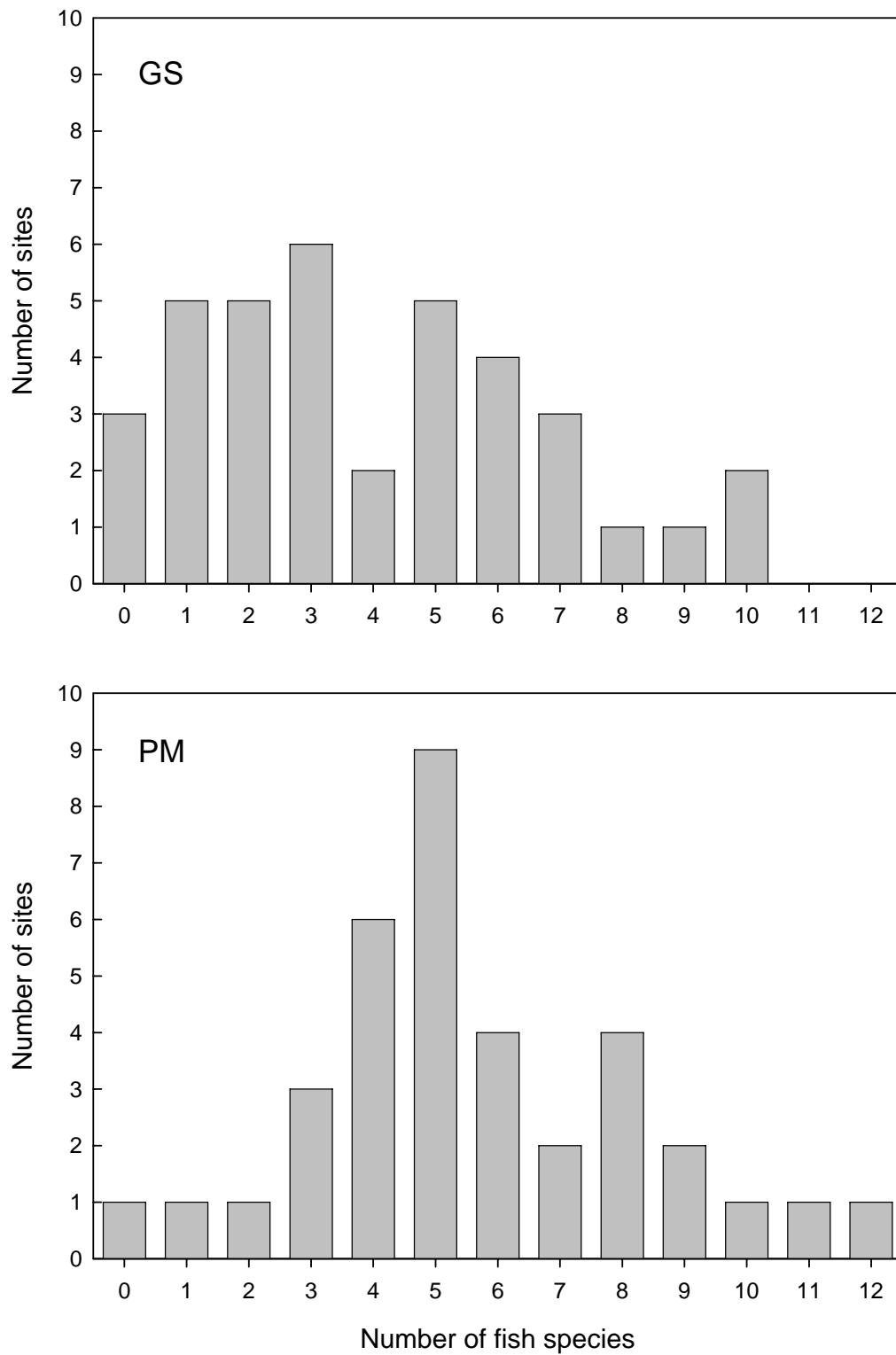


Figure 2. Frequency distribution showing fish species richness across study sites located within the Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas.

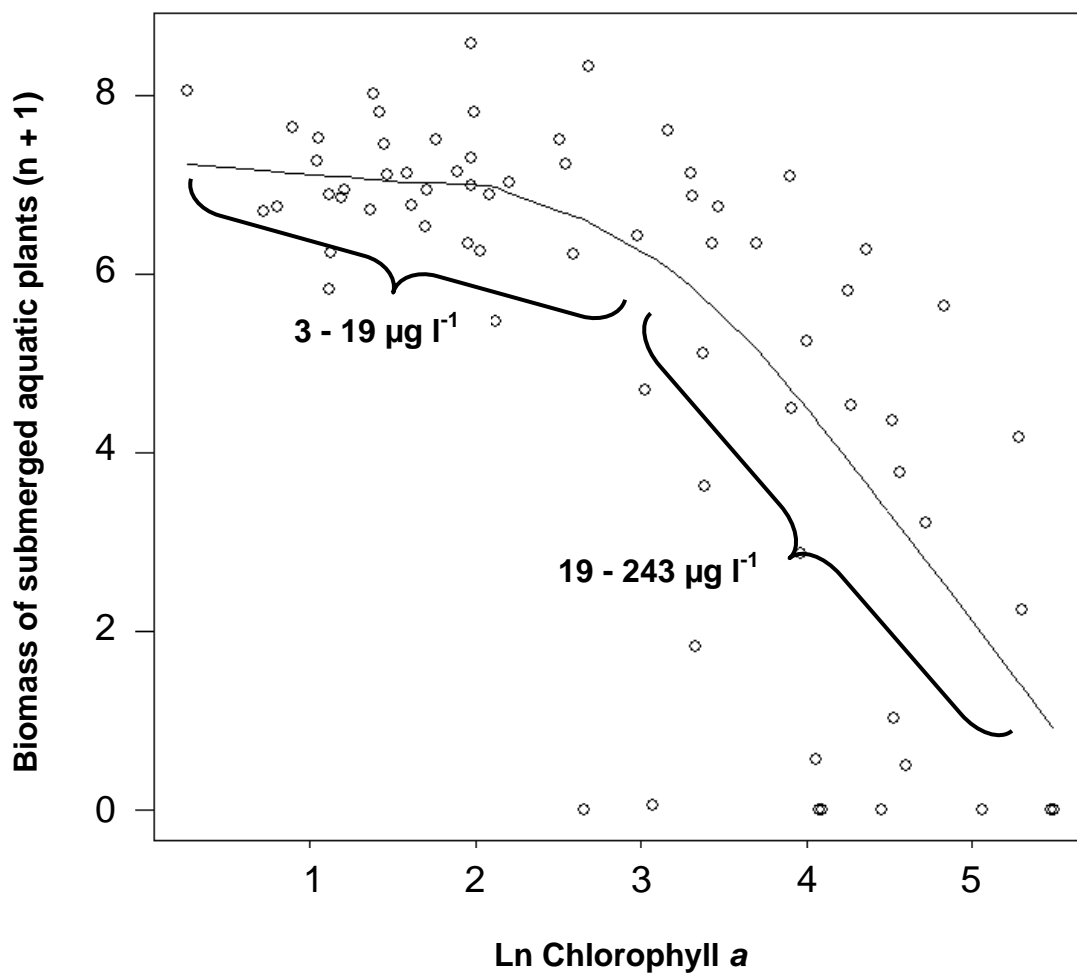


Figure 3. Lowess regression relationship between chlorophyll a (natural log) and biomass of submerged aquatic plants (natural log [n+1]) sampled in wetland study sites during July and August 2005.

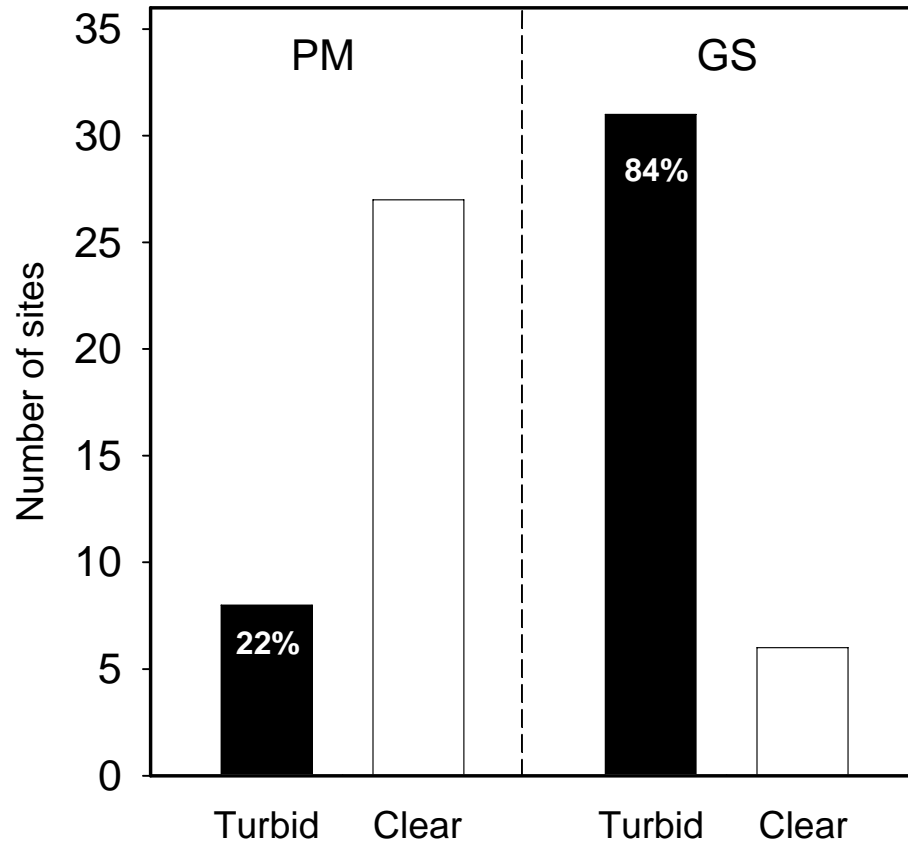


Figure 4. Proportion of turbid vs. clear wetland sites (based on threshold of $19 \mu\text{g l}^{-1}$) sampled in Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas during 2005.

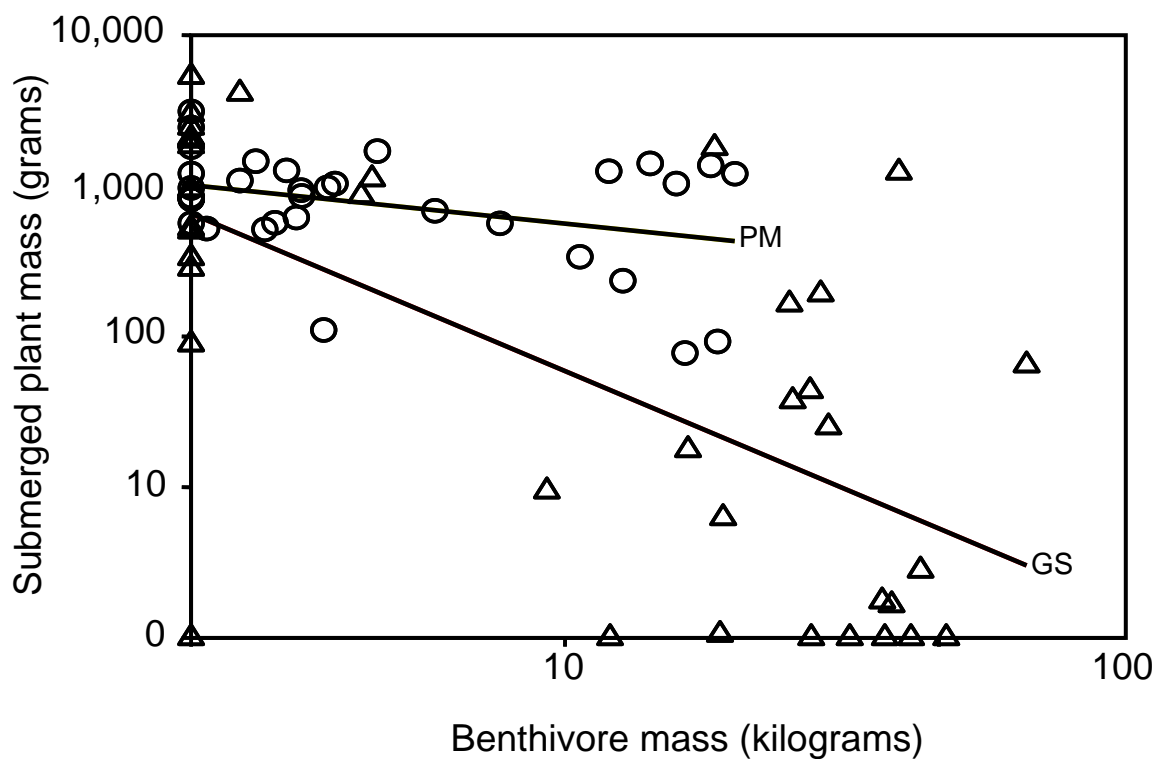


Figure 5. Relationships among submerged aquatic plants and biomass of benthivorous fishes sampled in Polk/Mahnomen (PM) and Grant/Stevens (GS) focus areas during 2005; open circles = PM, Open triangles = GS. Lines indicate relationships fitted separately to data from PM and GS focus areas using ANCOVA. Note that the slope of best-fit lines differed from 0 ($P < 0.01$; $R^2 = 0.43$) for GS sites, but not for PM sites ($P = 0.34$; $R^2 = 0.13$).

SEASONAL FOREST WETLANDS: CHARACTERISTICS AND INFLUENCES

Shane Bowe¹, Mark A. Hanson, Matt Bischof¹, and Rick Koch¹

SUMMARY OF FINDINGS

Seasonal forest wetlands are abundant and broadly distributed throughout aspen-dominated landscapes in Minnesota's Laurentian Mixed Forest. Interest in seasonal wetlands has increased in recent years due to increased awareness of their ecological significance, and because these habitats are often influenced by silviculture activities. It is evident that site-level characteristics of seasonal wetlands are functionally linked to adjacent forested uplands. Forest wetlands receive major energy inputs through deposition of leaf-litter from the adjacent forest (Oertli 1993). Clear-cut timber harvest may have unexpected consequences for adjacent wetlands including modified vegetation and hydrology, increased sedimentation, reduced evapotranspiration, and soil desiccation. It is likely that biotic communities and physical attributes of small wetlands are often altered, but to date, relationships between silvicultural activities and small wetlands are poorly known, and little information is available to guide forest and wildlife managers who are interested in conserving integrity of small riparian areas.

INTRODUCTION

Since 1999, we have studied 24 small, seasonally-flooded (≤ 0.6 ha) wetlands in aspen-dominated landscapes of the Buena Vista and Paul Bunyan State Forests in north central Minnesota. Study wetlands are assigned to one of three "age-class" levels of treatment, or identified as controls based upon adjacent forest (stand) age-since-harvest using natural breaks identified with Arcview. We blocked study sites based on proximity to account for local influences of soils, landforms, or other geophysical features. We assigned study wetlands to clusters,

each comprised of 4 adjacent wetlands (1 in each of 4 treatment groups) located within the same general state forest area. Each state forest (hence subsection of the Ecological Classification System [ECS], Almedinger and Hanson 1998) contained three clusters of four wetlands, including one control, 2 effect/recovery sites, and 1 clearcut treatment site (total of 12 sites per state forest). Control sites were those with no adjacent forest harvesting during the past 59+ years. Treatment sites included one 59+ year area that was harvested during the winter of 2000-2001 (clearcut treatment), and 2 effect/recovery sites consisting of wetlands in stands harvested 10-34 (young-age) and 35-58 (mid-age) years before present. Overall, our design included 6 replicate sites within these 4 age-class treatments. Data gathering and analyses associated with this initial phase of the research are well underway. These analyses will assess in more detail wetland characteristics and potential changes observed during 2001-2005, the initial period following clear-cutting in adjacent uplands (winter 2000/2001). Here, we report on preliminary analyses of invertebrate-community responses, and related environmental changes including leaf litter and duration of ponding (hydroperiod), both attributes likely to be influenced by timber harvest. Our objectives were to: 1) characterize community features and identify site-level environmental characteristics of seasonal wetland habitats in the Laurentian Forest, and 2) evaluate initial responses of aquatic invertebrate communities and other wetland features to clear-cut timber harvest.

METHODS

We sampled aquatic invertebrates using surface-associated activity traps (SAT; Hanson et al. 2000) deployed for 24

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hrs at random locations near the margin of each wetland. Five traps were used concurrently in each wetland. Aquatic macroinvertebrates were sampled during open-water periods, at approximately 3-week intervals during May, June, and July 1999-05. Vertical and horizontal leaf litter traps were implemented at one cluster (4 wetlands) in each forest during 2004-05. Leaf litter was collected every two weeks during September-mid-November. Litter samples were dried at 60°C for 24 hrs, weighed and combusted at 450°C for 4 hrs to determine organic matter content (ash-free dry weight). Site-level measurements of vertical distance to groundwater were obtained using networks of piezometers and monitoring wells (following methods of Sprecher 2000). Single wells were established in the deepest portion of all wetlands during 2004 to assess approximate distance to upper limits of groundwater. Additionally, during 2005, piezometer nests were deployed at 8 wetland sites to more accurately characterize relationships between groundwater movements and wetlands. Wetland maximum depth was recorded weekly from spring thaw until surface water disappeared, and every two weeks thereafter until frozen. Additional measurements were made at these wetland sites during 1999-2005 (Ossman 2001).

Invertebrate data were analyzed to identify potential patterns using Nonmetric Multidimensional Scaling (NMS). We used NMS to compare invertebrate community structure by ordination of site scores based on a dissimilarity matrix. Significance of patterns in our site scores were further assessed using Multi-response Permutation Procedures (MRPP; McCune and Grace 2002). Leaf litter data were assessed graphically and using independent samples t-tests (Green and Salkind 2005). Hydroperiod data (days of continuous inundation) were compared among treatments graphically and using ANOVA (Green and Salkind 2005). Results presented here are

preliminary; interpretations are likely to change following additional data analysis.

RESULTS AND DISCUSSION

Invertebrate community composition showed a notable shift during the first 5 years following clear-cut timber harvests (Winter 2000-01). For example, during 2005, invertebrate communities from clear-cut sites exhibited higher within-group similarity than did non-clear-cut sites (Figure 1). Clear-cut invertebrate communities comprised a distinct group that was clustered based on dissimilarity with uncut wetland sites. Similar contrasts between invertebrate communities of clear-cut and other treatment sites also were observed during 2001-2004. Our results reflect patterns of change in wetland invertebrate communities, apparently in response to clear-cut timber harvest in adjacent uplands. Comparison of NMS site scores using MRPP indicated that dissimilarity between invertebrate communities of clear-cut wetlands and other treatment groups was greater than expected by chance ($T = -1.8$; $P < 0.05$). This is not surprising given the widely held view that biological processes and communities in small, seasonal wetlands are functionally linked to adjacent upland areas (Palik et al. 2001).

Clear-cut harvesting modifies wetland hydroperiods (Verry 1997, Roy et al. 2000), leaf-litter inputs, light availability at the wetland surface, and water temperature, among other things. We observed obvious differences in wetland hydroperiods among our forest-age treatments; clear-cut wetlands maintained standing water longer than did all other groups ($F_{(3,20)} = 3.14$; $P < 0.05$). During 2004, on average, study wetlands embedded in clear-cut harvests remained flooded approximately 45 days longer than did sites in old-growth aspen stands (Figure 2). Following adjacent clear-cut harvest, litter inputs to our wetland sites diminished ($T = 3.02$; $P < 0.05$; Figure 3), concurrent with sharp decreases in canopy closure.

Observed patterns in invertebrates are consistent with Church (2006) who also reported changes in similar communities following clear-cutting adjacent to seasonal ponds in north central Minnesota. Changes in invertebrates probably reflect cumulative influences of shifts in site-level environmental characteristics during periods immediately following clear-cutting. Oertli (1993) suggested that leaf litter constitutes the major source of energy for macroinvertebrate production in small wetlands, thus reductions in leaf litter inputs to our sites are likely associated with observed changes in invertebrates. Batzer et al. (2004) reported weak associations between wetland invertebrate communities and hydroperiods in seasonal ponds in north central Minnesota. Relationships between hydrology of small seasonal wetlands and clear-cut timber harvest are poorly understood. Some previous research indicates that tree removal has the potential to elevate water tables (Verry 1997, Roy et al. 2000) and modify local hydrology (Roy et al. 2000). Other unanticipated ecological responses to timber harvest are also possible. For example, extending hydroperiods of small forest wetlands may allow vertebrate and invertebrate predators to persist and disrupt natural community dynamics. Hence, other animals including amphibians and early arriving birds and waterfowl, may face added competition for food resources before larger water bodies become ice-free. We expect that subsequent data and analyses will provide better characterization of these wetlands and help clarify specific relationships between wetland communities and clearcut timber harvest.

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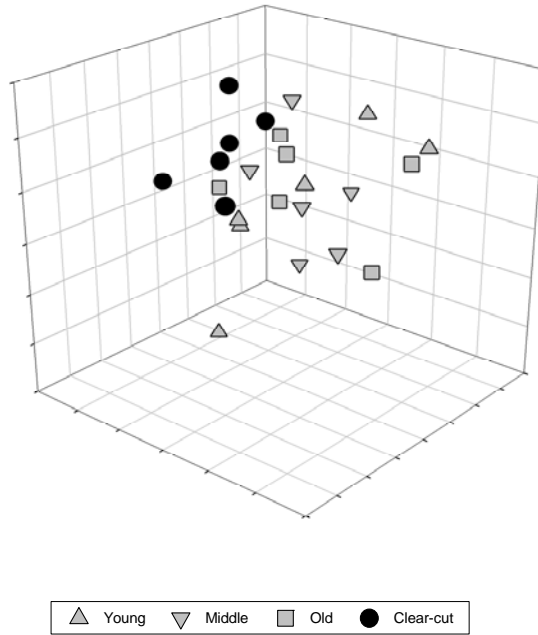


Figure 1. A three-dimensional NMS ordination of site scores based on dissimilarity in invertebrate communities among wetland study sites during 2005. Distances between plotted site scores illustrate extent of dissimilarity in invertebrate species composition. Symbols represent age-structure characteristics of adjacent uplands.

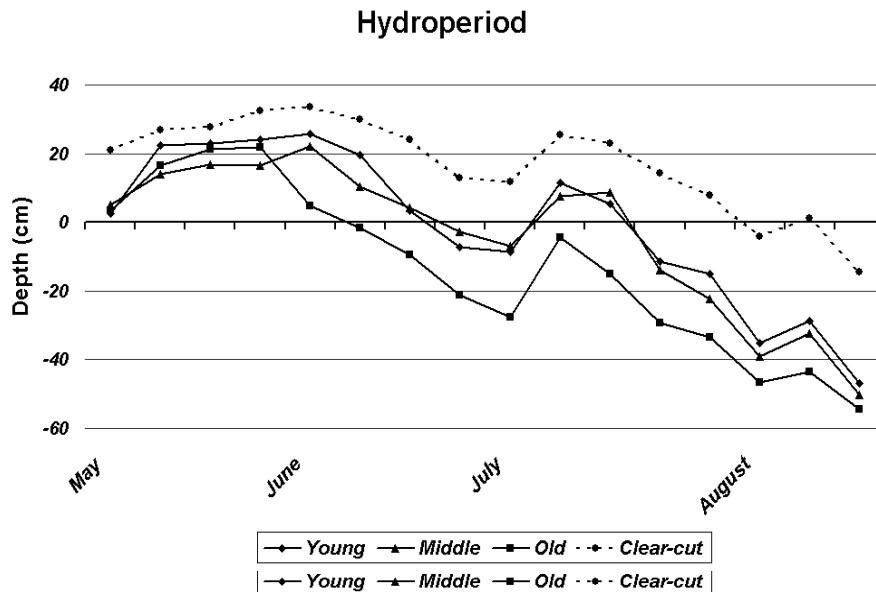


Figure 2. Average maximum depth (standing waters) and distance to groundwater for 24 wetland sites during 2004. A value of 0 cm indicates lack of standing water within the deepest portion of the wetland basin; negative values reflect approximate distance to upper limits of groundwater.

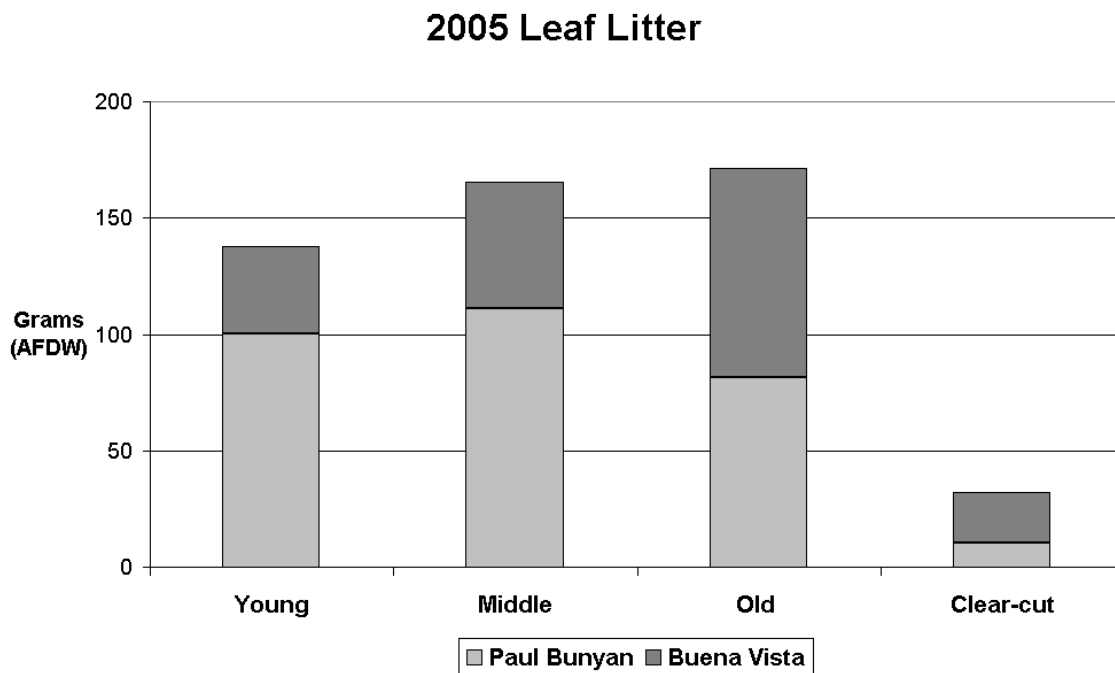


Figure 3. Total grams (Ash Free Dry Weight) of leaf litter collected from 4 sites in the Paul Bunyan State Forest and 4 sites in the Buena Vista State Forest during 2005.