# ESTIMATING WHITE-TAILED DEER ABUNDANCE USING AERIAL QUADRAT SURVEYS

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## SUMMARY OF FINDINGS

I estimated white-tailed deer (*Odocoileus virginianus)* abundance in select permit areas (PA) using quadrat surveys to recalibrate deer population models and evaluate the impact of deer season regulation changes on population size. With rare exception, precision of population estimates was similar among permit areas. However, because population estimates were not corrected for sightability, estimates represent minimum counts and are biased low. In 2009, I began testing a sightability estimator to adjust estimates for animals missed during surveys. Undocumented deer movement made analysis difficult. Additional trials are needed to develop sightability estimates.

## INTRODUCTION

Management goals for animal populations are frequently expressed in terms of population size (Lancia et al. 1994). Accurate estimates of animal abundance allow for documentation of population trends, provide the basis for setting harvest quotas (Miller et al. 1997), and permit assessment of population and habitat management programs (Storm et al. 1992).

The Minnesota Department of Natural Resources (MNDNR) uses simulation modeling to estimate and track changes in deer abundance and, subsequently, to develop harvest recommendations to keep deer populations within goal levels. In general, model inputs include estimates of initial population size and spatial/temporal estimates of survival and reproduction for various age and sex cohorts. Because simulated population estimates are subject to drift as model input errors accumulate over time, it is imperative to periodically recalibrate the starting population within these models with independent deer population estimates (Grund and Woolf 2004).

Minnesota's deer numbers are managed according to numeric population goals within 125 PAs. MNDNR recently revised deer population goals within each PA using a consensus-based, roundtable approach consisting of 15-20 citizens representing varied interest groups (e.g. deer hunters, farmers, foresters, environmental groups, etc.; Stout et al. 1996). Revised goals are used to guide deer-harvest recommendations. Currently, deer populations exceed management goals in many PAs. A conventional approach of increasing the bag limit within the established hunting season framework has failed to reduce deer densities. As a result, MNDNR began testing the effectiveness of 3 non-traditional harvest regulations to increase the harvest of antlerless deer and reduce overall population levels (Grund et al. 2005). Accurate estimates of deer abundance are needed to evaluate these regulations.

My objective in this investigation is to provide independent estimates of deer abundance in select PAs that are within 20% of the true mean with 90% confidence (Lancia et al. 1994). Abundance data will be used to recalibrate population models to improve population management and to evaluate impacts of deer season regulation changes on deer abundance.

## METHODS

I estimated deer populations in selected PAs using a quadrat-based, aerial survey design. Quadrat surveys have been used to estimate populations of caribou (*Rangifer tarandus*; Siniff and Skoog 1964), moose (*Alces alces*; Evans et al. 1966), and mule deer (*O. heimonus*; Bartmann et al. 1986) in a variety of habitat types. Quadrats were selected using 1 of 3 sampling designs: (1) stratified random (StRS; Cochran 1977); (2) 2-dimensional (2-D) systematic (Cressie 1993, D'Orazio 2003); or (3) generalized random-tessellation stratified (GRTS; Stevens and Olsen 2004). I used a StRS sampling design in PAs where the local wildlife manager had prior knowledge about deer abundance and distribution. Quadrats were stratified into 2 abundance classes (low, high) based on relative deer densities. Occasionally, additional strata were constructed to encompass management boundaries (e.g., park boundaries). I used a 2-D systematic sampling design in other areas. Systematic designs are typically easier to implement and maximize spatial distribution of the sample. Beginning in 2008, I used the GRTS design to obtain spatially balanced stratified and random samples. This design improves the spatial distribution of StRS and permits replacement of sample quadrats that are lost due to navigation hazard or high human development. Previously, replacement quadrats were unavailable in systematic PAs because of the rigid, 2-D design.

Within each PA, quadrats were delineated by Public Land Survey section boundaries and a 20% sample was selected for surveying. Sample size calculations indicated this sampling rate was needed to meet accuracy and precision objectives. I used OH-58 helicopters during most surveys and attempted to maintain a flight altitude of 60 m above ground level and an airspeed of 64-80 km/hr. A Cessna 182 airplane was used in 3 PAs dominated by intensive row-crop agriculture. To increase visibility, I completed surveys after leaf-drop and when snow cover measured at least 15 cm. A pilot and 2 observers searched for deer along transects spaced at 270-m intervals until they were confident all deer were observed. When animals fled the helicopter, direction of movement was noted to avoid double counting. I used a real-time, moving-map software program (DNR Survey; MNDNR 2005), coupled to a global positioning system receiver and a tablet-style computer, to guide transect navigation and record deer locations and aircraft flight paths directly to ArcView GIS (Environmental Systems Research Institute 1996) shapefiles. I estimated deer abundance from StRS surveys using PROC SURVEYMEANS (SAS 1999). I used the R programming language (RDCT 2009) and formulas developed by D'Orazio (2003) for 2-D systematic surveys and the R package SPSURVEY (ver. 2.0; RDCT 2009) for GRTS surveys. I evaluated precision using coefficient of variation (CV), defined as standard deviation of the population estimate divided by the population estimate, and relative error (RE), defined as the 90% confidence interval bound divided by the population estimate (Krebs 1999).

During 2009, I conducted a pilot study in 2 PAs (240, 345) to test the repeatability (temporal variation) of our survey protocol and to evaluate if the Gasaway et al. (1986) resurvey method could be used to adjust population estimates for sightability. Deteriorating snow conditions precluded data collection in additional PAs. After the initial survey was completed in each PA, I selected 10 quadrats where at least 20 deer had been previously observed. I then resurveyed each quadrat twice in succession before moving to subsequent quadrats. In the first replicate (temporal survey), I followed the same survey protocol as the initial survey. However, in the second replicate (sightability survey), flight speed was reduced to 48-64 km/hr to allow observers additional time to locate deer. Replicate surveys were flown 1-4 days and 11 days after completion of initial surveys in PA 345 and 240, respectively. The pilot and observers were the same for all surveys within each PA. I compared deer counts during initial and temporal surveys to evaluate repeatability. To evaluate sightability, I examined deer locations from survey-specific shapefiles and compared deer counts during temporal and sightability surveys. I defined sightability as number of deer observed / number of deer available. Animals observed departing a quadrat were excluded from analysis if they were only observed during 1 survey. Deer groups ($n$=1, 2, 3, etc.) observed during both surveys and located in close proximity were assumed to be replicate sightings. If group size varied between surveys, I used the higher value as the "available" group size in computations. When deer groups had no obvious match between surveys, I assigned each group to 2 classes: (1) minimum estimate - groups were treated as independent observations, available during both surveys, but not observed; and (2) maximum estimate - groups were pooled together and group size was summed by survey; the higher count value between surveys defined available group size and the lower value defined observed group size. Once all deer groups were classified, I calculated minimum and maximum sightability estimates by quadrat and mean sightability by PA.

**RESULTS AND DISCUSSION**

I completed 4-8 surveys each winter (December-March, 2005-2009; Table 1). Stratified fixed-wing surveys were conducted in PAs 270 and 272. Based on long-term deer harvest metrics, population estimates in these areas were biased low. Several possibilities may explain this result: (1) deer were clustered in unsampled quadrats; (2) deer were wintering outside PA boundaries; (3) sightability was biased using fixed-wing aircraft; and/or (4) kill locations from hunter-killed deer were reported incorrectly. Land cover in these PAs was dominated by intensive row-crop agriculture. After crops were harvested each fall, deer habitat was limited to riparian areas, wetlands, abandoned farm groves, and undisturbed grasslands, including those enrolled in state and federal conservation programs. Although recreational feeding of deer could influence distribution, wildlife managers believed it was not a common practice in these PAs. Thus, I had no evidence to support non-traditional deer distribution in these units. I also had no reason to believe hunter registration errors had greater bias in these units than in other PAs. Although it was possible that deer occupied unsampled quadrats by chance, the use of optimal allocation to increase sampling effort in high strata quadrats because of expected higher deer densities should minimize this possibility. Furthermore, we surveyed 100% of the high-strata quadrats in PA 421, resulting in no unsampled quadrats. Sightability bias, however, is greater in fixed-wing aircraft than helicopters (LeResche and Rausch 1974, Kufeld et al. 1980, Ludwig 1981) and likely explained much of the bias I observed in these PAs. Consequently, all surveys have subsequently been conducted using a helicopter.

With the exception of PAs 270, 272, and 201, precision (CV, RE) of the population estimates was similar among PAs (Table 1). High precision in PA 270 was, in part, an artifact of sample design. Based on optimal allocation formulas, we selected and surveyed all high strata quadrats. Thus, because no sampling occurred within the high stratum (100% surveyed), sampling variance was calculated only from low strata quadrats. We observed few deer in these low strata quadrats, which resulted in low sampling variance and high precision of the population estimate. It is unlikely that this design (i.e., sampling 100% of high strata quadrats) will be feasible in all areas, especially if deer are more uniformly distributed throughout the landscape.

In contrast, survey precision in PAs 272 and 201 was poor. We observed few deer during either survey ($n$=144 and 56, respectively) and nearly all observations occurred within 1 or 2 quadrats. As a result, associated confidence intervals exceeded 60% of the population estimate (Table 1). Kufeld et al. (1980) described similar challenges with precision due to nonuniformity of mule deer distribution within strata in Colorado.

The number of deer observed between initial and temporal surveys varied by 2% and 18% in PAs 240 and 345, respectively (Table 2). Although Potvin et al. (2002) suggested values <30% were not biologically significant, my assessment of repeatability was derived from small sample sizes (sample rate=1-3%) and needs further evaluation. Repeatability of surveys is important if deer distribution varies significantly during static weather conditions. Random movement of deer across quadrat boundaries likely has little impact on population estimates if the sampling design is robust and incorporates habitat stratification and randomization. However, in situations where significant non-random movements are likely to occur in response to changing conditions (e.g., weather events), replicate surveys should be considered.

I did not correct population estimates for sightability. Thus, estimates represent minimum counts and are biased low. However, a sightability survey protocol was examined in sample quadrats during 2009. Mean sightability ranged from 0.68 to 0.72 in PAs 345 and 240, respectively. Estimated sightability per quadrat was also similar between PAs (Table 2). Despite greater search intensity during sightability surveys, temporal survey counts were higher in 8 of 19 quadrats (Table 2). Sightability data were excluded from 1 quadrat because survey protocol was not maintained during inclement weather. This count discrepancy between surveys may be the result of undocumented deer movement across quadrat boundaries. Deer response to aerial surveys ranged from apparent disregard to escape (running away). Although observers were instructed to make mental note of directional movement of deer groups to minimize double counting, implementation of consecutive surveys will mandate documentation of movement because of observer memory-bias between surveys and the unknown fate of moving deer. When deer are sedentary during surveys, the resurvey method (Gasaway et al. 1986) shows promise. However, as deer movement increases, confidence in count data decreases.

Additional sightability trials are needed to improve our understanding of the applicability and limitations of the resurvey method for adjusting population estimates for sightability. Future analysis will also include *post-hoc* evaluation of habitat features present in quadrats containing deer. This will provide additional empirical data for use in quadrat stratification. In addition, the impact of winter feeding on deer distribution will be examined to determine if pre-survey stratification flights (Gasaway et al. 1986) are warranted.

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Table 1. Deer population and density estimates derived from aerial surveys in Minnesota, 2005-2009.

| Sampling design | Year | Permit area | Population estimate *N* | Population estimate 90% CI | CV (%) | Relative error (%)[a] | Density estimate (deer/mi$^2$) Mean | Density estimate (deer/mi$^2$) 90% CI | Model estimate (deer/mi$^2$) |
|---|---|---|---|---|---|---|---|---|---|
| Systematic | 2005 | 252 | 2,999 | 2,034 – 3,969 | 19.5 | 32.2 | 2.9 | 2.0 – 3.9 | 2 |
| | | 257 | 2,575 | 1,851 – 3,299 | 16.9 | 28.1 | 6.2 | 4.4 – 7.9 | 7 |
| | 2006 | 204 | 3,432 | 2,464 – 4,401 | 17.0 | 28.2 | 4.6 | 3.3 – 5.9 | 5 |
| | | 209 | 6,205 | 5,033 – 7,383 | 11.4 | 18.9 | 9.7 | 7.9 – 11.5 | 5 |
| | | 210 | 3,976 | 3,150 – 4,803 | 12.5 | 20.8 | 6.3 | 5.0 – 7.6 | 7 |
| | | 256 | 4,670 | 3,441 – 5,899 | 15.9 | 26.3 | 7.1 | 5.3 – 9.0 | 5 |
| | | 236 | 6,774 | 5,406 – 8,140 | 12.1 | 20.2 | 16.8 | 13.4 – 20.2 | 37 |
| | 2007 | 225 | 5,341 | 4,038 – 6,645 | 14.7 | 24.4 | 8.0 | 6.0 – 9.9 | 24 |
| | | 227 | 5,101 | 4,245 – 5,960 | 10.1 | 16.8 | 9.8 | 8.2 – 11.5 | 13 |
| | | 346 | 7,896 | 5,736 – 10,062 | 16.4 | 27.4 | 22.7 | 16.5 – 29.0 | 31 |
| | 2008 | 266 | 3,853 | 2,733 – 4,977 | 17.5 | 29.1 | 6.2 | 4.4 – 8.0 | n/a[b] |
| Stratified | 2005 | 206 | 2,486 | 1,921 – 3,051 | 13.7 | 22.5 | 5.2 | 4.0 – 6.4 | 5 |
| | | 270 | 631 | 599 – 663 | 3.0 | 5.0 | 0.8 | 0.8 – 0.9 | 5 |
| | | 342 | 3,322 | 2,726 – 3,918 | 10.8 | 17.7 | 9.1 | 7.5 –10.7 | 10 |
| | 2006 | 201 | 274 | 100 – 449 | 37.6 | 61.9 | 1.6 | 0.6 – 2.7 | 6 |
| | | 269 | 1,740 | 1,301 – 2,180 | 15.2 | 25.1 | 2.6 | 2.0 – 3.3 | 3 |
| | | 272 | 472 | 179 – 764 | 37.4 | 61.5 | 0.9 | 0.3 – 1.4 | 5 |
| | 2007 | 343 | 6,982 | 5,957 – 8,006 | 8.9 | 14.6 | 10.1 | 8.6 – 11.6 | 29 |
| | | 344 | 4,116 | 3,375 – 4,857 | 10.7 | 17.7 | 19.7 | 16.1 – 23.2 | 49 |
| | | 347 | 5,482 | 4,472 – 6,492 | 11.1 | 18.2 | 12.6 | 10.3 – 14.9 | 13 |
| | | 349 | 10,103 | 8,573 – 11,633 | 9.1 | 15.0 | 20.4 | 17.3 – 23.5 | 35 |
| | 2008 | 262 | 2,065 | 1,692 – 2,437 | 10.9 | 17.9 | 3.0 | 2.5 – 3.6 | n/a[b] |
| | | 271 | 1,019 | 848 – 1,189 | 10.1 | 16.6 | 1.6 | 1.3 – 1.8 | 8 |
| GRTS[c] | 2008 | 265 | 4,575 | 3,766 – 5,384 | 10.7 | 17.7 | 9.2 | 7.6 – 10.9 | n/a[b] |
| | 2009 | 240 | 11,041 | 9,799 – 13,003 | 8.5 | 14.1 | 16.7 | 14.4 – 19.1 | 28 |
| | | 261 | 1,721 | 1,450 – 1,992 | 9.6 | 15.7 | 2.2 | 1.8 – 2.5 | 4 |
| | | 345 | 4,247 | 3,678 – 4,806 | 8.0 | 13.2 | 12.8 | 11.1 – 14.5 | 21 |
| | | 348 | 5,717 | 4,953 – 6,480 | 8.1 | 13.4 | 17.8 | 15.4 – 20.1 | 13 |

[a]Relative precision of population estimate. Calculate as 90% CI bound/*N*.
[b]Permit area boundaries were recently modified. No model estimate is available.
[c]Generalized Random-Tessellation Stratified sample design.

Table 2. Deer observed and estimated sightability during multiple aerial surveys in Minnesota, 2009.

| Permit area | Quadrat | Survey[a] 1 | 2 | 3 | Minimum estimated value Observed | Available | Sightability | Maximum estimated value Observed | Available | Sightability |
|---|---|---|---|---|---|---|---|---|---|---|
| 345 | 45 | 27 | 23 | 27 | 23 | 27 | 0.85 | 23 | 27 | 0.85 |
| | 48 | 42 | 54 | 72 | 49 | 77 | 0.64 | 49 | 77 | 0.64 |
| | 57 | 24 | 27 | 26 | 23 | 30 | 0.77 | 23 | 26 | 0.88 |
| | 16 | 20 | 20 | 4 | 7 | 14 | 0.50 | 7 | 12 | 0.58 |
| | 18 | 18 | 14 | 14 | 12 | 16 | 0.75 | 12 | 16 | 0.75 |
| | 19 | 66 | 36 | 41 | 27 | 48 | 0.56 | 30 | 46 | 0.65 |
| | 23 | 29 | 22 | 20 | 18 | 24 | 0.75 | 18 | 21 | 0.86 |
| | 22 | 19 | 24 | 29 | 21 | 32 | 0.66 | 23 | 30 | 0.77 |
| | 34 | 48 | 32 | 22 | 17 | 37 | 0.46 | 21 | 33 | 0.64 |
| | 65 | 38 | 18 | —[b] | —[b] | —[b] | —[b] | —[b] | —[b] | —[b] |
| | Total | 331 | 270 | 255 | 197 | 305 | 0.65 | 206 | 288 | 0.72 |
| 240 | 11 | 51 | 45 | 48 | 40 | 53 | 0.75 | 44 | 49 | 0.90 |
| | 42 | 33 | 60 | 56 | 52 | 65 | 0.80 | 52 | 61 | 0.85 |
| | 81 | 31 | 11 | 10 | 7 | 10 | 0.70 | 7 | 8 | 0.88 |
| | 18 | 45 | 33 | 49 | 33 | 49 | 0.67 | 33 | 49 | 0.67 |
| | 97 | 49 | 57 | 47 | 39 | 66 | 0.59 | 41 | 52 | 0.79 |
| | 128 | 22 | 27 | 33 | 19 | 33 | 0.58 | 19 | 33 | 0.58 |
| | 15 | 50 | 60 | 58 | 51 | 67 | 0.76 | 51 | 61 | 0.84 |
| | 32 | 26 | 26 | 28 | 17 | 35 | 0.49 | 21 | 27 | 0.78 |
| | 52 | 23 | 27 | 28 | 20 | 35 | 0.57 | 20 | 35 | 0.57 |
| | 106 | 54 | 45 | 57 | 42 | 60 | 0.70 | 45 | 57 | 0.79 |
| | Total | 384 | 391 | 414 | 320 | 473 | 0.68 | 333 | 432 | 0.77 |

[a]Survey 1 = initial survey; Survey 2 = temporal survey; Survey 3 = sightability survey.
[b]Data excluded because deteriorating weather compromised survey protocol.

# ESTIMATING WHITE-TAILED DEER DENSITY USING TRAIL CAMERAS

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## SUMMARY OF FINDINGS

White-tailed deer (*Odocoileus virginianus*) densities in the farmland zone of Minnesota are estimated using simulation modeling and aerial surveys. Simulation modeling is not well suited for modeling population dynamics in small areas, such as Itasca State Park (Permit Area 287). In 2005, Itasca State Park was chosen as a study area to test alternative deer hunting regulations. Deer density estimates were needed to evaluate the effect of antler-point restriction regulations (>3-points-on-a-side) on the deer population in the park. A trail camera study was initiated in 2006 to monitor the population. Forty-two cameras were systematically placed at a density of 1 camera/130 ha. The ratio of legal bucks to sub-legal bucks (fork and spike bucks) and buck:antlerless deer was calculated for 2, 3-week sampling periods before and after the hunting season. During 2006, cameras captured 12,486 images of deer, and in 2007 cameras captured 11,326 images of deer over the 6-week sampling period. The study was continued in 2008; data entry was not complete at the time this report was written.

## INTRODUCTION

In 2005, Itasca State Park was chosen as a study area to test a 3-points-on-a-side antler-point restriction regulation for deer hunting. Deer density estimates were needed to evaluate the effect of the antler-point restriction on the density and demographics of the deer population. The primary management objective associated with the antler-point restriction was to reduce deer density by increasing the antlerless harvest via reductions in the antlered harvest (Grund et al. 2005).

Deer densities in Minnesota have traditionally been estimated using simulation modeling (Grund 2007, Lenarz 2007). Aerial surveys have been used in some farmland permit areas to provide an independent field estimate for correcting population models (Haroldson and Giudice 2006). However, due to errors caused by demographic stochasticity and seasonal movement patterns, simulation modeling is not recommended for small areas (Grund 2001). The small size of Itasca State Park (approximately 130 km$^2$) made population modeling impractical. Also, aerial surveys were not feasible due to dense coniferous cover that exists in parts of the park. While deer density estimates were not available for the park, the simulated deer density immediately north of the park was estimated at 65 deer/km$^2$ (25 deer/mi$^2$) in spring 2007 (Lenarz 2007).

Infrared-triggered cameras have been used to estimate deer populations in a variety of habitat types and study area sizes (Moore 1995, Jacobson et al. 1997, Koerth et al. 1997, Warlock et al. 1997, and Roberts et al. 2006). Jacobson et al. (1997) developed a camera technique to estimate deer density using known numbers of individually identifiable mature bucks and associated age and sex ratios from the deer herd. In Texas, Koerth et al. (1997) compared camera population estimates to helicopter counts and concluded that both techniques provided reliable deer density estimates.

In Fall 2005, a pilot study, initiated at Itasca State Park using infrared-triggered cameras, determined that: (1) more sampling effort was needed, (2) a systematic sampling design should be used, and (3) pre-baiting of sites was needed. In 2006, the study was adjusted to accommodate the pilot study findings. The study was continued in 2007 and in 2008; data entry from 2008 was not complete at the time this report was written.

## OBJECTIVE

1. To determine the density and demographics of the deer herd to assess effects associated with the antler-point restriction regulation at Itasca State Park.

## METHODS

The trail camera study was conducted at Itasca State Park, located in northwestern Minnesota in 2006-2008 from September to December. The park is approximately 130 $km^2$. The study area we used was approximately 6,400 ha located in the interior portion of the park in order to minimize effects that movement patterns would have on deer observations along the perimeter of the park. Following the protocol developed by Jacobson et al. (1997), 42 trail cameras were systematically placed at a density of 1 camera/130 ha throughout the study area using the Systematic Point Sample tool in ArcView 3.3. Minor adjustments were needed to avoid wetland areas (Figure 1).

Each site was located using a global positioning system unit and marked using flagging material. Cameras were in the field for 3 weeks before and after the regular firearms season for a total of 6 weeks each year. For the first sampling period (before the regular firearms deer hunting season), sites were baited with 23 kg (50 lbs) of shelled corn 3 weeks prior to placing the cameras in the field. An additional 11 kg (25 lbs) of corn was added to each site 1 week before camera sampling began. Sites were baited with 23 kg (50 lbs) of shelled corn 1 week before the second sampling period (after the regular firearms season).

A Bushnell TrailScout Pro 2.1 Mega Pixel (MP) or 3.0 MP trail camera was used at each site. Cameras were attached to a nearby tree at a height of 1.5 m. Each camera faced north and was 4-6 m from the established bait pile. Cameras were angled slightly downward to aim the infrared beam to a height approximately 1 m above the bait pile. In 2007, wooden boxes were constructed to house cameras to protect them from precipitation and damage by bears. Cameras were programmed to take pictures day and night with a 1-minute delay between pictures in 2006 and a 30-second delay between pictures in 2007. Batteries and memory cards were replaced on a weekly basis. Corn (11 kg) was added to the baited area on a weekly basis for both sampling periods in 2006, the first sampling period in 2007, and 23 kg was added to each site on a weekly basis for the second sampling period in 2007. These adjustments were made based on observed feeding patterns each season.

Each image was examined using Adobe Photoshop 3.0 or Microsoft Photo Editor, and only images of deer within the sampling time frame were used in the analysis. We classified each deer as legal buck (>3-points-to-a-side), sub-legal buck, or antlerless deer. Legal bucks were individually identified using number, size, and arrangement of points. We excluded images if we were unable to classify a deer to an appropriate category.

## RESULTS AND DISCUSSION

In 2006, trail cameras captured 16,682 images during the 2, 3-week sampling periods. More images were captured during the postseason (9,346) than during the preseason period (7,336). Approximately 75% of the images contained a photo of a deer. Other species we observed included black bear (*Ursus americanus*), raccoon (*Procyon lotor*), bobcat (*Lynx rufus*), snowshoe hare (*Lepus americanus*), a variety of avian species, gray wolf (*Canis lupus*), mice (*Peromyscus* spp.), squirrels (*Sciurus* spp. and *Tamiasciurus hudsonicus*), chipmunks (*Tamias striatus*), fisher (*Martes pennanti*) and humans. Some images (16%) were eliminated due to: (1) images contained no visible animal, (2) the image was distorted, (3) the distance from the camera to the deer was too great, and (4) vegetation obstructed the view of the deer. Thus, 11,526 images containing 14,115 deer observations were useable for project purposes in 2006. During the preseason period in 2006, we observed 1,507 legal bucks, 811 sub-legal bucks, and

3,430 antlerless deer. During the postseason period in 2006, we observed 1,772 legal bucks, 1,519 sub-legal bucks, and 5,078 antlerless deer.

In 2007, trail cameras captured 21,486 images during the 2, 3-week sampling periods. Nearly equal numbers of images were captured during the postseason (10,754) and the preseason period (10,732). Approximately 53% of the images contained a photo of a deer. Species other than deer that were observed in 2007 included those seen in 2006 (except for fisher) and red fox (*Vulpes vulpes*) and porcupine (*Erethizon dorsatum*). Some images (27%) contained no visible animal, and distortion of the image, distance from the camera, or vegetation obstructing the view also caused some deer to be unidentifiable (6%). Thus, 11,326 images containing 13,380 deer observations were useable for project purposes in 2007. During the preseason period in 2007, we observed 440 legal bucks, 41 sub-legal bucks, and 3,294 antlerless deer. During the postseason period in 2007, we observed 2,218 legal bucks, 1,159 sub-legal bucks, and 6,228 antlerless deer.

Future work includes finishing data entry for 2008 and comparing population estimates for all three years (2006-2008) using Jacobson's (1997) mark-recapture technique and change-in-ratio formula (Paulik and Robson 1969).

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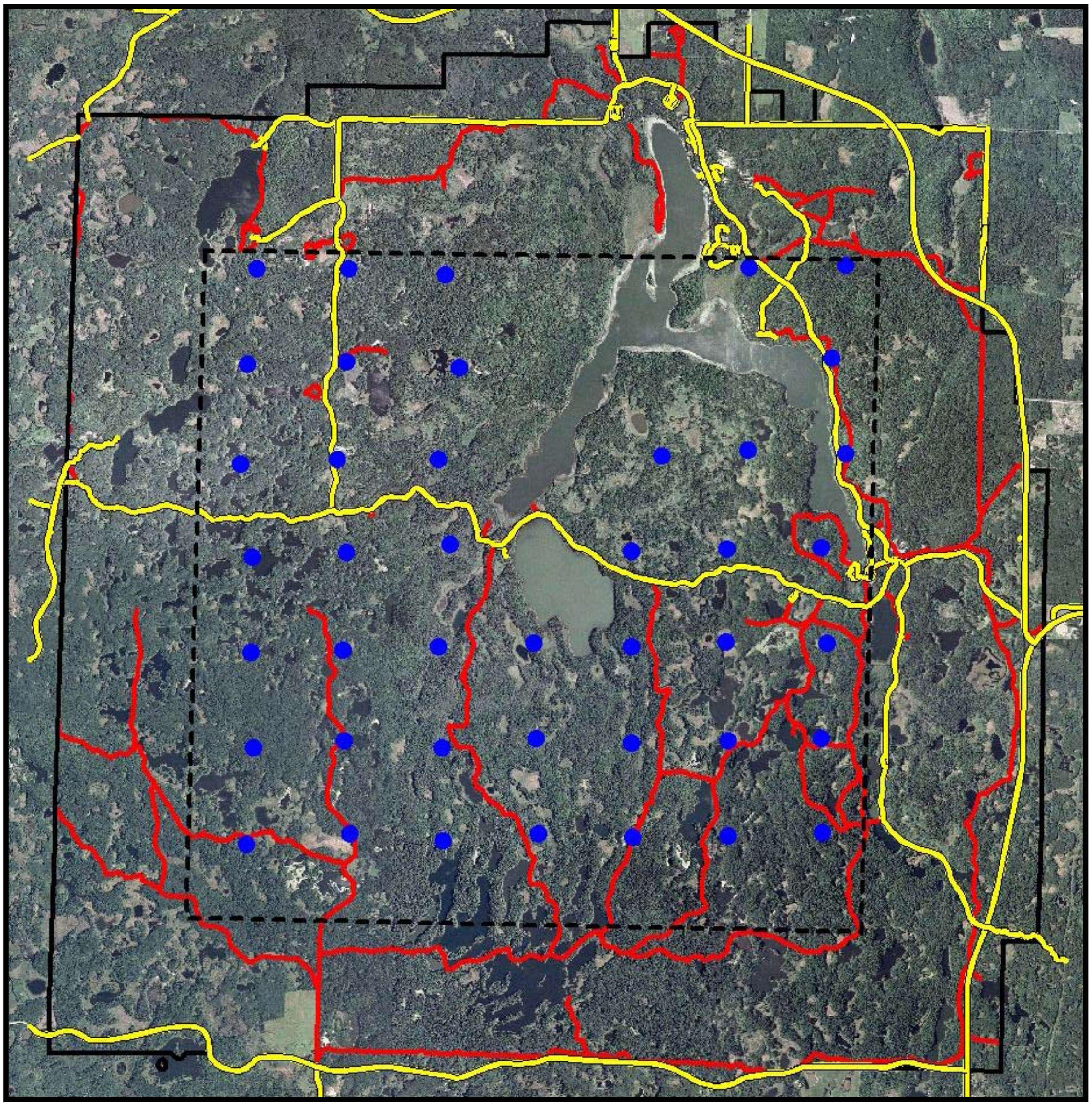

Figure 1. Locations of trail cameras (dots) in the study area (dashed line) at Itasca State Park, Minnesota in 2006-2008.

# ECOLOGY, HUNTING SEASONS, AND MANAGEMENT OF GRAY AND FOX SQUIRRELS IN MINNESOTA

Emily J. Dunbar, Eric M. Walberg, and Richard O. Kimmel

## SUMMARY OF FINDINGS

Southeast Asian Hmong hunters expressed concern about perceived low populations of squirrels (*Sciurus* spp.) on public hunting lands near population centers. The Section of Wildlife initiated a pilot study to identify potential problems for squirrel hunters and to determine if squirrel-hunting opportunities would benefit from harvest regulation changes or management activities. A survey of Minnesota squirrel hunters was conducted April-May 2009 to provide an understanding of how hunting opportunities could be improved and to determine if perception of squirrel hunting problems differed between hunter groups. A survey of state/provincial wildlife agencies was conducted April-May 2009 to gain knowledge about squirrel hunting seasons and management in other jurisdictions. Data collection is currently ongoing and results will be reported in a future report.

## INTRODUCTION

Gray and fox squirrel (*Sciurus carolinensis* and *S. niger*) hunting provides recreational opportunities for an estimated 26,000 hunters in Minnesota (Dexter 2008). Hunters are not required to report harvest, but numbers of squirrels harvested are estimated through annual small game hunter surveys (Dexter 2008). Recently, southeast Asian Hmong hunters expressed concern about perceived low populations of squirrels on public hunting land near population centers. The Minnesota Department of Natural Resources (MNDNR) does not estimate squirrel populations, which are only managed through harvest regulations. MNDNR Section of Wildlife initiated a pilot study to determine if squirrel hunting opportunities would benefit from harvest regulation changes and/or management activities and which changes could potentially increase the huntable squirrel population on public land. We conducted a literature review of fox and gray squirrel ecology and management, a survey of squirrel hunters, and a survey of squirrel hunting season and management programs of state and provincial wildlife agencies. Data collection is currently ongoing and results will be provided in a future report.

## OBJECTIVES

1. Conduct a literature review of squirrel ecology and management.
2. Survey squirrel hunters to provide an understanding of their experiences and thoughts on how squirrel hunting and management can be improved.
3. Survey other state and provincial wildlife agencies regarding squirrel hunting seasons and management programs.
4. Provide recommendations for squirrel hunting season management for Minnesota and potential research and/or management activities.

## METHODS AND RESULTS

Manuscripts relating to the ecology, management, and hunting mortality of gray and fox squirrels are being compiled and selected manuscripts summarized.

A survey of Minnesota squirrel hunters was conducted April-May 2009 to provide an understanding of how hunting opportunities can be improved to determine if perceptions about squirrel populations and hunter satisfaction differ between hunter groups. We collected names and addresses of hunters that had indicated they had harvested squirrels on small game hunter surveys from 2005-2008. Additional names and addresses of assumed Hmong hunters were included from the 2008-2009 small game surveys. (Note – Hmong hunters had requested and provided funding for this project.) Surveys were mailed to 100 randomly selected assumed non-Hmong hunters from the 7-county metro region (Anoka, Carver, Dakota, Hennepin, Ramsey, Scott, and Washington), 100 randomly selected assumed Hmong hunters from the metro region, and 200 randomly selected assumed non-Hmong hunters from non-Metro regions of Minnesota. The survey instrument consisted of questions relating to harvest, counties and land ownership of properties hunted, hunter experiences, hunter access, hunter perception of squirrel populations, and suggestions for improving squirrel hunting experiences (Appendix 1). Surveys were mailed on 10 April 2009 with a second mailing to non-respondents on 1 May 2009. At this writing, 300 completed surveys have been returned from the first 2 mailings for an initial response rate of 79%.

A survey of state/provincial wildlife agencies was conducted April-May 2009 to gain knowledge about squirrel hunting regulations and management. An email survey was sent to 47 state wildlife agencies (Alaska, Hawaii, and Minnesota were excluded) and 4 Canadian provincial wildlife agencies (Quebec, Ontario, Manitoba, and Saskatchewan). The survey instrument consisted of questions relating to season opening/closing dates, bag and possession limits, management, research, population estimation activities, and issues concerning squirrel hunting (Appendix 2). Surveys were emailed on 27 April 2009 with a second emailing to non-respondents on 15 May 2009. At this writing, 41 completed surveys have been returned for an initial response rate of 80%.

## ACKNOWLEDGMENTS

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Appendix 1.

**Minnesota Fox and Gray Squirrel Hunter Survey**

You have been selected from a group of hunters that harvested squirrels as indicated on past Small Game Hunter surveys. Because this survey is only being sent to a small number of hunters, your input is extremely valuable, please complete and return the following survey as soon as possible. Your identity will be kept confidential.

1. Did you hunt fox and/or gray squirrels last season (September 2008-February 2009)?

   Yes ___ No* ___

   *If No, then you do not need to continue the survey; please answer question 1 and return the survey.

2. In which county/counties did you hunt squirrels? ______________________________

3. Approximately, how many squirrels did you harvest last season?

   1-5 ____ 6-10 ____ 11-15 ____ 16-20 ____ 21-25 ____ 26-30 ____ 31+_____

4. Approximately, how many days did you spend hunting squirrels last season?

   1-7 days ____ 8-14 days ____ 15-21 days _____ 22-28 days ____ 29+ days ____

5. How long have you hunted squirrels in Minnesota?

   1-5 years ____ 6-10 years ____ 11-15 years _____ 16-20 years ____ 21+ years ____

6. Do you hunt squirrels on public land ___, private land ___, or both ___?

7. In the past 5 years, have you hunted squirrels:

   on the same properties ___, on new properties ___, or both ___?

(Survey continues on next side)

8. What are the main obstacles for gaining access to property for squirrel hunting? (check all that apply)

 ____ I have not encountered obstacles

____ Land is posted as no hunting or trespassing

____ Not sure how to find additional public hunting lands

____ Not comfortable asking for permission to access private land from landowner

____ Denied access by landowner(s) in the past

____ Difficulty finding owners of private land

____ Other (please specify) ________________________________________________

9. Over the past 5 years, do you think squirrel populations in areas where you hunt are:
   decreasing____, about the same____, or increasing_____?

10. Based on your perception of squirrel population trends, would you recommend changes to
(check all that apply):
____ Hunting regulations
____ Habitat management
____ Enforcement of regulations
____ Other (Please Specify) ________________________________________________
____ I don't recommend any changes

11. How could the DNR improve your squirrel hunting experience?

______________________________________________________________________

______________________________________________________________________

______________________________________________________________________

**Thank you for completing the survey! Please return the survey in the enclosed, postage-paid envelope.**

Appendix 2.

**Wildlife Agency Squirrel-Hunting Survey**

1. Your name________________________
   Your position title __________________________
   Your email address_________________________________
   Your phone number _________________________________

2. Does your state/province have a fox and/or gray squirrel hunting season?
   Yes ____ No____
   If No, then the survey is complete. Please send this survey back.

3. When does the hunting season open?

4. When does the season close?

5. What is the daily bag limit and possession limit?

6. What are the shooting/hunting hours?

7. Does your state/province manage specifically for fox and/or gray squirrels ?
   Yes ____ No____
   If Yes, what are the management activities?

8. Does your state/province estimate fox and/or gray squirrel populations?
   Yes ____ No____
   If Yes, what techniques is used?

9. Is there currently any research being conducted by your agency on fox and/or gray squirrels?
   Yes _____ No______
   If so, please describe the study/studies:

10. Are there any issues surrounding squirrel populations or squirrel hunting in your state/province?

11. Other things we should know about your squirrel season:

# CONDITIONAL PROBABILITY OF DETECTION OF RING-NECKED PHEASANTS IN CROWING MALE SURVEYS

Alison L. Harwood[1], Brock R. McMillan[1], Kurt J. Haroldson, and John H. Giudice.


## SUMMARY OF FINDINGS

Because population-estimation methods are not practical for ring-necked pheasants (*Phasianus colchicus*), managers typically use indices of population size. A common criticism of indices is that constant probability of detection is assumed but not estimated. We used an auditory, mark-recapture (MR) method to estimate the conditional probability of detection ($\hat{\rho}$) in replicate crowing surveys of male pheasants on 18, 23-km² sites in southern Minnesota. Probability of detection varied by study site (range $\hat{\bar{\rho}}$ = 0.382 – 0.731) and was negatively associated with disturbance (including initial observer disturbance) and positively associated with crowing frequency and intensity. We also fit a mixture model to the data, which suggested there may be 2 groups of male pheasants with different detection probabilities (e.g., possibly due to different vocalization rates). Mark-recapture methodology may only be practical for intensive studies on relatively small areas, but crowing male indices adjusted for probability of detection could be used as a standard from which to evaluate the more common roadside index.


## INTRODUCTION

Ring-necked pheasants are a species for which formal population-estimation methods are not practical. Pheasants are secretive and mobile (i.e., they tend to hide or move away from observers), and difficult to capture (Thomas 1996, Giudice and Ratti 2001, Lancia et al. 2005). Pheasants also do not have the flocking or territorial habits that increase the availability of some species for detection (Brown 1947, Eberhardt and Simmons 1987, Gibbs et al. 1998, Lancia et al. 2005). Because of these difficulties in detecting pheasants, populations are commonly monitored using indices based on crowing calls of males (Brown 1947, Kimball 1949, Kozicky 1952, Rice 2003).

In a crow-count survey, an observer stops at predetermined stations along a route and records the number of pheasant crowing calls heard during a specified time period (usually 2 minutes). The standard index is crows/stop, which is assumed (but not known) to be proportional to the abundance of male pheasants (Kimball 1949, Kozicky 1952, Luukkonen et al. 1997, Rice 2003). The validity of crow-count surveys as an index of male pheasant abundance requires that probabilities of detection are similar among survey periods and locations (i.e., the proportion of males detected is constant across time and space). Fisher et al. (1947), Luukkonen et al. (1997), and Anderson (2001, 2003) have criticized use of such indices because they are highly variable and probabilities of detection are generally unknown.

A variety of factors may affect the probability of detecting crowing male pheasants. Some factors (e.g., calendar period, time of day, weather, observer skill) can be controlled through survey design, but others cannot. Observer presence may cause nearby birds to stop calling at least temporarily (Johnson 2008). Background noise may mask the sound of calling birds (Simons et al. 2007). In addition, the distribution of pheasants in relation to the survey route may affect their availability for detection (Fisher et al. 1947, Hutto and Young 2003). Variability may be controlled by conducting replicated counts (Kimball 1949, Kozicky 1952, Gibbs et al. 1998, Johnson 2008), but the assumption that the expected probability of detection E($\hat{\rho}$) is similar among comparison groups or years remains untested.

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Alldredge et al. (2007) used a time-of-detection method to estimate probability of detection in songbirds. In this method, information about whether a bird is detected during separate time intervals is used to create a detection history. The detection history can be treated like a MR history in a closed population model (Alldredge et al. 2007, Riddle et al. 2008). In this study, we applied time-of-detection methods to replicate surveys of crowing male pheasants from 18 study sites in southern Minnesota to estimate mean detection probability and evaluate the assumption that $E(\hat{\rho})$ was similar among study sites. Our objectives were to: (1) evaluate use of an auditory MR technique to estimate detection probability of crowing male pheasants; and (2) evaluate factors that may influence probability of detection and overall accuracy of crowing male indices.

## STUDY AREA

We conducted this study on 18 sites in southern Minnesota as part of a concurrent study on pheasant habitat needs (Haroldson et al. 2007). Nine study sites were selected in each of 2 regions located near Faribault and Windom, Minnesota. Study sites averaged 23 $km^2$ (9 $miles^2$) in size. The primary land use in all study sites was agriculture, but sites varied in the amount and distribution of grassland habitat, pheasant winter habitat (e.g., emergent wetlands, shrub swamps, and planted shelterbelts), roads, and relative pheasant density (Haroldson et al. 2007). The percent of grassland habitat within the study sites varied from 3.4% to 31.6%, with most grassland habitat idled under the Conservation Reserve Program.

At each study site, we attempted to establish 9 listening stations that were evenly distributed across the site, based on an estimated 0.8 km (0.5 mile) auditory radius, to achieve maximum possible coverage of the study site and minimum overlap among stations (Figure 1). Where possible, we located stations on roads to facilitate convenient access. Where roads were not available, we located stations up to 0.4 km (0.25 mile) from roads. Due to road coverage and landscape obstacles (e.g., lakes), 2 study sites had only 8 stations.

## METHODS

### Crowing Male Surveys

We conducted 10 replicate surveys at each study site between 20 April and 31 May 2007 during mornings that met standardized weather conditions (wind speeds <16 km/hour and no precipitation at the beginning of the survey). Two trained observers performed surveys on each study site, dividing the 8-9 stations between them (4-5 stations/observer). The starting location for each survey route was selected randomly, and direction of travel was selected to minimize travel time and observer overlap. Observers rotated systematically among 2-4 study sites throughout the survey season.

Crowing male surveys began 45 minutes before sunrise and were completed by sunrise. Observers recorded wind speed, temperature, and amount of dew at the beginning and end of each survey and percent of sky covered by clouds at the end of the survey. Upon arrival at a listening station, observers immediately began to count the number of crowing males and the number of times each male crowed for 2 minutes. Observers attempted to identify each individual crowing male by marking their relative distance and direction from the observer on a map of the listening station. Observers documented potential misidentifications of individual crowing males by classifying the encounter history for each presumed individual as certain or uncertain. Observers also classified disturbance affecting their ability to hear crowing pheasants into 4 categories: none, low (e.g., distant tractor noise), medium (e.g., intermittent traffic), or high (e.g., constant background noise).

**Mark-Recapture**

We used extended listening intervals at 4 stations/study site to evaluate whether a closed population capture-recapture approach (Huggins 1989) could be used to estimate the mean detection probability of crowing pheasants. We randomly selected the first MR station on each study site, and then systematically selected the remaining 3 MR stations to ensure an even distribution across the study site, facilitate the assignment of 2 MR stations/observer, and ensure the completion of the survey within the 45-minute survey period. The same MR stations were sampled in each survey. Observers at MR stations continued to survey for 2 additional 2-minute intervals immediately following the first listening period. The second and third listening periods identified which birds heard during the first period were heard again, and also birds that had not previously been detected.

**Analysis**

We used Huggins (1989) closed-capture models to estimate conditional detection probability (ρ) and to evaluate a small set of covariates that could plausibly influence heterogeneity in detectability among study sites, listening stations, and survey days. The beta vector ($\beta$) for the most complex model included an intercept, constrained time variation (t = 1 versus t = 2, 3), and covariate effects. The constrained time variation evaluated the potential change in probabilities of detection between the first 2-minute listening interval and the second and third 2-minute listening intervals. The initial group covariates considered were study site (*SITE*) and road type (*RTYPE*) nearest each listening station. Study site was selected as a possible cause of variation because the amount and type of habitat within each site could possibly affect the distribution of pheasants, which in turn could affect the availability of male pheasants for detection. Road type (paved versus gravel) was selected as a surrogate for traffic volume, reasoning that traffic noise may negatively affect perceptibility of crowing calls. Individual covariates included relative disturbance (*DISTURB* [1-3 = none, low, moderate]) and total crows heard in the first listening interval (*CROWS*). Disturbance reflected the ability of observers to hear crowing pheasants over background noise. High disturbance sampling events were excluded from the MR analysis because they would typically be excluded in operational surveys. The variable *CROWS* was viewed as an index of crowing frequency and intensity and, indirectly, local density of male pheasants (at the listening station scale). We postulated that availability and perceptibility (Marsh and Sinclair 1989, Johnson 2008) would be positively correlated with crowing intensity (proportion of roosters that crow) and frequency, respectively, which may inflate the crowing male index if crowing incites other birds to crow more frequently (Gates 1966). We also evaluated a quadratic term for *CROWS* (which modeled a plausible non-linear relationship between detectability and *CROWS*) and an interaction term (*CROWS*×*SITE*) to permit the estimated slope to vary among sites.

Our primary research question was whether $E(\hat{\rho})$ varied substantially among study sites. Consequently, *SITE* was a reasonable covariate to consider in the initial model set. However, we were also interested in understanding what explanatory factors may be correlated with *SITE* effects. Thus, we also evaluated models where *SITE* was replaced with 2 explanatory variables: (1) percent of each study area composed of grassland habitats (*GRASS*); and (2) observer group (*OBSG* = 6 clusters of observers that reflected the restricted assignment of observers to study sites). We postulated that spring pheasant abundance (males and females) on each study site would be positively correlated with *GRASS* (Haroldson et al. 2006), and that bird density and distribution (at the study-site scale) may in turn influence $E(\hat{\rho})$. Finally, we constructed a model that included weather and date effects to evaluate our assumption that survey protocols minimized heterogeneity due to these covariates.

We used the R programming language (R Development Core Team 2008) and the RMark package (Lake and Rexstad 2008) to construct MR models, and the optimization routine in program MARK (White and Burnham 1999) to obtain parameter estimates ($\hat{\beta}$ and $\hat{\Sigma}$). We compared models using Akaike's Information Criterion ($AIC_c$) and estimated model weights (Burnham and Anderson 2002). Unfortunately, there is no unique way to compute goodness-of-fit for closed-population models with individual covariates (Cooch and White 2008). Therefore, we used estimability and precision of parameter estimates and consistency of covariate effects (observed versus predicted direction) to evaluate model fit and adequacy. We also developed a jackknife assessment of predictability by removing observations from one sampling event (site-station-date), refitting the best-approximating model to the reduced data set, and predicting the conditional probability of detection in the first listening interval (t1) for the held-out event. We repeated this process for each sampling event and then compared observed (proportion of birds in group *i* detected in t1) and predicted detectability. Based on the jackknife comparisons, we constructed an additional post-hoc model that assumed the bimodal distribution in observed detectability was the result of one group or mixture of birds being more detectable than a second group (Pledger 2000, Cooch and White 2008). More specifically, we constructed an additive model where the probability of belonging to a mixture ($\pi$) was constant across time and space, which resulted in 2 additional link-function parameters compared to the best-approximating *a priori* model where $\pi$ was fixed (= 1). We used 2 mixtures, defined the probability of detection in listening period 1 as $\hat{p}_{.1} = \hat{\pi}_1 \hat{p}_{11} + (1-\hat{\pi}_1)\hat{p}_{21}$, and computed $\hat{\mathrm{var}}(\hat{p}_{.1})$ using the delta method (Seber 2002). We also considered more complex mixture models (e.g., where $\pi$ was modeled as function of covariates), but these models failed to converge, had non-positive-definite variance-covariance matrices, or generated imprecise parameter estimates. Thus, we excluded these models from further consideration.

## RESULTS

Although weather varied throughout the survey season, 92% of crowing male surveys were conducted on mornings meeting standardized conditions (i.e., wind <16 km/hour and no precipitation). Fifty percent of surveys were conducted on mornings with ≤ 20% cloud cover, whereas 5% of surveys were conducted under complete cloud cover.

Observers completed 177 crowing male surveys which acquired data at 1586 of 1600 potential sampling events (site-station-date). Pheasants were heard crowing on all 18 study sites. Crowing male indices ranged from 1.2 to 6.4 males/station (Table 1). Mean number of males detected/station declined from 4.5 to 3.0 when disturbance increased from none to high. Likewise, variation in crowing frequency was higher during high-disturbance sampling events (SD = 1.40) than those with moderate to no disturbance (SD ≤ 0.99), although only 4% of sampling events were classified as high disturbance.

The MR dataset consisted of 647 sampling events (site-station-date) and 3,849 encounter histories. The number of males, crowing calls, and calls/male detected in the first listening period varied more among study sites than between MR and non-MR stations within study sites (Figure 2). Total number of crowing male pheasants detected over the 3 listening periods ranged from 0-26 (median = 7) per sampling event. Eighty percent of encounter histories were classified as certain, 17% as uncertain, and 3% were missing quality-control information. The proportion of encounter histories classified as uncertain increased slightly with an increase in total birds detected (Cochran-Armitage Trend Test, Z = -10.087, $P$ < 0.001, Figure 3) especially at densities >10 birds. However, 75% of encounter histories were from site-station-dates with ≤10 total male pheasants detected.

We initially considered a set of 14 MR models to estimate probability of detection of crowing male pheasants. Of these 14 models, only model 13 {$\pi$(1) *p* (*TIME* + *SITE* + *CROWS* + *DISTURB* + *CROWS*×*SITE*)} was supported by the data ($w_i$ = 1.0). Post-hoc models where *SITE* was replaced with *GRASS* and *OBSG* were not supported by the data. Model 13 suggested that conditional probability of detection varied among sites, was negatively associated with *DISTURB*, and positively associated with *TIME* (first versus second and third listening periods) and the number of crows (*CROWS*) heard per site (but the effect of *CROWS* varied by site). The amount of variation within sites was generally low with the exception of the site with the lowest pheasant counts (Figure 4, study site 36). The jackknife comparison of observed versus predicted detectability indicated that model 13 fit the data reasonably well (Spearman's *rho* = 0.5; Figure 5). However, detectability approached unity for some sampling events regardless of covariate values, which suggested the presence of some unexplained heterogeneity in ρ. We attempted to explain additional heterogeneity by fitting a finite mixture model (model 17). The mixture model improved model fit compared to model 13 ($\Delta AIC_c$ = 210.3) and was clearly the best-approximating model in the final model set ($w_i$ = 1.0). Based on model 17, the estimated proportion of individuals belonging to mixture 1 (group with lower detectability) was 40% (SE = 2.4) and the odds of being detected were 21 times greater for individuals in mixture 2 (Table 2). The signs of the relationships between conditional probability of detection and covariates were consistent with model 13 (Table 2) and relative differences in $E(\hat{\rho})$ among study sites were similar between the 2 models (Figure 4). On the other hand, site-specific estimates of $E(\hat{\rho})$ were less and estimated variances (of $\hat{\rho}$) were greater in model 17 (Figure 4).

## DISCUSSION

Rates of detection are influenced by both the availability of animals to be detected and the perceptibility of observers in detecting animals (Marsh and Sinclair 1989, Johnson 2008). In crowing male surveys, a pheasant becomes available for detection when it crows. In order to be detected, however, the observer must perceive and correctly identify the crowing call. This study considered only detectable animals (i.e., male pheasants that emitted at least 1 crowing call during the survey period). Thus, variability in $\hat{\rho}$ as reported here pertains to perceptibility, which is primarily a function of observer skill and survey conditions (Johnson 2008).

We considered several factors that may influence perceptibility of crowing pheasants and lead to heterogeneous detection probabilities. For example, our survey protocol constrained surveys to mornings with weather conditions favorable to pheasant crowing activity and perceptibility. Thus, not surprisingly, wind was not identified during model selection as an important covariate explaining probability of detection. Similarly, training was provided to observers to correctly identify male pheasant crowing calls and to distinguish individual males. Although uncertainty in distinguishing individual males increased with total number of males detected (especially in the minority of encounter histories when >10 males were detected), observer group was not identified during model selection as an important covariate explaining detection probability. These examples provide indirect evidence that our survey protocols reduced effects of weather and observer skill on perceptibility of crowing pheasants.

Increasing disturbance (e.g., background noise) was associated with reduced probability of detection. Road type may contribute to disturbance (e.g., greater traffic noise on paved roads); however, models containing disturbance were better supported by the data. The negative effect of disturbance may be minimized by selecting listening stations where high disturbance is unlikely. Although some types of disturbance (e.g., birds calling near a wetland) may be difficult to avoid, it may be possible to locate survey routes away from roads that receive a high amount of traffic.

We found that detection probabilities were lower during the first listening period than during the second and third listening periods, which likely reflected temporary disturbance caused by the presence of the observer. This result may indicate the need for a short, 2-3 minute waiting period prior to beginning the survey to allow pheasants to acclimate to the presence of an observer and resume crowing. In addition, these data suggest that a single 2-minute listening period was sufficient to detect an average of only 60% of available males. Extending the listening period to 6 minutes would increase probability of detection, but also reduces the number of stations that could be surveyed.

Some behavioral traits may affect availability of pheasants for detection. Rosenstock et al. (2002) suggested that the presence of non-territorial males may affect calling frequency of some birds. Leonard and Horn (1995) found that dominant domestic roosters (*Gallus gallus domesticus*) crowed more frequently than subordinate roosters. Non-territorial or subordinate males may not crow as often as other males, or may not crow at all, making them unavailable for detection. In this study, model 17 provided evidence of 2 groups of males with dramatically different probabilities of detection, which may reflect different crowing behavior between dominant and subordinate males.

Pheasant density may also affect crowing frequency, which in turn affects probability of detection. Gates (1966) reported significantly lower crowing frequency of individual males at low pheasant densities, presumably because of reduced need to defend territories from competing males. Therefore, a larger proportion of males may be unavailable for detection and crowing male indices may underestimate relative abundance at low pheasant densities. This result may explain the low $E(\hat{p})$ and high variance for study site 36, which had the lowest pheasant count.

Rosenstock et al. (2002) indicated that for indices to be reliable, rates of detection must remain constant among sites. We found that $E(\hat{p})$ varied among sites, which suggests that reliability of crowing male indices could be improved by adjusting counts for $E(\hat{p})$. However, Johnson (2008) cautioned that the cost to adjust counts may outweigh the benefits. We suggest that MR methodology may only be practical for intensive studies on relatively small areas with moderate to high densities of males. For operational surveys, managers may increase probability of detection by locating listening stations away from areas of high disturbance and allowing pheasants to acclimate to observer presence before counting crowing calls. For intensive studies, crowing male indices adjusted for probability of detection could be used as a standard from which to evaluate the more common pheasant population index derived from roadside surveys.

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Table 1. Pheasant population indices from replicate (*n*) crowing surveys on 18 study sites in southern Minnesota during spring 2007.

| | | | | Males/station | |
|---|---|---|---|---|---|
| Region | Study site | Stations | *n* | Mean | SE |
| Windom | 19 | 9 | 10 | 4.6 | 0.28 |
| | 20 | 9 | 10 | 6.4 | 0.37 |
| | 21 | 8 | 10 | 3.5 | 0.22 |
| | 22 | 9 | 10 | 5.5 | 0.26 |
| | 23 | 9 | 9 | 5.0 | 0.29 |
| | 24 | 9 | 9 | 4.7 | 0.22 |
| | 25 | 9 | 10 | 3.9 | 0.23 |
| | 26 | 9 | 10 | 5.5 | 0.27 |
| | 27 | 9 | 9 | 2.7 | 0.17 |
| Faribault | 28 | 8 | 10 | 3.0 | 0.21 |
| | 29 | 9 | 10 | 3.9 | 0.25 |
| | 30 | 9 | 10 | 2.7 | 0.15 |
| | 31 | 9 | 10 | 4.2 | 0.22 |
| | 32 | 9 | 10 | 3.1 | 0.18 |
| | 33 | 9 | 10 | 3.7 | 0.21 |
| | 34 | 9 | 10 | 3.7 | 0.17 |
| | 35 | 9 | 10 | 3.5 | 0.24 |
| | 36 | 9 | 10 | 1.2 | 0.10 |

Table 2. Link-function parameter estimates for study site 19 (reference group) and 33 based on Model 17.

| Parameter | MLE | 95% CI | Odds |
|---|---|---|---|
| $\pi$ (mixture1) | -0.4080 | -0.6056 – -0.2104 | ----- |
| *p*: intercept (site19) | -2.793 | -3.568 – -2.017 | ----- |
| *p*: mixture2 | 3.041 | 2.733 – 3.349 | 20.92 |
| *p*: time (t2,3) | 0.3143 | 0.2272 – 0.4013 | 1.369 |
| *p*: site33 | -1.116 | -1.938 – -0.2930 | 0.3277 |
| *p*: crows | 0.1585 | 0.0935 – 0.2236 | 1.172 |
| *p*: site33*crows | 0.2492 | 0.1275 – 0.3709 | 1.283 |
| *p*: disturb | -0.1298 | -0.2450 – -0.0147 | 0.8782 |

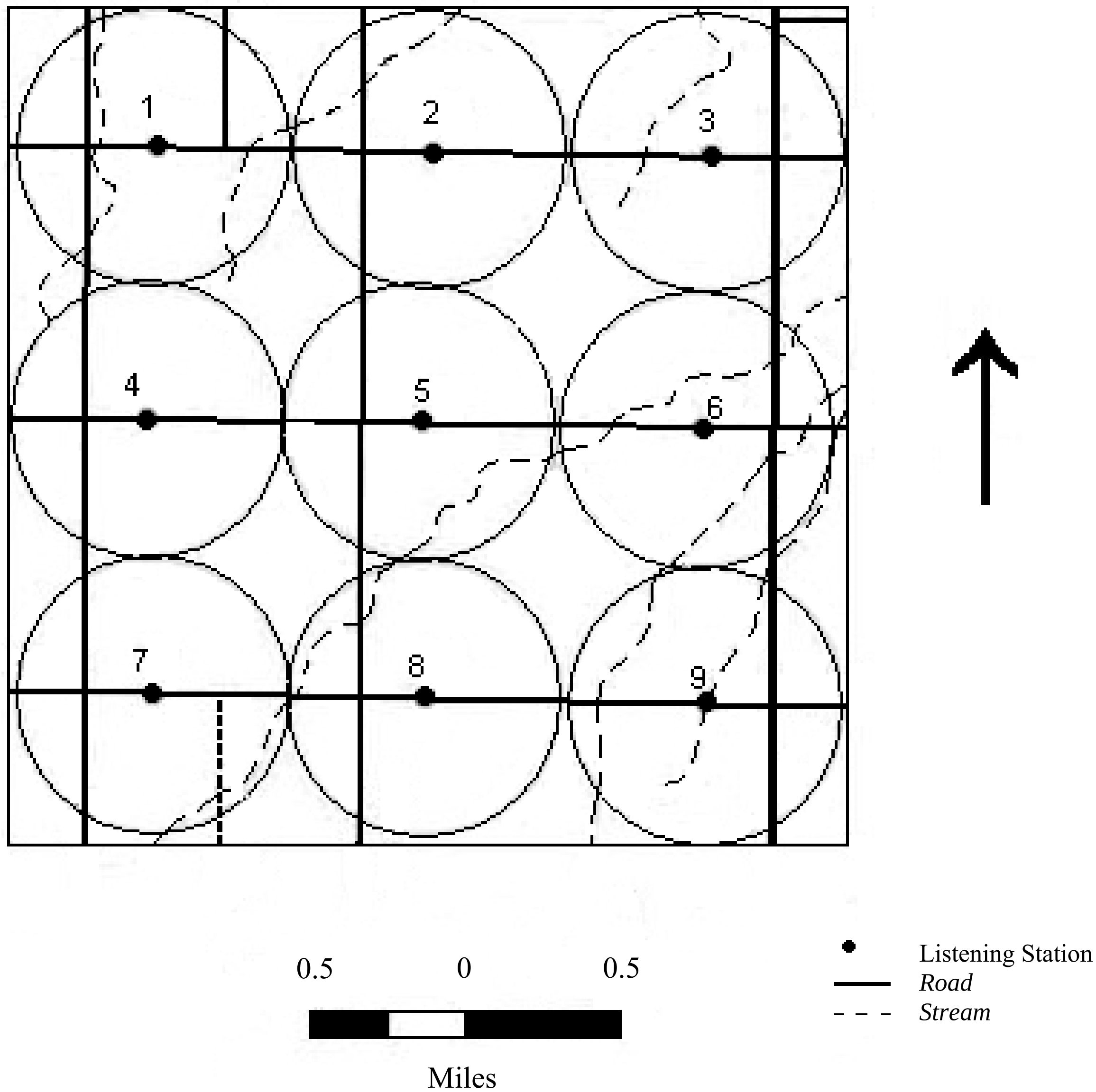


Figure 1.  Typical study site showing 9 crow-count listening stations and estimated 0.8 km (0.5 mile) auditory radii, Minnesota, spring 2007.

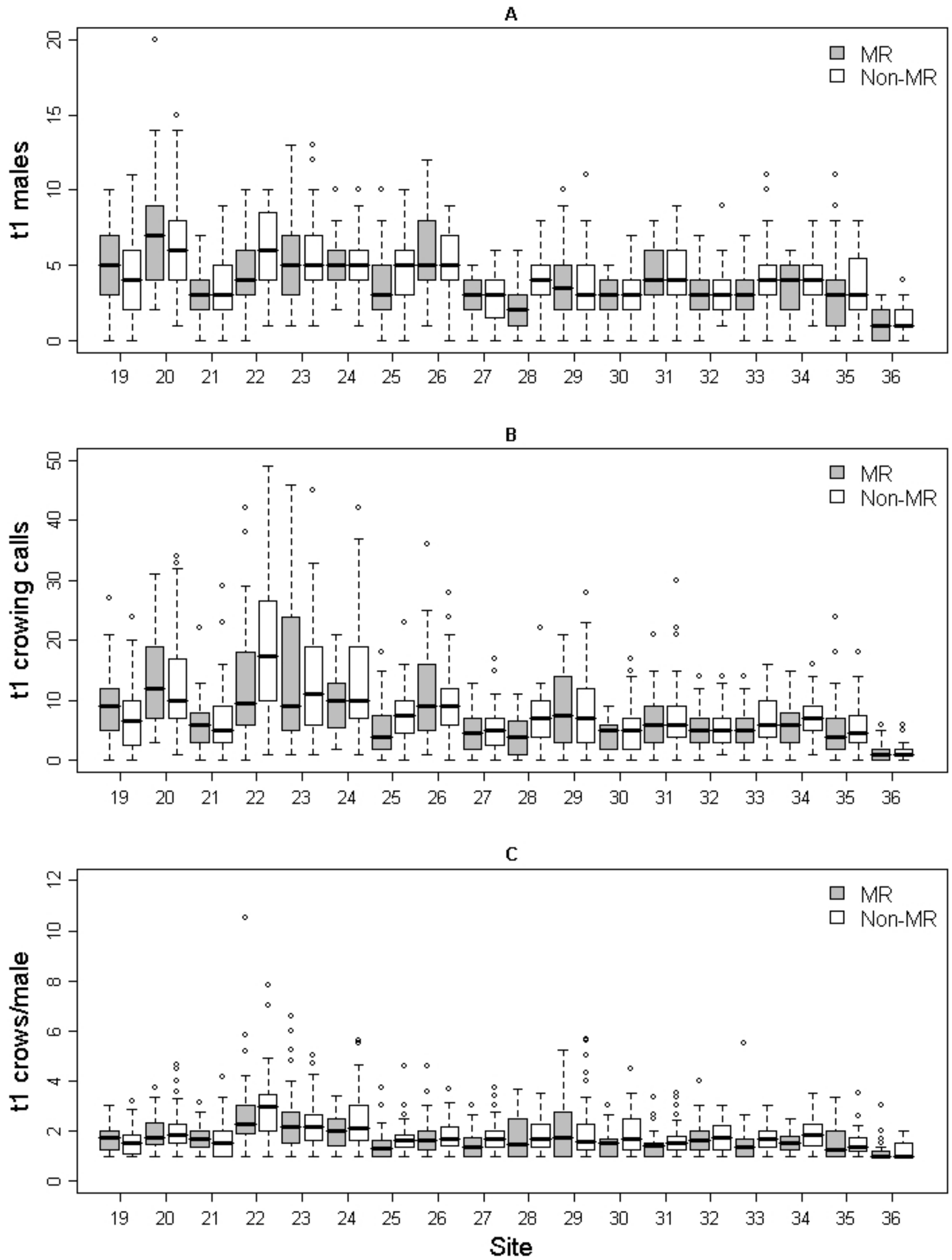


Figure 2. Number of male pheasants (A), crowing calls (B), and crows/male (C) detected in the first listening interval by study site and station type (mark-recapture [MR] versus non-MR). Boxplots do not include 65 sampling events (40 MR, 25 non-MR) with high disturbance.

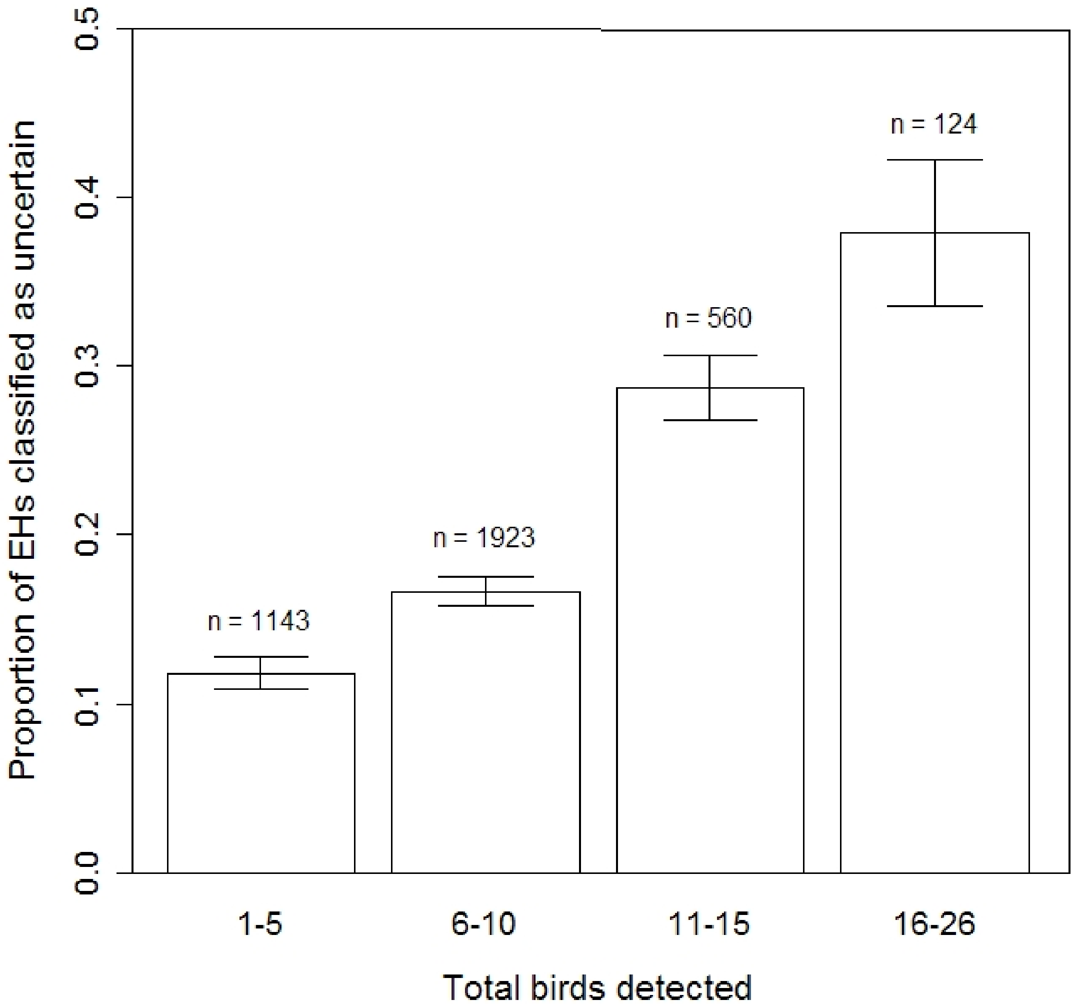


Figure 3. Proportion of encounter histories (EHs) classified as ‘uncertain’ as a function of total male pheasants detected at each site-station-date.

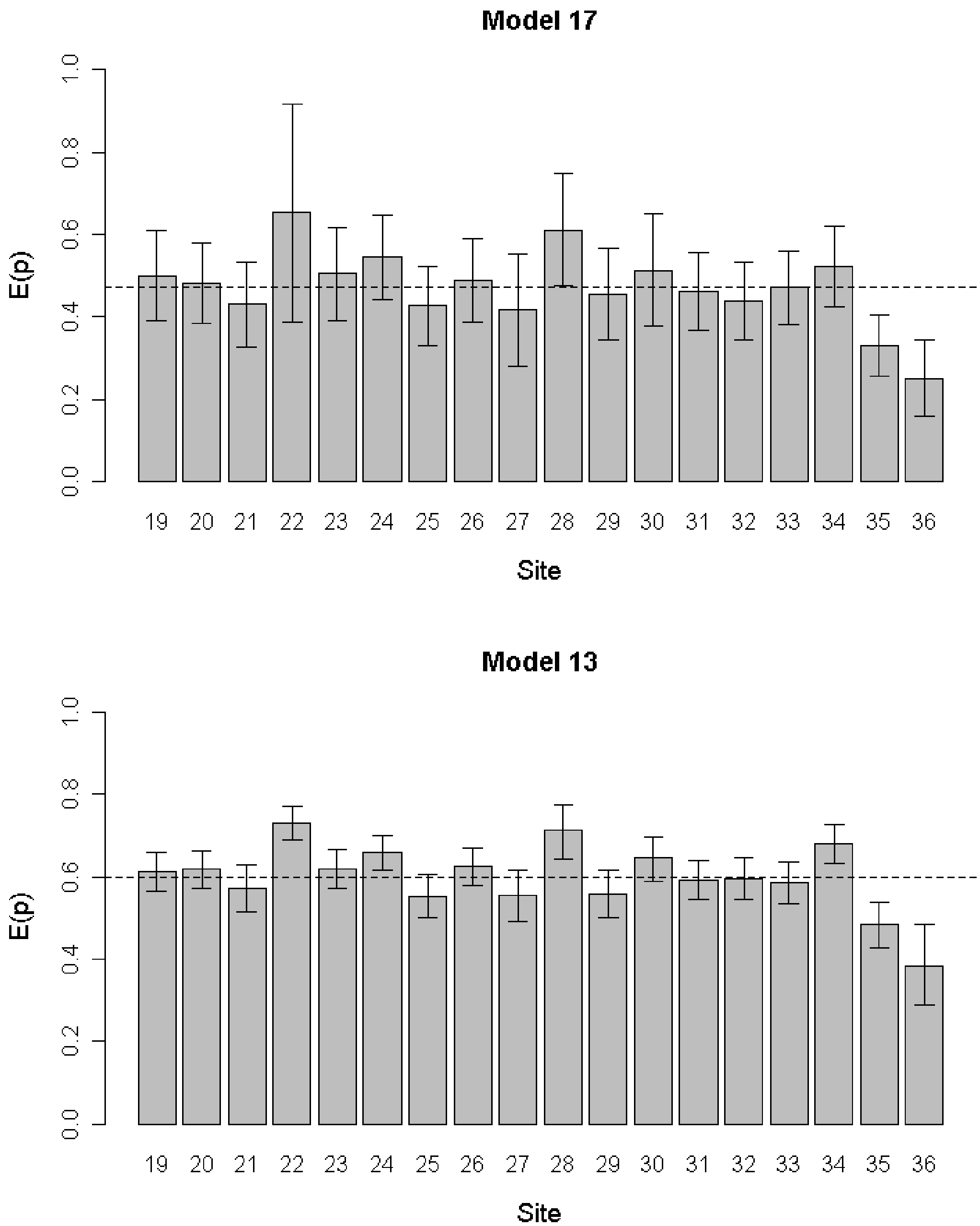


Figure 4. Expected probability of detection $E(\hat{\rho})$ in the first 2-minute listening interval based on model 13 and model 17 (mixture model) given median covariate values (for *DISTURB* and *CROWS*) observed at each site. Dashed lines are the overall mean probability of detection, and error bars denote asymptotic 95% confidence intervals.

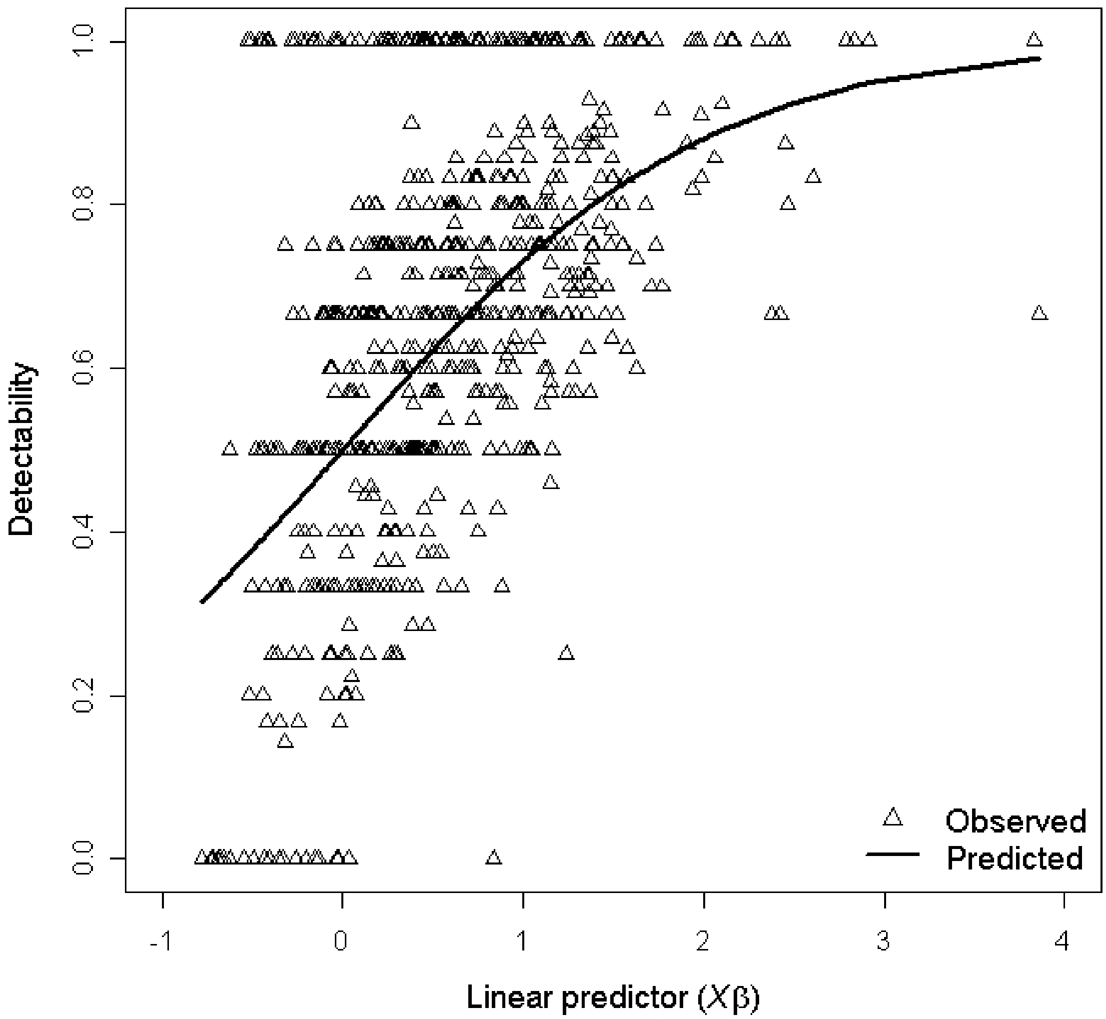


Figure 5. Jackknife estimate of predictability for model 13, where the linear predictor for detection probability contained *TIME*, *SITE*, *DISTURB*, *CROWS*, and *CROWS×SITE* effects. Observed detectability was the proportion of crowing males detected in the first 2-minute listening period of each sampling event (site-station-date). Predicted detectability was the predicted probability of detection for each event (site-station-date) based on fitting model 13 to all other sampling events.

# RESEARCH PROPOSAL: ESTABLISHMENT AND MAINTENANCE OF FORBS IN EXISTING GRASS STANDS

Molly Tranel

## INTRODUCTION

MNDNR Managers have requested information on establishing and maintaining an abundance and diversity of forbs in grasslands. A diversity of forbs in native grass stands provides better habitat for invertebrates and grassland birds, creates more heterogeneous vegetation structure, and may resist invasion from non-native invasive plants. Interseeding a diverse mix of forbs directly into existing vegetation, followed by management treatments that reduce competition with established grasses may provide a means for forbs to be reintroduced into species-poor grass stands. I developed a study to investigate the effects of 2 mowing and 2 herbicide treatments on: (1) the diversity and persistence of forbs interseeded into established grasslands in the farmland region of Minnesota, and (2) the abundance and community response of insects to interseeding treatments. Forbs will be interseeded in fall 2009 on 15-20 sites located throughout the farmland region that contain >10 acres of uniform, native grasses with few to no forbs present. During summer 2010, 4 management treatments of mowing (once or twice per season) and grass-selective herbicide application (at a high and low rate) will be randomly applied to plots within each site. Beginning in summer 2011, I will evaluate treatment effects by estimating plant species richness based on the presence/absence of 30 planted species and volunteer species. I will also assess vegetation structural characteristics using measures of visual obstruction, litter depth, and percent cover. Insects will be collected prior to vegetation sampling using sweep nets and identified to the lowest practical level. Monitoring will continue through summer 2013, or longer if funding allows.

## JUSTIFICATION

Although methods of restoring and maintaining grasslands are fairly well known, 82% of Minnesota Department of Natural Resources (MNDNR) wildlife managers responding to a survey on grassland information needs (Tranel 2008) indicated a lack of information on maintaining plant species diversity in restored grasslands. In particular, managers wanted more information on establishing and maintaining an abundance and diversity of forbs in grasslands. Research on forb establishment could provide managers with the most effective management techniques to improve grassland wildlife habitat.

One important function of forbs in grasslands is to provide habitat for invertebrates, an essential food for grassland birds and their broods (Buchanan et al. 2006). Insect abundance is strongly associated with forb abundance (Jones 1963) and diversity (Haddad et al. 2001), and insect abundance in chick diets has been positively correlated with growth rates and survival in gallinaceous birds such as grouse (Park et al. 2001, Huwer et al. 2008), gray partridge (*Perdix perdix;* Sotherton and Robertson 1990), and pheasants (*Phasianus cholchicus*; Hill 1985). Broods of gallinaceous birds such as the prairie chicken (*Tympanuchus cupido*) move directly from nests to brood habitat (Svedarsky 1979), and habitats with high forb abundance were preferred (Jones 1963, Drobney and Sparrow 1977).

A diversity of forbs also creates more heterogeneous vegetation structure, which some birds require for nesting and brooding (Volkert 1992). Sample and Mossman (1997) reported "several bird species such as dickcissel (*Spiza americana*) and savannah sparrow (*Passerculus sandwichensis*) are most abundant in fields with a strong forb component." In addition to the wildlife benefits, a diverse vegetation structure may better resist invasion of non-native invasive species (Pokorny et al. 2005, Sheley and Half 2006) and increase overall ecosystem health and function.

Regenscheid et al. (1987) found that insect availability and feeding rates of partridge and pheasant chicks were lowest in monotypic switchgrass stands compared to other cover types, and Leathers (2003) reported that mean invertebrate biomass was higher in interseeded Conservation Reserve Program (CRP) fields than controls. A number of MNDNR grasslands that were restored 10-20 years ago were seeded at heavy grass rates, with little or no forb component. In other grass stands, forbs were seeded but are now absent or present in low numbers. As a result, grasslands with little or no forbs are common in the farmland region of Minnesota. Over 60,000 acres of CRP contracts in Minnesota are scheduled to expire in 2009 (U.S. Department of Agriculture 2009). With this potential loss of grassland habitat, restoring the remaining grasslands to their full potential by increasing forb diversity and abundance will become more essential to wildlife.

Managers interested in increasing the diversity and quality of species-poor stands are faced with the costly option of completely eliminating the existing vegetation and planting into bare ground, or attempting to seed forbs directly into that vegetation. The latter technique, known as interseeding, could potentially reduce labor and fuel costs and protect from soil erosion (Packard and Mutel 1997) and was recommended by Rodgers (1999) to improve pheasant habitat. However, MNDNR managers report having limited experience and poor success with a few early interseeding attempts.

Management techniques that reduce competition from established grasses may provide an opportunity for forbs to become established in existing grasslands (Collins et al. 1998). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008). Mowing and application of selective herbicides are familiar techniques for controlling grasses. Williams et al. (2007) found that grasslands mowed frequently in the first growing season increased interseeded forb emergence and reduced forb mortality. Similarly, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, I will investigate the effects of 2 mowing and 2 herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota. In addition, I will monitor insect abundance in response to interseeding treatments. Finally, I will track the cost of implementing each management technique and conduct a cost-benefit analysis.

## STUDY AREA

The study will be conducted on 15-20 sites located throughout the farmland region (Figure 1). I have identified potential sites located on state, federal, and privately owned properties. Only sites with ≥10 uniform acres of native grass with similar soils and no or few forbs will be used. The number of sites will be determined based on a power analysis of pilot study results, conformity of sites to criteria described above, willingness of property managers to fully participate in the study design, and available budget.

### Pilot Site

I will assess feasibility of treatments and potential for identifying forb seedlings on 1 pilot site during summer 2009. This site is located on the Wood Lake Wildlife Management Area (WMA) in Redwood County and is a new MNDNR acquisition currently enrolled in the CRP. It was planted to 8 native grass species in 1999, burned in October 2008, and frost seeded with forbs in January 2009 (Figure 2). In addition to the pilot site, I will estimate number of vegetation and insect samples needed by sampling established grasslands with varying forb abundance throughout the farmland region during summer 2009.

## METHODS

Prior to interseeding, all sites will be surveyed to determine plant species already present and general condition of each site. Sites will then be prepared in the following manner: mowed in August 2009, burned in October 2009, and interseeded with native forbs during fall 2009 - winter 2010 (dormant season). For the purpose of this study, interseeding is defined as dormant season broadcast seeding directly into existing grass stands without prior tilling or herbiciding. The same seed mix (Table 1), consisting of 29 native forb species and 1 sedge species, will be interseeded at each site.

### Treatments

After each site is prepared and seeded, I will select a block with relatively uniform soil type, topography, and other physical characteristics and divide it into 10 plots (Figure 3) of equal size. I will then randomly assign 1 of the 4 treatments and the control to each of the 10 plots. Each of the 4 treatments and control will be replicated twice within each block. The following treatments, which are designed to suppress grass competition, will be applied during the 2010 growing season while the forbs are becoming established:

- Mow to a height of 4-6 inches once when vegetation reaches 10-12 inches in height.
- Mow to a height of 4-6 inches twice when vegetation reaches 10-12 inches in height.
- Apply grass herbicide Clethodim (Select 2 EC) at an 8 oz/acre rate in late May - early June.
- Apply grass herbicide Clethodim (Select 2 EC) at a 16 oz/acre rate in late May - early June.

### Vegetation Sampling

I will permanently mark the corners of treatment plots with metal stakes and record their coordinates using a Global Positioning System. I will locate sampling frames along a transect running through the center (excluding edges) of each treatment plot (Figure 3). I will estimate number of transects, transect length, and number of sampling frames needed from the pilot study based on number of plots required to detect >80% of the forb species present in the treatment parcel. I will sample vegetation twice/year, once in the early part of the growing season (June) and once in the late part of the growing season (September), beginning the year after treatment application.

I will estimate species richness by counting the presence/absence of the 30 planted species (Table 1), species present prior to the study, and volunteer species in each sampling frame. I will use Simpson's diversity index (Simpson 1949) to account for species richness and the proportion of each species present. Simpson's diversity index (*D*) is:

$$D = \frac{1}{\sum_{i=1}^{S} p_i^2}$$

*D* Simpson's diversity index

*S* total number of species in the community (richness)

$p_i$ proportion of *S* made up of the *i*th species (# species ***i* / ** total # of species )

Managers are interested in knowing which planted forb species become established and tend to persist over time. Therefore, I will primarily focus my vegetation sampling on species richness and diversity. However, structural measurements are useful in determining the value of a vegetation community in providing wildlife habitat (Herrick et al. 2005). Sample and Mossman (1997) categorized grassland habitat for birds based on measures of vegetation height-density, litter depth and cover, cover of standing residual vegetation, cover of standing live vegetation, cover of bare ground, and ratio of grass to forb cover. I will assess structural characteristics using measures of visual obstruction (Robel et al. 1970), litter depth, and percent cover of forbs, grass, and bare ground (Daubenmire 1959).

### Insect Sampling

If time and budget allow, I will assess the response of the insect community to management treatments. Immediately prior to vegetation sampling, I will collect insects using a 15 inch sweep net along randomly selected segments of the transect through each plot (Leathers 2003). Insects will also be sampled from control plots on each site located in areas that have not been interseeded. Data from the pilot study will be used to determine insect sampling intensity. Collected insects will be frozen, sorted by size, and identified to the lowest taxonomic level.

### Timeline (Figure 2)

- **Spring-Summer 2009:** Determine sample size, variance, and sampling techniques from pilot sites. Ground truth study sites and prescribe management treatments.
- **Fall 2009 – Winter 2010:** Initiate full study, collect pre-treatment measurements, mow, burn, and interseed study sites.
- **Summer 2010:** Apply management treatments to study sites.
- **Summer 2011:** Sample vegetation and insects during June and September.
- **Summer 2012:** Sample vegetation and insects during June and September.
- **Summer 2013:** Sample vegetation and insects during June and September

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Table 1. Prairie plant species and seeding rates selected for interseeding into established grasslands in southern Minnesota, 2009. Prices are estimates. Total cost is the total cost of each species per acre. It was determined by multiplying cost per ounce multiplied by the ounces per acre for each species.

| Common name | *Scientific name* | Ounce/ acre | Seeds / acre | % of mix[1] | Total cost[2] | Tolerance to transline |
|---|---|---|---|---|---|---|
| Leadplant | *Amorpha canescens* | 0.50 | 8,000 | 0.79% | $ 6.50 | Unknown |
| Black eyed Susan | *Rudbeckia hirta* | 1.20 | 110,400 | 10.92% | $ 4.80 | Susceptible |
| Maximilian Sunflower | *Helianthus maximilianii* | 0.50 | 6,500 | 0.64% | $ 2.50 | Susceptible |
| Yellow Coneflower | *Ratibida pinnata* | 1.35 | 40,500 | 4.01% | $ 6.75 | Susceptible |
| Golden Alexanders | *Zizia aurea* | 1.00 | 11,000 | 1.09% | $ 7.00 | Tolerant |
| Sky Blue Aster | *Aster oolentangiensis* | 0.85 | 68,000 | 6.73% | $ 6.80 | Moderate |
| Canada Milk Vetch | *Astragalus canadensis* | 1.75 | 29,750 | 2.94% | $ 8.75 | Unknown |
| Prairie Cinquefoil | *Potentilla arguta* | 0.85 | 195,500 | 19.35% | $ 8.50 | Tolerant |
| White Prairie Clover | *Dalea candida* | 1.50 | 28,500 | 2.82% | $ 6.00 | Moderate-Susceptible |
| Purple Prairie Clover | *Dalea purpurea* | 1.50 | 22,500 | 2.23% | $ 6.00 | Moderate-Susceptible |
| False Sunflower | *Heliopsis helianthoides* | 1.25 | 7,875 | 0.78% | $ 5.00 | Moderate |
| Alumroot | *Heuchera richardsonii* | 0.05 | 35,000 | 3.46% | $ 3.00 | Moderate |
| Narrow Leaf Purple Coneflower | *Echinacea angustifolia* | 0.85 | 5,950 | 0.59% | $ 10.20 | Unknown |
| Virginia Mountain Mint | *Pycnanthemum virginianum* | 0.20 | 44,000 | 4.35% | $ 6.00 | Unknown |
| Common Milkweed | *Asclepias syriaca* | 1.00 | 4,000 | 0.40% | $ 5.00 | Tolerant |
| Blue Vervain | *Verbena hastate* | 0.75 | 69,750 | 6.90% | $ 3.75 | Tolerant |
| Rough Blazingstar | *Liatris aspera* | 0.15 | 2,400 | 0.24% | $ 7.50 | Moderate |
| New England Aster | *Aster novae-angliae* | 0.65 | 42,900 | 4.25% | $ 5.85 | Moderate |
| Prairie Onion | *Allium stellatum* | 0.70 | 7,700 | 0.76% | $ 5.60 | Tolerant |
| Hoary Vervain | *Verbena stricta* | 0.65 | 18,200 | 1.80% | $ 3.90 | Tolerant |
| Heath Aster | *Aster ericoides* | 0.15 | 30,000 | 2.97% | $ 4.50 | Moderate |
| Stiff Goldenrod | *Oligoneuron rigidum* | 0.75 | 30,750 | 3.04% | $ 7.50 | Moderate |
| Culver's Root | *Veronicastrum virginicum* | 0.10 | 80,000 | 7.92% | $ 2.00 | Unknown |
| Showy Tick Trefoil | *Desmodium canadense* | 0.85 | 4,675 | 0.46% | $ 7.65 | Moderate-Susceptible |
| Wild Bergamot | *Monarda fistulosa* | 0.70 | 49,000 | 4.85% | $ 7.00 | Tolerant |
| Prairie Coreopsis | *Coreopsis palmate* | 0.25 | 2,500 | 0.25% | $ 6.25 | Unknown |
| Partridge Pea | *Chamaechrista fasciculate* | 2.00 | 5,400 | 0.53% | $ 6.00 | Unknown |
| Closed Bottle Gentain | *Gentiana andrewsii* | 0.08 | 22,400 | 2.22% | $ 4.80 | Unknown |
| Heart Leaf Golden Alexander | *Zizia aptera* | 0.20 | 2,400 | 0.24% | $ 5.00 | Tolerant |
| Brown Fox sedge | *Carex vulpinoidea* | 0.25 | 25,000 | 2.47% | $ 1.88 | Unknown |

[1] % of Mix = (Ounce per Acre * Seeds per Acre) / ∑ Seeds per Acre
[2] Total Cost = Ounce per Acre * Price per Ounce

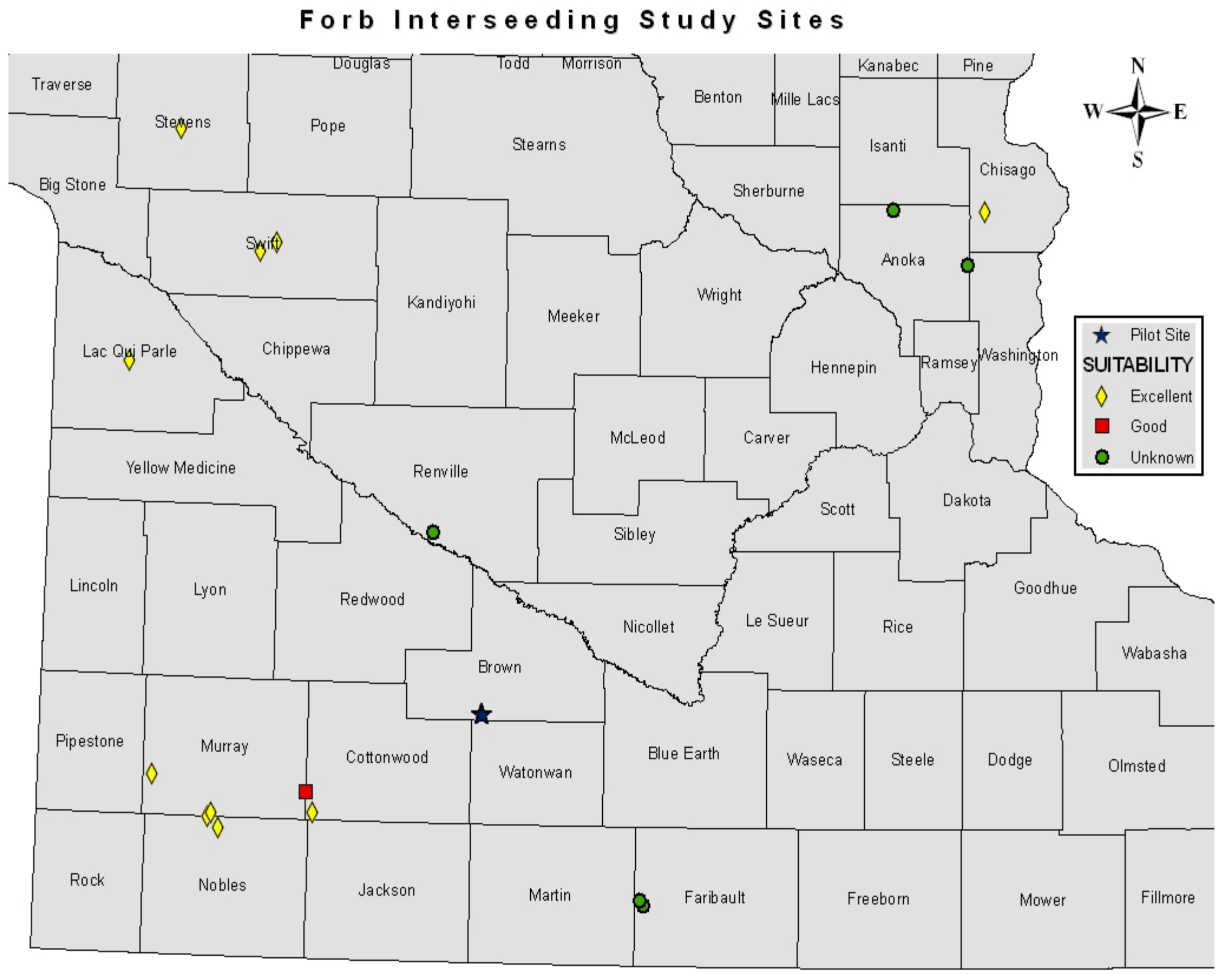


Figure 1. Locations of potential study sites for the forb interseeding study, categorized by suitablility. Sites of excellent suitability were flat, uniform, and contained similar vegetation and soils on >10 contiguous acres. Unknown sites have not been visited yet.

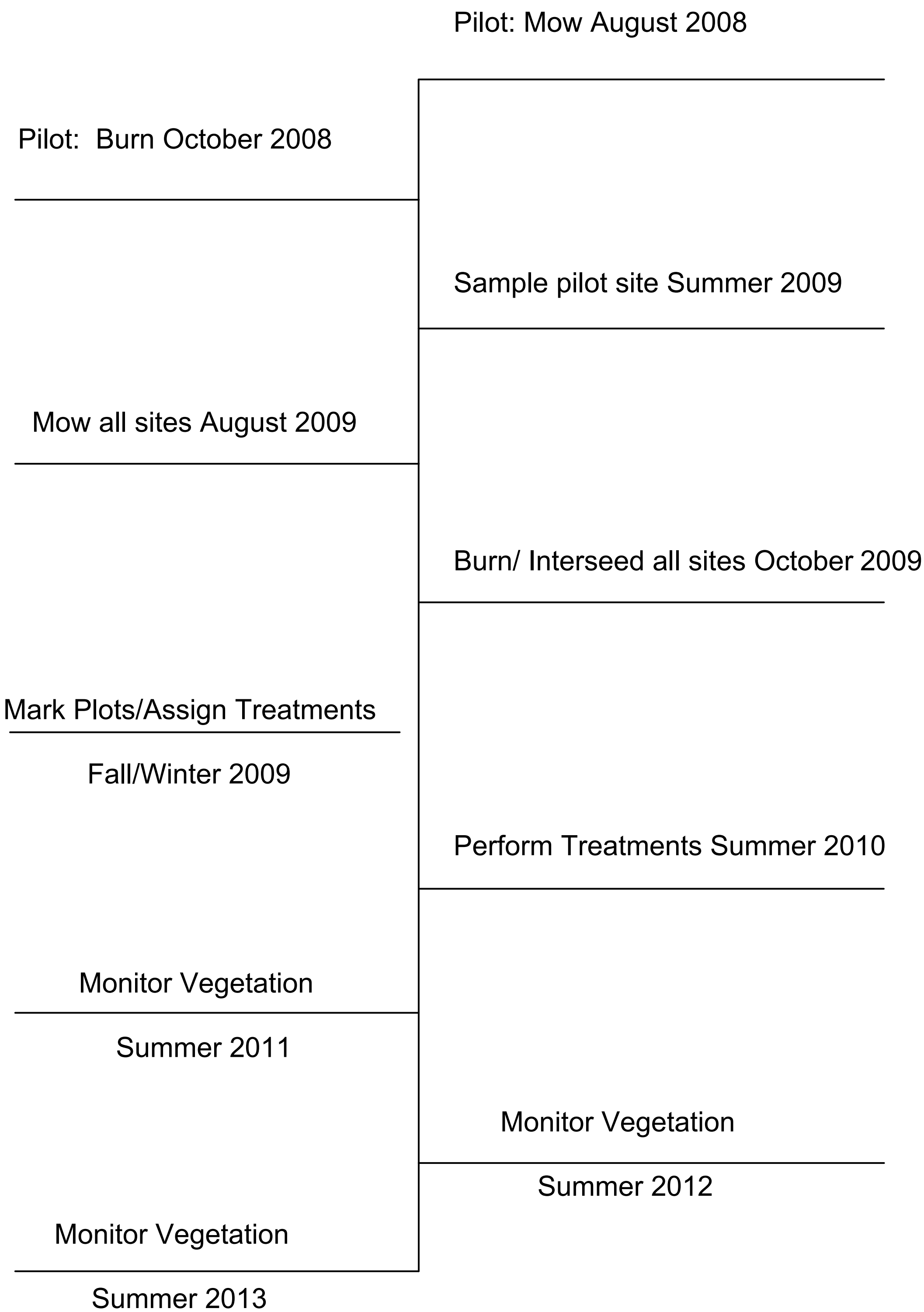


Figure 2. Timeline for forb interseeding study in southern Minnesota, 2009-2013.

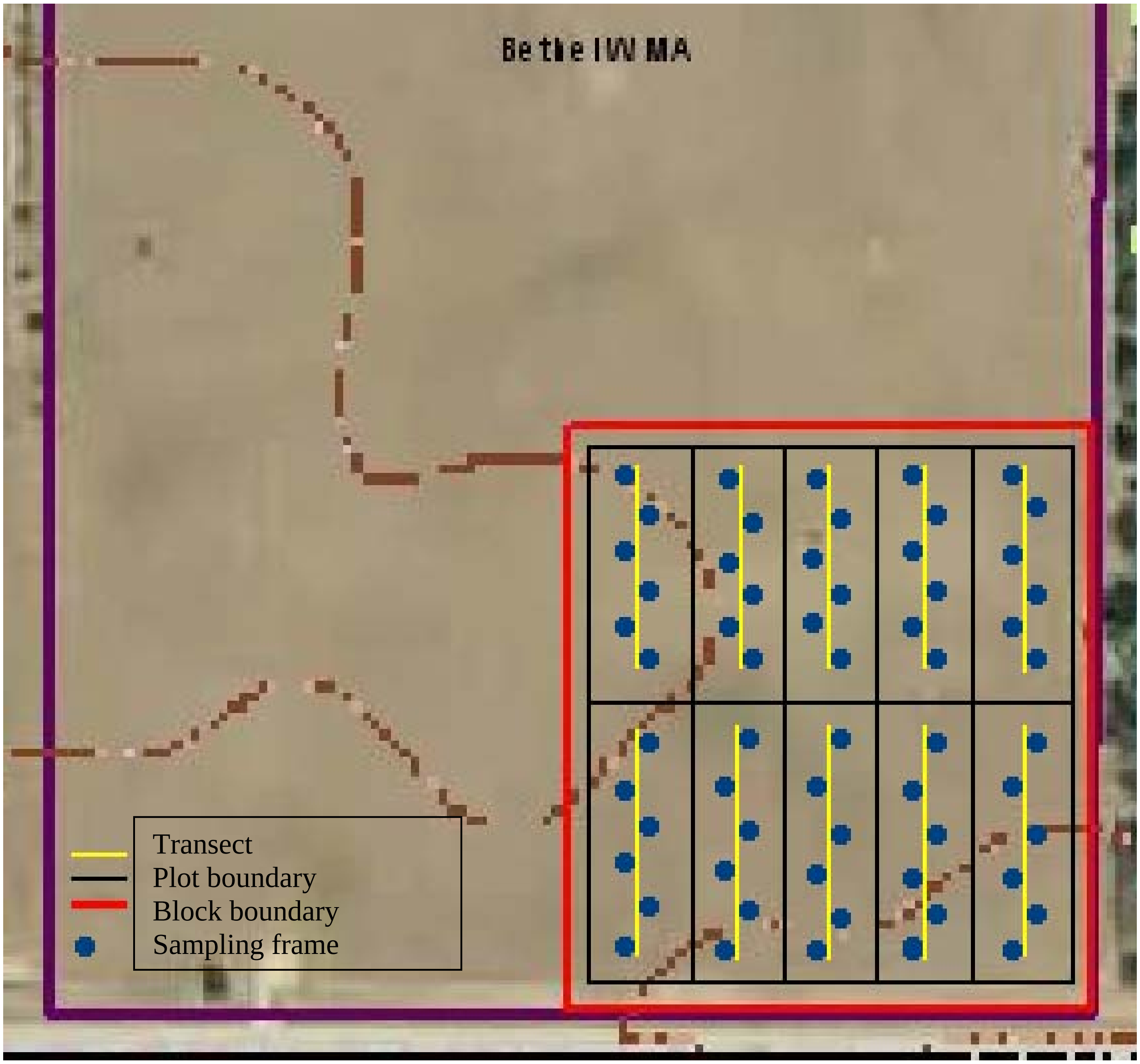


Figure 3. Example of a representative study site on the Bethel WMA. The purple line delineates the entire WMA, and the red line delineates the site (block) that meets study criteria. The black rectangles represent the plots that will receive 1 of 4 management treatments or serve as a control. Each of these treatments and control will be replicated twice within a block. Vegetation will be sampled within frames (blue circles) along transects (yellow lines) running through the center of each plot.

# WILD TURKEY FOOD HABITS ON THE NORTHERN FRINGE OF THEIR RANGE IN MINNESOTA

Eric M. Dunton, Joshua T. Ream, John Fieberg, and Kurt J. Haroldson

## SUMMARY OF FINDINGS

The purpose of this study was to evaluate diet selection and body condition of eastern wild turkeys (*Meleagris gallopavo silvestris)* in agricultural and forested areas on the northern fringe of their range in Minnesota. During winter 2009, 7 turkeys were collected in forested habitat and 24 in agricultural habitat. Adult females in forested habitat had 32% less body weight, 72% less total body fat, and were assigned to lower body condition classes than adult females from agricultural habitats. Forested turkeys' diets consisted of a mixture of high energy (acorns) and low energy (grass, smooth rose, leaf litter) food items, while diets of turkeys located in agricultural habitats consisted primarily of high energy (corn) food items.

## INTRODUCTION

The current range of the eastern wild turkey extends far north of what was identified by Schorger (1966) as their historical range. This northern expansion has been associated with increased availability of food during winter (Wunz 1992, Wunz and Pack 1992, Kubisiak et al. 2001), which was considered limiting prior to settlement by European farmers. Wild turkey range in Minnesota and throughout the northeastern United States and southeastern Canada is currently expanding northward beyond agricultural areas (Kimmel and Krueger 2007). It is unknown how far turkeys will expand outside of mixed forest-agriculture areas into northern forest areas, and what their diet will include. Understanding winter diet selection of turkeys on the northern periphery of their range and the interaction of agriculture, snow conditions, and food habits will provide management tools to enhance turkey survival outside of an agriculturally dominated landscape.

The eastern wild turkey is a food generalist with a winter diet ranging from >20 species (Korschgen 1967) to a restricted diet of only corn (Porter et al. 1980). As wild turkey range expanded north through mixed forest-agricultural habitats, Porter (2007) stated, “Looking back at the field studies of the 1970s, it is clear that they were telling us more than we realized: snow and cold are not the issue, the key is food.” Adequate information is available on turkey foraging behavior and survival in northern turkey habitats with access to agricultural foods (Porter et al. 1980, Vander Haegen et al. 1989, Kassube 2005, Kane et al. 2007), but information is lacking on turkey food habits in northern non-agricultural areas.

Our objectives were to: (1) determine winter foods used by wild turkeys on the northern fringe of their range in Minnesota; (2) describe diet as a function of agriculture and snow condition; and (3) compare body condition of wild turkeys with access to high-energy agricultural diets to those without.

## STUDY AREA

We conducted this study north of Minnesota’s historical wild turkey range (Leopold 1931, Schorger 1966, Snyders 2009) where wild turkey populations were established by translocation during 1990s - 2008. This region is located within the Western Superior Uplands and Northern Minnesota Drift and Lake Plain Ecological Sections of the Laurentian Mixed Forest Ecological Province (MNDNR 2003). The 25,959 $km^2$ study area is comprised of 35% upland deciduous forest, 31% crop/grass, 16% aquatic environment, 10% shrubland, 4% upland conifer forest, 2% lowland conifer forest, 2% lowland deciduous forest, and 1% non-vegetated (GAP Analysis Program MNDNR 2008).

To aid in locating wild turkeys with and without access to agricultural foods, we stratified the study area using a 500 ha grid and classified each cell to 1 of 3 turkey habitat categories based on reclassified GAP land cover data: agricultural cells contained ≥ 30% cropland and ≥ 20% forested habitat; forested cells contained ≥ 50% forested habitat and 0% cropland; and other cells contained all other combinations of habitats (Figure 1). We used a focal sum analysis to identify forested cells that were surrounded by other forested cells. We focused surveys on clusters of forested cells, but we also surveyed some forested cells that were surrounded by agricultural cells.

## METHODS

Using fixed wing aircraft we attempted to locate an equal number of wintering flocks of turkeys in agricultural and forested strata. We used real-time, moving-map software (MNDNR 2005) coupled to a global positioning system receiver and a tablet-style computer to guide transect navigation and record turkey locations and aircraft flight paths directly to a geographic information system (Haroldson 2007). Turkeys were then relocated on the ground within 1-3 days. We then attempted to collect 1-5 turkeys from each flock in late afternoon or early evening, when crops are most likely to be full (Hillerman et al. 1953), by shooting.

At each collection site, we recorded date, snow depth, snow condition (e.g., crusted vs. powder snow), temperature, habitat class (agricultural versus forested), and geographic coordinates. We verified habitat class by plotting collection sites on Farm Service Agency 2008 aerial imagery, and identifying presence or absence of cropland within a 1,545-m radius buffer (based on the 750 ha winter home range of wild turkeys in Minnesota reported by McMahon and Johnson (1980)). Habitat was classified as forested if no cropland was located within the buffer; otherwise habitat was classified agricultural.

We evaluated body condition of wild turkeys collected in forested and agricultural habitats based on relative body weight and 3 estimates of body fat. We estimated total body fat of adult hens using a formula from Pekins (2007). We also assigned turkeys to 1 of 4 body condition classes based on amount and color of visible fat (Carter 1970). Finally, we assigned turkeys to 1 of 3 classes based on the amount of fat visible on the gizzard. We tested for differences in estimated weights and in the distribution of body condition classes among adult females in agricultural and forested habitats using a t-test and Monte Carlo $\chi^2$ test (Hope 1968), respectively. Tests were conducted using the t.test and chi square.test functions in the R programming language (R Development Core Team 2008).

We determined frequency of occurrence and weight of food items present in the crops and gizzards according to the methods of Korschgen (1967). We determined dry matter content of foods by drying to a constant weight at 50°C (Decker et al. 1991). We assigned each food item 1 of 3 classes (high, medium, and low) based on energy content (Decker et al. 1991).

## RESULTS

During 2 January – 24 February 2009, we aerially surveyed 122 forested strata and 103 agricultural strata. We located 0 turkeys in forested strata and 1,130 turkeys (mean flock size = 23) in agricultural strata. We collected 31 turkeys; 7 from forested habitats (i.e., no agricultural foods present within 1545 m), and 24 from agricultural habitats. The 7 turkeys collected in forested habitat were all found by MNDNR staff on roads and trails that were plowed for logging, military training, or state park use. Snow plowing exposed snow-covered ground in forested habitats and turkeys were found foraging on the newly plowed roads. Adult females from forested habitats had 32% less body weight ($P < 0.001$), 72% less total body fat, and were assigned to lower body condition classes than adult females from agricultural habitats ($P = 0.0012$) (Table 1). One juvenile female and 1 adult male collected in forested habitats weighed 3% and 30% less, respectively, than their counterparts in agricultural habitats (Table 1). We classified gizzard fat from 6 forest habitat birds as no fat and 1 as fat. In agricultural habitats, we classified 14 turkey gizzards as very fat and 10 as fat.

We classified snow conditions at all forested collection sites as powder, and snow depth averaged 39.2 cm. We classified snow conditions at 10 agricultural sites as powder and 14 as crusted. Snow depth at agricultural sites averaged 27.4 cm across both snow condition classes. Mean collection date of forested birds was 24, January 2009 (range 14 January – 7 February) and 30 January 2009 (range 8 January – 24 February 2009) for agricultural birds. Temperature and monthly snow depth data from the Minnesota Climatology Working Group were not available at the time of this report.

High energy food (i.e., acorn [*Quercus spp.*]) was found in 86% of the crops from forest-habitat turkeys but formed only 47% of the crop contents by weight (Table 2). For agricultural habitat turkeys, high energy foods (e.g., corn [*Zea mays*]) formed 86% of the crop contents by weight (Table 3). Crops from 8% of turkeys collected in agricultural habitats and 14% of turkeys collected in forested habitats were empty or nearly empty.

## DISCUSSION

Butler et al. (2007) found that turkey sightability was primarily influenced by flock size and vegetative cover. Our ability to detect birds was also influenced by snow conditions, in addition to type of cover (agricultural versus forested habitat). When snow depth was >25 cm, large wintering flocks of wild turkeys were often congregated near agricultural food sources and were easily detected using fixed wing aircraft. However when snow depth diminished in late February, aerial surveys in agricultural areas became less effective. Aerial surveys in forested strata were ineffective both with and without deep snow even though we searched forested strata that were known to have established populations. We suspect that some turkeys in forested areas moved to areas with supplemental food sources or remained in forested habitats in small groups that were hard to detect by aerial survey.

The forested turkeys we collected were all found along roads and trails where snow plowing occurred. Snow plowing permitted ground foraging on only a small fraction of forested habitats, which may account for the high frequency of occurrence (86%) of high energy food items (acorns) found in crops of turkeys collected in forested habitats, but the lower proportion of total weight (47%). Not surprisingly, turkeys collected in agricultural habitats had been feeding on corn and other high energy foods that were not available in forested habitats (e.g., (Porter et al. 1980, 1983; Vander Haegen et al. 1989, Healy 1992). Although, Hurst (1992) considered crop content analysis the best technique for evaluating wild turkey food habits, it is negatively biased toward succulent foods and soft-bodied invertebrates, which are digested more rapidly than hard and fibrous food items. In this study, bias is assumed minimal because few succulent foods were available during winter

Body weights of adult and juvenile hens collected in forested habitats in this study were below average whereas body weights of adult and juvenile hens collected in agricultural habitats were within the average range reported by Porter (1980) in Minnesota, Vander Haegen et al (1989) in Massachusetts, and Coup and Pekins (1999) in New Hampshire. Pekins (2007) suggested that adult hens weighing < 3.0 kg have minimal body fat and were approaching a critical threshold of malnutrition. Thus, most adult hens collected from forested habitats in this study were showing signs of food deprivation. As supporting evidence, we frequently observed turkeys in forested habitats remaining in their roosts late in the morning. This behavior is generally considered an indication of stress (Hayden and Nelson 1963).

Findings from this study indicate that turkeys in agricultural areas were able to find sufficient food (primarily corn) to maintain energy balance and fat reserves throughout the winter, even when snow depth was >25 cm. In contrast, turkeys using exclusively forested habitats in deep snow were in poor body condition with little to no fat reserves. Powder snow >15-20cm hinders mobility, and >30cm can prevent movement of wild turkeys (Austin and DeGraff 1975, Porter 1977, Healy 1992). Deep persistent snow cover can ultimately result in starvation. Wild turkeys began starving when snow depth was >30cm for >2 weeks in Pennsylvania (Wunz and Hayden 1975), 49 days in Wisconsin (Wright et al. 1996), and 40-59 days in New York (Roberts et al. 1995). Wright et al. (1996) documented starvation when deep snow restricted movements even though food was available within 0.8 km.

Further range expansion of wild turkeys in Minnesota's northern forests may be limited by availability of supplemental food during winter. Wild turkey range may expand during periods with consecutive mild winters and then contract during severe winters. Because opportunities for agriculture are limited in this region, unharvested crops and livestock feeding operations may attract large concentrations of wintering turkeys, resulting in depredation complaints.

## ACKNOWLEDGEMENTS

Funding for this project was provided by MNDNR with contributions from Minnesota State Chapter of the National Wild Turkey Federation. R. Wright, MNDR, provided GIS support. MNDNR wildlife, parks, and forestry staff and Camp Ripley staff provided locations of forested turkeys and assisted with data collection. Private landowners in the study area allowed access to their lands. T. Buker, MNDNR enforcement pilot, flew surveys. B. Haroldson provided assistance with survey software. W. Smith, MNDNR Ecological Resources, provided assistance with food item identification. P. Pekins, University of New Hampshire, provided comments on the proposal.

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Table 1. Estimates of body fat for 31 wild turkeys collected in forested versus agricultural habitats on the northern fringe of their range in Minnesota during winter, 2009.

| | | | | | Visual fat score | | | | Estimated total body fat[e] | |
|---|---|---|---|---|---|---|---|---|---|---|
| Habitat | Gender | Age | n | Average weight (kg) | Very fat[a] | Fat[b] | Lean[c] | Thin[d] | kg | % |
| Forest | Female | Adult | 5 | 3.24 | 0 | 0 | 2 | 3 | 0.21 | 5.80 |
| Forest | Female | Juvenile | 1 | 3.81 | 0 | 0 | 1 | 0 | - | - |
| Forest | Male | Adult | 1 | 6.48 | 0 | 0 | 1 | 0 | - | - |
| Forest | Male | Juvenile | 0 | 0 | 0 | 0 | 0 | 0 | - | - |
| | | | | | | | | | | |
| Ag | Female | Adult | 14 | 4.75 | 5 | 5 | 4 | 0 | 1.02 | 20.57 |
| Ag | Female | Juvenile | 3 | 3.91 | 0 | 2 | 1 | 0 | - | - |
| Ag | Male | Adult | 3 | 9.26 | 2 | 1 | 0 | 0 | - | - |
| Ag | Male | Juvenile | 4 | 6.41 | 0 | 4 | 0 | 0 | - | - |

[a] Large deposits of fat on mid-line of breast, thighs, back, around crop, at the posterior of the body cavity, and immediately beneath skin. Fat is bright yellow (Carter 1970).
[b] Large fat deposits on back and thighs and reduced deposits elsewhere. Fat may be orange in color. (Carter 1970).
[c] Fat deposits are completely resorbed. Breast muscle has “normal” contour. Dark orange color in cellular framework of resorbed fat deposits (Carter 1970).
[d] Breast muscle attains wedge-like appearance (“hatchet-breast”). Skin resembles parchment (Carter 1970).
[e] Body fat (g) = 571.3 x (kg body weight) – 1696; $R^2 = 0.59$, $P < 0.05$ (Pekins 2007)

Table 2. Crop contents and energetic value of food items for 7 wild turkeys collected in forested habitats on the northern fringe of their range in Minnesota during winter, 2009.

| | Weight | | | |
|---|---|---|---|---|
| Food item | Total (g) | % of total | Frequency (%) | Energetic diet class |
| Acorn (*Quercus spp.)* | 158.9 | 46.6 | 86 | High |
| Grass (*Poa spp.*) | 98.7 | 28.9 | 86 | Low |
| Smooth rose (*Rosa blanda*) | 48.5 | 14.2 | 14 | Medium |
| Leaf litter | 30.3 | 8.9 | 100 | Low |
| Hazel catkins (*Corylus spp.*) | 2.7 | 0.8 | 29 | Low |
| Unknown seed | 0.8 | 0.2 | 14 | Medium |
| Feathers | 0.5 | 0.2 | 14 | Unknown |
| Sumac (*Rhus spp.*) | 0.4 | 0.1 | 29 | Medium |
| Poison ivy (Toxicodendron spp.) | 0.3 | 0.1 | 14 | Medium |
| Beetle (*Coleoptera spp.*) | 0.05 | TR[a] | 14 | Unknown |
| | | | | |
| Total | 341.2 | | | |
| Mean forested turkey crop weight | 48.7 | | | |

[a] Trace (TR) amount of food item present in diet < 0.1

Table 3. Crop contents and energetic value of food items for 24 wild turkeys collected in agricultural habitats on the northern fringe of their range in Minnesota during winter, 2009.

| Food item | Weight | | Frequency (%) | Energetic diet class |
|---|---|---|---|---|
| | Total (g) | % of total | | |
| Corn kernels (*Zea mays*) | 821.0 | 79.8 | 92 | High |
| Corn parts (*Zea mays*) | 30.3 | 3.0 | 29 | Low |
| Sunflower seed (*Helianthus spp.*) | 38.3 | 3.7 | 17 | High |
| Grass (*Poa spp.*) | 33.3 | 3.2 | 42 | Low |
| Millet (*Panicum spp.)* | 17.8 | 1.7 | 13 | High |
| Cow manure | 16.8 | 1.6 | 8 | Unknown |
| Club moss (*Lycopodium spp.*) | 14.7 | 1.4 | 17 | Low |
| Sensitive fern (*Onoclea sensibilis*) | 12.2 | 1.2 | 17 | Low |
| Unknown seed | 10.2 | 1.0 | 4 | Medium |
| Oats (*Avena sativa*) | 8.5 | 0.8 | 13 | High |
| Leaf litter | 7.0 | 0.7 | 33 | Low |
| Pine needle (*Pinus spp.*) | 7.1 | 0.7 | 8 | Low |
| Poison ivy (*Toxicodendron spp.*) | 3.6 | 0.3 | 4 | Medium |
| Unknown Forb seed | 3.1 | 0.3 | 4 | Medium |
| Acorn (*Quercus spp.)* | 2.3 | 0.2 | 8 | High |
| Hazel catkins (*Corylus spp.*) | 1.2 | 0.1 | 13 | Low |
| Thistle seed (*Guizota spp.*) | 0.6 | 0.1 | 4 | High |
| Aven seed (*Geum spp.*) | 0.4 | TR[a] | 8 | High |
| Unknown tree/shrub bud | 0.1 | TR[a] | 13 | Medium |
| Feather | 0.3 | TR[a] | 4 | Unknown |
| Snail | 0.1 | TR[a] | 4 | Unknown |
| Unknown insect cocoon | 0.0 | TR[a] | 4 | Unknown |
| Beetle (*Coleoptera spp.*) | 0.0 | TR[a] | 4 | Unknown |
| Total | 1028.8 | | | |
| Mean agricultural turkey crop weight | 42.9 | | | |

[a] Trace (TR) amount of food item present in diet < 0.1

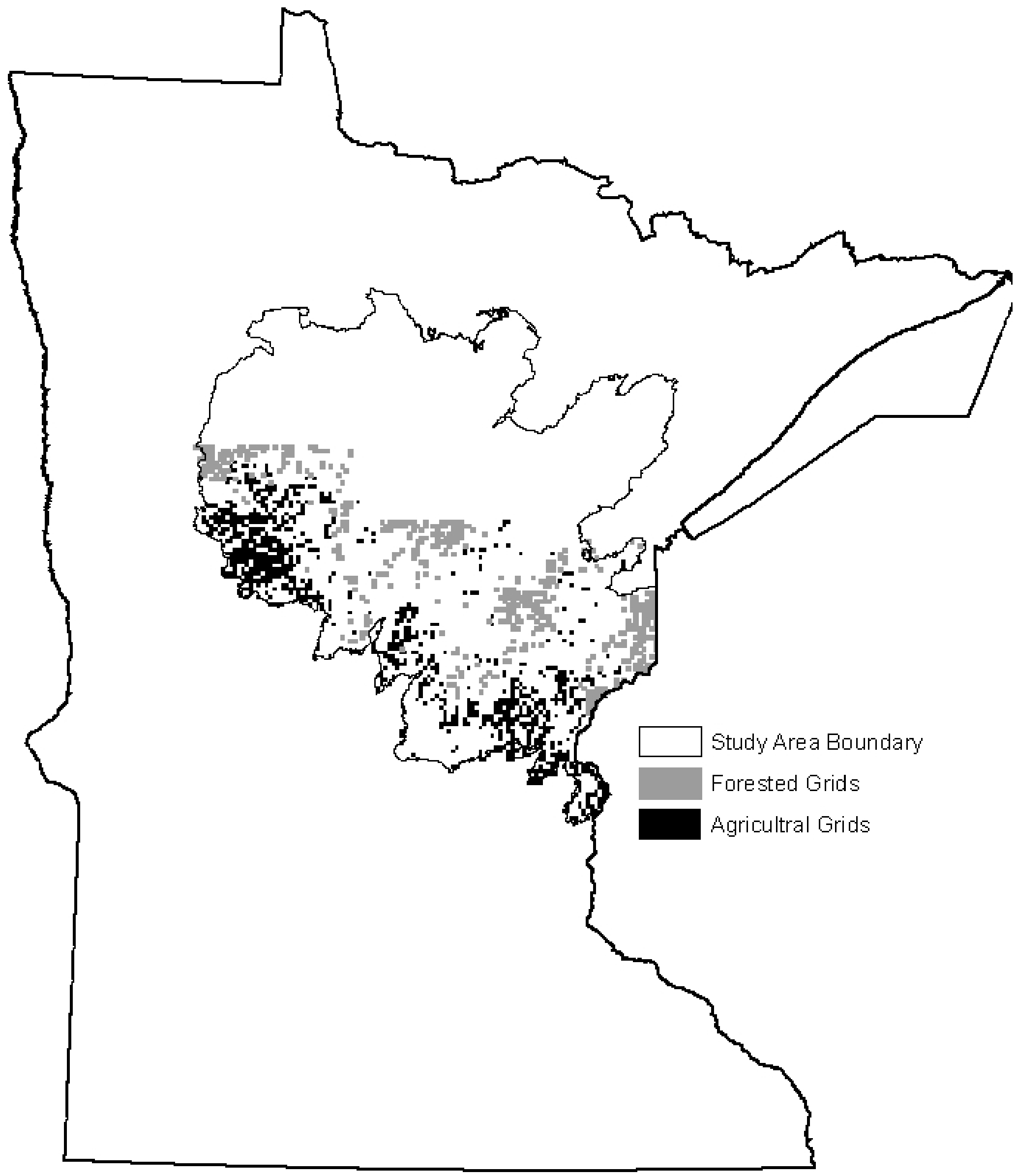


Figure 1. Study area location, forested and agricultural grids surveyed to determine wintering wild turkey flock location for winter food habits project on the northern fringe of wild turkey range in Minnesota, 2009.