HABITAT USE AND HOME RANGE OF AMERICAN BITTERNS (*BOTAURUS LENTIGINOSUS*) AND MONITORING OF INCONSPICUOUS MARSH BIRDS IN NORTHWEST MINNESOTA.

A Dissertation

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Of the Requirements for the Degree

Doctor of Philosophy

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by

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The undersigned, appointed by the dean of the Graduate School, have examined the or dissertation entitled

HABITAT USE AND HOME RANGE OF AMERICAN BITTERNS

(BOTAURUS LENTIGINOSUS) AND MONITORING OF

INCONSPICUOUS MARSH BIRDS IN NORTHWEST MINNESOTA

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and hereby certify that, in their opinion, it is worthy of acceptance.

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I dedicate this work to my parents, Chhoeurk Mak, who sacrificed everything for so many years, and to my father, Pao Lor, who sacrificed his life, so that their children can have better lives.

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TABLE OF CONTENTS

TABLE OF CONTENTS (continued)

TABLE OF CONTENTS (continued)

LIST OF TABLES

CHAPTER 1

LIST OF TABLES (continued)

CHAPTER 2

CHAPTER 3

LIST OF TABLES (continued)

LIST OF FIGURES

CHAPTER 3

HABITAT USE AND HOME RANGE OF AMERICAN BITTERNS (*BOTAURUS LENTIGINOSUS*) AND MONITORING OF INCONSPICUOUS MARSH BIRDS IN NORTHWEST MINNESOTA.

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ABSTRACT

Information on habitat use of the American Bittern (*Botaurus lentiginosus*) and a statistically valid survey design for monitoring changes in populations of inconspicuous marsh birds, which include American and Least Bitterns (*Ixobrychus exilis*), Pied-billed Grebes (*Podilymbus podiceps*), Soras (*Porzana carolina*), and Virginia Rails (*Rallus limicola*)) is needed to inform conservation and management actions. My research, from 1999 – 2002, examined breeding habitat use and home range of American Bitterns. Also, I used pilot survey data to guide design options to meet objectives for monitoring marsh bird occupancy rates in association with habitat changes. Nest sites of American Bitterns in wetlands (n = 47) and grasslands (n = 33) were positively associated with percent dead vegetation cover and density and negatively associated with vegetation height. Foraging sites of American Bitterns were negatively associated with distance to small water openings and vegetation height. Daily survival rate was 0.96 (95% CI 0.930 – 0.979) and nest survival rate of American Bitterns was 0.35 (95% CI = 0.15 – 0.58). The average core home range size (50%) was 18.08 ha (± 6.38) and the 95% home range was 109.28 ha (± 38.47) using the fixed-kernel estimator. Results from occupancy analyses of pilot data and evaluation of a set of *a priori* candidate models provide the needed guidance for reliable marsh bird monitoring programs.

CHAPTER 1

HABITAT USE AND NEST SURVIVAL OF AMERICAN BITTERNS (*BOTAURUS LENTIGINOSUS*) IN NORTHWESTERN MINNESOTA

ABSTRACT

The American Bittern (*Botaurus lentiginosus*) is a large, inconspicuous marsh bird that breeds from the mid- and northern United Unites to the northern provinces of Canada. Information on life history strategies, breeding biology, and habitat use of the American bittern is scarce, but provides important baseline data for more in-depth studies. The objectives of this study are to: 1) assess nest survival of American bitterns nesting in wetland habitats and compare with grassland nesting bitterns, 2) assess habitat use of wetland habitat and grassland nesting sites of American Bitterns. Between 1999 and 2002, 47 nests of American Bitterns were located and habitats at the nest site and at random sites (n = 105) were measured to construct predictive models of nest sites. Daily nest survival for American bittern nest was 0.96 (95% CI 0.930 – 0.979) and the conditional interval nest survival rate was 0.35 (95% CI = 0.152 – 0.576). In wetland habitats, the odds of encountering a nest increased with higher percentage of dead vegetation and vegetation density. In grassland habitats nest site selection decreased almost 35% with every 10 cm increase in vegetation height. Vegetation species and habitat types may not be critical factors compared to vegetation structure (e.g., height and density).

INTRODUCTION

The American Bittern (*Botaurus lentiginosus*) is a large, inconspicuous marsh bird that breeds throughout mid- and northern United States to the northern provinces of Canada. Like many marsh and wetland bird species, the American Bittern is being affected by the loss and degradation of wetlands and associated grassland habitats (Gibbs et al. 1992, Dechant et al. 1999). From 1966-2004, the breeding population was estimated to have declined 1.36% per year ($p \leq 0.1$) in the United States and Canada and 6.23% ($P \leq 0.10$) per year in the midwestern region of the United States (USGS 2006). The USFWS (2002) designated the American bittern as a species of "Resource Conservation Priority" throughout its midwestern region (Region 3). This concern was again emphasized in 2005, when the species was ranked as "highly imperiled" in the Upper Mississippi Valley/Great Lakes Waterbird Conservation Plan (USFWS 2005a).

Numerous ornithologists and biometricians have devoted considerable time to studying and quantifying nest success and factors that affect nest success, as summarized by Dinsmore et al. (2002), Hazler (2004), Shaffer (2004), and most recently by Jones and Geupel (2007). Mayfield's nest success rate (Mayfield 1961, 1975) is commonly reported as estimates of nest survival. However, this method has been contentious because of the assumption that daily nest survival is constant in time and that the date of a hatch or loss is known exactly. Recently, a number of alternative methods have been developed to estimate unbiased nest survival rates, and also relating nest survival rates to factors (e.g, habitat features, climatic conditions, etc.) that may contribute to those rates (Dinsmore et al. 2002, Hazler 2004, and Shaffer 2004).

Large wetlands and grasslands that are indicative of bittern habitats are limited primarily to lands owned by federal, state, and non-profit organizations (Azure 1998, Laney 2003). Consequently, it is crucial that biologists acquire sufficient and accurate information to properly conserve and manage the remaining habitats for not only bitterns but also for other marsh bird species (Hand et al. 1989, USFWS 2005a). However, the low densities and elusive nature of the American Bittern makes it difficult to obtain more detailed information on the habitat features most important to the species. The objectives of this study are to: 1) assess nest survival of American bitterns nesting in wetland habitats, 2) compare wetland habitat characteristics with those of grassland nesting habitats of bitterns. Specifically, my hypotheses for nest-habitat relationships are:

1) Habitat characteristics, particularly vegetation cover, water depth, and vegetation density vary among the annual life history events of American bitterns. Predictions:

A) Based on a previous model and previous observations, American bitterns that occupy wetland habitats select dense stands of emergent vegetation, greater cover:water ratio, and shallow water levels (<20 cm).

B) Based on previous observations, American bitterns nesting in upland habitats select sites that are associated with dense and tall grass (100% visual obstruction readings of >50 cm).

STUDY SITES

This study was conducted in northwestern Minnesota, at the Agassiz NWR, located in Marshall County, and Red Lake Indian Reservation, located in Clearwater county. Agassiz NWR is a part of a large complex of land specifically managed for migratory birds by the USFWS and the Minnesota Department f Natural Resources (MNDNR). The refuge comprises 24,888 ha of which 15,135 ha are impounded wetland habitats, including permanent, semi-permanent, sedge meadows, and raised bog; 4006 ha aspen woodland, 4715 shrubland (primarily *Salix* spp.), and 761 ha grassland habitats. There are 26 impoundments on the refuge; each has a control structures used to manipulate water levels. These impoundments range in size from approximately 75 – 4000 ha (USFWS Agassiz NWR 2005).   Immediately adjacent to Agassiz NWR on the southern, eastern, and northern boundaries are the Elm Lake (~6356 ha), Eckvoll (~2632 ha), and Thief Lake (~22,267 ha) Wildlife Management Areas (WMAs), which are composed mainly of permanent and semi-permanent wetland habitats.

Prior to refuge acquisition in 1937, many attempts were made to convert the muck land into arable agricultural fields by developing a series of drainage ditches. However, these drainage ditches did not sufficiently drain the land because of vegetation growth in the bottom of the ditches (USFWS 2005). Climatic conditions at the refuge are typical of the region, where variations in temperatures are wide and extreme, having severe winters with moderate snowfall and annual precipitation of approximately 56 cm. Because of the variation in temperatures and rapid snowmelt in the spring combined with the flat terrain, flooding occurs frequently and is often detrimental to the farming efforts in the area (USFWS 1978).

The small lakes (Mud, Kuriko, Webster, Whiskey, and Elm Lakes) located within Agassiz NWR were formed in depressions in the bed of Glacial Lake Agassiz that were not entirely filled with sediment (Minnesota Conservation Department 1968). A landscape feature on the refuge that reveals the substratum underlying the refuge is a sandy beach ridge located along the northwestern portion of Agassiz Pool. Agassiz Pool surrounds Mud Lake and is the largest (ca. 4000 ha) impounded lake on the refuge. Wave-action from glacial activity deposited the sand which formed the beach ridges that lie within Agassiz NWR. These geologic formations and anthropomorphic alterations of the landscape shaped a complex of wetlands, raised bog, and grassland that is now protected and managed for wildlife habitats.

Red Lake Indian Reservation contains 1031 ha of wild rice paddies and upland areas that are managed as dense nesting cover for ducks. Grassland habitats are managed for waterfowl nesting habitats. These fields, ranging in size from 8 – 51 ha, were planted with a combination of alfalfa and smooth brome or timothy, red top, red clover, and alsike clover and are hayed and burned on a rotational basis (J. Huseby, Red Lake Band Dept. of Natural Resources, pers. com.). The reservation is owned by the Red Lake Band of the Chippewa Indians and is located approximately 40 km southeast of Agassiz NWR.

Both study areas lay on the southeastern portion of the prehistoric Glacial Lake Agassiz, within the geomorphic region classified as the Agassiz Lacustrine Plain, Red River Valley area (University of Minnesota 1980a) and Agassiz Lacustrine Plain, Red Lake area (University of Minnesota 1980b). Quaternary geologic sediment at Agassiz NWR is Holocene peat, and at Red Lake Indian Reservation is lake-modified till (Hobbs and Goebel 1982). The landscape position in which Agassiz NWR is located is characterized as shallow depressions on lake plain, and the soil series is mainly Cathro Haug, which consists of organic soils. The rooting zone (0.3 – 1.5 m) consists of muck and peat, and the substratum (1.5 - >7 m) is loam and sandy loam (University of Minnesota 1980a).

The study area at Red Lake Indian Reservation is comprised of the Roliss and Rockwell soil series, which are poorly drained, mildly alkaline loam and sandy clay loam soils, respectively. The landscape is level to depressed lake plain and level lake plain, with clay loam, sandy loam, loamy fine sand, and loam in the rooting zone (0.6 – 1.5 m), and loam and clay loam in the substratum (>1.5 m) (University of Minnesota 1980b).

Climate of northwestern MN is characterized by broad range of temperatures with late springs, early fall frosts, and 115 frost-free days (USFWS 2002). Annual average precipitation over the past 30 years was 56 cm and during the duration of the study period (1999-2002) was 57.35 cm. Winter is relatively dry at both study sites, with average snowfall of approximately 99 cm. The wettest months are June, July, and August. During 2002, almost 24 cm of rain fell within 9-11 June, raising water levels in wetland units of greater than one meter in some pools, causing major flooding in the area, including nesting habitats.

METHODS

Nest Search

All areas of potential nesting sites where American Bitterns were detected were searched for nests. Several ways of recording potential nesting sites included calls or pumps detected during morning surveys using call – broadcasting techniques (Chapter 3, Conway 2005), from flushing events while traveling in wetlands on airboats, or from observing bitterns land in potential nesting site. Only portions of wetlands were searched because of the expansive size of the entire wetland area and also because of the large size of each wetland unit. Walking searches were conducted with 1 – 5 people walking in transects, arms width apart, through an area of up to 200 m^2 or until emergent vegetation ended (i.e., to edge of water and/shrubs or trees). Airboats were used in situations where vegetation was too thick to efficiently walk through or when the water depth and/or substrate prevented walking. An airboat team typically consisted of 2-3 people on the airboat, depending on whether we used a 2- or 3- person airboat. Sometimes, two airboats were used simultaneously. The size of the area searched depended on the habitat type; in areas where emergent vegetation was expansive, an area of approximately 200-m^2 was searched, 100-m from the center of where birds were either flushed or were heard calling. Only emergent vegetation and sedge meadows were searched.

Nests in grassland habitat were discovered during a nest-dragging study for duck nesting activities using the chain-drag method with two ATVs dragging a 150-m chain link (Klett et al. 1986, Armour 2002).

Nests were marked with fluorescent flagging tied to a dead willow stick placed approximately 4-m north of the nest. The number of eggs was recorded and eggs were floated using methods of Hays and LeCroy (1971) to estimate hatching date. Dimensions of nests and nesting material content were measured and recorded along with habitat characteristics around the nest. Nests were revisited every 1 - 7 days to determine nest fate. Because American Bittern chicks are highly active at ages >7 days, nests were only monitored until at least one egg hatched, which is a measurement of successful nest (Klett et al. 1986, Brininger 1996). Unsuccessful nests included those that were depredated, abandoned or damaged by researchers.

Habitat Measurements

Wetland Habitat

Macro- and micro-habitat variables were measured at each nest site and random sites. The scale at which variables were measured from the nest were under the assumption that birds select habitats on a hierarchical level based on macro-habitat features and then micro-habitat features. Variables at nest sites were measured immediately after the nest was discovered and re-measured at the time of random site measurements, between 26 June – 13 Aug. Random sites were selected from a grid of 100-m squares, overlaid over each wetland unit in which bittern nests were found, using Geographic Information System (GIS) covertypes in ArcView 3.1 (ESRI, Redlands, CA). Universal Transverse Mercator (UTM) coordinates of the center of each randomly selected point were obtained from the GIS and used for navigation to the point of random site. In the event that the random site was not accessible or was not in wetland habitat (i.e., island), I traveled to the next closest random site. Macro-habitat variables included distances to nearest land, small water opening (<0.4 ha), large water opening (>0.4 ha), distance to nearest dominant standing cover change, ditch, edge, and land. These variables were verified by examining the refuge's habitat GIS covertypes in ArcView 3.1 because in some instances, the features were difficult to discern in the field.

At the micro-habitat scale, vegetation composition was estimated in a 10-m radius circle from the nest or random site, dominant vegetation species, relative density of the vegetation, percent live and percent dead of the dominant vegetation, percent open water, percent emergent vegetation, and percent shrub/scrub vegetation. Further, within the 4-m radius circle, vertical and horizontal vegetation structures were measured at four axis points, located at the four cardinal directions.

Measurements of vertical and horizontal habitat structures were adopted from Hays et al. (1981). Vegetation structure was measured by using a Vegetation Profile Board (Hays et. al 1981). The dimension of the board was 0.3 m x 2.0 m, constructed from 0.95 cm (3/8 inch) plywood, and 5 x 5 cm black and white checkers were painted on one side to facilitate estimations of percent coverage of the board by vegetation. Height intervals (at 10, 20, 30, 50, 70, 100, 150 cm) were marked on the board with bright red paint. The board was held upright at the nest or random site, flushed with the ground or at the water surface. An observer stands 4-m from the board, at one of the four cardinal directions, while facing the board, the observer estimated by ocular estimation, the extent (in percentage) at which the board was obscured by vegetation at each marked height interval. Water depth (cm) and vegetation height (cm) were also recorded at each of the four cardinal points.

Grassland Habitat

Thirty-three nest sites and 53 random sites were measured in grassland habitats using the same method as those in wetland habitats except for the addition of obtaining Robel pole readings (Robel et al. 1970) in grassland habitats, where 100% obstruction height-density readings were recorded. Vegetation covertypes were not available on GIS, so aerial photos were used to overlay 100-m grids over each field where nests were discovered to select random sites. Habitat variables at nest and random sites were measured within one week of nest discovery, with exception of two nests that were measured three weeks after discovery because of logistical difficulties. Nest and random sites were measured on the same day.

Data Analysis

Nest Survival

I used the logistic-exposure approach (Shaffer 2004) to model nest survival as a function of nest-specific predictor variables and to estimate daily nest survival rates. This approach assumes survival and predictor variables to be constant within a nest-observation interval, but does not assume constant daily survival and does not require exact date of loss or hatch as in Mayfield's method. It accommodates varying exposure periods, continuous, categorical, and time-specific predictor variables. A modified logit link function ($\log_e(\theta^{1/t})/[1 - \theta^{1/t}]$), where θ is the interval survival rate and t is the interval length in days (Shaffer 2004, Knutson et al. 2007). Candidate models that were constructed after data collection but prior to data analysis were fitted using the SAS generalized linear modeling procedure (PROC GENMOD; SAS Institute 2004). I evaluated model fit by examining the Hosmer and Lemeshow (2000) goodness-of-fit for the global model. Candidate models included one with edge-effect, which contained the predictor variables distance to small water opening (SMALL) and distance to edge (EDGE); temporal model which included Julian dates (I did not include year because it was highly correlated with Julian dates); and a nest site effect that included vegetation height (VEGHT), percent live vegetation (LIVE), horizontal cover at 0.5 m. above the nest (HC5).

Estimates for daily nest survival rates were calculated using the model with the smallest AIC_c value. Each interval between visits to a nest was treated as one observation in the analysis. The predicted probabilities represent the probability of nest surviving 1 day and are comparable to Mayfield daily nest success rates (Mayfield 1961, 1975). These predicted probabilities were conditional on the mean of the covariate values that were used in the models (Grant et al. 2005, Knutson et al. 2007). In addition, I estimated conditional interval nest success (percentage surviving the incubation period = 26 days for American Bittern) using the most-supported models, assuming constant survival during incubation period. This survival estimate is similar to Mayfield (1961) estimates.

Wetland Habitat

In the model development process, I first developed *a priori* hypotheses for nesting habitats; I predicted that nesting sites of American Bitterns were: A) In wetland habitats, American Bitterns select dense stands of emergent vegetation, greater cover:water ratio, and shallow water levels (<20 cm). In the model construction process, 47 nest sites and 105 random sites were used. A global model was constructed from "biological important" variables that were associated with the hypothesis. Several predictor variables were duplicates, such as percent open water and percent emergent vegetation because they were complimentary of each other (i.e., 30% open water:70% emergent vegetation cover). In this case, I excluded one (open water) from the models. Other candidate models were subsets of the global model, which represent various aspects of American Bittern nesting habitat features that correspond to the methods of data collection: 1) habitat features within 4-m of nest; 2) habitat features within 10-m of the nest; 3) Edge "effect", which includes measures of distance to nearest edge, ditch, and land; 4) Macro model, which includes variables distance of nest/available site to large and small water openings. I further screened the variables using paired *t*-test and retained variables that were significant at $P < 0.25$ (Hosmer and Lemeshow 2000). I plotted the standardized deviance residuals for the global model against the explanatory values and found no outliers (all values <3) or patterns that required transformation. I then conducted multi-collinearity analysis to check for highly related variables using PROC REG on the predictor variables in the global model and excluded highly related (TOL < 0.40) variables from further analysis (Allison 1999). Tolerances (TOL) measures the strength of inter- relationships among the explanatory variables in the model. Tolerance is $1 - R^2$ for the R^2 that results from the regression of the explanatory variable on the other explanatory variables in the model. If a variable is closely related to other variables, the tolerance goes to 0 and the variance inflation becomes large (SAS Institute Inc. 2004). The 11 variables that were retained for analysis included: DEAD = % dead vegetation of the dominant species; LARGE = distance to large water openings (>0.4 ha); SMALL = distance to small (<0.4 ha) water opening; LAND = distance to nearest land; EDGE = distance to vegetation edge; DITCH = distance to ditch; HT = average vegetation height within 4 -m of nest sites; HC5 = horizontal cover on the density board below 0.5 m, average of readings from four cardinal directions ; AWD = average water depth within 4 m of the nest site; DEN = relative density of dominant vegetation within 10-m of nest, classified into four categories (0 = no vegetation, 1 = rank [water not visible through base of stems at water level and one cannot easily push hands through the stems], 2 =

moderate [anything that falls between category 1 and 3], 3 = sparse [water easily visible through base of widely scattered stems]); DOM = dominant vegetation within 10-m of nest. Overdispersion occurs when the sampling variance exceeds the theoretical or model-based variance and is caused by the positive correlation between the binary responses or variation between the response probabilities (SAS Institute, Inc. 1995, Burnham and Anderson 2002). Overdispersion is common in most real data and causes underestimation of the variance of the parameter estimates (SAS Institute, Inc. 1995). Overdispersion can be modeled with variance inflation factor ($\hat{c}$), which can be estimated from the goodness-of-fit chi-square statistic (χ^2) of the global model and its degrees of freedom ((Burnham and Anderson 2002). Model structure is acceptable when $1 \leq \hat{c} \leq 4$, but inadequate when $\hat{c} > 6$ (Burnham and Anderson 2002:68). I checked for overdispersion in the global model by using the AGGREGATE and SCALE options in PROC LOGISTIC to obtain an estimate for variance inflation factor $\hat{c}$.

I used the information theoretic approach (Burnham and Anderson 2002) to fit models and evaluate support for each of the models. All analyses were performed using PROC LOGISTIC in SAS (SAS Institute 2002-2003) unless otherwise noted. Interaction terms were not included because of the small sample size of nest sites (B. Gray, pers. comm.). Because of the disparity between the number of nest sites (47) and number of random sites (105), an OFFSET option for the parameter estimates was specified as a correction factor in the SAS procedure. I evaluated the models by using Akaike's Information Criterion (AIC), modified for small sample size (AIC_c; Burnham and Anderson 2002). I used AIC_c to rank models from most to least supported given the data. I calculated ΔAIC_c (the difference between lowest observed AIC_c value and value for the next model) and Akaike weight of evidence (w_i, a measure of model support based on ΔAIC_c that sums to 1 across all candidate models) as measures of model support. In the event that the model fits the data and model predictive power (>60% concordance) was adequate, odds ratios and 95% confidence intervals were calculated. Confidence intervals for parameter estimates were based on profile likelihood function rather than the asymptotic normality because they provide more accurate estimates for small sample sizes (SAS Institute 2004; Burnham and Anderson 2002). In the event of model-selection uncertainty (w_i <0.90), I calculated odds ratios based on model-averaged coefficients and 95% confidence intervals (CI) based on unconditional standard errors (Burnham and Anderson 2002). Only odds ratios with CIs that did not include 1 were interpreted.

Grassland Habitat

For model construction 33 nest sites and 53 available sites were used. The same analysis approach was used as the wetland nest habitats. Two variables (average water depth and Robel pole readings) were removed from all stages of the analysis because only seven observations had measurable water depth and Robel readings were duplicates of the density cover board. Density category level (1, 2, 3) were collapsed to alleviate model convergence problems; only one observation (random site) had a category 3 (sparse), thus it was collapsed into a category 2 (moderate). Dominant vegetation species also caused model convergence problems because seven nest site observations were in the collapsed category level called “miscellaneous” grass species (timothy, quack grass, red top) and there were no observations for random sites in this category. I did not feel comfortable collapsing levels further because it was not biological meaningful, thus, I excluded DOM from the models. Instead, I report simple statistics for this covariate.

RESULTS

Nest Survival

Hosmer and Lemeshow goodness-of-fit test on the global model indicated good model fit (χ^2 = 2.761, df = 7, P = 0.906). I examined plots of standardized deviance residuals for the global model plotted against the explanatory values and found no large values that suggested outliers or patterns required transformations. I examined tolerance values to diagnose for multi-collinearity among variables in the global model by using PROC REG (SAS Institute Inc. 2004). The diagnostic suggests that several variables (Julian date, vegetation height, percent live vegetation, and horizontal cover) were closely correlated to each other (TOL = 0.365 – 0.489). However, the variables distance to small water open and distance to edge had TOL $\geq$ 0.70. Given results of this diagnostic, I assessed for possible overdispersion in the data by computing $\hat{c}$ from the chi-squared goodness-of-fit test of the global model. Because $\hat{c}$ = 2.90 for the global model, suggesting some overdispersion exist in the data, I used the small sample, corrected for overdispersion variant of the Akaike information criterion ($QAIC_c$) and Akaike weights, w_i (Burnham and Anderson 2002) to evaluate the candidate models. To guard against model uncertainty, I calculated odds ratios based on model-averaged coefficients and 95% confidence intervals (CI) based on unconditional standard errors (Burnham and Anderson 2002).

Model selection indicated that the null model received the most support and the edge effect model and nest-site model received adequate support (Table 1). Daily nest survival rate estimated from the null model was 0.962 (95% CI: 0.930 – 0.979). Daily nest survival from the temporal model (Julian date) was 0.960 (0.928 – 0.979). Daily nest survival rate estimated from the edge effect model was similar, 0.963 (0.931 – 0.981) and also for the nest site model, 0.961 (0.927 – 0.980). Daily survival rates for all candidate models were similar (0.96), and the conditional interval nest survival rate was 0.35 (95% CI = 0.152 - 0.576) for all three models.

Wetland Nest Habitats

At Agassiz NWR, 47 American Bittern nests were located between 1999 – 2002. I initially removed five variables (interspersion ratio, percent total cover, percent open water, edge type, distance to nearest cover change) because they were complimentary to other variables (i.e., % open water and % emergent and % total cover measured same things). All combinations of the paired *t*-test were significant ($P = <0.25$). Multi-collinearity check showed several variables (horizontal cover at 0.1, 0.2, 0.7, 1.0, 1.5 m, % shrub cover, % dead vegetation) were highly related (TOL <0.40) and thus they were excluded from further analysis. After diagnostic evaluations, 11 variables were retained for further analysis. Further problems were encountered with two categorical variables, dominant vegetation species and vegetation density class (1, 2, 3) within 10-m of the nest, which caused quasi-complete separation of the data. The problem was due to the unbalanced number of observations and lack of observations in one or both of the response variables (e.g., there were 0 observations of the stinging nettle under nest sites, but three observations in random sites). In an attempt to alleviate the problem, I collapsed vegetation with similar structures (e.g., softstem bulrush and hardstem bulrush = bulrush, and stinging nettle and impatiens into a category called "herbs"). This exercise reduced the number of levels in dominant species from 11 to 5. For the four category levels in density (DEN), there were no nest sites with level 0 (no vegetation) nor in level 3 (sparse vegetation), so eight observations of random sites were collapsed into level 2 (moderate density). This exercise alleviated the problem.

The global model, which includes all covariates in the submodels, show adequate model fit (Hosmer and Lemeshow $\chi^2 = 5.53$, df = 8, $p = 0.70$), and the estimated overdispersion parameter ($\hat{c} = 0.832$) suggests no overdispersion. The weight w_i of evidence shows clear support for the Nest10 model ($w_i = 0.986$). The model correctly classified 73% of the observations and had a high predictive power (81.6% concordance; Table 2). All other models were >7 ΔAIC_c from the best approximating model (Table 2). The model suggests that the odds of nest site selection increased with increase in percent dead vegetation, decreased in moderate vegetation density relative to dense or "rank" vegetation density, and decreased in *Carex* stand relative to *Phragmites* stands (Table 3). Percent dead vegetation at nest sites was indeed almost twice as high compared to random sites (Table 4). A higher proportion of random sites %26%) were in *Carex* stands compared to nest sites (6%), and a higher proportion of nests (72%) were located in *Typha* spp. stands compared to random sites (55%) (Table 4).

Grassland Nest Habitats

Between 1999 and 2002, 33 American Bittern nests were located in grassland habitats at the Red Lake Indian Reservation. Similar approach was taken to reduce the number of predictor variables due to possible highly related variables as the wetland nest habitat procedures. Of the 16 variables, I initially excluded percent total vegetation cover, distance to nearest cover change, edge type, water depth, and Robel pole readings because they duplicate other variables, or in the case of water depth, there were only seven observations that had negligible measurable water depth in upland. Estimate of the overdispersion parameter of the global model ($\hat{c}$ = 1.240) showed little overdispersion in the sampling variance. Hosmer and Lemeshow (2000) goodness-of-fit test showed the global model had adequate fit (χ^2 = 7.698, df = 8, P = 0.464).

The best approximating model for nest sites in grassland habitats was the nest features within 4-m (Nest4) model (Table 5). The model had moderate predictability (67% concordance) and correctly classified almost 70% of the observations (Table 5). The odds of finding a nest in grassland habitats increased by 35% with every 10 cm decrease in vegetation height (Table 6). Average vegetation height at nests was 53.86 cm (± 2.84, range 20 – 85 cm) compared to random sites of 69.36 cm (± 3.32, range 30 – 150 cm, Table 7). Mean height of reed canarygrass and smooth brome, the two most dominant (79%) vegetation species at Red Lake, was 58.08 cm (±4.28, n = 17; Table 7). However, only 52% of the nests were found in smooth brome (*Bromus inermis*) and reed canarygrass (*Phalaris arundinacea*), while the rest were in a mixture of other species, which included *Agrostis* spp., clover (*Trifolium* spp.), goldenrod (*Solidago* spp.), alfalfa

(*Medicago sativa*), and timothy (*Phleum pratense*) which tend to be shorter ($\bar{x}$ = 49.38 cm ± 3.47, *n* = 16) compared to smooth brome and reed canarygrass.

DISCUSSION

Nest Survival

Between 1999 and 2002, 47 nests of American Bitterns were located. Average clutch size of wetland nesting individuals was 4.03 (SE = 0.12, *n* = 30). Estimated nest initiation was as early as 12 May in 2000 and as late as 4 June in 1999 during a normal climatic year. In 2002, a major flood event on 9 - 11 June inundated and caused all nests to fail. Re-nesting attempts (second nest initiation) began on 10 June and as late as 29 June. Successful nests were defined as those with at least one egg hatched, and twenty-nine of the 47 nests were hatched successfully; 15 were either destroyed or abandoned or ran over with the airboat (2), and fate was unknown for three nests.

I found partial support for the edge and temporal effects on nest success and little support for nest site effect. There is some evidence to suggest that distance to edge negatively affects nest success and distance to small water opening positively affects nest success. Nests close to edge, typically dry land in the study area would be more vulnerable to predators and nests close to small water openings are less accessible by mammal, terrestrial predators, such as raccoons and skunks that were observed at AGNWR. Also, there is some evidence in the models that the later the nests were discovered (Julian dates), the more successful. However, these interpretations need to be verified with more nest data.

Nest success estimate (until >1 egg hatching) was lower than those reported elsewhere (Armour 2002, Lor and Malecki 2006). However, there are few reports of nest success for American Bitterns, and existing reports may be biased estimates of nest success rates, especially when sample sizes are small. Other nest success rates reported for American Bittern was by Armour (2002) in grassland habitats in western MN, 54.13% (n = 47, 95% CI = 38.91, 75.01). Apparent nest success rates calculated for American Bitterns in the same study area were 50% (n = 4) for nests in wetland habitats and 70% (n = 7) for grassland nests (Brininger 1996, Azure 1998), however, samples sizes were small.

Causes for unsuccessful nesting attempts ranged from depredation to disturbance from researchers. There were a number of possible predators in northwest MN, from mammalian (raccoons, mink, river otters, fishers) to avian (black-crowned night herons, ravens and crows), but we did not observe predation directly because such observations were out of the scope of this study. Another potential cause for unsuccessful nesting attempt, that warrant further discussion, was the effects of researchers on nest success. Several studies reviewed and reported various levels of investigator disturbance on nest attendance and nest success rates for a number of species of birds (Westtemeier et al. 1998, Bêty and Gauthier 2001, and Sandvik and Barrett 2001). Of the 47 nests found, two nests were destroyed when crushed by an airboat, because bitterns were tenacious incubators and did not flush easily. Furthermore, nests and adults are difficult to detect because of the well camouflaged plumage and the limited visibility in the dense vegetation communities where bitterns typically nest. Sample size was too small to determine the cause of nest abandonment or depredation. To what extent researcher activities have on nest success and behavior of American Bitterns is unknown compared to the disturbance from natural events, such as predation or weather conditions, as were encountered during the flood of 2002. Driving an airboat or walking to nests to check on status may leave paths in the vegetation that is more open because of the vegetation has been flattened. Whether these paths attract predators to the vicinity of the nests is undocumented. I observed that paths developed from walking to the nests were more permanent than airboat paths because nests were checked up to several times during the season. In addition, although efforts were made to prevent as much vegetation destruction as possible when measuring habitat characteristics around nests, this activity further contributes to leaving a path around the nests. Airboat paths can last until well into the following nesting season, thus, changing the vegetation for at least one nesting period, and how these paths affect behavior and nesting activities of American Bitterns is undocumented. However, paths created by airboats leads less directly to nests; they are in a more systematic transect, as was the method of searching. Furthermore, it was observed that bitterns were more likely to flush from human presence than they were from airboats and vehicles (driving vs. walking).

Wetland Nest Habitats

The models partially supported the hypothesis that *American Bitterns occupying wetland habitats select dense stands of emergent vegetation, greater cover:water ratio, and shallow water levels (<20 cm)* in that the probability of encountering a nest site increased with an increase in percent dead vegetation cover and in dense stands of vegetation compared to moderately dense stands. Results are similar to those of

American Bittern nest site selection models in western New York, where an increase in percent horizontal cover and percent emergent vegetation cover increased the odds of nest site selection (Lor and Malecki 2006). However, the sample size (n = 12) was small in New York, hence comparisons may not be reliable. A number of issues make modeling and model validation of habitat selection by American Bitterns a challenge. American Bitterns are habitat generalists, occupying a variety of wetlands and upland areas. Small sample size is an issue connected to studying secretive marsh birds. Bittern nests are difficult to locate and densities are low, which attributes to low detection rates. These difficulties have been the case for other bitterns in the genus *Botaurus*, as in studies of European Bitterns (*Botaurus stellaris*; Puglisi et al. 2003, Adamo et al. 2004). Common vegetation characteristics between the study sites in western New York and northwest Minnesota included dense vegetation (e.g., cattail) as the dominant emergent species, and actively managed impoundments either in drawdown and/or with a prescribed burn. However, the larger macro habitat characteristics were quite different. New York wetlands were much smaller in size, ranging from 2 – 155 ha, many emergent wetlands consisted of standing dead timber from flooded forests, and the surrounding landscape consisted of more deciduous hardwoods (Lor and Malecki 2002). In contrast wetlands at Agassiz NWR ranged in size from 75 – 4000 ha, consisting of emergent cattail as the dominant species, with small areas of bulrush (*Scirpus acutus* and *S.validus*), sedge (*Carex* spp.) meadows, with willow shrub/scrub habitats, Aspen (Popular spp.) islands, and raised black spruce-tamarack bog. Given these differences, coupled with the small sample size of nest sites and the opportunistic nature of American Bitterns, it is not surprising that the models were different. From observations, data, and literature review, it is difficult to quantify and validate predictive models of nesting sites of American Bitterns at a micro-habitat scale between two different geographic areas.

Grassland Nest Habitats

The models did not support the hypothesis that *American Bitterns nesting in upland habitats select sites that are associated with dense and tall grass (100% visual obstruction readings of >50 cm).* In fact, the model with the most support suggested the probability of nest site selection decreased with increase in vegetation height. Vegetation height at nest sites ranged from 20 – 85 cm, compared to 30 – 150 cm at random sites. This means that when vegetation composition gets to thick and tall beyond some threshold (perhaps 85 cm?), American Bitterns were less likely to use the site. I found a disproportionate number of nest sites compared to random sites in dominant vegetation species; although a higher percentage of nests were found in reed canarygrass and smooth brome (52%) compared to the mix of grasses and forbs (48%), a much higher percentage (79%) of random sites was in reed canarygrass and smooth brome and lower percentage (21%) was in mix-grass and forbs. Brininger (1996) reported reed canarygrass was the predominant vegetation species in grassland habitats used by American Bitterns. Armour (2002) reported similar results, American Bitterns nested in vegetation with 100% visual obstruction reading higher than 5 dm, with highest nest density (0.08 nest/ha) in idle fields which consisted primarily of reed canarygrass, Kentucky bluegrass (*Poa pratensis*) and willow (*Salix* spp.), or in unmowed field (0.079 nests/ha), which was predominately alfalfa and smooth brome. These findings were consistent with a review by Dechant et al. (1999), where nests of American Bitterns were observed in both grassland and shrubland. In the Great Plains region (North and South Dakota, Minnesota, and Montana), bittern nests were located in mid to tall, dense idle grassland ranging in height from 30-99 cm, where 100% vertical visual obstruction (Robel et al. 1970) was typically > 50 cm. Grassland plant species at American Bittern nests included smooth brome, wheatgrass (*Agropyron* spp.), alfalfa, big bluestem (*Andropogon gerardii*), redtop (*Agrostis stolonifera*), quackgrass (*Agropyron repens*), and switchgrass (*Panicum virgatum*).

The American Bittern is a unique marsh bird because it utilizes wetland and upland habitats for nesting and foraging. At Agassiz NWR, semi-permanent wetlands with dense vegetation is the predominant habitat type (>60%), surrounded by agricultural land, with very little grassland, thus making the refuge attractive to American Bittern nesting. On the other hand, Red Lake Farms is predominantly grassland habitat surrounded by hay fields and wild rice paddies with little emergent wetlands. Thus, American Bitterns used grassland habitat for nesting. Consequently, habitat types may not be as important as vegetation structure and juxtaposition of fields and wetlands to provide nesting habitats. Nesting cover requires sufficient residual vegetation and sufficient foraging habitats consisting of ditches, edges, or areas with small water openings within a larger landscape scale that are important factors to American Bittern.

MANAGEMENT IMPLICATIONS

American Bitterns nest successfully in wetland and grassland habitats, both of which have declined drastically since the early 1900's. Results of this and other studies show that vegetation structure may be a more important factor than habitat type for nesting American Bitterns. Because bitterns nest in a wide range of water depth, from dry grassland to semi-permanent wetlands, water depth does not appear to be a significant factor in nesting site selection, but water depth does play an important role in foraging habitats (chapter 2) because foraging bitterns frequented sites close to ditches and open water bodies. Based on these observations and data, wetland and grassland management practices should include a variety of habitat types (wetland, grassland, shrubs) and conditions that emulate natural fluctuations, such as drought (drawdowns), flooding, and fires (prescribed fires), which rejuvenate habitats crucial to providing nesting and foraging conditions for marsh birds and other water dependent birds. It is critical that these management actions are monitored and evaluated to determine at what point in time and under what management conditions these actions achieve the threshold (e.g., vegetation height of 20 – 85 cm or $\bar{x}$ = ~50 cm for American Bitterns) that are required by bird species. The timing of these management actions is also critical because timing of American Bitterns nesting is typically later than nesting by most dabbling ducks in the same habitat. Management actions such as haying, mowing, and prescribed burns should carefully consider the timing and development of American Bittern nesting activities before they are implemented. General rules of thumb and regulations concerning current habitat management such as water-level manipulation, prescribed fire, mowing, and haying of fields were set based on duck nesting activities and dates. In many instances, these dates do not coincide with marsh bird nesting, such as American and Least Bitterns or Pied-billed Grebe. Based on my data and observations in previous studies, implementation of haying or mowing after 15 July were based on dabbling ducks hatching period, which essentially ignores life histories of other wetlands birds. Dabbling ducks are precocious species and usually leave the nesting area to forage in wetlands elsewhere. Dates set to protect or benefit dabbling ducks do not necessarily provide the same degree of protection or benefits to American Bitterns because they are altricial species and chicks remain on and around the nest sites until they are fledged. According to Brininger (1996) the average number of days from hatching to fledging for American Bitterns at Agassiz NWR was 41 days (range 37 - 43 days). In actuality, the nesting period was much longer for the species because Azure (1998) found that nest initiation began during the first week of May through first week of July (Armour 2002), and chicks were in the field and in the wetlands well past 15 July (Armour 2002, and per. obs.). At Red Lake Farms, I either observed or were informed that nests were present, but haying activities on the farm destroyed the nests. In one instance, a nest found in mid-July was destroyed within two hours of discovery by a haying operation. Thus, the American Bittern is a species that has adapted to a variety of habitat types and can probably do well by its own nature, given that management activities of remaining wetlands and grassland habitats accommodate the life history events that are not quite the same as dabbling ducks, in which they share habitats.

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Table 1. *A priori* candidate models for explaining nest survival of American Bitterns in wetland habitats at Agassiz National Wildlife Refuge in northwestern Minnesota, 1999 - 2002. Number of parameters (K) in each model includes intercept and the explanatory variables; n = 56 (number of nest observations), odds ratios from profile likelihood parameter estimates only reported for the models with most support (first four models); $\hat{c}$ is the correction for overdispersion calculated from the goodness-of-fit χ^2/df from the global model. Odds ratios and 95% confidence intervals (CI) for predictor variables based on model-averaged results from the five *a priori* candidate models predicting nest survival of American Bitterns.

Model	**Parameter**	**K**	**-2log**	$\hat{c}$	**QAIC$_c$**	**ΔQAIC$_c$**	w_i	**β**	**95% CI (β)**	**Odds Ratio***	**95% CI***
Null		1	67.19	2.92	25.12	0.00	0.60	3.222	2.644, 3.923		
Edge-effect	EDGE	3	66.30	3.18	27.30	2.18	0.20	-0.003	-0.008, 0.003	0.997	0.990, 1.000
	SMALL							0.001	-0.008, 0.013	1.001	0.990, 1.010
Temporal effect	JDATE	2	66.11	2.80	27.82	2.70	0.16	-0.014	-0.07, 0.039	0.986	0.940, 1.040
Nest-site effect	LIVE	4	64.00	2.92	30.73	5.61	0.04	-0.006	-0.044, 0.044	0.994	0.950, 1.040
	HT							-0.032	-0.067, 0.011	0.968	0.930, 1.000
	HC5							0.028	-0.004, 0.059	0.971	0.865, 1.090
Global	JDATE	7	63.16	2.90	38.08	12.97	0.00	7.943	-7.058, 23.802	0.986	0.940, 1.040
	EDGE							-0.002	-0.114, 0.059	0.997	0.990, 1.000
	SMALL							0.001	-0.052, 0.061	1.001	0.990, 1.010
	LIVE							-0.026	-0.010, 0.014	0.994	0.950, 1.040
	HT							0.001	-0.009, 0.004	0.968	0.930, 1.000
	HC5							-0.033	-0.070, 0.010	0.971	0.865, 1.090

Table 2. A set of *a priori* models to explain nest site selection in wetland habitats by American Bitterns (n = 47) in wetlands in northwest Minnesota (1999-2002). *Nest4: nest habitat variables measured within 4-m of nest; Nest10: nest habitat variables measured within 10-m of nest; Macro: macro-habitat variables in relation to nest; Edge: variables related with nest distance to edge.

Model *	**Predictor Variables[a]**	**K^b**	**AIC_c**	**ΔAIC_c**	**w_i^c**	**% concord.**	**Correct class.**
NEST10	DEAD, DOM, DEN	6	150.86	0.00	0.986	81.6	73.0
NEST4	HT, HC5, H2O	4	159.42	8.56	0.014	79.8	75.7
DOMIN.	DOM, DEN	6	171.60	20.74	0.000	64.3	70.4
MACRO	LARGE, SMALL	3	189.47	38.61	0.000	58.9	67.8
EDGE	LAND, EDGE, DITCH	4	189.79	38.93	0.000	67.0	67.8
NULL	INTERCEPT	1	190.04	39.18	0.000		

[a]DEAD = % dead vegetation cover of the dominant species; SMALL = distance to small (<0.4 ha) water pool (1 = 0 - 20 m, 2 = 21 - 100 m, 3 = >100 m); LARGE = distance to nearest large water opening; LAND = distance to nearest land; EDGE = distance to vegetation edge; DITCH = distance to ditch; HT = average vegetation height within 4-m of nest sites; HC5 = horizontal cover on the density board below 0.5 m, average of readings from four cardinal directions; H2O = average water depth within 4-m of the nest; DOM = dominant species within 10-m of nest; DEN = density category 1=rank, 2=moderate (see text for full explanation).

[b]K = number of parameters in the model.

[c]w_i = AIC_c weight of the model *i*.

Table 3. Habitat parameters and associated statistics from the best approximating AIC-selected models of American Bittern nests (n = 47) in wetlands in northwestern Minnesota, 1999 – 2002.

Model	Predictor Variables[a]	K[b]	β	Confidence Interval (β)	Odds Ratio[c]	Confidence interval (OR)
NEST10	Intercept	6	-2.416	-3.628, -1.366		
	DEAD		0.042	0.024, 0.062	1.043	1.024, 1.064
	DEN-Moderate		-0.488	-0.949, -0.044	0.377	0.150, 0.916
	DEN-Dense		Refer.			
	DOM-BULL		1.465	-0.142, 3.270	2.025	0.202, 24.709
	DOM-CAREX		-1.812	-3.267, -0.563	0.076	0.009, 0.551
	DOM-CATT		-0.146	-0.987, 0.731	0.405	0.091, 1.975
	DOM-HERBS		-0.266	-2.784,1.496	0.359	0.013, 4.366
	DOM-PHRAG		Ref.			
NEST4	INTERCEPT	4	-7.060	-10.735, -4.225	0.973	0.958, 0.986
	HT		-0.028	-0.043, -0.014	1.085	1.050, 1.130
	HC5		0.081	0.049, 0.122	1.034	1.015, 1.056
	H2O		0.034	0.015, 0.054		
DOMIN.	INTERCEPT	6	-0.653	-1.388, -0.023		
	DEN-Mod.		-0.717	-1.141, -0.322	0.238	0.102, 0.525
	DOM-BULL		1.995	0.590, 3.684	6.165	0.828, 62.786
	DOM-CAREX		-1.616	-2.926, -0.500	0.167	0.026, 0.951
	DOM-CATT		0.194	-0.548, 1.003	1.019	0.284, 4.196
	DOM-HERBS		-0.749	-3.145, 0.768	0.397	0.018, 3.680
	DOM-PHRAG		Ref.			
MACRO	INTERCEPT	3	-0.174	-0.765, 0.406		
	LARGE		-0.001	-0.002, 0.000	0.999	0.998, 1.000
	SMALL		0.002	-0.002, 0.005	1.002	0.998, 1.005
EDGE	INTERCEPT	4	-0.543	-1.246, 0.133		
	LAND		0.001	-0.001, 0.003	1.001	0.999, 1.003
	EDGE		-0.002	-0.004, 0.000	0.998	0.996, 1.000
	DITCH		0.001	-0.001, 0.003	1.001	0.999, 1.003
NULL	INTERCEPT	1	-0.594	-0.946, -0.256		

[a]DEAD = % dead vegetation; SMALL = distance to small (<0.4 ha) water pool; LAND = distance to nearest land; LARGE = distance to nearest large water opening; EDGE = distance to vegetation edge; DITCH = distance to ditch; HT = average vegetation height within 4-m of nest

Table 3. continued.

sites; HC5 = horizontal cover on the density board below 0.5 m, average of readings from four cardinal directions; H2O = average water depth within 4-m of nest; DOM = dominant species at 10-m within nest; DEN = density category 1=rank, 2=moderate, see text for full explanation).

[b]K = number of parameters in models, not including intercept term.

[c]Profile Likelihood Confidence Interval for Adjusted Odds Ratio.

Table 4. Summary statistics (range, $\bar{x}$, SE) for variables associated with nests of American Bitterns in wetland habitats at Agassiz National Wildlife Refuge, in northwest Minnesota, 1999-2002. *See Table 2 for descriptions of variable codes.

	Nest n = 47		**Random** n = 105	
VARIABLE	Range	$\bar{x}$ (SE)	Range	$\bar{x}$ (SE)
DEAD (%)	30 - 100	61.91(3.21)	0 - 100	32.84 (2.51)
DEN	70:30 (dense:mod)		40:60(dense:mod)	
Carex	3/47 (6%)		27/105 (26%)	
Phragmites	4/47 (9%)		10/105 (10%)	
Cattail (*Typha* spp.)	34/47 (72%)		58/105 (55%)	

Table 5. A set of *a priori* candidate models to explain nest sites (n = 33) of American Bitterns in grassland habitats at Red Lake Farm Indian Reservation, in western Minnesota, between 1999 - 2002. *Nest4, Nest10: nest habitat variables measured within 4-m and 10-m of nests; Edge: variables related with nest distance to edge.

Model*	**Predictors variables**	**K^b**	**AIC_c**	**Δ AIC_c**	**w_i^c**	**% concord.**	**Correct class.**
Nest4	HT HC5	3	109.02	0.00	0.963	66.6	69.8
Null	Intercept	1	116.55	7.54	0.022		
Nest10	DEN1	2	117.84	8.83	0.011	27.5	61.6
Edge	DITCH EDGE	3	120.61	11.60	0.003	50.2	61.6

[a]HT = average vegetation height within 4-m of nest sites; HC5 = horizontal cover on the density board below 0.5 m, average of readings from four cardinal directions; DEN = relative density of dominant vegetation within 10-m of nest, classified into four categories (0 = no vegetation, 1 = rank [water not visible through base of stems at water level and one cannot easily push hands through the stems], 2 = moderate [anything that falls between category 1 and 3], 3 = sparse [water easily visible through base of widely scattered stems]); DITCH = distance to ditch; EDGE = distance to vegetation edge.

[b]Number of parameters.

[c]AIC weight of the model *i*.

Table 6. Habitat parameters and associated statistics from the best approximating AIC-selected models of American bittern nests (n = 33) in Red Lake Farm Indian Reservation, in western Minnesota, 1999-2002.

Model	**Predictor variables**	**K**	**β**	**CI (β)**	**Odds Ratio***	**CI**
Nest4	Intercept	3	4.855	-3.143, 20.064		
	HT		-0.036	-0.067, -0.010	0.965	0.935, 0.990
	HC5		-0.037	-0.194, 0.049	0.964	0.823, 1.050
Null	Intercept	1	-0.975	-1.418, -0.546		
Nest10	Intercept	2	-0.912	-1.376, -0.459		
	DEN1		-0.204	-0.662, 0.255	0.665	0.266, 1.666
Edge	Intercept	3	-0.869	-1.802, 0.038		
	DITCH		-0.001	-0.006, 0.005	0.999	0.994, 1.005
	EDGE		0.000	-0.004, 0.003	1.000	0.996, 1.003

Table 7. Summary statistics (range, $\bar{x}$, SE) for variables were associated with nests of American Bitterns in grassland habitats at Red Lake Indian Reservation, in Minnesota, 1999-2002. *See Table 5 for description of variable codes.

Variables	**Nest** n = 33		**Random** n = 53	
	Range	$\bar{x}$ (SE)	Range	$\bar{x}$ (SE)
HT (cm)	20 - 85	53.86 (2.84)	30 - 150	69.36 (3.32)
HC5 (%)	47.5 - 100	96.63 (1.76)	91.25 - 100	99.20 (0.26)
Reed canarygrass/ smooth brome	17/33 (52%)		42/53 (79%)	
Mix-grass/forbs	16/33 (48%)		11/53 (21%)	

CHAPTER 2

HOME RANGE AND FORAGING SITE SELECTION OF AMERICAN BITTERNS (*BOTAURUS LENTIGINOSUS*) IN NORTHWEST MINNESOTA

ABSTRACT

Populations of American Bitterns (*Boataurus lentiginosus*) have undergone dramatic decline over the past three decades because of loss and degradation of wetland and grassland habitats. In this study, home ranges from ten years of data and foraging habitats of American Bitterns were assessed using radio-telemetry techniques and habitat modeling procedures. Using fixed kernel home range estimator with least-squares cross-validation bandwidth selection method, the average core area was 18.08 ha (± 1.68) and the 95% home range was 109.28 ha (± 38.47). Habitat variables were collected and analyzed for radio-tagged American Bitterns. The odds of predicting foraging sites of American Bitterns increased with distances closer to small water openings and increase in percent live vegetation cover. Conservation and management activities that would benefit American Bitterns at the study areas would involve restoring deepwater wetland areas with more dynamic water regimes, reducing rank stands of *Typha* and *Phragmites*, and creating shallower, open water to vegetation interfaces. Further, timing of management activities in wetlands need to consider life history events of American Bitterns to ensure sufficient habitats exist during post-breeding and molting periods.

INTRODUCTION

The American Bittern (*Botaurus lentiginosus*) is a large, inconspicuous marsh bird that breeds throughout mid- and northern United States to the northern provinces of Canada. Like many marsh and wetland bird species, the American Bittern is affected by the loss and degradation of wetlands and associated grassland habitats (Gibbs et al. 1992, Dechant et al. 1999). From 1966 - 2004, the breeding population was estimated to have declined 1.36% per year ($P \leq 0.1$) in the United States and Canada and 6.23% ($P \leq 0.10$) per year in the midwestern region of the United States (USGS 2006). The USFWS (2002) designated the American Bittern as a species of "Resource Conservation Priority" throughout its midwestern region (Region 3). This concern was again emphasized in 2005, when the species was ranked as "highly imperiled" in the Upper Mississippi Valley/Great Lakes Waterbird Conservation Plan (USFWS 2005).

Quality bittern habitats are limited primarily to lands owned by federal, state, and non-profit organizations (Azure 1998, Laney 2003, Lor and Malecki 2006). Consequently, it is crucial that biologists acquire sufficient and accurate information to properly conserve and manage the remaining habitats for not only bitterns but also for other associated waterbird species (Hand et al. 1989, Dechant et al. 1999, USFWS 2005). However, low densities and elusive nature of American Bitterns makes it difficult to obtain more detailed information on the habitat characteristics most important to the species. Radio-telemetry technology and studies have proven to be an effective method of studying resource selection by animals (Millspaugh and Marzluff 2001).

An animal's use of space or home range, typically described in the context of time, is central to understanding habitat requirements to meet life history needs. The concept of an animal's home range is critical in efforts to quantify ecological characteristics required by animals to survive, via resource selection studies. In this study, the home range and foraging habitats of American Bitterns were assessed during the post-breeding season at AGNWR. The study objectives were to 1) determine post-breeding habitat requirements of American Bitterns and how these relate to movement and size of their home range, 2) to assess foraging habitats of American Bitterns, which tested the hypothesis that *foraging sites of American Bitterns are associated with vegetation fringes and shorelines, greater water:cover ratio, and shallow water areas.*

STUDY SITE

This study was conducted in northwestern Minnesota, at the AGNWR, located in Marshall county, and Red Lake Indian Reservation, located in Clearwater county. AGNWR is a part of a large complex of land specifically managed for migratory birds by the USFWS and the Minnesota Department of Natural Resources (MNDNR). The refuge comprises 24,888 ha of which 15,135 ha are impounded wetland habitats, including permanent, semi-permanent, sedge meadows, and raised bog; 4006 ha aspen woodland, 4715 shrubland (primarily *Salix* spp.), and 761 ha grassland habitats. There are 26 impoundments on the refuge, each has control structures used to manipulate water levels. These impoundments range in size from approximately 75 – 4000 ha (USFWS Agassiz NWR 2005). Immediately adjacent to Agassiz NWR on the southern, eastern, and northern boundaries are the Elm Lake (~6356 ha), Eckvoll (~2632 ha), and Thief Lake (~22,267 ha) Wildlife Management Areas (WMAs), which are composed mainly of permanent and semi-permanent wetland habitats.

Prior to refuge acquisition in 1937, many attempts were made to convert the muck land into arable agricultural fields by developing a series of drainage ditches. However, these drainage ditches did not sufficiently drain the land because of vegetation growth in the bottom of the ditches (USFWS 2005). Climatic conditions at the refuge is typical of the region, where variations in temperatures are wide and extreme, having severe winters with moderate snowfall and annual precipitation of approximately 56 cm. Because of the variation in temperatures and rapid snowmelt in the spring combined with the flat terrain, flooding occurs frequently and is often detrimental to the farming efforts in the area (USFWS 1978).

AGNWR lies on the southeastern portion of the prehistoric Glacial Lake Agassiz, within the geomorphic region classified as the Agassiz Lacustrine Plain, Red River Valley area (University of Minnesota 1980a) and Agassiz Lacustrine Plain, Red Lake area (University of Minnesota 1980b). Quaternary geologic sediment at Agassiz NWR is Holocene peat, and at Red Lake Indian Reservation is lake-modified till (Hobbs and Goebel 1982). The landscape position in which Agassiz NWR is located is characterized as shallow depressions on lake plain, and the soil series is mainly Cathro Haug, which consists of organic soils. The rooting zone (0.3 – 1.5 m) consists of muck and peat, and the substratum (1.5 - >7 m) is loam and sandy loam (University of Minnesota 1980a).

The small lakes (Mud, Kuriko, Webster, Whiskey, and Elm Lakes) located within AGNWR were formed in depressions in the bed of Glacial Lake Agassiz that were not entirely filled with sediment (Minnesota Conservation Department 1968). A landscape feature on the refuge that reveals the substratum underlying the refuge is a sandy beach ridge located along the northwestern portion of the largest impoundment on the refuge

(Agassiz Pool). Wave-action from glacial activities deposited the sand which formed the beach ridges that lie within AGNWR. These geologic formations and anthropomorphic alterations of the landscape shaped a complex of managed wetlands, raised bog, and grassland within the refuge boundaries that reflect past management dogma related to development and management. The current conditions on the refuge resulted from the past history of development and management in combination with current practices. The site is now protected in the sense that it will not be converted to other uses, but managing habitats and emulating natural processes that are required by waterbird species is more problematic. Developing monitoring programs for waterbirds and habitats within such a highly modified landscape will require much patience and is a daunting task.

Climate of northwestern MN is characterized by temperatures ranging from -35°C March to 36°C in August, with an average of 115 frost-free days (USFWS 2002). Annual average precipitation over the past 30 years was 56 cm. During the duration of this study (1999-2003), average precipitation was 57.35 cm. Winter is relatively dry, with average snowfall of approximately 99 cm. The wettest months are June, July, and August. During 2002, almost 24 cm of rain fell between 9 and 11 June, raising water levels in wetland units of >1 m in some pools, causing major flooding in the area.

METHODS

Radio-telemetry

Modified ruffed grouse (*Bonasa umbellus*) traps were used to capture male American Bitterns during 1999 – 2001 (Brininger 1996). Typically, a trap site consisted of two mirror traps set either back to back or side by side with the entrances facing opposite each other. These were covered with vegetation for camouflage. A tape player, which broadcast the American Bittern courtship call (*"pump-per-lunk"*) was placed on top of the traps, secured with a bungee cord. A variety of small and lightweight "boomboxes" made by Sony Corporation were used to broadcast bird calls at approximately 50 - 80 db at 1-m distance. Females were captured at the nest by using a long-handle dip net.

Captured bitterns were weighed, measured, fitted with a USFWS leg band size 6 (females) or 7A (males), and a whip-antennae radio-collar transmitter. Measurements were taken of the culmen, bill, wing chord, head dimensions, and toe, as described in Azure et al. (2000). Each single-pulse transmitter (AVM Instrument Co., Livermore, CA) was sewn with dental floss and super glued to a collar made of herculite material (Herculite Protective Fabrics Corp., New York, New York; Brininger 1996). Each transmitter was powered by a lithium battery, encased in dental acrylic, and operated within the 148 – 152 MHz range; life expectancy ranged from 12 - 16 months, and each weighed between 15 – 17 g. Once processed, all bitterns were checked for signs of distress or abnormal behavior while they were held in the release box and were released at the original trap site, typically within 1 – 4 hours after capture.

Radiotracking of bitterns began immediately after the bird was released to check on status of the bitterns, but formal, interval tracking to assess home range during the post-breeding period began typically during the last week of May through mid-September each year. A pick-up truck (½ ton 1993 Chevrolet), with a double-mounted 4-element yagi antennae was used to track all birds. The antennae mast was approximately 1.8 m tall above the top of the truck. Radio signals were received with an Advanced Telemetry

Systems TLR-4000 (Isanti, MN) receiver with a null-peak switch box, where only the nulls were recorded. During 2000 and 2001, each radio-collared American Bittern was tracked once every two days, throughout the summer using triangulation methods described by White and Garrott (1990). In 1999 and 2002, radio-tagged bitterns were located only once or twice per week because of time constraints. Directional bearings were obtained from at least 3 locations along dikes and roads. For efficiency in travel time and to make tracking of all birds possible, the refuge was divided into 3 tracking routes. Starting time interval of each route was randomly selected and then each bird was systematically monitored chronologically based on the time intervals. Radio tracking time was divided into 4 4-hr intervals (i.e., 0600 - 1000, 1000 - 1400, 1400 - 1800, 1800 - 2200) during daylight hours. No tracking was conducted at nighttime because Brininger (1996), Azure (1998), and personal observations showed that American Bitterns were inactive at night. Tracking time for each radio-collared bird was systematically rotated so that each bird was tracked at different times of the day throughout the season.

Telemetry Accuracy and Bias

A beacon study, where transmitters were hidden in known locations and observers who were unaware of the locations, triangulated to the transmitter, was conducted to measure telemetry accuracy by determining the difference between the actual location of a radio transmitter and its estimated location. Observers were told that the hidden transmitters were newly radio-collared birds so that no bias from potentially extra effort was placed into obtaining these locations. Mean absolute angular error calculation, with extreme outliers (> 25°) removed (Lee et al. 1985), mean estimated distance error from plotted to actually transmitter location (n = 40 random locations), mean error polygon (with six outliers removed in 2001), observer error, and bearing accuracy (White and Garrott 1990, Gould and Jenkins 1993) were also calculated to assess home range accuracy and bias.

Home Range Analysis

The program Locate II (Nams 1990-2000) was used to estimate locations from triangulation azimuths. Estimates of home range size vary widely by the estimator, the bandwidth selection method, and by telemetry accuracy and biases (Seaman et al. 1999, Garton et al. 2001, and Gitzen et al. 2006, Moser and Garton 2007). As suggested in the literature (Seaman et al. 1999), only individual birds with >30 locations were used to estimate home ranges.

The Home Range Extension in ArcView 3.X (Rogers et al. 1998) was used to calculate the 95% utilization distribution, referred to here as the bittern's "home range" and the 50% utilization distribution, referred to as the "core area." Locations of bitterns from the 4 tracking time-intervals were pooled for home range analysis because of small sample sizes. Least-squares cross-validation (LSCV, option h_cv), with "unit variance" option to standardize the data, and a 70 x 70 grid cell size were used to estimate home range sizes. I obtained locations of radio-tagged bitterns from 1994-1997 from W. Brininger and D. Azure and calculated home range sizes using the same method as above and report home range sizes separately for this dataset because telemetry methods were different and could bias comparisons.

Foraging Sites

In the model development process, a hypothesis of foraging habitats, based on information from the literature (Gibbs et al. 1992) and from personal observations was developed: *foraging sites of American Bitterns were associated with vegetation edge, greater open water:cover ratio, and shallow water areas.* Habitat features based on this hypothesis were collected at foraging sites and random sites. Flushed sites of American Bitterns were presumed to be foraging sites because birds were flushed in the day time, when bitterns were actively foraging (Gibbs et al. 1992, Azure 1998, pers. obs.).

Each radio-collared bittern was flushed from 1 - 4 ($\bar{x}$ = 1.7, SE = 0.23) times between 22 June through 30 August to both verify the location and condition (dead or alive) of the bird, and to measure foraging habitat.

Macro- and micro-habitat variables were measured at each foraging site and random sites. Variables at foraging sites were measured immediately after the bird was flushed. Random sites were selected from a grid of 100-m squares, overlaid over each wetland unit in which bitterns were flushed, using Geographic Information System (GIS) covertypes in ArcView 3.1 (ESRI, Redlands, CA) and were measured between 26 June and 30 August to coincide with measurements of foraging sites. Universal Transverse Mercator (UTM) coordinates of the center of each randomly selected point were obtained from the GIS and used for navigation to the point of random site. In the event that the random site was not accessible or was not in wetland habitat (i.e., island), I traveled to the next closest random site to collect data. Macro-habitat variables included distances to nearest land, small water opening (<0.4 ha), large water opening (>0.4 ha), ditch, edge, and land. These variables were verified by examining the refuge's habitat GIS covertypes in ArcView 3.1 because in some instances, these features were difficult to discern in the field.

Micro-habitat features measured at a 10-m radius circle from the foraging or random sites included percent live and percent dead vegetation of the dominant species, percent open water, percent emergent vegetation, percent shrub/scrub vegetation, and relative density (1 = rank, 2 = moderate, 3 = sparse; Hickey 1996, Hickey and Malecki 1997). Also, vertical and horizontal structures of the vegetation were measured with methods adopted from Hays et al. (1981). Within the 4-m radius circle, vertical and horizontal vegetation structures were measured at four axis points, located at the four cardinal directions. Vegetation structure was measured by using a Vegetation Profile Board (Hays et. al 1981). The dimension of the board was 0.3 m x 2.0 m, constructed from 0.95 cm (3/8 inch) plywood, and 5 x 5 cm black and white checkers were painted on one side to facilitate estimations of percent coverage of the board by vegetation. Height intervals (at 10, 20, 30, 50, 70, 100, 150 cm) were marked on the board with bright red paint. The board was held upright at foraging or random site, flushed with the ground or at the water surface. An observer stands 4-m from the board, at each of the four cardinal directions, while facing the board, the observer estimated by ocular estimation, the extent (in percentage) at which the board was obscured by vegetation at each marked height interval. Water depth (cm) and vegetation height (cm) were also recorded at each of the four cardinal points.

Data Analysis

In the analysis process, I screened predictor variables in several steps. Several predictor variables were duplicates, such as percent open water and percent emergent vegetation because they were complimentary of each other (i.e., 30% open water:70% emergent vegetation cover). In this case, I excluded one (e.g., emergent vegetation cover) from further analysis because greater open water compared to vegetation cover was part of the hypothesis. Four variables were eliminated in this way: percent emergent vegetation, percent scrub because there were only 6 (3 foraging observations and 3 random observations) out of 158 total observations, percent dead vegetation cover because it was complimentary to percent live cover, and interspersion category because it was the ratio of percent open water and percent emergent vegetation (30% open:70% emergent vegetation as in example above). A global model was constructed from remaining 17 habitat features related to the hypothesis. I developed a set of *a priori* candidate models that were subsets of the global model which represent various aspects of American Bittern foraging habitat features: 1) Global model, 2) Site-specific model; 3) Edge "effect", which includes measures of distance to nearest edge, ditch, and land; 4) Cover-effect model, which includes vegetation height and density, 5) Null model. I further screened the variables using paired *t*-test and retained variables that were significant at $P < 0.25$ (Hosmer and Lemeshow 2000); only one variable (distance to ditch) was dropped from this process where ditch was not significantly ($p = 0.852$) different from distance to edge. I plotted the standardized deviance residuals for the global model against the explanatory values and found no outliers (all values <3) or patterns that required transformation. I then conducted multi-collinearity analysis to check for highly related variables using PROC REG on the predictor variables in the global model and excluded highly related (TOL < 0.40) variables from further analysis (Allison 1999). Tolerances (TOL) measures the strength of inter- relationships among the explanatory variables in the model. Tolerance is 1 - R^2 for the R^2 that results from the regression of the explanatory variable on the other explanatory variables in the model. If a variable is closely related to other variables, the tolerance goes to 0 and the variance inflation becomes large (SAS Institute Inc. 2004). Multi-collinearity check eliminated six more variables all related to % horizontal density, which were measured at 0.1, 0.2, 0.5, 0.7, 1.0, and 1.5 m. height on the density profile board. The 10 variables that remained included: OPEN = % open water area within 10-m of foraging/random site; LARGE = distance to large water openings (>0.4 ha); SMALL = distance to small (<0.4 ha) water opening; LAND = distance to nearest land; EDGE = distance to vegetation edge; HT = average vegetation height within 4 -m of foraging/random sites; H2O = average water depth within 4-m of the foraging/random site; DEN = relative density of dominant vegetation within 10-m of foraging or random site, classified into four categories (0 = no vegetation, 1 = rank [water not visible through base of stems at water level and one cannot easily push hands through the stems], 2 = moderate [anything between category 1 and 3], 3 = sparse [water easily visible through base of widely scattered stems]); DOM = dominant vegetation within 10-m of foraging or random site. To alleviate model convergence problems, I collapsed vegetation species with similar structures and small number of observations into one level (e.g., softstem bulrush, hardstem bulrush, *Phragmites,* reed canarygrass, J*uncus, Salix* = "MISC") and reduced nine species-level to three (MISC, CATTAIL, CAREX) levels to achieve a more balanced distribution of observations in each category. For the four categories in density (DEN), there were no foraging or random sites with level 0 (no vegetation) and only one random observation with level 3 (sparse vegetation), and it was collapsed into DEN level 2 (moderate density).

Overdispersion occurs when the sampling variance exceeds the model-based variance and is caused by the positive correlation between the binary responses (foraging/random) or variation between the response probabilities (SAS Institute, Inc. 1995, Burnham and Anderson 2002). Overdispersion is common in most real data and causes underestimation of the variance of the parameter estimates (SAS Institute, Inc. 1995). Overdispersion can be modeled with variance inflation factor ($\hat{c}$), which can be estimated from the goodness-of-fit chi-square statistic (χ^2) of the global model and its degrees of freedom ((Burnham and Anderson 2002). Model structure is acceptable when $1 \leq \hat{c} \leq 4$, but inadequate when $\hat{c} > 6$ (Burnham and Anderson 2002:68). I checked for overdispersion in the global model by using the AGGREGATE and SCALE options in PROC LOGISTIC to obtain an estimate for variance inflation factor $\hat{c}$.

I used the information theoretic approach (Burnham and Anderson 2002) to fit models and evaluate support for each of the models. All analyses were performed using PROC LOGISTIC in SAS (SAS Institute 2002-2003) unless otherwise noted. I examined interaction terms in the models and they did not appear to improve the models nor did they change the ranking order of the models. Therefore, for sake of parsimony, I did not describe them further. The OFFSET option for the parameter estimates was included in the modeling procedure to account for the disparity between the number of foraging sites (65) and number of random sites (93). I evaluated the models by using Akaike's

Information Criterion (AIC), modified for small sample size (AIC_c; Burnham and Anderson 2002). I used AIC_c to rank models from most to least supported given the data. I calculated ΔAIC_c (the difference between lowest observed AIC_c value and value for the next model) and Akaike weight of evidence (w_i, a measure of model support based on ΔAIC_c that sums to 1 across all candidate models) as measures of model support. However, in the event that the model fits the data and model predictive power (>60% concordance) was adequate, odds ratios and 95% confidence intervals were calculated. Confidence intervals for parameter estimates were based on profile likelihood function rather than the asymptotic normality because they provide more accurate estimates for small sample sizes (SAS Institute 2004; Burnham and Anderson 2002). In the event of model-selection uncertainty (w_i <0.90), I calculated odds ratios based on model-averaged coefficients and 95% confidence intervals (CI) based on unconditional standard errors (Burnham and Anderson 2002). Only odds ratios with CIs that did not include 1 were interpreted.

RESULTS

From 1999 – 2002 64 adult males, five adult females, and one juvenile female American Bittern were captured and radio-marked at Agassiz NWR. Four males in 1999 and 10 males in 2000 were re-captured during the molting period (July) to replace VHF transmitter collars with satellite (PTT) transmitters. This was part of a separate migration study (J. Toepfer and G. Huschle, unpubl. data). A total of 15 male bitterns from 2000 and 2001 had >30 locations and were used in the home range estimates. The number of locations per animal ranged from 31 - 58 ($\bar{x}$ = 43.53 ± 1.68). Between 1994 – 1997, 10 bitterns had >30 locations ($\bar{x}$ = 39.80 ± 2.05, range 31 – 46) and were used in the home range estimation.

Home Range

The average fixed-kernel estimates of American Bittern 95% home range was 109.28 ha (± 38.47) in 2000 and 2001 combined and the 50% core area was 18.08 ha (± 1.68; Table 1). Home ranges for 10 bitterns during the period of 1994 – 1997 was 147.06 ha (± 22.93) and 50% core area was 28.68 ha (± 5.18; Table 1). Because telemetry methods were different (tracking truck, receiving system, observers) and the associated telemetry errors, in addition to the differences in the number of locations, comparisons of home range areas for these two studies are not reliable.

The within year average maximum distance traveled by individual male American Bitterns was significantly (Kruskal-Wallis Test, χ^2 = 6.96, DF = 1, P = 0.008 [exact test, 2-sample]) greater in 2001 ($\bar{x}$ = 2845.65, SE = 354.55) compared to 2000 ($\bar{x}$ = 1542.59, SE = 238.42). The differences in maximum distance traveled coincided with larger home range size in 2001, even with 1-2 outliers removed from 2001 locations and no outliers in 2000. In addition, more bitterns in 2000 had overlapping home ranges (all but two) compared to those of 2001, where only two birds had overlapping ranges.

Telemetry Accuracy and Bias

The mean absolute angular error was 4.57° (SE = 0.29, n = 219 bearings). The mean animal-to-antennae distance was 655.51 m (SE = 80.99, n = 40 locations; Gould and Jenkins 1993). The mean estimated distance error from plotted to actual transmitter location, when mean angular error and mean animal-to-anteannae distance were used, was 51.36 m, and bias was 0.46° (White and Garrott 1990). Mean error polygon was 1.85 ha (SE = 0.44, $n = 388$) for 2000 and 3.88 ha (SE = 0.50, $n = 324$) for 2001.

Results from Tukey's test for observer difference in beacon study was not significant in either 2000 ($F_{2,109} = 0.83$, $p = 0.44$) or 2001 ($F_{2,103} = 0.18$; $p = 0.839$), nor was there any significant differences among observers at $p = 0.05$. Overall differences in mean bearing errors (from animal locations) were not significant in either 2000 ($F_{3,377} = 2.43$; $p = 0.065$) or 2001 ($F_{2,333} = 2.00$; $p = 0.138$), nor were there any differences among observers at $p = 0.05$.

Of the 70 bitterns captured, two were confirmed dead; one died in 1999 and was too decomposed to determine cause of death; one in 2000 where the bird "strangled" itself by hooking its bill through the hole of the herculite collar.

Site Fidelity

Nine different individual males were recaptured in the same general area in which they were previously captured, four of which returned to the same corner of the marsh and four others returned to within 3 km of their original capture site (Table 2). The farthest relocation distance was by bittern #82098; it was originally captured in 2000 in the northwest corner of South Pool and recaptured in 2001 in the northeast corner of Farmes Pool, a distance of approximately 3 km. Two individuals were captured three times; #82006 was originally captured in 1996, was recaptured in the same location in Dahl pool in 1998, 2000, and 2001; the other (# 82057) was originally captured in the southeast corner of Pool 8 in 1998, then in the southwest corner in Pool 8 in 1999, and returned to southeast corner in 2000 ((Table 2).

Bird Measurements

From 1999 – 2002, nine bitterns were recaptured at least one year later, two of the nine were recaptured twice. Measurements and weight of when the bird was originally caught and recaptured were not statistically significant at $p = 0.05$, except for wing chord length where recaptured birds ($\bar{x}$ = 28.65cm, SE = 0.22) had a slightly longer wing chord compared to when they were first captured ($\bar{x}$ = 27.75 cm, SE = 0.21). Overall, males were much heavier ($\bar{x}$ = 916.99 g, SE = 8.17, $n = 73$) than females ($\bar{x}$ = 526.66 g, SE = 22.26, $n = 5$) and had longer and larger bills, tarsus, toe length and width and also longer wing chord (Table 3).

Habitats at Foraging Sites

The global model had good fit ($\chi^2 = 4.642$, df = 8, $p = 0.795$) and the data did not appear to be overdispersed ($\hat{c} = 1.104$). The best supported model for predicting foraging sites of American bitterns was the "site-specific" model (Table 4). There were no competing models that were within 7 ΔAIC_c value (Table 4). Habitat features that had the most association with predicting foraging sites of American Bitterns included distance to small water openings(SMALL) and percent live vegetation cover (Table 5). The odds of predicting foraging site selection increased by almost 20% with every decrease of 10 m in distance from small water openings. Distances to small water pool was much further in random sites ($\bar{x}$ = 104.19 ± 20.08) compared to foraging sites ($\bar{x}$ = 20.70 ± 3.85); the odds of a foraging site being selected by a bittern increased slightly with increase in percent live vegetation. The average and range of distances to small water openings were much smaller in foraging sites compared to random sites (Table 6). However, percent live vegetation cover at foraging sites compared to random sites was not much different (Table 6). Furthermore, foraging sites were more highly associated with *Carex* dominated areas relative to cattail and other miscellaneous species such as *Juncus,* reed canarygrass, or *Phragmites* stands.

DISCUSSION

Information on home range and habitat selection of American Bitterns is scarce, compared to other wetland birds. The ten-year study at Agassiz NWR is important in that it began with the basic technique of developing trapping methods, then radiotelemetry and determining movement and home range sizes, then habitat selection of foraging and nesting sites.

Radio-telemetry

The goal for the radio-telemetry work was to obtain a larger sample size to calculate a more accurate and reliable home range sizes for the American Bittern at Agassiz NWR, and also to quantify foraging habitats during the post-breeding season to inform management and conservation of American bittern habitats. The variability in home range size from the two different data sets (1994 – 1997 and 2000 – 2001) may be attributed to a variety of factors ranging from differences in tracking and receiving systems used, telemetry errors, observers, and sample sizes to habitat and climatic conditions among years. In southcentral MN in the Prairie Pothole region, Laney (2003) found American Bitterns had an average home range of 156.73 ha (n = 16). On the other hand, Armour's (2002) estimates of home range sizes ($\bar{x}$ = 42.84 ha, core area $\bar{x}$ = 3.69 ha, n = 6 using Adaptive Kernel and $\bar{x}$ = 17.06 ha using MCP; SE = not reported) were much smaller in the wild rice paddy complex in northcentral MN. A study in Italy on a closely related bittern species, the Eurasian Bittern (*Botaurus stellaris*), revealed smaller home range size (core area = 31.7 and 95% UD = 215.7 ha, n = 11 males, 1 female) during the post-breeding season (Puglisi et al. 2003) compared to the American Bittern. Furthermore, Eurasian Bitterns also exhibited site fidelity within year and among years. The study area in Italy was small (800 ha of marsh area) compared to the marsh complex at Agassiz.

Several reasons that may have contributed to larger home range size at AGNWR include: 1) large wetland complex compared to other study sites of American and Eurasian Bitterns and 2) water-level management conducted at the refuge. Large wetland complexes such as AGNWR have more available habitats and can accommodate more American Bitterns. When more habitats are available, birds tend to move more, thus contributing to larger average home range size. Another factor that attributes to larger home range size of American Bitterns is water management scheme conducted by refuge staff. In 2000 and 2001, precipitation patterns were similar, increased water depth in March, peaked in mid-June and tapered off in mid-July. Bitterns were first captured and radio-tagged throughout May and early June, when water level was rising and flooding areas that may not have flooded previously, in turn, creating more foraging sites. These newly flooded sites were typically shallow areas that were presumably very productive, containing tadpoles, small fish, crayfish, and dragonfly larvae. However, beginning in May and continuing throughout the summer, drawdown activities by refuge staff resulted in birds being attracted to concentration and exposure of food resources as a result of the dewatering of wetland units. For instance, in 2000, only one unit, Agassiz Pool was dewatered and bittern home range sizes were smaller compared to 2001. Although Agassiz Pool was a large pool (ca. 4,000 ha), the wetland was >1 m deep and open in large portions of the unit, in other portions are deciduous tree islands, dry Phragmites and sedge meadows, but little gradual shoreline or edges. Thus, bitterns in 2000 were located on smaller wetland units that were adjacent to Agassiz pool, perhaps taking advantage of the natural dewatering process of wetlands. In 2001, seven wetland units were drawn down, and because pools were smaller, food resources were more concentrated and accessible to bitterns, thus American Bitterns were located using more wetland areas.

Habitat requirements during the post-breeding period, which for male American Bitterns is also the molting period, includes areas that are large enough to support shelter from predators and adverse climatic conditions and areas that have sufficient food resources. In previous studies on American Bitterns at Agassiz NWR, Azure (1998) and Brininger (1996) suggested that bitterns moved into the interior of the wetlands because it provided more protection from predators (presumably mammalian) and these areas were in deeper, more permanent water during the summer months, where prey items should be plentiful. I did not observe American Bitterns moving into interior portions of the marsh. In fact, they were located more along edges and ditches, as indicated by the models and personal observations.

Effects of Transmitters on American Bitterns

Two male radio-transmittered bitterns were confirmed dead during the period of this study. Many instances of observing radio-collared birds in flight indicated that birds were able to fly normally. Also, individuals that were recaptured, either the following year or during the same trap season showed birds in good condition. Two recaptured birds developed calluses on the back of the necks from the herculite collar and the breast feathers of two different recaptured birds were frayed at the ends resulting from the transmitter package rubbing on the feathers. In a study that was done in conjunction with this one (J. Toepfer and G. Huschle, unpublished data), male American Bitterns that were fitted with satellite transmitters, which were approximately 30% heavier, had a survival and returning rate of 60% (n = 20) and the remaining eight (40%) died on the wintering ground. This information provides supporting evidence that transmitters did not greatly affect the behavior and nesting efforts of American Bitterns, because male American Bitterns were captured where they were calling during the breeding season, the majority stayed in the captured location, which were areas where nests were found during searches. Of the five females, three that were radio-tagged in 1999 behaved as if they were brooding and rearing young as they were observed making feeding trips throughout the summer. One female that was captured on the nest in 2002 abandoned it two days after it was recaptured, probably because it was sensitive to disturbance from this possible re-nesting attempt after the flooding event.

Site Fidelity

Our data show that site fidelity in American Bitterns was relatively high when given natural survival rates of wetland birds. Nearly 13% (9 of 70) of the captured birds returned to the original capture site. In the Prairie Pothole Region in southcentral MN, four of 13 (31%) male American Bitterns returned to the study area (2003). Documented return rate for male Eurasian Bitterns was 27% (3 of 11 males) in a study in Italy by Puglisi et al. (2003).

Foraging Habitats

Results of the analyses partially supported the original hypothesis that foraging sites of American Bitterns were associated with small water openings, which is related to percent open water. However, the covariates percent open water, by itself, and distance to edge were not helpful habitat features in predicting foraging sites of American Bitterns. Although food resources were not examined in this study, based on previous experience and observations, numerous American Bitterns were flushed and frequently observed foraging along agricultural ditches and along edges of wetland dikes and borrow ditches where water depth was shallow. Radio-tagged birds that were flushed typically occupied areas adjacent to small open water, and more likely in *Carex* stands versus *Typha* stands.

Results of the modeling efforts provide good insight into the complex but opportunistic nature of American Bitterns, which occupy habitats ranging from dry grassland, to sedge meadows, to emergent marshes. Most herons (including bitterns) of the world do not feed in their nesting territory (Kushlan and Hafner 2000) and their nesting sites typically have adequate water stability and materials to support and construct nests, and is accessible to feeding areas within foraging range (Kushlan and Hafner 2000). In freshwater wetland systems, the most critical factors to herons and bitterns are presence of surface water and availability of suitable prey organisms, which includes a wide variety of wetland types (Kushlan and Hafner 2000). Therefore, it is essentially simpler to characterize wetlands that are not suitable for bittern foraging areas, which include areas that are too deep, too salty, or too acidic to either have a sufficient prey base or are not economically within foraging range (Kushlan and Hafner 2000).

The highly variable wetland and grassland types used by bitterns present a difficult challenge in developing predictive models of foraging sites of American Bitterns. More specific habitat characteristics that are important to bitterns are areas with sufficiently “less thick herbaceous vegetation” (Kushlan and Hafner 2000), shallower and drier habitats where sufficient prey are accessible. At the study site, these habitats included artificial habitats such as ditches adjacent to the marshes at Agassiz NWR and small water openings in sedge (*Carex*) meadows. Kushlan and Hafner (2000) mentioned that ditches are perhaps the most overlooked habitat of herons and bitterns. Water levels in ditches fluctuate seasonally and these ditches are perhaps abundant in food resources ranging from small fish, mammals, amphibians, and invertebrates. Ditches that are within and adjacent to Agassiz NWR are flooded during the spring season (March – June) by run-off from refuge drawdown activities and by drainage from farmers fields. During the summer time (June-August) these ditches are shallower, thus food resources are more accessible to foraging bitterns. In fact, Armour (2002) reported that male and female radio-tagged American Bitterns were observed foraging along ditches and stated that male bitterns had small home range sizes because they stayed in the same area or ditch for days and weeks at a time. In a study of Eurasian Bitterns, water depth was the only significant predictive variable for the presence of male bitterns, where it was negatively affected by increasing water levels (Adamo et al. 2004). Old and young stem density and stem diameter of emergent vegetation (primarily *Phragmites australis*) were not good predictors of presence of Eurasian Bitterns. Dominant vegetation species is perhaps not as important as water depth because water or hydro – period affects vegetation structure, species, and food availability (Weller 1999, Adamo et al. 2004).

IMPLICATIONS

Home range of American Bitterns can vary significantly from home range estimators, equipment used, sample sizes, variation in habitat conditions, and because of the opportunistic nature of the species. For these reasons, it is advisable to obtain home range estimates in site specific locations, depending on the objective of the home range study. The size of the home range appears to be greatly associated with habitat conditions and hydrologic regimes within season. This aspect of habitat association with home range size needs to be investigated further. In particular, specific covariates that may influence home range size and American Bittern movement patterns during the breeding and post-breeding (molt) season should include those that I have found in the modeling process.

Results of the modeling work, which suggests American Bitterns forage near small water openings and selected more live vegetation areas provide good information to guide future investigations into foraging behaviors. American Bitterns are wetland obligate birds like other Ardeidae because they depend on wetland habitats to at least complete one life history cycle. The primary life stage on which American Bitterns depend on wetlands is molting, a vulnerable period when bitterns are semi-flightless. However, it appears that for most other life stages, they appear to be the most opportunistic and generalist of all the Ardeidae, not depending solely on wetland habitats nor on grassland habitats as they nest and forage on either wetlands and/or uplands. When habitat requirements and resource selection are examined in this perspective, the ideal habitat management practices for American Bitterns would be to provide a complex of wetlands that functions naturally (Weller 1999), with shallow emergent marsh interspersed with open-water areas, with gradual sloping edges that provides sufficient cover and water depth for accessibility to food. Also, a complex of grassland and wetlands that ranges from upland grassland, to sedge meadows and emergent wetlands, both complexes should have water levels at different stages that facilitate foraging activities and nesting activities. The infrastructures (water control structures, levees/dikes, and ditches) that were constructed to hold or move water presented great challenges to refuge staff to manage shallower water levels and to emulate natural water level fluctuations. The consequences are deep water (0.5 - 3 m) areas and drier, rank monotypic stands of emergent vegetation areas, with very little diverse shallow (<0.5 m) wetlands.

Further insight that would benefit the conservation and management of habitats for American Bitterns involves examining effects of restoring the hydrologic regime of wetlands, reducing rank stands of *Typha* and *Phragmites*, creating shallower, open water to vegetation interfaces where accessibility to food resources is facilitated.

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Table 1. Home range of male American bitterns at Agassiz National Wildlife Refuge, in northwest Minnesota, using fixed kernel estimates with least-squares cross-validation (LSCV) bandwidth selection method. (*Locations obtained from W. Brininger and D. Azure who used different radio-tracking methods from this study).

Year	***n***	**# of locations ($\bar{x}$)**	**50% Core area (ha)**	**95% Home range (ha)**
1994-1997*	10	39.80 (±2.05)	28.68 (±5.18)	147.06 (±22.93)
2000-2001	15	43.53 (±1.68)	18.08 (±6.38)	109.28 (±38.47)

Table 2. Distances moved by recaptured American bitterns at Agassiz National Wildlife Refuge, northwestern MN, 1996-2001 (Azure 1998[1], G. Huschle @ Agassiz NWR[2]).

Bird #	First captured		Recaptured		Distance moved
	Year	Location	Year	Location	
2397-82004[1]	1996	?	2000	E. Dahl Pool	?
2397-82006[1]	1996	?	1998[2]	E. Dahl Pool	?
2397-82006		E. Dahl Pool	2001	E. Dahl Pool	0
2397-82039[2]	1998	SW East Pool	2000	SW East Pool	0
2397-82057[2]	1998	SE Pool 8	1999	SW Pool 8	1.09 km
2397-82057		SW Pool 8	2000	SE Pool 8	1.09 km
2397-82060[2]	1998	SW Agassiz Pool	2000	SW Agassiz Pool	0
2397-82088	2000	SW East Pool	2002	Maakstad, Agassiz Pool	1.96 km
2397-82091	2000	Hairpin Turn, Agassiz Pool	2001	Hairpin Turn, Agassiz Pool	0
2397-82098	2000	NW South Pool	2001	NE Farmes Pool	3.00 km
947-79440	2001	Maakstadd, Agassiz Pool	2002	SC Agassiz Pool	2.28 km

Table 3. Measurements of American bitterns captured in northwest Minnesota (*n* = 69 at Agassiz NWR, *n* = 2 Twin Valley, MN) and central WI (*n* = 5 at Horicon NWR) between 1999 and 2002.

Variables	**Female**			**Male**		
	$\bar{x}$	**SE**	***n***	$\bar{x}$	**SE**	***n***
Weight (g)	525.66	22.26	5	916.99	8.17	73
Bill length 1 (cm)	4.89	0.08	7	5.51	0.03	73
Bill length 2 (cm)	6.38	0.08	7	7.16	0.04	73
Bill length 3 (cm)	6.64	0.09	7	7.67	0.05	71
Bill width 1 (cm)	0.90	0.02	7	1.01	0.01	74
Bill width 2 (cm)	1.33	0.07	7	1.50	0.02	74
Head width (cm)	2.43	0.06	7	2.58	0.03	74
Tarsus joint (cm)	8.21	0.27	7	9.50	0.04	70
Tarsus entire (cm)	9.25	0.19	7	10.54	0.05	61
Toe length (cm)	6.80	0.10	7	7.71	0.05	74
Toe width (cm)	0.44	0.04	7	0.40	0.001	72
Nail length (cm)	1.99	0.06	7	2.15	0.16	74
Nail width (cm)	0.26	0.006	7	0.64	0.34	73
Tail length (cm)	8.03	0.35	4	9.04	0.07	73
Wing (cm)	25.09	0.27	7	28.24	0.17	74

Table 4. Statistics from models of data from radio-tagged American Bittern foraging sites at Agassiz National Wildlife Refuge in NW Minnesota, 2000-2001 (*parameter explanation = see text; [a]K = # of parameters includes intercept.

Model	**Predictive variables**[a]	**K**[b]	**AIC$_c$**	**Δ AIC$_c$**	w_i^c	**% concord.**	**Correct class.**
SITE-SPECIFIC	H20 LIVE SMALL	4	196.80	0.00	0.976	72.3	70.9
EDGE	LARGE LAND EDGE	4	204.22	7.41	0.024	69.6	63.3
COVER-EFFECT	HT DEN	3	212.57	15.76	0.000	61.4	57
NULL	Intercept	1	214.07	17.27	0.000	91.8	82.9

aOPEN = % open water area within 10-m of foraging/random site; LARGE = distance to large water openings (>0.4 ha); SMALL = distance to small (<0.4 ha) water opening; LAND = distance to nearest land; EDGE = distance to vegetation edge; HT = average vegetation height within 4 -m of foraging/random sites; H2O = average water depth within 4-m of the foraging/random site; DEN = relative density of dominant vegetation within 10-m of foraging or random site, classified into four categories (0 = no vegetation, 1 = rank [water not visible through base of stems at water level and one cannot easily push hands through the stems], 2 = moderate [anything between category 1 and 3], 3 = sparse [water easily visible through base of widely scattered stems]); DOM = dominant vegetation within 10-m of foraging or random sites.

bK = number of parameters in the models.

*c*AIC = w_i weight of evidence for model *i*.

Table 5. Habitat parameters and profile-likelihood odds ratios and confidence intervals for the most supported models predicting foraging sites of radio-tagged American Bittern at Agassiz National Wildlife Refuge in NW Minnesota, 2000-2001.

Model	Predictor Variables[a]	K[b]	β	Confidence Interval (β)	Odds Ratio	Confidence Interval (OR)
SITE-SPECIFIC EFFECT	INTERCEPT	4	-1.290	-2.481, -0.136		
	H2O (cm)		-0.018	-0.038, 0.001	0.982	0.962, 1.001
	LIVE (%)		0.015	0.000, 0.032	1.016	1.000, 1.032
	SMALL (m)		-0.016	-0.027, -0.007	0.984	0.973, 0.993
EDGE-EFFECT	INTERCEPT	4	-0.199	-0.858, 0.490		
	LARGE (m)		-0.002	-0.003, -0.001	0.998	0.997, 0.999
	LAND (m)		-0.001	-0.002, 0.001	0.999	0.998, 1.001
	EDGE (m)		0.000	-0.001, 0.002	1.000	0.999, 1.002
COVER-EFFECT	INTERCEPT	3	-0.593	-1.215, 0.037		
	HT (cm)		-0.012	-0.022, -0.003	0.988	0.978, 0.997
	DEN-1		-0.055	-0.428, 0.314	0.897	0.425, 1.872
NULL	INTERCEPT	1	-1.200	-1.521, -0.886		

Table 6. Mean values of three habitat features of biological importance at foraging sites of American Bittern, compared with random sites and used in foraging habitat selection models, collected at Agassiz National Wildlife Refuge, 1999-2001.

Habitat Features	Foraging sites n = 65			Random sites n = 93		
	Range	$\bar{x}$	SE	Range	$\bar{x}$	SE
SMALL (m)	0 – 150	20.70	3.85	0 – 1000	104.19	20.08
LIVE (%)	9 – 100	71.60	2.60	0 – 100	68.98	2.56

CHAPTER 3

A MONITORING PROGRAM TO DETECT CHANGES IN OCCUPANCY OF MARSH BIRDS IN RELATION TO HABITAT MANAGEMENT

ABSTRACT

An effective and efficient bird monitoring program is challenging to develop, given the different level of intensity of monitoring and the statistical vigor that monitoring programs must meet to effectively address specific objectives. I evaluated the marsh bird monitoring program established at Agassiz National Wildlife Refuge (AGNWR) within the framework of the three primary elements (WHY, WHAT, HOW) described by Yoccoz et al. (2001). The objective (WHY) of the monitoring program was to monitoring long-term changes and habitat management of marsh bird species at AGNWR. A set of *a priori* models were developed and the program PRESENCE 2.0 was used to model occupancy and detection probabilities of five marsh bird species with wetland management actions (wetland drawdown and prescribed burn) and wind speed as covariates. Results show that occupancy probability varied among surveys within year and among year, but occupancy rate was not associated with management actions. Estimated occupancy rates for American Bittern were highest (ψ = 0.82 – 0.96) and lowest for Least Bittern (ψ = 0.06 – 0.40). American Bittern and Sora had the highest estimated detection probabilities (p = 0.85, 0.83, respectively) compared to Least Bittern (p = 0.00 – 0.58) and estimated detection probabilities varied by surveys for American Bittern, Least Bittern, and Sora, by survey and wind speed for Pied-billed Grebe, and by year for Virginia Rail. Estimated occupancy probabilities were not related to the number of years since the target wetland was drawn down or burned for any of the target species.

General management actions information, such as the number of years since burn, without more specific habitat information (species composition, water depth, vegetation height and density) was not helpful in evaluating site occupancy rates of marsh birds. Recommendations to improve monitor changes in occupancy probabilities in relation to habitat management include clearly stated objectives and updating *a priori* models, developing better assessment techniques for habitat conditions at survey points and designing sampling techniques to include probabilistic methods of selecting survey points located along roads and also in interior of wetlands.

INTRODUCTION

An effective and efficient bird monitoring program is challenging to develop and implement. Thompson et al. (1998:3) defines monitoring, in its most general form as "*a repeated assessment of status of some quantity, attribute, or task within a defined area over a specified time period.*" Implicit in this definition is that the monitoring program will detect important changes in the status of the state variable of which the program was developed (Thompson et al. 1998). Monitoring serves four primary roles: 1. Evaluation of management objectives, 2. State-dependent decision – to assess current state of the system to determine which action to take, 3. Learning - to increase understanding of the ecological dynamics and the effects of management actions, 4. Future Modeling – to develop new system models (Kendall 2001). In ecological studies, monitoring is referred to in various contexts, from gathering baseline ecological information, such as to determine a species presence/absence in a particular area or habitat, and to assessing the effectiveness of management programs (Thompson et al. 1998). Monitoring is also referred to in the context of “science-based or “management-based” monitoring, depending on the underlying objectives (Yoccoz et al. 2001, Coordinate Bird Monitoring team 2004). The challenge is to sort through the myriad of publications and discussions on monitoring and develop a program that effectively and efficiently meets the program’s objective(s).

Over the last two decades, there are a number of publications on monitoring issues that discuss the relevance of index estimations, abundance, population trend, sampling design (Seber 1982, Thompson et al. 1998, Williams et al. 2002, Royle and Nichols 2003, MacKenzie et al. 2006) and a number of analysis programs have been developed to facilitate data analyses (see http://www.mbr-pwrc.usgs.gov/software.html, http://www.phidot.org/software/, http://www.warnercnr.colostate.edu/~gwhite/ software.html). Further, a recent influx of publications emphasizes the necessity of detection probability in monitoring programs (MacKenzie et al. 2002, Royle and Nichols 2003, MacKenzie et al. 2006, Bailey et al. 2007), and analysis programs (i.e., programs PRESENCE and GENPRES) were developed to facilitate analyses. Additionally, a number of biometricians, scientists, biologists and working groups (Williams et al. 2002, Coordinated Bird Monitoring Group 2004, Holthausen et al. 2005, Nichols and Williams 2006) urge that monitoring programs are conducted in the context of an informed decision making process to guide management, because monitoring and the decision process are critical components of adaptive management (Williams 2003).

Monitoring is an integral component of a resource biologist’s job. However, the problem is that most of these monitoring programs are inadequate in many ways. These inadequacies have been identified in several published papers. Most notably are those by Yoccoz et al. (2001) and Nichols and Williams (2006), which identify the inadequacies of clearly thinking about the objective of the monitoring program – essentially, "WHY" monitor? (Yoccoz et al. 2001). Biologists often focus on a form of omnibus or surveillance monitoring (Nichols and Williams 2006), or convenient or opportunistic monitoring where survey points commonly are placed along roadsides or in areas where objects are more highly detected. Or worse, monitoring for monitoring sake in hopes of someday perhaps the data will be analyzed and that some meaningful results appear. The disconnect that exists in the "investigatory process" as emphasized by Skalski and Robson (1992:3), where the process should be "*an unbroken flow from the statement of the research objectives to the design, analysis, and interpretation of results…*" results in wasted time, effort, and money.

Given the range of responsibilities in the current state of reduced budgets and staffing limitations it is critical to ensure that the time invested in monitoring activities is well spent. On National Wildlife Refuges (NWR), where monitoring and managing habitats for migratory birds in particular, is of great interest, biologists conduct many surveys as part of monitoring programs. A growing number of NWRs are implementing surveys for marsh bird monitoring. Currently, approximately 100 NWRs in the contiguous 48 states are conducting marsh bird surveys. One of the NWRs is Agassiz NWR (AGNWR).

In this paper, I evaluated the marsh bird monitoring program on AGNWR in the context of the three main elements (e.g., *WHY, WHAT, HOW*) of a monitoring program as discussed above. I examined the pilot data collected from 2000 - 2003 putatively to develop an effective monitoring program that meets refuge objectives. Using this case study, I will first describe the "initial" *WHY*, *WHAT*, and *HOW* for marsh bird monitoring on refuges. Then, I will evaluate the initial *WHY*, *WHAT*, and *HOW* in terms of the guidance of Yoccoz et al. (2001) and Nichols and Williams (2006). Lastly, I will re-examine how well the *WHY*, *WHAT* and *HOW* provide recommendations for future monitoring program for marsh birds.

1) *WHY* monitor? Most programs lack clear and explicit objective (s). There are two categories of objectives: scientific objectives focus on gaining reliable knowledge of the behavior and dynamics of a system, whereas monitoring for management objectives identifies the system state (e.g., population size, available habitat, vital rates, etc.) and provides useful information on a response to management actions, which feed into informed management decision making (Nichols and Williams 2006). A monitoring program for management objectives considers *a priori* hypotheses of management effects, which contain explicit variables, such as density or demographic rates of focal or indicator species and groups, along with habitat variables. Additionally, if the objective is to understand the processes behind the observed changes in the managed system, monitoring the rates of change of some system state variables, such as the rate of species loss or the proportion of new species, can be informative, but may be a long-term monitoring because of the inherent variability in natural systems as well as the behavioral responses to this variability on a daily, seasonal, and long-term basis (Nichols and Williams 2006).

2) *WHAT* should be monitored? Once the objectives of a monitoring program are clearly identified, the state variables to be monitored or measured will be determined by the objectives. For monitoring programs to address management objectives, these state

variables would be population size, trend, and habitat conditions, such as number of wetlands, percent vegetation cover, water depth, etc.

3) *HOW* should monitoring be carried out (survey design)? Having explicit objectives and state variables to measure will facilitate the sampling or survey design. In designing a monitoring program, two primary sources of error commonly ignored are 1) detection error, which occurs because many surveys assume that individuals of all species are detected equally well and that all species present are detected, but in fact, few survey methods permit the detection of all individual animals, or even all species of animals, within the survey area (Yoccoz et al. 2001). To obtain unbiased parameter estimates detection probabilities must be estimated (Mackenzie et al. 2006). The second potential source of error is spatial variation: this error is common among monitoring programs because of the inability of a monitoring program to survey large areas completely, but the desire to draw inferences to the large areas exists. To account for spatial variation, sample locations need to be selected to permit inferences to that larger area of interest. For instance, even though only a subsample of locations in the area has been surveyed, the intent is often to make statements about the entire area of interest, such as in the entire emergent wetland unit on a particular National Wildlife Refuge.

WHY monitor marsh birds?

Many inconspicuous marsh bird species in North America have undergone population declines, presumably because of decline in wetland habitats. Although the dataset is small, the Breeding Bird Survey (BBS) provides the best available long-term data on population trends: Soras (*Porzana carolina*), declining surveywide at 0.5%

annually, (P = 0.39, n = 509) and in the USFWS midwest region at -2.5% annually (P = 0.09, n = 86); American Bitterns (*Botaurus lentiginosus*), declining surveywide at -1.5% annually (P = 0.07, n = 623), USFWS R3 at -5.0% annually (P = 0.01, n = 116); Pied-billed Grebes (*Podilymbus podiceps*), increasing at 1.2% annually (P = 0.19, n = 505), USFWS R3 declining at -2.7% annually (P = 0.07, n = 86), and BBS has insufficient data to project trends for Least Bitterns (*Ixobrychus exilis*) and Virginia Rails (*Rallus limicola*) (U. S. Geological Survey 2007a). The BBS dataset for these species are flagged with "important deficiency," indicating "very low abundance" (<0.1 bird/route) and "very small sample size" (<5 routes) or with "deficiency", which means the data reflect "low abundance" (<1.0 bird/route) and "small sample size" (<14 routes; U. S. Geological Survey 2007b). A factor that contributes to the low abundance and small sample size of marsh birds is that BBS survey routes do not adequately sample wetland habitats. Hence, besides presence-absence information from incidental observations and state atlases, data on marsh bird status are essentially lacking. Because refuges were established and most were mandated to protect manage wetland and grassland habitats and migratory birds, including marsh birds, I, like many refuge biologists, set out to survey marsh birds in an attempt to fill some of the data needs.

The goal of marsh bird monitoring on AGNWR was to establish a long-term program to determine how marsh birds respond to wetland management activities. Objectives and questions include:

A. Estimate relative abundance of five marsh bird species at AGNWR.
    a. Relative abundance was defined as the average number of birds detected at a point or route over time (within year among surveys,

across 4 years.). This information was used to determine what species were present and obtain a rough estimate of, relatively, how many were present in a wetland unit.

B. Estimate annual population trend at survey points and across survey points.

C. Determine relative distribution of marsh birds in the wetlands on the refuge: in which wetlands are marsh birds found?

D. Determine habitat management associations with marsh bird relative abundance at points/routes.

E. When (time of day and year) is the best time of detection?

F. Are there sufficient survey points (sample units) and precision to establish a long-term monitoring program to detect meaningful changes in population over time?

STUDY SITE

This study was conducted in northwestern Minnesota, at the AGNWR, located in Marshall county, and Red Lake Indian Reservation, located in Clearwater county. AGNWR is part of a large complex of land specifically managed for migratory birds by the USFWS and the Minnesota Department f Natural Resources (MNDNR). The refuge comprises 24,888 ha of which 15,135 ha are impounded wetland habitats, including permanent, semi-permanent, sedge meadows, and raised bog; 4006 ha aspen woodland, 4715 shrubland (primarily *Salix* spp.), and 761 ha grassland habitats. There are 26 impoundments on the refuge, each has a control structures used to manipulate water levels. These impoundments range in size from approximately 75 – 4000 ha (U.S. Fish and Wildlife Service 2005). Immediately adjacent to Agassiz NWR on the southern, eastern, and northern boundaries are the Elm Lake (6356 ha), Eckvoll (2632 ha), and Thief Lake (22,267 ha) Wildlife Management Areas (WMAs), which are composed mainly of permanent and semi-permanent wetland habitats.

Prior to refuge acquisition in 1937, many attempts were made to convert the muck land into arable agricultural fields by developing a series of drainage ditches. However, these drainage ditches did not sufficiently drain the land because of vegetation growth in the bottom of the ditches (U. S. Fish and Wildlife Service 2005). Climatic conditions at the refuge is typical of the region, where variations in temperatures are wide and extreme, having severe winters with moderate snowfall and annual precipitation of approximately 56 cm. Because of the variation in temperatures and rapid snowmelt in the spring combined with the flat terrain, flooding occurs frequently and is often detrimental to the farming efforts in the area (U. S. Fish and Wildlife Service 1978).

AGNWR lies on the southeastern portion of the prehistoric Glacial Lake Agassiz, within the geomorphic region classified as the Agassiz Lacustrine Plain, Red River Valley area (University of Minnesota 1980). Quaternary geologic sediment at AGNWR is Holocene peat, and at Red Lake Indian Reservation is lake-modified till (Hobbs and Goebel 1982). The landscape position in which AGNWR is located is characterized as shallow depressions on lake plain, and the soil series is mainly Cathro Haug, which consists of organic soils. The rooting zone (0.3 – 1.5 m) consists of muck and peat, and the substratum (1.5 - 7 m) is loam and sandy loam (University of Minnesota 1980).

The small lakes within AGNWR were formed in depressions in the bed of Glacial Lake Agassiz that were not entirely filled with sediment (Minnesota Conservation

Department 1968). A landscape feature on the refuge that reveals the substratum underlying the refuge is a sandy beach ridge located along the northwestern portion of the largest impoundment on the refuge (Agassiz Pool). Wave-action from glacial activities deposited the sand which formed the beach ridges that lie within AGNWR. These geologic formations and anthropomorphic alterations of the landscape shaped a complex of managed wetlands, raised bog, and grassland within the refuge boundaries that reflect past management dogma related to traditional development and management of habitats. The current conditions on the refuge resulted from the past history of development and management in combination with current practices. The site is now protected in the sense that it will not be converted to other uses, but managing habitats and emulating natural processes that are required by waterbird species is more problematic because of the drastic disruption of ecological processes. Developing monitoring programs for waterbirds and habitats within such a highly modified landscape will require much patience and is a daunting task.

Climate of northwestern MN is characterized by temperatures ranging from -35°C March to 36°C in August, with an average of 115 frost-free days (U. S. Fish and Wildlife Service 2002). Annual average precipitation over the past 30 years was 56 cm. During the duration of this study (1999-2003), annual average precipitation was 57.35 cm. Winter is relatively dry, with average annual snowfall of approximately 99 cm. The wettest months are June, July, and August. During 2002, almost 24 cm of rain fell between 9 and 11 June, raising water levels in wetland units of >1 m in some pools, causing major flooding in the area, including nesting habitats.

METHODS

Marsh Bird Surveys

Field methodologies in the North American Standardized Marsh Bird Monitoring Protocol (Conway 2005) were used to conduct marsh bird surveys. Three surveys were conducted each year during the breeding season at each of the 43 survey points, which were located along five “routes.” A “route” was a group of points, ranging from 4 – 12 survey points, along a refuge dike or refuge road, and each route provided immediate accessibility to the point. The starting point along the route was selected randomly and subsequent points were systematically located at 1.26-km intervals along the route. A call-broadcasting system was used to conduct the surveys, which included broadcasting a CD that when played in the Sony Sport Series boombox, consisted of a silent “passive” segment (3 or 5 min), followed by a “broadcast” segment of 30-sec of each species call and 30-sec silent period. In 2000 - 2002 the broadcast segment was followed by another passive period (Table 1). CDs were obtained from Dr. Courtney Conway, U. S. Geological Survey, Arizona Cooperative Research Unit, Tucson, AZ). The order in which bird species was broadcast was from softest sounding call to loudest call as suggested by Ribic et al. (1999). The CD player was placed on the hood of the vehicle (pickup truck or sports utility vehicle), pointing in the direction of the target wetland unit, and the observer stood approximately 2 m behind or to the side of the CD player. The CD player broadcast bird calls at a sound level of approximately 90 db at 1-m distance in front of the CD player. Because the survey protocol was still in testing phase, some adjustments were made among years to accommodate testing methods for the national protocol, and these adjustments included differences in length of passive and broadcast segments and distance recording (Table 1). The same four observers conducted surveys at each of their respective “routes” within and among years, with the exception of on rare occasions, I substituted. All observers were trained on bird identification (visual and sound) and practiced distance estimation prior to surveys. In addition, I conducted field check of observers, where she accompanied the observer and compared survey results.

Weather variables (temperature, cloud cover, wind speed, precipitation) known to affect bird response and detection were recorded at the first survey point. The history of the drawdown and prescribed burn activities of the target wetland units were obtained from refuge records.

Data Analysis

Occupancy and Detection Probabilities

The program PRESENCE 2.0 (Hines 2006) was used to model occupancy and detection probabilities based on detection histories, site covariates (prescribed burn history, drawdown history) and a sampling covariate (wind speed during survey) of each marsh bird species. Occupancy models that accounts for detection probabilities have the following assumptions: 1) within a given survey season, sites are assumed to be occupied by the target species (sites are “closed” to changes in occupancy, but this assumption may be relaxed if changes in occupancy occur at random); between seasons/year, changes in occupancy may occur due to processes such as colonization and local extinction; 2) detections occur independently at sites; 3) occupancy and detection probabilities are similar across sites and time, except when differences can be modeled with covariates, such as habitat characteristics; and 4) the target species is identified correctly.

Analyses below are intended to inform the development of a more robust monitoring program. For the following analyses, data from the surveys were not separated into passive and broadcast segments. These separations were for a different objective (to accommodate testing the national protocol) and should not effect the "pooling" of count data into 10-min surveys (Conway and Nadeau 2005., C. J. Conway, pers. comm.). In 2003, where the Black Rail (*Laterallus jamaicensis*) was added at the beginning of the broadcast segment, all bird observations detected during the 1-min segment of the Black Rail were excluded from these analyses, thus, making all surveys 10-min long. Also, I did not conduct analysis on different detection rates between passive and broadcast segments because there are sufficient data and published literature (Gibbs and Melvin 1993, Bogner and Baldassarre 2002, Lor and Malecki 2002, Conway and Gibbs 2005) that demonstrate broadcast calls increase detection rates of most marsh bird species.

A set of *a priori* candidate models for each of the five species (Table 7) was developed prior to analysis but after data collection. The candidate models included those that assumed a species had a constant occupancy rate ψ, occupancy varying across years, or occupancy that vary with drawdown (DD) or prescribed burn (PB). I also included models with linear (L) and quadratic (Q) effects ψ (DDL) and ψ (DDQ), ψ (PBL) and ψ (PBQ), respectively. Drawdown and prescribed burn covariates were transformed to quadratic term because from observations and published literature, marsh bird response to vegetation changes in a quadratic relationship – one of the reasons for drawdown and prescribed burn activities (Lor and Malecki 2006, L. L. Bailey, pers. comm.). Further, bird population trend is not typically a linear relationship, but rather quadratic (Thompson et al. 1998). Detection probability was either constant across year, *p* (.), varied among year, *p* (yr) and surveys *p* (SURV), or varied among surveys and wind speed, *p* (SURV + WIND). Models for Least Bittern did not include drawdown or prescribed burn effects as sample size was small. Models for Sora and Virginia Rail included *p* (YR+BURN-L) and *p* (YR+BURN-Q) because detection probability may be a function of the change in vegetation density due to prescribed burn activities. Colonization γ and local extinctions ε were modeled for each species. Local extinction and colonization processes may cause a form of temporal autocorrelation because wetland units that were occupied in one season are more likely to be occupied in the next season. Program PRESENCE exploits temporal autocorrelation in the calculations of local extinction and colonization rates (L. L. Bailey, pers. comm.), and analyses that incorporate the processes of local extinction and colonization are likely to provide more reliable results than those that do not (MacKenzie 2005). I used Akaike information criterion adjusted for small sample size (AIC_c) for model selection and chose the model with the smallest AIC_c value as the top model, and models within 7 ΔAIC value from the top model were considered candidate models that had sufficient support for the data (Burnham and Anderson 2002:71). Estimates of abundance for each species were obtained from program PRESENCE, using models developed by Royle (2004). In addition, I plotted frequency of response during different survey times to examine the best time for detection.

Allocation of Effort

In discussing survey design options, it is necessary to emphasize that imperfect detection of the target species results in substantial effects on bias and precision of parameter estimates (i.e., occupancy probabilities), and thus, reduce statistical power to detect trends in populations. Statistical power is a function of sample size, effect size, and significance level (α). There is a direct trade-off between power and significance level; more stringent significance value (e.g., 0.01) or precision results in lower power estimates. One way to achieve high power with stringent significance level is to have large samples (Williams et al. 2002). The precision in the estimates should depend in part on the cost associated with failing to detect a trend. In a monitoring program for endangered species for instance, one needs to design a monitoring program that is statistically powerful and has a high likelihood of detecting changes in populations (Gibbs and Ramirez de Arellano 2007). As part of the objective, the effect size (i.e., I want to estimate within 10% of the true value or SE = 0.05 of occupancy estimates) should be identified. Demonstrating trends reliably in survey data is challenging and have lead to a slough of designs and analysis methods (reviewed by Gerrodette 1993, Thomas and Krebs 1997, Gibbs and Ramirez de Arellano 2007). In this paper, I will examine precision in several scenarios dealing with tradeoffs between spatial and temporal replications (number of sites vs number of survey replications). These scenarios are optimized allocations of resources that field biologists and resource managers must deal with under budget and personnel constraints and are discussed in more detail in Field et al. (2005) and MacKenzie and Royle (2005).

I used results from my pilot data to calculate recommended sample sizes (MacKenzie and Royle 2005) using ψ and p estimates at specified precision (SE, CI or CV) levels for the monitoring program. Based on their simulations investigating the most efficient sampling design for a monitoring program using occupancy, MacKenzie and Royle (2005) recommended optimum numbers of survey replications for efficient designs. Using ψ and p estimates that I obtained for each species (American Bittern ψ = 0.89, p = 0.52,; Least Bittern ψ = 0.22, p = 0.17; Pied-billed Grebe = 0.88, p = 0.62; Sora ψ = 0.79, p = 0.59; Virginia Rail ψ = 0.67, p = 0.45) that MacKenzie and Royle (2005) recommended, the number of survey replicates for the five target marsh bird species was from 3 – 5 surveys/season. Hence, my calculations of sample sizes to obtain the specified precision were based on three and five survey replicates.

RESULTS

Marsh Bird Surveys

Over the 4-yr period, during the breeding season, 36 - 43 stations were surveyed, on average 3 times per season for a total of 491 points surveyed. Sora responded most frequently (n = 527, 33%), followed by Pied-billed grebe (n = 434, 27%), American Bittern (n = 432, 27%), Virginia Rail (n = 184, 11%), and Least Bittern (n = 31, 1.9%).

Occupancy and Detection Probabilities

American Bittern

The proportion of sites occupied for American Bitterns ranged from ψ = 0.82 – 0.96 among years (Table 2). The best approximating model suggests a constant occupancy rate, fixed at 1, and shows that detection probabilities (p = 0.05 - 0.85) was a function of year and survey windows (Tables 7 and 8). Other candidate models had ΔAIC_c >7 from the top model thus did not have sufficient support for consideration (Table 7). The detection probability-adjusted counts for American bitterns ranged from 54 – 76 birds detected each year across all survey points (Table 2). Management actions (neither drawdown nor prescribed burn) did not clearly explain variations in occupancy or detection probabilities. An overall 4-yr average estimate of abundance (λ) from Royle's (2004) model was 44 American bitterns (SE = 6.72, 95% CI = 30.82 - 57.18) across survey points.

Least Bittern

The total 4-yr responses from Least Bitterns were few (<2%), so only simple occupancy models were analyzed. The model that best fitted the Least Bittern data assumed a constant occupancy rate and detection probability that varied by year and survey windows (Table 7 and 8). Competing models included one where detection probability varied by year and survey windows, and the model that included constant occupancy rate, colonization, local extinction, and detection probability varying by year. Occupancy probability ranged from 0.06 – 0.40 among year, while detection probability ranged from 0.00 – 0.58 among surveys (Table 3). Detection probability adjusted-counts resulted in eight (in 2000) to 26 (2002) Least Bitterns detected (Table 3). Sample size was too small to fit models including management actions or to estimate abundance.

Pied-billed Grebe

Pied-billed Grebes comprised 27% of the total number of birds detected at survey points over 4 years, the second highest response rate recorded of the five target species. The best model assumed constant occupancy rate, colonization and extinction rates, and detection probability varied by survey windows and wind speed (Table 7 and 8). Adjusted counts estimated a detection of 49 – 68 Pied-billed Grebes each year (Table 4). Royle (2004) estimates of average abundance over 4 yrs was 49 individuals (SE =7.45, CI = 34.18 - 63.37; ψ = 0.68, SE = 0.06, 95% CI = 0.57 - 0.79; p = 0.46, SE = 0.031, CI = 0.40 - 0.52).

Sora

Sora was the most frequently detected of all the species (33%). The best approximating model was one that included detection probability varying among years and survey windows, while occupancy, colonization, and extinction rates were held constant; wind speed may have affected detection probability as well, as suggested by a competing model (Table 7 and 8). Variability among detection probability was highest in 2000 ($\bar{x}$ = 0.51, SE = 0.12) compared to other years (Table 5). Occupancy probability was highest in 2000 (ψ = 0.91, SE = 0.00) and lowest in 2001 (ψ = 0.68, SE = 0.00). Average counts of each year's survey, adjusted for detection probability indicated 62 – 83 Soras were detected per year. Average abundance estimate over 4-yrs was 43.55 individuals (SE = 6.65, 95% CI = 30.52 - 56.58).

Virginia Rail

Six competing models appear to have support for occupancy and detection probabilities of Virginia Rail (Table 7 and 8). Because it was not clear which model best represented the data, model averaging of the top three models suggested that occupancy was 0.67 and detection probability was 0.39. Two of the three top models suggested that detection probability varied among year and among surveys (Table 7). Adjusted average counts yield 28 - 39 Virginia Rails detected during 2000 – 2003 (Table 6). Average abundance estimate was 53 Virginia rails over the 4-yr period (SE = 11.60, 95% CI = 30.35 - 75.82).

As part of this pilot effort, I needed to determine the best period and the best time in the morning during the breeding season to conduct the surveys. Analysis shows that detection probability, where surveys (survey window 1) began during 6 – 12 May and survey window 3 ending around 8 – 13 June, was reasonable for all but the Least Bittern (Figure 1). For the Least Bittern, detection probability was highest ($p = 0.58$) during survey window 2 in 2001, when this survey was conducted two weeks later (6.12.01) compared to other years. Furthermore, detection probability for American Bitterns ($p = 0.05$) and Soras ($p = 0.19$) dropped precipitously during survey window 3 (6.27.01) in 2001, compared to survey windows 1 and 2, and survey windows 3 in other years (Figure 1). A summary of the raw counts (unadjusted for detection probability), showed the detection of all five marsh bird species was highest from 0500 to 0630, and then drops off progressively after 0800 (Figure 2).

Allocation of Effort

Results of precision and sample size calculation from occupancy and detection probabilities using MacKenzie and Royal (2005) formula showed that the difference in the number of sites between conducting 3 surveys vs. 5 surveys per site was substantial and change in variance (precision) of occupancy estimates changed the required number of survey sites significantly (Table 9). At precision level SE = 0.10 or effect size of 20% within the true value, the current monitoring program at AGNWR falls substantially short for Least Bittern, but was adequate or exceeded the requirements for the other four species of marsh birds (Table 9).

DISCUSSION

Generally, I believe the results of the analysis reflect my observations of the "relative" abundance of marsh birds on the refuge, with the exception of Least Bitterns. More Least Bitterns occupy the refuge wetlands than the numbers being detected during surveys because Least Bitterns typically stop calling initially after being "disturbed" (by broadcast calls) and thus miss detection by the observer (Lor and Malecki 2000, Arnold 2005, pers. obser.). Based on observations, I believe that Least Bitterns are more common than American Bitterns at AGNWR.

Occupancy and Detection Probabilities

Occupancy rates modeled for each year suggested that a high proportion of survey sites were occupied by American Bitterns, Soras, and Pied-billed Grebes compared to Least Bitterns and Virginia Rails. Detection probability and adjusted counts coincided with occupancy rates for the same species. From observations, territorial and courtship calls of American Bitterns, Soras, and Pied-billed Grebes are much louder than Least Bitterns and Virginia Rails, making them more detectible, and because detection probability is a function of occupancy rate, higher occupancy and detection probabilities are not surprising. In addition, survey dates may have missed the peak period of Least Bittern calls, resulting in low detections. MacKenzie et al. (2006) suggest that occupancy rates of 0.2 – 0.8 would provide reliable estimates for a monitoring program given a specified area. Thus, it appears that occupancy rates of four species are sufficient for a good monitoring program to detect changes in occupancy rates in this study site. To increase the occupancy and detection probabilities of "rare" species, such as the Least Bittern, MacKenzie and Royal (2005) and MacKenzie et al. (2006) suggested less intensive visits (i.e., fewer survey replicates) to more survey sites. Gibbs and Melvin (1993) and Bogner and Baldassarre (2002) showed that >9 visits per site were needed to detect >1 Least Bittern in Maine and New York, respectively.

Adjusted-counts

The detection probability adjusted-counts over the 4-yr period are good indices to the relative abundance of the five target marsh bird species. Based on the results, Sora is probably the most common of all five species, followed by Pied-billed Grebe, American Bittern, Virginia Rail, and Least Bittern. Few studies have examined detection probabilities of secretive marsh birds, particularly one using call broadcast (Gibbs and Melvin 1993, Bogner and Baldassarre 2002). Gibbs and Melvin (1993) calculated the probability of detection of the same five marsh bird species ranged from 0.56 (Least

Bittern) to 0.86 (Pied-billed Grebe) for broadcast surveys. A detection rate of 25.5% reported by Bogner and Baldassarre (2002) from radio-tagged Least Bittern during the nesting season in western New York was similar to the estimated rates of this study.

Effects of Habitat Management

Given the study design and duration of the study, it was not surprising that occupancy probabilities were not associated with the number of years since the target wetland was drawn down or burn for any of the species. General management actions information, such as the number of years since burn, without more specific habitat information (i.e., species composition, water depth, vegetation height and density) was not of sufficient detail to show relationship to the changes in species occupancy rates. The survey points were biased towards roadsides and marsh bird occupancy rates, also habitats were not representative of the habitats in the entire unit. In general, habitats along roadsides consisted of thicker vegetation and more shallow water compared to interior of the marsh, hence, management treatments (drawdown or burn) probably influence habitats along the roads differently compared to the interior portions of the wetlands. The duration of the study, four years, was probably not sufficiently long to detect a relationship between occupancy rate and habitat changes, particularly when sample sizes of marsh birds, namely Least Bittern and Virginia Rail, were barely adequate. Based on experience and observations, the pattern of marsh bird response to water and vegetation manipulation is as follows: American Bitterns, Soras, and Virginia Rails are more habitat generalists compared to Least Bitterns and Pied-billed Grebes, so management activities, such as drawdown or burn would have less effect on the former species than the latter species. American Bitterns, Soras, and Virginia Rails use thick vegetation and shallow water to dry areas, but Least Bitterns and Pied-billed Grebes are more restricted to deeper and open areas (Weller and Spatcher 1965, Lor and Malecki 2006), thus a shift in occupancy rates of these two species was expected from higher occupancy rate during the first 1 - 3 years since drawdown and prescribed burn and gradually decreasing with longer time since drawdown or burn as the vegetation recovers to “rank” stands. At AGNWR, given the large wetlands and level of historic disturbance and management of habitats, effects of habitat management on bird communities may take decades of repeated treatments to detect significant effects. In other words, a typical one-year or season treatment, such as drawdown or prescribed burn of surface vegetation may not have any measurable effect on the habitats or on bird communities.

Survey Windows and Survey Time

Detection probability during Survey 3 was lower in 2001 for American Bitterns, Pied-billed Grebes, and Soras because it was later in the breeding season. For American Bitterns, this pattern corresponded with observations and capture results (another portion of this study), which showed that American Bittern calls and response to capture tapered off during the end of May, and detection rates were highest during the nest initiation stage (Brininger 1996, Azure 1998, Lor, unpublished data). On the other hand, Least Bittern response was almost five times higher during Survey 2 (6.12.01) compared to other survey periods, within and among years. This result corresponds to the nest initiation time and observations that the Least Bittern is a later nesting species compared to other marsh birds (Bogner and Baldassarre 2002, Arnold 2005, Lor and Malecki 2006).

However, total response was small (*n* = 31 or 1.8%) compared to other species. In contrast to 2001, the spring rain event of 2002, which occurred during 9 - 11 June with 24 cm of rainfall, resulted in a higher detection probability during Survey 3 (6.13.02) compared to other years because nests that were initiated prior to 9 June were lost and birds were re-nesting, thus going through courtship again at that time.

Recommendations in the North American Standardized Marsh Bird Monitoring Protocol suggest conducting surveys 30 min prior to sunrise and ending four hours after sunrise, based on previous information on response rates of birds. However, there may be some geographic variations in response rates, even with the same species. The trade-off between conducting surveys for longer period in the morning is that more survey points can be surveyed but trading off higher detection rates. At AGNWR, detection rates dropped off noticeably after 0800, hence surveys should end approximately 0800 in order to obtain an effective detection probability. This effectively means that surveys of five species takes 10 min./survey point, estimating 5 min travel time between points along roads, approximately 8-10 points could be surveyed each morning. In less accessible sites or points, more travel time will be required and thus less points will surveyed with the time frame. As for survey dates, as suggested by the recent modification in the survey protocol (USFWS Marsh Bird Committee, unpublished info.), a 47-day survey window that includes 30 Apr. – 15 June would work best for all species, except perhaps for Least Bitterns, where detection probability may be better during the latter part of June (Lor and Malecki 2000, Rehm and Baldassarre 2006).

Allocation of Effort

As previously mentioned allocation of effort depends on the survey objective and finding a balance between precision and risk of the monitoring program, which means finding a balance between the required number of survey sites and the number of survey replicates per site. It has been demonstrated that increasing the number of sites (sample size) will improve the precision of the occupancy estimates (MacKenzie and Royle 2005, Bailey et al. 2007). However, advantages of more survey sites does not necessarily outweigh the benefits of increasing survey replicates because the bias associated with uncertainty due to imperfect detection will increase and could in effect negate benefits from surveying more sites (MacKenzie et al. 2006).

Biologists should determine specific objective for the monitoring program, set the precision level they are able to accept for the program, while considering the status and ecology of marsh birds and the consequences of the risk of the precision levels, then calculate the appropriate number of survey sites and replications. For instance, as mentioned above, marsh birds are inconspicuous and difficult to detect compared to other bird guilds, their population status (abundance and trend) is relatively unknown in their breeding and migration range, and given these reasons and localized information, most marsh bird species have a special protected status (e.g, threatened or species of special concern) in many states. Habitats at AGNWR is unique in that it consists of a large expanse of emergent vegetation and sedge meadows that are managed specifically for migratory bird species and the occupancy rates of the five marsh bird species are relatively high. Based on this knowledge, two main objectives at AGNWR of long-term monitoring population changes and effects of habitat management actions on the occupancy rates of breeding marsh bird species would probably be off to a good start with precision level at 20% within the true occupancy rate, thus conduct surveys at ≥ 48 survey sites, surveyed three times/site during the breeding season to account for American Bitterns, Pied-billed Grebes, Soras, and Virginia Rails. More than three survey replications per season may be required to obtain adequate occupancy estimates for Least Bitterns to meet an adequate precision level. Another option is to coordinate with other national wildlife refuges or conservation areas to establish as many survey points as possible to obtain a decent estimate of occupancy rates for Least Bitterns.

RECOMMENDATIONS

Preliminary analyses of the pilot data suggest several improvements for a better monitoring program for marsh birds. The most important one is to more critically re-examine the first of the three main elements presented by Yoccoz et al. (2001), and re-iterated many times in MacKenzie (2005), MacKenzie and Royle (2005), and MacKenzie et al. (2006), "*WHY*". Subsequent to forming clear and well thought-out objective(s) for monitoring marsh birds, then "*WHAT*" and "*HOW*" will be guided by *WHY* monitoring is needed.

WHY

Objectives summary: Assess relative abundance and distribution, assess management effects on marsh bird number changes, and to use the pilot data to guide establishment of a long-term monitoring program. The original objectives were ambitious for the amount of effort (staff number, time, and cost) and experience. A study designed for long-term monitoring program is drastically different from a study investigating the effects of wetland management on the number of marsh birds occupying wetlands during the breeding season, or for that matter, the effects of wetland management on breeding activities on marsh birds.

A long-term population monitoring program that occurs only on the refuge is valuable only from the perspective that the trend in bird abundance is linked to habitat changes, whether general, large scale change in habitat types, i.e., from sedge meadow to tall semi-permanent wetland types dominated by invasive cattail, or reed canary grass, or decline or increase in wetland habitats. In addition, information on bird trends on the refuge in relation to a larger landscape scale is more beneficial; comparing trends on the refuge to trends around the refuge, to other refuges, within the region or ecoregion (i.e., Bird Conservation Region, state, etc.) or to national-landscape scale is more informative than looking at trend on a single refuge. From the perspective of examining the effects of management actions on changes in marsh bird community during the breeding season, more critical thinking is needed at the level of detail that biologists and managers require to improve management of wetlands for the purpose of the marsh bird community. In most instances, an objective that is in the context of management effects on the wetland bird community as a whole, waterfowl, shorebirds, and marsh birds is the ultimate goal, because typically, biologists manage wetlands for the habitat communities required by birds to meet life history needs (breeding or migration), not typically aimed at any one species or guild (with exception of waterfowl at some stations). That is, unless, a refuge has an Endangered species that requires specific habitats to meet the objective of population recovery.

Although results of the data analyses are not conclusive, from observations and available wetland habitats, AGNWR contributes significantly to the breeding population of marsh birds in the region, state of Minnesota, and among national wildlife refuges. Thus, management of wetland habitats and monitoring marsh bird community, in addition to the current monitoring programs of waterfowl and shorebirds, is important to the refuge. With that said, below are general recommendations to the refuge:

***WHY monitor (objectives)*:**

Objective 1: Long-term monitoring of occupancy rates on the refuge: conduct long-term (10-20 yrs) monitoring program to detect changes in occupancy rates of breeding marsh bird species, within 20% of the true occupancy rate (SE =0.10 (ψ)), at $\geq$ 48 survey sites, surveyed three times/site during the breeding season. It would be ideal to cooperate with other refuges and wildlife management areas in a larger area or regionwide survey effort so that results can be compared more meaningfully and to obtain better occupancy estimates for Least Bitterns.

Objective 2: Investigate the effects of management actions, wetland drawdown and prescribed burn activities on changes in occupancy rates of marsh birds: A more specific investigation on how wetland management activities affect processes that influence occupancy rates of the target marsh bird species will provide valuable information into the adaptive management process of wetland management. Initially, perhaps a precision level of SE $\geq$ 0.20 for occupancy estimates and a change in occupancy rate of >50% should trigger management action to reverse the declining trend. Updating models to evaluate uncertainties in changes in occupancy probability, colonization, and local extinction with changes in wetland habitat characteristics due to management actions would be most informative for the refuge.

***WHAT* – metrics to measure for each objective**

Occupancy rates, as opposed to relative abundance are easier to obtain and should provide sufficient information to inform management. The state variable in which most managers and biologists are typically interested is "abundance" or some count of animals or plants. However, obtaining accurate counts or abundance is difficult, time consuming and potentially costly, thus the level or detail of information that is required to make management decisions needs to be clearly identified. Are occupancy rates and obtaining detection probability to adjust survey counts sufficient to take action if a trend in occupancy rates is detected? At AGNWR, changes in occupancy rates and adjusted counts in relation to changes in habitat characteristics due to management actions is appropriate for management purposes because refuge objectives are habitat-based, targeted at providing habitats to benefit a suite of wetland birds, from waterfowl, to marsh birds, or shorebirds.

The best approach to investigating relationship between bird number changes and management actions is to develop *a priori* models and then validating the models with real data. For instance, biologists drain wetlands to emulate the natural drying cycle of wetlands, and sometimes the drained wetlands are burned to rejuvenate vegetation, reduce vegetation structure and biomass, and to restore nutrient cycling to improve habitats for wetland birds. From this knowledge and information in the literature, *a priori* models would include:

For objective 1: Long-term monitoring of changes in marsh bird occupancy should include changes in vegetation composition at the macro-scale (at wetland unit level and at entire refuge habitat level) within and among wetland units. At this larger scale, covertype maps derived from aerial photos, in ArcGIS, could be used to measure the changes in proportion of vegetation species composition, water depth, open water areas, and then used as covariates in occupancy models. A long-term monitoring program is beneficial to compare AGNWR to other managed sites, such as other refuges or a group of refuges or wildlife management areas.

For objective 2: Counts of individuals of five target species using the North American Standardized Marsh Bird Monitoring Protocol and habitat characteristics in relation to management histories in each target wetlands should be recorded. Field methods for recording the number of birds detected at the survey point have essentially been finalized. However, the more challenging tasks of measuring habitat variables and recording survey variables, such as climatic conditions, time of detection, etc., that influences occupancy and detection probabilities are more challenging and needs to be examined more closely.

Changes in occupancy rates of marsh birds are affected by the changes in management actions: drawdown eliminates or reduces water level during the year of drawdown, gradually improves conditions (vegetation species and structure increase) in subsequent years, then eventually, decreases the required vegetation composition and structure. Occupancy rate is expected to drop during the year of the draw down, rise subsequently with recovery of vegetation composition and structure. Thus, measured variables would include vegetation species composition, vegetation density, height, water level, and distance to small water openings, along with counts of birds (for occupancy estimates). Global model: ψ = drawdown, burn, vegsp., veg. density, veg. ht, water level, vegsp.*density, vegsp.*ht., density*ht. Then, develop a set of candidate submodels from the global model that best explain changes in occupancy rate and the number of birds (adjusted counts) using the wetlands. Also, survey covariates that may affect detection probability should be recorded, which include wind speed, temperature, precipitation, cloud cover, and observer. The study design for this objective is more complex and a biometrician should be consulted!

HOW – Survey Design

A sampling design for any monitoring program needs to consider detection probability and need to account for errors over space and time (Yoccoz et al. 2001, MacKenzie 2005, Nichols and Williams 2006). Generally, because interest is in making inferences to the entire potential marsh bird habitats on the refuge, survey points should be placed in a random manner that includes available and potential marsh bird habitats, which would allow inferences to the entire area of interest. A stratified-random sampling method using ArcGIS was developed for refuges (or any sampling unit) to facilitate point selection, which incorporates random selection of points within each at roadside and non-roadside areas from a set of all potential points to permit comparisons of occupancy rates and detection probability of roadside and non-roadside points (Conway, Lor, Knutson, and Nelson, unpublished). Spatial variation will be accounted for by surveying birds at randomly placed survey points and temporal variation will be accounted for by conducting multiple surveys within and among years and by examining colonization and local extinction rates in the analyses (MacKenzie 2005). Additionally, a shorter survey interval should be considered to reduce the temporal variation in detection and occupancy probabilities. For instance, since peak detection periods for all five marsh bird species fall within the end of May (survey window 2) and early June, perhaps all three survey windows/replications should be conducted within a two-week period between late May and early June. Further analysis of existing data from this study and marsh bird monitoring programs on other national wildlife refuges should provide further and valuable insight into considering this option.

Allocation of Effort

Based on results from preliminary analysis of pilot data, AGNWR's objectives should be to:

1) Conduct long-term (10 - 20 yrs) monitoring program to detect changes in occupancy rates of breeding marsh bird species, within 20% of the true occupancy rate (SE =0.10 (ψ)), at $\geq$ 48 survey sites, surveyed three times/site during the breeding season. These survey sites should be allocated in the stratified-random manner mentioned above. However, because the pilot data were biased towards roadsides, it would be ideal to allocate 48 sites along roadsides, and additional 11 - 48 points in less accessible areas for one year and re-analyze the data to obtain accuracy and guide long-term monitoring program. Collaboration with partners, such as other national wildlife refuges/wildlife

management areas is important for reasons of comparing trends and also obtaining sufficient sample sizes for difficult to detect species such as the Least Bittern.

2) A monitoring program to detect changes in occupancy rates of marsh birds in response to management actions that were geared towards increasing the occupancy rate in particular wetland unit or habitat type may need a more precise estimate because management actions, such as prescribed burns, are costly. Here, biologists and managers need to not only decide on the threshold of occupancy estimates at which they are willing to take action, but also decide on the precision of the estimates from the true value. Thus, at what threshold level (20, 30, 40, 50%) of decline in occupancy rate and at what threshold in precision in the estimates (10, or 20, or 40% of the true occupancy rate) will management action be taken to reverse the trend in decline of occupancy rate? Part of the answer to these questions is the objective of the refuge – whether or not management of marsh bird populations and in turn their habitats, is a high priori for the refuge. Given refuge goals and objectives identified in AGNWR's Comprehensive Conservation Plan (USFWS 2005), providing breeding habitat for marsh birds and continuation of the marsh bird monitoring are high priorities on the refuge. Thus, precise (i.e., SE = ≥ 0.20) and unbiased estimates of occupancy rates to inform management actions are good starting points. With a management objective such as this one, a sampling design in the context of bird response (occupancy rate) to management treatments using a model-based approach (Burnham and Anderson 2002, MacKenzie et al. 2006) would be more practical to field biologists and resource managers. Carefully developing a set of candidate models that predicts relationships between occupancy rates and habitat features/management effects (e.g., site covariates) should facilitate the process of teasing out the complexity of

the treatment and response by birds. The set of *a priori* candidate models would include hypotheses of the expected changes in occupancy rates associated with various habitat features related to management actions (site covariates) and sample covariates that are not related to management actions, such as climatic variables. Careful consideration must be taken here to identify habitat features and data collection that will meet the study objectives.

The number of sampling points per management unit and per management treatment will depend on the size of the unit and how many units the refuge is able to treat, etc. A statistician should be consulted after *a priori* hypotheses are developed.

Data Management and Analysis

Another aspect of monitoring that is often neglected until the end of the study or program is data management and analysis. It is critical that the data are transferred from paper copy or personal digital assistants (PDAs) and checked for errors to a database. A web-based Marsh Bird Database has been developed and released to the public at: http://www.pwrc.usgs.gov/point/mb/ (accessed 10.17.07) for the purpose of storing and archiving data, conducting simple analyses, and producing simple reports. Data analysis is not an easy task for most biologists. Analysis is time consuming and most biologists either do not have the time or the skills required. Moreover, monitoring program tend to overlook the time and budget for design, implementation, and evaluation. It is ironic because obviously, without results, the objective to make more informed decisions cannot be met. Unfortunately, this is more common than not (Lovett et al. 2007, pers. obs.).

IMPLICATIONS

Monitoring is an integral component of a resource biologist's job. Under current circumstances of reduced budgets and staffs it is critical to ensure that the time invested in monitoring programs is well spent. Prior to implementing a monitoring program, at any level, it is critical to invest the time up-front to identify and address issues related to the "investigatory process", beginning with the most important element, which is identifying clear objectives or answering WHY monitor? Clearly stated objective (s) will facilitate identification of subsequent elements WHAT variables to measure and HOW to measure. Furthermore, monitoring programs without clear objectives will undoubtedly result in wasted time and effort. Monitoring in the context of informed decision process, where *a priori* models are tested and updated with new data are most effective and efficient in management-based monitoring programs.

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Table 1. Changes in protocol to survey marsh birds on Agassiz National Wildlife Refuge, northwestern, Minnesota, 2000-2003. (*Distances to birds detected were recorded in either categories of 0-10 m, 11-20, 21-30, 31-40, 41-50, 51-60, 61-70, 71-80, 81-100, 101-120, 121-150, 151-200, >200 m or were variable, continuous distances).

Year	Passive	Broadcast	Passive	Distance*
2000	3 min	5 min (LEBI, SORA, VIRA, AMBI, PBGR)	2 min	Categories
2001	3 min	5 min (LEBI, SORA, VIRA, AMBI, PBGR)	2 min	Categories
2002	5 min	6 min (BLRA, LEBI, SORA, VIRA, AMBI, PBGR)	1 min	Variable
2003	5 min	5 min (LEBI, SORA, VIRA, AMBI, PBGR)	0 min	Variable

Table 2. Summary of raw counts in three survey windows, detection probability (*p*), *p*-adjusted counts, and occupancy rate (ψ) for American Bitterns surveyed during the breeding season (early May – mid-June) in 2000-2003 at Agassiz NWR, Minnesota.*

Year	Parameters	Raw counts	*p* (SE)	*p* - adjusted counts	ψ (SE)**
2000	Survey 1	35	0.56 (0.08)	62.99	0.96 (0.00)
	Survey 2	32	0.46 (0.08)	69.34	
	Survey 3	23	0.43 (0.08)	54.12	
	$\bar{x}$ (SE)			62.15 (4.41)	
2001	Survey 1	47	0.76 (0.07)	62.16	0.91 (0.00)
	Survey 2	27	0.47 (0.08)	58.05	
	Survey 3	2	0.05 (0.03)	42.02	
	$\bar{x}$ (SE)			54.08(6.15)	
2002	Survey 1	21	0.42 (0.08)	49.89	0.86 (0.00)
	Survey 2	50	0.63 (0.08)	79.59	
	Survey 3	50	0.64 (0.08)	77.68	
	$\bar{x}$ (SE)			69.05 (9.60)	
2003	Survey 1	74	0.85 (0.06)	87.41	0.82 (0.00)
	Survey 2	43	0.56 (0.08)	77.30	
	Survey 3	28	0.44 (0.08)	64.31	
	$\bar{x}$ (SE)			76.34(6.69)	

**p* (detection probability) from top ranked model; for all species except Virginia Rail, top model included *p* as a function of surveys.

p – adjusted counts: raw counts/*p* (MacKenzie et al. 2002).

$\bar{x}$ (SE): average adjusted counts with SE for the year.

**ψ values obtained from model ψ (YR) ε p; SE was not calculated because numerical convergence was not reached, but parameter estimates converged to approximately 6.03 significant digits – estimates are good (L. Bailey, Patuxent WRC, pers. comm.).

Table 3. Summary of raw counts in three survey windows, detection probability (*p*), *p*-adjusted counts, and occupancy rate (ψ) for Least Bitterns surveyed during the breeding season (early May – mid-June) in 2000-2003 at Agassiz NWR, Minnesota.*

Year	Parameters	Raw counts	*p* (SE)	*p* - adjusted counts	ψ (SE)
2000	Survey 1	2	0.30 (0.20)	6.69	0.19 (0.19)
	Survey 2	2	0.28 (0.19)	7.16	
	Survey 3	4	0.40 (0.24)	10.05	
	$\bar{x}$ (SE)			7.97 (7.16)	
2001	Survey 1	0	0.00 (0)	0	0.40 (0.26)
	Survey 2	11	0.58 (0.25)	19.06	
	Survey 3	2	0.14 (0.11)	14.60	
	$\bar{x}$ (SE)	4.33		11.22 (5.76)	
2002	Survey 1	1	0.05 (0.05)	19.08	0.06 (0.05)
	Survey 2	1	0.05 (0.05)	19.96	
	Survey 3	2	0.05 (0.05)	38.17	
	$\bar{x}$ (SE)			25.74 (6.22)	
2003	Survey 1	0	0.00 (0.00)	0	0.21 (NA)**
	Survey 2	3	0.13 (0.11)	23.49	
	Survey 3	3	0.13 (0.11)	23.49	
	$\bar{x}$ (SE)			15.66 (7.83)	

**p* (detection probability) from top ranked model; for all species except Virginia Rail, top model included *p* as a function of survey windows.

p – adjusted counts: raw counts/*p* (MacKenzie et al. 2002).

$\bar{x}$ (SE): average adjusted counts with SE for the year.

**ψ calculated from equation $\psi_{t+1} = \psi_t (1 - \varepsilon_t) + (1 - \psi_t)\gamma_t$ because the high variance among surveys and years caused numerical convergence problems in the matrix of both single-season models, which yield ψ = 1.0 (SE = 0); this is most likely due to unmodeled heterogeneity (MacKenzie 2002, 2006, and 2007*).

Table 4. Summary of raw counts in three survey windows, detection probability (*p*), *p*-adjusted counts, and occupancy rate (ψ) for Pied-billed Grebes surveyed during the breeding season (early May – mid-June) in 2000-2003 at Agassiz NWR, Minnesota.*

Year	**Parameters**	**Raw counts**	***p* (SE)**	***p* - adjusted counts**	**ψ (SE)**
	Survey 1	35	0.47 (0.06)	75.12	0.88 (0.06)
2000	Survey 2	42	0.70 (0.07)	60.10	
	Survey 3	62	0.89 (0.07)	69.28	
	$\bar{x}$ (SE)			68.17 (4.37)	
2001	Survey 1	28	0.56 (0.11)	49.80	0.81 (0.10)
	Survey 2	30	0.57 (0.04)	52.97	
	Survey 3	21	0.47 (0.07)	45.07	
	$\bar{x}$ (SE)			49.28 (2.29)	
2002	Survey 1	20	0.31 (0.13)	65.45	0.85 (0.07)
	Survey 2	31	0.58 (0.06)	53.13	
	Survey 3	48	0.65 (0.05)	74.12	
	$\bar{x}$ (SE)			64.23 (6.09)	
2003	Survey 1	34	0.80 (0.07)	42.59	0.80 (0.07)
	Survey 2	37	0.68 (0.08)	54.41	
	Survey 3	46	0.79 (0.16)	58.52	
	$\bar{x}$ (SE)			51.84 (4.77)	

**p* (detection probability) from top ranked model; for all species except Virginia Rail, top model included *p* as a function of survey windows.

p – adjusted counts: raw counts/*p* (MacKenzie et al. 2002).

$\bar{x}$ (SE): average adjusted counts with SE for the year.

ψ values obtained from model ψ (YR) ε p

Table 5. Summary of raw counts in three survey windows, detection probability (*p*), *p*-adjusted counts, and occupancy rate (ψ) for Sora surveyed during the breeding season (early May – mid-June) in 2000-2003 at Agassiz NWR, Minnesota.*

Year	**Parameters**	**Raw counts**	***p (SE)***	***p* - adjusted counts**	**ψ (SE)**
	Survey 1	62	0.81 (0.07)	76.96	0.91 (0.00)
2000	Survey 2	52	0.70 (0.07)	74.29	
	Survey 3	47	0.55 (0.08)	85.45	
	$\bar{x}$ (SE)			78.90 (3.37)	
2001	Survey 1	24	0.52 (0.09)	46.35	0.68 (0.00)
	Survey 2	45	0.57 (0.09)	79.14	
	Survey 3	8	0.19 (0.06)	43.08	
	$\bar{x}$ (SE)			56.19 (11.51)	
2002	Survey 1	36	0.47 (0.08)	76.19	0.78 (0.05)
	Survey 2	45	0.57 (0.08)	79.65	
	Survey 3	69	0.79 (0.07)	87.61	
	$\bar{x}$ (SE)			81.15 (3.38)	
2003	Survey 1	74	0.83 (0.07)	89.04	0.77 (0.05)
	Survey 2	38	0.64 (0.08)	59.06	
	Survey 3	41	0.51 (0.08)	80.49	
	$\bar{x}$ (SE)			76.20 (8.92)	

**p* (detection probability) from top ranked model; for all species except Virginia Rail, top model included *p* as a function of survey windows.

p – adjusted counts: raw counts/*p* (MacKenzie et al. 2002).

$\bar{x}$ (SE): average adjusted counts with SE for the year.

ψ values obtained from model ψ (YR) ε p

Table 6. Summary of raw counts in three survey windows, detection probability (*p*), *p*-adjusted counts, and occupancy rate (ψ) for Virginia Rail surveyed during the breeding season (early May – mid-June) in 2000-2003 at Agassiz NWR, Minnesota.*

Year	**Parameters**	**Raw counts**	***p* (SE)**	***p* - adjusted counts**	**ψ (SE)**
	Survey 1	21	0.51 (0.07)	36.34	0.74 (0.09)
2000	Survey 2	21			
	Survey 3	14			
	$\overline{x}$ (SE)	18.67 (2.33)			
2001	Survey 1	16	0.38 (0.07)	39.17	0.52 (0.08)
	Survey 2	14			
	Survey 3	15			
	$\overline{x}$ (SE)	15 (0.58)			
2002	Survey 1	9			
	Survey 2	4			
	Survey 3	19	0.33 (0.07)	31.87	0.44 (0.08)
	$\overline{x}$ (SE)	10.67 (4.41)			
2003	Survey 1	22	0.59 (0.07)	28.96	0.51 (0.08)
	Survey 2	10			
	Survey 3	19			
	$\overline{x}$ (SE)	17 (3.61)			

**p* (detection probability) from top ranked model; for all species except Virginia Rail, top model included *p* as a function of year.

p – adjusted counts: average of raw counts/*p* (YR) (MacKenzie et al. 2002).

$\overline{x}$ (SE): average (SE) of raw counts for the year.

ψ values obtained from model ψ (YR) ε p

Table 7. Occupancy models of five marsh bird species surveyed between 2000 – 2003 at Agassiz National Wildlife Refuge, northwestern, Minnesota. *-2 Log Likelihood.

Species	**Models**	**# Para**	**ψ (SE)**	**AIC_c**	**ΔAIC_c**	**w_i**	**-2 *l* ***
American Bittern							
	ψ γ ε *p*(YR + SURV)	15		626.54	0.00	0.98	596.54
	ψ γ ε *p*(SURV+WINDSD)	15		634.11	7.57	0.02	604.11
	ψ γ ε *p*(YR)	7		684.07	57.53	0.00	670.07
	ψ γ ε *p*	4		688.21	61.67	0.00	680.21
	ψ ε *p(YR + BURN-L)*	6		688.48	61.94	0.00	676.48
	ψ (YR) ε p	6		705.59	79.05	0.00	693.59
	ψ (YR + BURN-L) ε p	6		752.37	125.83	0.00	740.37
	ψ (YR+DRAWDOWN-L) ε p	6		752.62	126.08	0.00	740.62
	ψ (YR+BURN-Q) ε p	6		772.69	146.15	0.00	760.69
	ψ (YR+DRAWDOWN-Q) ε p	6		796.28	169.74	0.00	784.28
Least Bittern							
	ψ γ ε *p*(YR + SURV)	15	0.20 (0.10)	203.03	0.00	0.71	173.03
	ψ γ ε *p*(SURV+WINDSPD)	15	0.26 (0.14)	205.34	2.31	0.22	175.34
	ψ γ ε *p*(YR)	7	0.21 (0.11)	208.43	5.40	0.05	194.43
	ψ γ ε *p*	4	0.41 (0.20)	210.16	7.13	0.02	202.16
	ψ (YR) ε p	6		213.40	10.37	0.00	201.40
	Model-averaged		0.22				
Pied-billed Grebe							
	ψ γ ε *p*(SURV+WINDSPD)	15	0.88 (0.06)	644.79	0.00	0.86	614.79
	ψ γ ε *p*(YR+SURV)	15	0.88 (0.06)	648.67	3.88	0.12	618.67
	ψ γ ε *p*(YR)	7	0.89 (0.06)	654.57	9.78	0.01	640.57
	ψ ε *p(YR + BURN-L)*	6	0.85 (0.04)	654.62	9.83	0.01	642.62
	ψ γ ε *p*	4		659.66	14.87	0.00	651.66
	ψ (YR) ε p	6		661.73	16.94	0.00	649.73
	ψ (YR+DRAWDOWN-L) ε p	6		705.99	61.20	0.00	693.99
	ψ (YR + BURN-L) ε p	6		707.91	63.12	0.00	695.91
	ψ (YR+BURN-Q) ε p	6		739.94	95.15	0.00	727.94
	ψ (YR+DRAWDOWN-Q) ε p	6		740.01	95.22	0.00	728.01
	Model-averaged		0.88				
Sora							
	ψ γ ε *p*(YR + SURV)	15		640.54	0.00	0.92	610.54
	ψ γ ε *p*(SURV+WINDSPD)	15		645.46	4.92	0.08	615.46
	ψ γ ε *p*(YR)	7		662.15	21.61	0.00	648.15
	ψ γ ε *p*	4		671.40	30.86	0.00	663.40
	ψ ε *p(YR + BURN-L)*	6		686.59	46.05	0.00	674.59

Table 7. continued.

ψ (YR) ε p	6		691.98	51.44	0.00	679.98
ψ ε p (YR+BURN-Q)	6		693.04	52.50	0.00	681.04
ψ (YR+DRAWDOWN-L) ε p	6		747.12	106.58	0.00	735.12
ψ (YR + BURN-L) ε p	6		747.31	106.77	0.00	735.31
ψ (YR+BURN-Q) ε p	6		747.52	106.98	0.00	735.52
ψ (YR+BURN-Q) ε p	6		750.36	109.82	0.00	738.36
Virginia Rail						
ψ γ ε *p*(YR)	7	0.71 (0.09)	512.48	0.00	0.42	498.48
ψ γ ε *p*	4	0.72 (0.09)	513.84	1.36	0.21	505.84
ψ γ ε *p*(SURV)	15	0.70 (0.09)	514.54	2.06	0.15	484.54
ψ (YR) ε p	6		515.12	2.64	0.11	503.12
ψ γ ε *p*(SURV + WINDSPD)	15	0.71 (0.09)	516.55	4.07	0.05	486.55
ψ ε *p(YR + BURN-L)*	6	0.57 (0.06)	517.59	5.11	0.03	505.59
ψ (YR + BURN-L) ε p	6		520.02	7.54	0.01	508.02
ψ (YR+DRAWDOWN-L) ε p	6		520.15	7.67	0.01	508.15
ψ (YR+DRAWDOWN-Q) ε p	6		521.85	9.37	0.00	509.85
ψ (YR+BURN-Q) ε p	6		522.23	9.75	0.00	510.23
Model-averaged		0.67				

Table 8. Parameters for top ranked model for five marsh bird species surveyed during the breeding season from 2000 – 2003 at Agassiz National Wildlife Refuge, northwestern, Minnesota. *American Bittern model – estimates of ψ and γ encountered numerical convergence problem (was not reached) because American Bitterns were detected at every site (thus, no site variation in ψ and γ estimates). However, parameter estimates converged to approximately 6.03 significant digits, thus, estimates of detection probabilities *p* are valid.

American Bittern	**β**	**SE (β)**	**(*p*)**	**SE (*p*)**	**LCI**	**UCI**
ψ γ ε (SURV)*						
ψ	43.47	NA	1.00	0.00	0.00	1.00
γ	-22.60	NA	0.00	0.00	0.00	1.00
ε	-5.25	1.56	0.01	0.01	0.00	0.10
p - 2000 (SURV1)	0.22	0.34	0.56	0.08	0.39	0.71
p - 2000 (SURV2)	-0.15	0.32	0.46	0.08	0.31	0.62
p - 2000 (SURV3)	-0.30	0.32	0.43	0.08	0.28	0.58
p - 2001 (SURV1)	1.13	0.36	0.76	0.07	0.60	0.86
p - 2001 (SURV2)	-0.14	0.31	0.47	0.08	0.32	0.61
p - 2001 (SURV3)	-3.00	0.72	0.05	0.03	0.01	0.17
p - 2002 (SURV1)	-0.32	0.32	0.42	0.08	0.28	0.58
p - 2002 (SURV2)	0.52	0.32	0.63	0.08	0.47	0.76
p - 2002 (SURV3)	0.59	0.33	0.64	0.08	0.49	0.78
p - 2003 (SURV1)	1.71	0.44	0.85	0.06	0.70	0.93
p - 2003 (SURV2)	0.23	0.31	0.56	0.08	0.40	0.70
p - 2003 (SURV3)	-0.26	0.31	0.44	0.08	0.29	0.59
Least Bittern						
ψ γ ε *p*(SURV)						
ψ	-1.38	0.63	0.20	0.10	0.07	0.46
γ	-1.20	0.71	0.23	0.13	0.07	0.55
ε	-2.41	4.33	0.08	0.33	0.00	1.00
p - 2000 (SURV1)	-0.85	0.97	0.30	0.20	0.06	0.74
p - 2000 (SURV2)	-0.95	0.98	0.28	0.20	0.05	0.73
p - 2000 (SURV3)	-0.41	0.99	0.40	0.24	0.09	0.82

Table 8. continued.

p - 2001 (SURV1)	-25.05	NA	0.00	0.00	0.00	1.00
p - 2001 (SURV2)	0.31	1.01	0.58	0.25	0.16	0.91
p - 2001 (SURV3)	-1.84	0.89	0.14	0.11	0.03	0.48
p - 2002 (SURV1)	-2.89	1.18	0.05	0.06	0.01	0.36
p - 2002 (SURV2)	-2.94	1.17	0.05	0.06	0.01	0.34
p - 2002 (SURV3)	-2.89	1.18	0.05	0.06	0.01	0.36
p - 2003 (SURV1)	-26.25	NA	0.00	0.00	0.00	1.00
p - 2003 (SURV2)	-1.92	0.96	0.13	0.11	0.02	0.49
p - 2003 (SURV3)	-1.92	0.96	0.13	0.11	0.02	0.49
Pied-billed Grebe						
ψ γ ε *p*(SURV, WINDSPD)						
ψ	1.98	0.55	0.88	0.06	0.71	0.96
γ	0.71	0.56	0.67	0.12	0.40	0.86
ε	-1.64	0.33	0.16	0.04	0.09	0.27
p - 2000 (SURV1)	-0.14	0.23	0.47	0.06	0.36	0.58
p - 2000 (SURV2)	0.42	0.17	0.70	0.07	0.54	0.82
p - 2000 (SURV3)	1.07	0.36	0.89	0.07	0.67	0.97
p - 2001 (SURV1)	0.13	0.23	0.56	0.11	0.34	0.76
p - 2001 (SURV2)	0.27	0.18	0.57	0.04	0.48	0.65
p - 2001 (SURV3)	-0.07	0.15	0.47	0.07	0.33	0.61
p - 2002 (SURV1)	-0.41	0.30	0.31	0.13	0.12	0.58
p - 2002 (SURV2)	0.34	0.24	0.58	0.06	0.47	0.69
p - 2002 (SURV3)	0.61	0.21	0.65	0.05	0.55	0.73
p - 2003 (SURV1)	1.38	0.45	0.80	0.07	0.62	0.91
p - 2003 (SURV2)	0.75	0.38	0.68	0.08	0.50	0.82
p - 2003 (SURV3)	0.33	0.23	0.79	0.16	0.37	0.96
Sora						
ψ γ ε *p*(SURV)						
ψ	25.42	NA	1.00	0.00	0.00	1.00
γ	1.20	1.22	0.77	0.22	0.23	0.97
ε	-2.35	0.50	0.09	0.04	0.03	0.20
p - 2000 (SURV1)	1.42	0.42	0.81	0.07	0.64	0.90
p - 2000 (SURV2)	0.85	0.35	0.70	0.07	0.54	0.82
p - 2000 (SURV3)	0.20	0.32	0.55	0.08	0.40	0.70
p - 2001 (SURV1)	0.07	0.36	0.52	0.09	0.35	0.68
p - 2001 (SURV2)	0.28	0.36	0.57	0.09	0.40	0.73

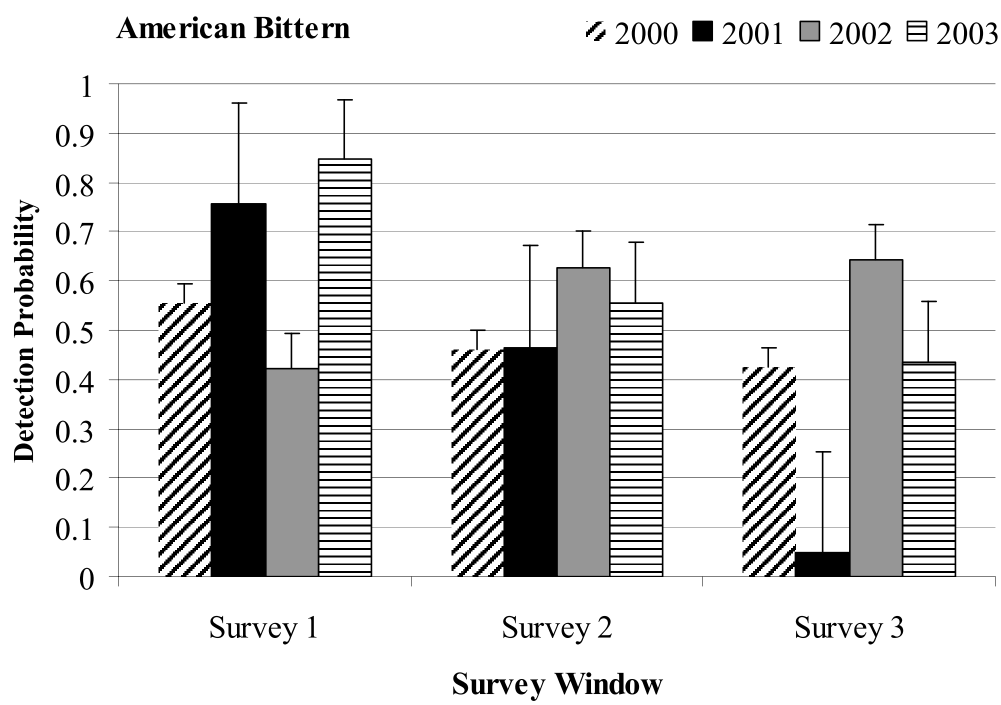
Table 8. continued.

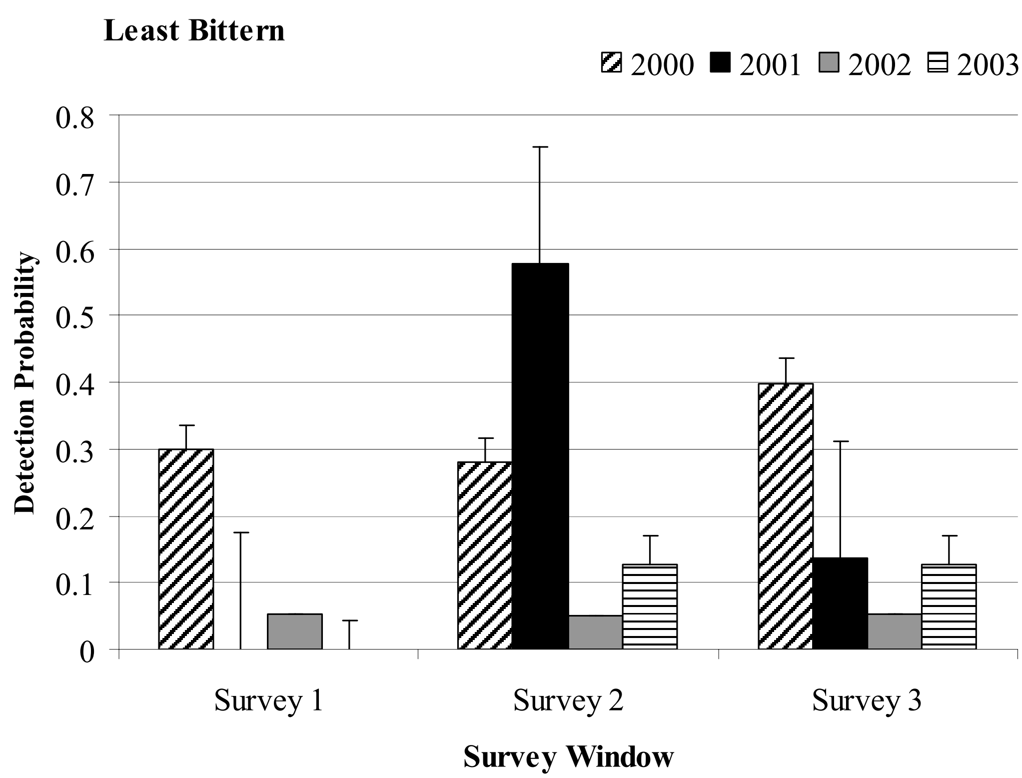
p - 2001 (SURV3)	-1.48	0.43	0.19	0.06	0.09	0.34
p - 2002 (SURV1)	-0.11	0.33	0.47	0.08	0.32	0.63
p - 2002 (SURV2)	0.26	0.33	0.57	0.08	0.41	0.71
p - 2002 (SURV3)	1.31	0.42	0.79	0.07	0.62	0.89
p - 2003 (SURV1)	1.59	0.48	0.83	0.07	0.66	0.93
p - 2003 (SURV2)	0.59	0.35	0.64	0.08	0.47	0.78
p - 2003 (SURV3)	0.04	0.33	0.51	0.08	0.35	0.67
Virginia Rail						
ψ γ ε *p*(YR)						
ψ	0.89	0.43	0.71	0.09	0.54	0.88
γ	-1.39	0.49	0.20	0.08	0.05	0.35
ε	-1.14	0.41	0.24	0.08	0.09	0.39
p - 2000	0.05	0.27	0.51	0.07	0.38	0.64
p - 2001	-0.48	0.31	0.38	0.07	0.24	0.53
p - 2002	-0.69	0.32	0.33	0.07	0.19	0.47
p - 2003	0.35	0.28	0.59	0.07	0.45	0.72

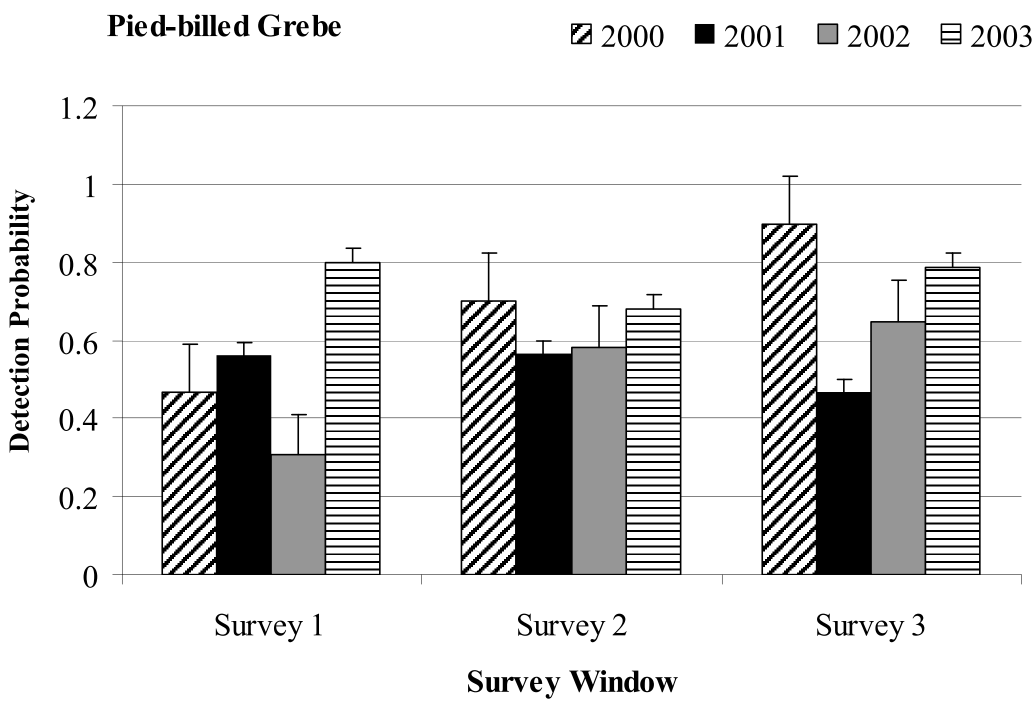
Table 9. Precision and sample sizes calculated from occupancy and detection probability estimates (MacKenzie et al. 2006) for five marsh bird species monitored during 3 survey replicates per season or 5 survey replicates per season per site (sample size) at Agassiz National Wildlife Refuge, northwestern, Minnesota. Estimates of ψ and p for each species, from top models, used in calculations are as follows: American Bittern $\psi = 0.89$, $p = 0.52$; *Least Bittern* $\psi = 0.22$, p = 0.17, *Pied-billed grebe = 0.88*, $p = 0.62$; Sora $\psi = 0.79$, $p = 0.59$; *Virginia Rail* $\psi = 0.67$, $p = 0.45$.

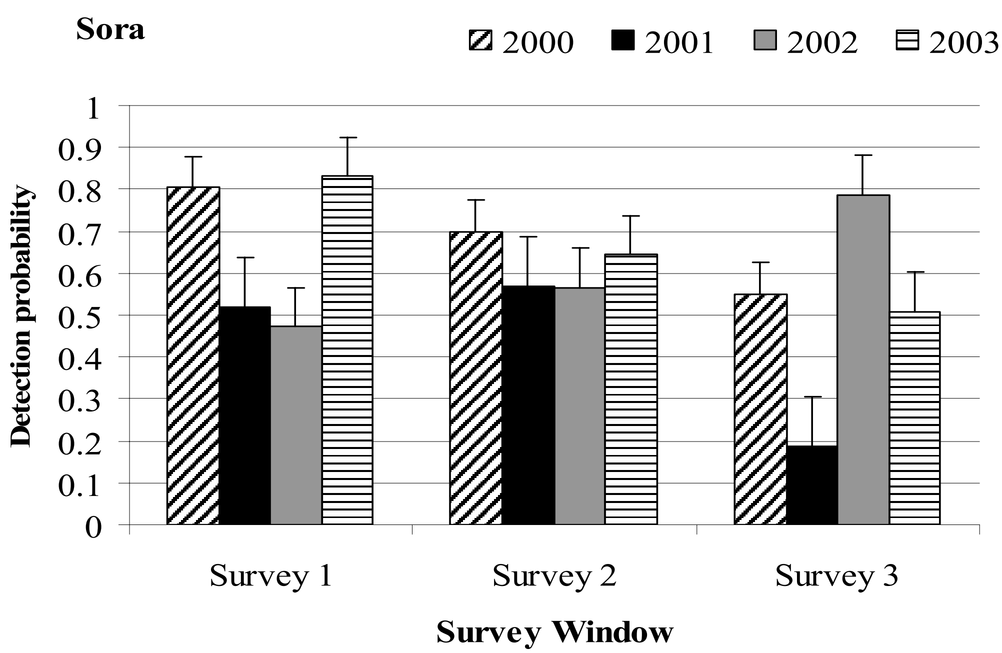
	American Bittern		Least Bittern		Pied-billed Grebe		Sora		Virginia Rail	
SE	3 replicates	5 replicates	3 replicates	5 replicates	3 replicates	5 replicates	3 replicates	5 replicates	3 replicates	5 replicates
0.05	113	50	723	240	71	45	101	70	193	107
0.10	**28**	**13**	**181**	**60**	**18**	**11**	**25**	**18**	**48**	**27**
0.15	13	6	80	27	8	5	11	8	21	12
0.20	**7**	**3**	**45**	**15**	**4**	**3**	**6**	**4**	**12**	**7**

Figure 1. Detection probability (+1 SE) from top ranking AIC_c models of five marsh bird species during surveys at Agassiz National Wildlife Refuge, MN from 2000 – 2003. (Dates for Survey Windows: 2000: Survey 1 = 5.06.00, Survey 2 = 5.25.00, Survey 3 = 6.08.00; 2001: Survey 1 = 5.28.01, survey 2 = 6.12.01, Survey 3 = 6.27.01; 2002: Survey 1 = 5.10.02, Survey 2 = 5.29.02, Survey 3 = 6.13.02; 2003: Survey 1 = 5.12.03, Survey 2 = 5.28.03, Survey 3 = 6.10.03). *(ΔAIC_c for Virginia Rail model *p*(Survey), where detection probability varied by survey windows and by year was $\Delta AIC_c = 2.06$; otherwise, $\Delta AIC_c = 0$ for other species).


American Bittern
2000
2001
2002
2003
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Detection Probability
Survey 1
Survey 2
Survey 3
Survey Window


Least Bittern
2000
2001
2002
2003
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Detection Probability
Survey 1
Survey 2
Survey 3
Survey Window


Pied-billed Grebe
2000
2001
2002
2003
1.2
1
0.8
0.6
0.4
0.2
0
Detection Probability
Survey 1
Survey 2
Survey 3
Survey Window


Sora
2000
2001
2002
2003
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Detection probability
Survey 1
Survey 2
Survey 3
Survey Window

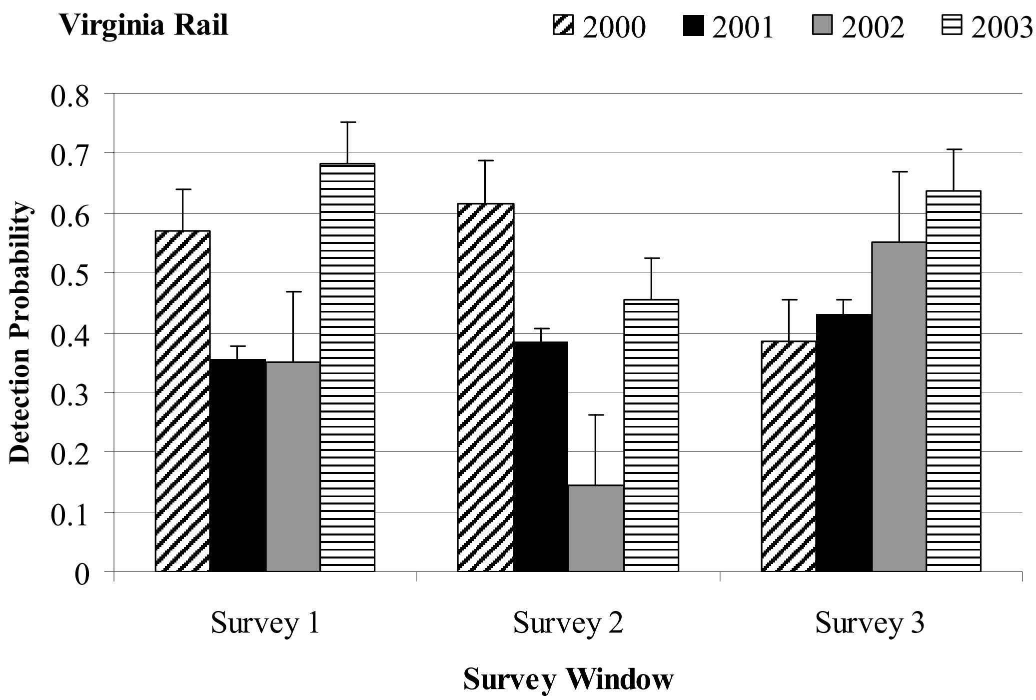
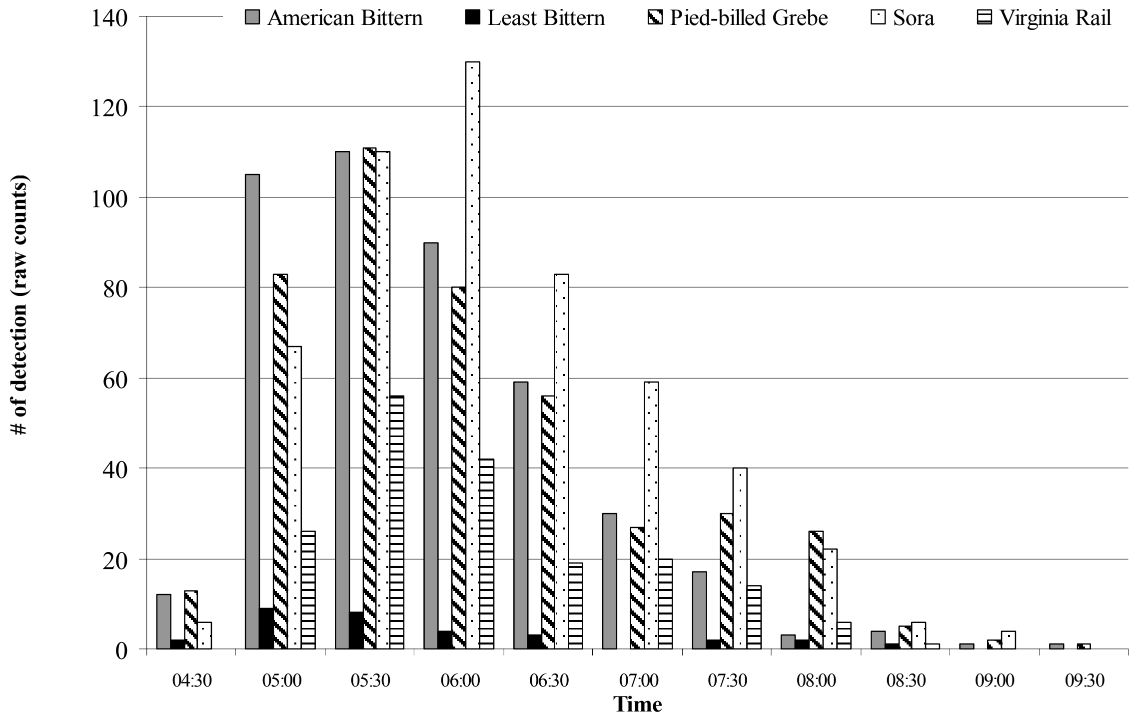

Virginia Rail
2000
2001
2002
2003
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Detection Probability
Survey 1
Survey 2
Survey 3
Survey Window

Figure 2. Frequency of detection, using raw counts, from surveys of five marsh bird species, during the breeding season (early May to mid-June) from 2000-2004 combined at Agassiz National Wildlife Refuge, in northwest Minnesota.



American Bittern
Least Bittern
Pied-billed Grebe
Sora
Virginia Rail
140
120
100
80
60
40
20
0
# of detection (raw counts)
04:30
05:00
05:30
06:00
06:30
07:00
07:30
08:00
08:30
09:00
09:30
Time

VITA

Socheata Lor was born in Cambodia some time in 1968, she and her surviving family members are unsure of the exact date. She was given a birth date of February 29, 1968 by her father while in a refugee camp in Thailand while in transit to the United States with her family. Her father passed away while in the refugee camp. Socheata, her mother, and six siblings immigrated to Libertyville, Illinois with the assistance of three families of sponsors in 1980. She eventually graduated in 1987 from Mundelein High School after which she pursued a Bachelor of Arts degree in biology at Ripon College, Ripon, Wisconsin, in 1991. While at Ripon College, she completed a long internship with the U. S. Fish and Wildlife Service, which lead to employment as a wildlife biologist at Erie National Wildlife Refuge (NWR) in Guys Mills, Pennsylvania, after she graduated. After two years at Erie, she transferred to Iroquois NWR, in Basom, New York, where she worked for five years prior to entering into a Master of Wildlife Science program at Cornell University, Ithaca, New York. She completed her degree in 2000 at Cornell University and began her doctoral program in Fisheries and Wildlife Sciences at the University of Missouri-Columbia.

Socheata remains employed with the U. S. Fish and Wildlife Service as a wildlife biologist in La Crosse, Wisconsin. She serves as a team member with the Biological Monitoring Team and also as the Assistant Regional Refuge Biologist in the Midwest and Great Lakes Region (Region 3).