Summaries of Wildlife Research Findings

2018

Minnesota Department of Natural Resources
Division of Fish and Wildlife
Wildlife Populations and Research Unit

SUMMARIES OF WILDLIFE RESEARCH FINDINGS
2018

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Printed in accordance with Minn. Stat. Sec. 84.03

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DEVELOPING METHODOLOGIES FOR PREDICTING THE LOCATIONS OF WOOD DUCK BREEDING HABITAT COMPONENTS IN MINNESOTA

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SUMMARY OF FINDINGS

There have been alterations to both aquatic and terrestrial habitats used by wood duck (*Aix sponsa*) hens and broods in Minnesota and the Upper Midwest during recent decades. We initiated this study to develop methodologies to predict the locations and monitor spatiotemporal changes in the areal extent of wood duck breeding complexes. Specifically, we want to use Light Detecting and Ranging (LiDAR) data to identify multiple habitat components and to monitor future changes in these components. We will provide better historical context regarding spatiotemporal changes in nesting habitat by analyzing Forest Inventory and Analysis (FIA) data with a quantitative method currently being developed to accurately estimate the population variance of stems that may have suitable nesting cavities. Our specific objectives are to (1) develop and evaluate spatial predictive models of habitat components that are important to breeding wood ducks (i.e., tree species [alternatively deciduous v. coniferous], diameter-at-breast height [DBH], tree canopy density, stand type, wetland type, water depth) based on LiDAR-generated metrics or other sources of spatial data [e.g., National Wetland Inventory (NWI), existing Geographic Information System (GIS) layers, aerial photographs], (2) ascertain the optimal pulse density of LiDAR needed to accurately measure or classify each habitat component of importance to wood ducks, (3) determine the generalizability of the LiDAR method for predicting the locations of habitat components by applying algorithms developed from data collected in the main study area (Cass County, Forest Ecological Province) to other sites in the Forest, Prairie, and/or Transition Provinces at which adequate LiDAR data have been obtained, (4) estimate the species- and DBH-specific proportions of trees with suitable cavities and detection probability of suitable cavities from empirical field data, and (5) determine whether there has been a change in the number of potential nest trees since the 1970s based on changes in FIA data.

We conducted vegetation surveys at 677 wetland plots during Summer 2016 and 2017, and 323 forest plots during Fall 2016, Spring 2017, Fall 2017, and Spring 2018. We assigned a habitat classification to 14 types of dominant emergent cover and 6 types of loafing structures during wetland surveys, 12 cover types to forest plots during nesting habitat surveys, and measured several other habitat variables in each survey. We examined 7,869 trees during forest surveys, and classified 223 cavities as suitable and 111 as marginally suitable for nesting wood ducks. Because data were sparse for relatively large DBH trees of multiple species (≥40 cm for early and mid-successional species, ≥50 cm for late successional species), we surveyed additional forest plots to obtain sufficient data on large-DBH trees with suitable cavities.

Flights to collect LiDAR data originally scheduled to occur during Fall 2016 were postponed until Fall 2017. This data became available during Summer 2018, and we began

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associating ground-level aquatic and forest vegetation measurements to LiDAR data during Winter 2019.

We began analyzing FIA data to estimate the change in population of 7 tree species- that were common in our study area and had some proclivity to produce suitable nesting cavities in the Laurentian Mixed Forest Province of Minnesota since 1990. We will use these population estimates and empirical knowledge of the influence of tree species, DBH, and health status to make inferences regarding temporal changes in suitable nesting cavities within this ecological province.

INTRODUCTION

Some terrestrial and aquatic habitats used by wood duck hens and broods during the pre-nesting, nesting, and brood-rearing life-cycle phases have been altered substantially in Minnesota and the Upper Midwest during recent decades. For example, there were decreases in the areal extent of some classes of aquatic habitats in northcentral Minnesota (Radomski 2006) and in the number of beaver impoundments in the forested portion of Minnesota between the early 1990s and 2002 (Dexter 2002, p. 52), both of which were used by wood duck broods (see McGilvery 1968, Bellrose and Holm 1994). Although the number of potential nesting trees for wood ducks was projected to increase both in Minnesota (Jaakko Pöyry Consulting, Inc. 1994) and the Upper Midwest (Denton et al. 2012*b*), there has been recent concern among Minnesota Department of Natural Resources (MNDNR) managers that harvesting relatively large-DBH trees of economically valuable species [e.g., aspen (*Populus* spp.)] in northern Minnesota will reduce the availability of cavity trees frequently used for nesting by some waterfowl (R. A. Norrgard and D. P. Rave, MNDNR, personal communication).

Thus, there is a need to develop methodologies that can be used to predict the locations of the habitat components that compose wood duck breeding complexes (i.e., important habitats used during the pre-breeding to brood-rearing life cycle phases). These methodologies should have the (A) flexibility to identify both forested and non-forested habitat components that occur at different spatial scales, (B) accuracy and precision to reliably quantify spatiotemporal changes in the characteristics (e.g., areal extent) of habitat components, and (C) efficiency to collect habitat data over large spatial scales. It also would be beneficial to develop such methodologies so that long-term trends in habitat characteristics could be analyzed in the future.

It is unlikely that all of these needs can be met with a single methodology or existing dataset. Consequently, we will develop 2 methodologies for obtaining better knowledge regarding spatiotemporal changes in wood duck breeding-habitat components. We propose to develop LiDAR methodology to identify multiple habitat components and to monitor changes in these components from the contemporary period forward. This methodology also could be used to provide habitat trend information that can be used in MNDNR administrative efforts [e.g., subsection planning) and research (e.g., estimating habitat availability in resource selection studies; see Aebischer et al. (1993)].

We also propose to provide better historical context regarding spatiotemporal changes in nesting habitat by analyzing Forest Inventory Analysis (FIA) data with a quantitative method currently being developed. Reliable FIA surveys have been conducted in Minnesota since the 1970s. We propose to conduct analyses of FIA data to identify spatiotemporal changes in nesting habitat components not characterized by LiDAR, at spatial scales smaller than those of previous investigations, and over a greater time period (i.e., since the 1970s). This methodology also will provide database queries that can be used in future monitoring efforts, and an insight of whether the predicted trend in the abundance of tree cavities (e.g., Denton et al. 2012*b*) is accurate.

GOALS AND OBJECTIVES

The ultimate goal of this project is to develop methodologies that can be used to predict the locations and monitor spatiotemporal changes in the areal extent of wood duck breeding complexes (i.e., important habitats during the pre-breeding to brood-rearing life cycle phases) and perhaps other species that use similar habitat components. Meeting this goal will require that we (1) identify the location and areal extent of breeding-habitat components in the main study area, (2) validate the predicted locations of wood duck breeding complexes with independent, empirical data from other sites, and (3) quantify the spatiotemporal trends in potential nesting trees in Minnesota over the long term. We will meet this goal using multiple sources of data (e.g., empirical field data, FIA, LiDAR, and associated remote sensing imagery). Our specific objectives are to:

1) Develop and evaluate spatial predictive models of habitat components that are important to breeding wood ducks [i.e., tree species (alternatively deciduous v. coniferous), DBH, tree canopy density, stand type, wetland type, water depth] based on LiDAR-generated metrics or other sources of spatial data [e.g., NWI, existing Geographic Information System (GIS) layers, aerial photographs]. This evaluation will include determining the accuracy with which each component can be predicted with LiDAR data.
2) Ascertain the optimal pulse density of LiDAR needed to accurately measure or classify each habitat component of importance to wood ducks.
3) Determine the generalizability of the LiDAR method for predicting the locations of habitat components by applying algorithms developed from data collected in the main study area (Cass County, Forest Ecological Province) to other sites in the Forest, Prairie, and/or Transition Provinces at which adequate LiDAR-cloud data have been obtained (e.g., J. Erb's study areas, MNDNR statewide elevation measurement project).
4) Estimate the species- and DBH-specific proportions of trees with suitable cavities and detection probability of suitable cavities from empirical field data.
5) Determine whether there has been a change in the number of potential nest trees since the 1970s based on changes in FIA data.

METHODS

Study Area

The primary study area encompasses 254,051 ha in northeastern Cass County, Minnesota (Figure 1). Parts of Chippewa Plains, Pine Moraines-Outwash Plains, and St. Louis Moraine Ecological Subsections (Hanson and Hargrave 1996) occur within this area. This study area occurs in BCR 12.

Wetland Surveys

In 2016, we used the available wetland spatial data from NWI (Cowardin et al. 1979, MNDNR 2009) to select 260 sampling plots in the study area. We stratified wetlands contained in the NWI GIS layer by NWI system, subsystem, and class (hereafter, wetland types). Unfortunately, information about NWI subclasses was not available for many wetland types. We calculated the proportion of the wetlands in the study area composed of 9 major wetland types: Lacustrine-Littoral-Emergent Vegetation (0.004), Palustrine-Emergent Vegetation (0.102), Lacustrine-Limnetic-Unconsolidated Bottom (0.522), Lacustrine-Littoral-Unconsolidated Bottom (0.020), Palustrine-Forested (0.191), Palustrine-Shrub Scrub (0.130), Palustrine-Unconsolidated Bottom (0.026), Riverine-Upper Perennial-Unconsolidated Bottom (0.003), and Riverine-Lower Perennial-Unconsolidated Bottom (0.002). We then randomly selected 260 2- X 2-m plots from these wetland types: 60 plots from both the Lacustrine-Littoral-Emergent Vegetation and Palustrine-Emergent Vegetation types, and 20 plots each from the remaining types. We selected more plots from the first 2 wetland types because we surmised that these habitats were more likely to be used by wood duck broods (e.g., Grice and Rogers 1965), and that there was a greater likelihood that these habitats would be structurally diverse and thus more difficult to identify from LiDAR signatures. We also specified that plots had to be ≥100 m apart to reduce the likelihood of non-independence among these sampling units (i.e., sampling plots with similar vegetation structure).

Many relatively small, isolated wetlands were not delineated in the NWI GIS layer, so we later selected 50 additional plots in these habitats from the MNDNR Hydrography GIS layer (MNDNR 2015). We randomly selected 1 plot per selected wetland if it was 0.81–8.09 ha, ≤402 m from a road, and adjacent to public land. After initially selecting plots from both layers, we examined aerial photos to assess the accessibility of these locations. We attempted to sample plots that initially appeared accessible.

We changed our approach to selecting wetland and plot locations for the 2017 field season to reduce number of plots located in wetland habitats not likely to be used by wood duck broods and to increase sampling efficiency. Specifically, we selected wetlands classified as either inundation or intermittent water; lake, pond or reservoir; river or stream; shallow water; or wetland from the MNDNR Hydrography GIS layer (MNDNR 2015) that either (1) had a public boat access site or (2) were on public lands and ≤100 m from both a public road and water feature. From sites that met these criteria, we then randomly selected ≤5 sampling locations per wetland that were ≥4.05 ha, with these points ≥100 m apart.

Because potential loafing sites were encountered infrequently at randomly selected plots during 2016, we chose to nonrandomly select and measure a variety of these structures as encountered so that we could observe the LiDAR signature for each. We also documented and measured these structures at randomly selected points during 2017.

We navigated to the approximate location of each plot center using a Garmin Montana Global Positioning System (GPS) unit, and established a plot center. If the plot center was difficult to access (e.g., because of soft bottom substrate that could not be traversed on foot, dense vegetation that could not be penetrated via boat) or on or near an ecotone, we moved the plot location to a site that was as close as possible to the initial location, accessible, and in the interior of a somewhat homogeneous vegetation patch. Moving plots away from ecotones reduced the likelihood of misclassifying habitats (i.e., habitat misclassifications are more likely to occur near ecotones because the exact location of a sampled plot is difficult to determine with somewhat imprecise GPS units). We also moved some plots located in open water to the nearest vegetated location within the wetland because the former habitat is simple and easily identified with LiDAR data. Instead, we chose to dedicate the greatest sampling effort to vegetated plots.

For each plot, we recorded the date, start time, observers, plot number, whether wood ducks were observed within 100 m of the plot, and if so, provided a count of individuals in each cohort (male, female, brood, unknown). We did not adjust wood duck counts for detectability. We ascertained whether the NWI classification (system, subsystem, class) available on our GIS layer was correct at each plot (i.e., some wetlands may have changed since the original classification or the original classification may have been incorrect), and recorded the appropriate NWI wetland classification to the level of subclass. We classified the types of wood duck loafing structures present within the plot (7 classes: none, rock, log or stump, muskrat lodge, beaver lodge or dam, small island or tussock, barely or lightly vegetated shoreline), as well as the type of beaver modification, if any that had some influence on the plot (6 classes: none, water level, runs, tree removal, dam or lodge, food cache). We also obtained location data for each plot center using a Geneq Sx Blue II GPS unit (15–20 cm accuracy in open habitats when data were obtained at 1 reading / second for 1 minute), and recorded the specific GPS unit used.

At each plot, we placed a 2- X 2-m Daubenmire square (Daubenmire 1959, Gilmore et al. 2008) so its center was located at plot center, and measured several habitat variables within the device. This square had 0.2 m delineations, which facilitated the measurement of several habitat variables. Specifically, we used these delineations to estimate the % coverage (5% increments) of 5 habitat classes [emergent, floating leaf, ground, open water, shrub (woody vegetation ≤1.37 m tall)] that were present at or above the water surface, and of submergent plants, when possible to make reliable observations (i.e., at locations in which water turbidity or sun glare did not substantially hinder observability). Within the Daubenmire square, we also documented the dominant emergent cover type (14 classes: none, alder [*Alnus spp*.], Canada bluejoint grass [*Calamagrostis canadensis*], giant bur-reed [*Sparganium eurycarpum*], cattail [*Typha spp*.], ericaceous shrub, floating-leaf, giant reed grass [*Phragmites spp*.], rush [*Scirpus spp*.], reed canary grass [*Phalaris arundinacea*], sedge [*Carex spp*.], willow [*Salix spp*.], wild rice [*Zizania aquatica*], other), and measured the minimum depth of submergent vegetation and the height of emergent vegetation and shrubs (0.1 m increments) with a 3-m ruler, tree canopy height (0.1 m increments for woody vegetation ≥1.37 m tall) with a Suunto clinometer or with a 3-m ruler, mean tree canopy closure with a spherical densiometer, and water depth with either a 3-m measuring pole (0.1 m increments) at relatively shallow plots or an Eagle FishEasy 245DS depth finder (0.03 m increments) at deeper locations.

Within the Daubenmire square, we also estimated vertical vegetation cover and structure using a round Robel pole (Robel et al. 1970) that had alternating 0.1-m white and black bands and narrow, vertical, and contrasting marks at the midpoint of each band. Because it was not possible for personnel to stand at plots in relatively deep water or where the soil substrate was soft, it was necessary to adapt this device so that it could be used by 2 people in a boat. This adaptation consisted of attaching a long wooden pole to the Robel pole in a perpendicular manner. One crew member extended the Robel pole to the corner of the Daubenmire square opposite the other crew member, and oriented this device upright to the water surface. The other crew member placed their sighting eye 0.8 and 1.6 m above the water surface with the aid of the 3-m ruler, and recorded the lowest decimeter or 0.5 dm mark that could be observed from diagonally across the Daubenmire square (2.8 m). Crew members switched assignments and took readings from across the opposite diagonal of the square. This approach generated 2 measurements from each observation height, all of which were averaged together.

Forest Surveys

We first obtained forest spatial data (e.g., forest cover type, stand age and location) of public forest lands from Cass County, State of Minnesota, and United State Department of Agriculture (USDA) Forest Service databases. There were slight differences in the manner that these agencies classified forest cover types, so we aggregated appropriate stands (i.e., likely to be used by nesting wood ducks) from each database into 5 basic cover types: aspen-birch, lowland hardwoods, mixed conifer-hardwood, northern hardwoods, and oak. We identified stands on public lands that were likely old enough to have developed cavities suitable for use by nesting wood ducks (i.e., aspen-birch ≥50 years, all other stand types ≥80 years), and constrained the potential sample to stands of these ages or greater. We then stratified stands by cover type and randomly selected 300 forest stands (60 stands of each of the 5 types) to be surveyed.

We then selected plots within these stands with the stipulations that (1) plot centers must be both ≥50 m apart and ≥30 m from the nearest stand boundary and (2) ≤2 plots per stand could be established. We used these selection criteria to increase the likelihood that plots adequately represented the diversity of vegetation structure of each forest type, thus facilitating the development of biologically realistic LiDAR models. We then randomly selected *n* = 563 plots to be surveyed. It was necessary to remove 19 plots from the sample because of nearby heritage sites or scheduled timber harvests (i.e., interpretation of habitat characteristics would be confounded if harvesting occurred between the times forest surveys were conducted and LiDAR data were collected).

We navigated to the selected plot centers using a Garmin Montana GPS, and established 20-m radius circular plots (0.126 ha) around those points. Plots located near ecotones not indicated on available GIS layers were moved sufficiently into the stand interior as to avoid potential edge effects on vegetation structure. We first recorded the plot identification number, date, start and end times of survey, visit number to the plot (first or second), observers, proportion of visible sky obscured by cloud cover (0.1 increments), and proportion of tree boles covered by snow or obscured by leaf-out (0, 0.01–0.10, 0.11–0.33, 0.34–0.66, 0.67–1.00). We obtained location data for each plot center using Geneq Sx Blue II (0.9–1.8 m accuracy under closed forest canopy when obtaining 1 reading / 5 seconds for approximately 15 min) and Geneq Sx Blue II + GNSS (0.5–0.9 m accuracy under closed forest canopy when obtaining 1 reading / 5 seconds for approximately 15 min) GPS units, and recorded the GPS make, model, and unit number used at each plot. We classified the stand structure following U.S. Department of Agriculture Forest Service methodology (2014; 5 classes: single story, two-storied, multi-storied, mosaic, unknown/unassessable). We assigned all plots to 1 of the 5 general forest cover types (Table 2) and to an Eyre (1980) cover type.

We then examined and measured individual tree stems within each plot following an established protocol (USDA Forest Service 2014), with some exceptions. Specifically, we surveyed only trees large enough to have cavities used by nesting wood ducks [i.e., ≥22.0 cm DBH (Haramis 1975)], and tall enough for the DBH to be measured (i.e., ≥1.37 m). Starting at the 0^o azimuth within each plot, we proceeded clockwise, numbering each suitable tree stem, and recording the following data for each stem: species, DBH (0.1 cm increments), distance (0.1 m increments) and direction (1^o increments that were not adjusted for declination) from plot center, health status (following Thomas 1979, Appendix 1), and crown class (5 classes: remnant, dominant, codominant, intermediate, overtopped; U.S. Department of Agriculture Forest Service 2014).

All field crew members then used binoculars to conduct a preliminary search of each tree ≥22.0 cm DBH in the plot to identify cavities that potentially were suitable for nesting by wood ducks. During the initial search, personnel ascertained whether the entrance dimensions likely were sufficient to permit a wood duck to pass through (i.e., 6 x 6 cm; Zwicker 1999, cited in Denton et al. 2012*b*) and the bottom of cavity entrance was high enough to be used by nesting wood ducks [i.e., ≥0.6 m above ground level (Strom 1969)]. When a potentially suitable cavity was encountered, we used a Pyle Model PLCM22IR remote camera attached via a stiff, braided wire to a 15.2 m Crain CMR Series Measuring Ruler (*sensu* Waldstein 2012) to perform a more careful examination of the entrance and interior of the cavity. We first determined whether cavity entrance dimensions were suitable by attempting to pass a cardboard cut-out of the minimum usable dimensions (i.e., 6 x 6 cm) through the cavity opening. This cut-out was placed on the wire connecting the camera to the measuring ruler. We then examined cavity interiors with the camera to ascertain whether the following conditions had been met: horizontal depth (approximately 10 cm from inner edge of the entrance opening toward the back of the cavity) appeared large enough for hens to move from the entrance to the interior of the cavity, vertical depth (from the bottom of the cavity to the bottom of the entrance) was ≥10.2 cm to 4.5 m; (Bellrose and Holm 1994 p. 176) and not hollow to the ground (Robb 1986, cited in Bellrose and Holm 1994, p. 178), nest platform dimensions were ≥14 x 15 cm (Boyer 1974, Haramis 1975, Denton et al. 2012*a*), and the cavity did not contain standing water or excess debris (Sousa and Farmer 1983).

Field personnel used this information to classify the suitability of each examined cavity for wood duck nesting (4 levels: suitable, marginal, unsuitable, unknown). We considered a cavity to be suitable if all these conditions were met. A cavity was classified as marginal if it was unclear whether all dimensional requirements were met (i.e., ≥1 dimensional measurement appeared to be close to some minimum or maximum value). Cavities typically were classified as unknown/unobservable if personnel were unable to completely observe the cavity, either because of cavity height or some structural attribute did not permit observation with the camera system. We considered a cavity to be unsuitable if any dimensional measurement was not met or if there was standing water or excess debris in the cavity. Field personnel also provided a cause for unsuitability (7 classes: entrance dimensions too small, insufficient horizontal depth, insufficient vertical depth, insufficient platform dimensions, too deep or hollow to the ground, standing water in the cavity, excessive debris in the cavity). We classified the reason that a cavity was unsuitable based on the order that structural restrictions would have been encountered as a wood duck entered a cavity (i.e., entrance dimensions, followed by horizontal depth, vertical depth, and finally, dimensions and other characteristics of the platform). Our assessment of the suitability of interior characteristics required some subjectivity because direct measurements could not be made with our camera system.

For each cavity inspected, we recorded tree number, cavity entrance type (3 classes: opening on the top, side, combination of top and side openings which are joined on the exterior of the tree), primary and secondary sources of cavity formation (11 classes: split, broken limb, broken top, woodpecker, fire, lightning, insect, logging wound, decay/rot, other, unknown), evidence of animal use (9 classes: eggshell/ membrane, nesting materials, hive or other insect structure, animal present, scratching at entrance, pecking at entrance, other, unknown, none), and animal taxa. We also measured cavity height with either a 15.24 m measuring ruler (±0.1 m), Leupold RX-800i rangefinder (±0.1 m), or Suunto clinometer (±0.5 m).

LiDAR Data Collection

The MNDNR Resource Assessment Program (RAP) originally planned to have LiDAR and associated remote sensing data collected during aerial flights conducted by a vendor during Fall 2016, but these efforts did not occur until Fall 2017. Data became available for analyses during late Summer 2018.

We clipped LiDAR data to our forest and wetland plot locations, and used Program FUSION/LDV version 3.80 (McGaughey 2018) to generate metrics for the LiDAR data associated with each forest plot. We performed a preliminary classification tree analysis using the R (R Core Team 2017) package randomForest (Liaw 2018), in which the LiDAR metrics and prior stand-type classifications associated with each plot were used to predict the presence or absence of a suitable cavity in those plots, and the proportion of plots that were classified correctly was ascertained. A similar approach will be used to analyze wetland plot data.

FIA Analysis

We initiated analyses of FIA data to gain an understanding of temporal changes in the potential number of nest trees of 7 tree species (American basswood, bigtooth aspen, northern red oak, paper birch, red maple, quaking aspen, and sugar maple) that are common in our study area and have some proclivity to produce cavities suitable for nesting wood ducks. We will use this information to make inferences about the temporal change in abundance of suitable nesting cavities within the Laurentian Mixed Forest Province of Minnesota (Hanson and Hargrave 1996) from 1977 to 2018. We limited our initial analyses to data from plots classified as ‘‘timberlands’ by the U.S. Department of Agriculture Forest Service, which is defined as ”forest land capable of producing in excess of 20 cubic feet per acre per year and not legally withdrawn from timber production, with a minimum area classification of one acre” (U.S. Department of Agriculture

Forest Service 2019). We are particularly interested in the temporal changes of 3 forest characteristics likely to be associated with the development of suitable cavities: species-specific temporal changes of the (1) number of stems ≥22.0 cm DBH of the target species, (2) mean DBH, and (3) proportions of stems with live-healthy, live-health impacted, and dead health status classifications.

Prior to extrapolating our empirical forest-survey results to FIA data, it was necessary to aggregate stems with a health status of 3–7 (Appendix 1) into a single 'dead' classification because of a sparseness of data. Stems with a health status classifications of 1 and 2 continued to be classified as 'live, healthy' and 'live, health-impacted', respectively. Unfortunately, some methodological differences with regard to the classification of health status may have occurred in FIA surveys since 1977, so we aggregated live-healthy and live-health impacted stems for this preliminary analysis. We also examined changes in the number of live stems beginning in 1977, but that of dead stems beginning in 1990 because of methodological changes that may have occurred between the 2 survey periods. Last, these surveys were conducted within a single year during 1977 and 1990, but a subset of plots have been surveyed annually beginning in 1999. Consequently, we averaged results for 4 periods after that: 1999–2003, 2004–2008, 2009–2013, and 2014–2018.

RESULTS

Wetland Surveys

We conducted surveys at 677 randomly selected wetland plots during the late summer and early fall of 2016 and 2017 (Table 1, Figure 2). We classified the dominant emergent cover as alder (0.7%), blue joint grass (0.6%), bur reed (0.3%), cattail *spp* (6.9%), ericaceous shrub (2.2%), floating leaf (18.0%), phragmites *spp* (2.5%), rush *spp* (20.7%), reed canary grass (2.2%), sedge *spp* (8.3%), willow (0.4%), wild rice (31.3%), other vegetation (0.9%), and none (4.9%). We also documented trees at 10 plots (1.5%), with canopy coverage ranging from 0.05 to 0.85. We observed that 12.3% of randomly selected plots were modified by beaver, wood ducks were present ≤100 m of 9.6% plots, and 4.4% of plots had potential wood duck loafing sites.

The potential loafing structures identified in randomly selected plots were 2 beaver lodges, 6 floating vegetation mats, 4 small islands or tussocks, 14 patches of bare or lightly vegetated shore, 5 logs or stumps, and 1 muskrat house in the randomly selected plots. We observed 6 beaver lodges, 2 logs or stumps, and 1 muskrat house in the 15 non-randomly selected plots.

Forest Surveys

We conducted surveys at 322 forest plots during fall 2016, spring 2017, fall 2017, and spring 2018 (Figure 3). We classified these plots to both general forest types and to Eyre (1980) types (Table 2). We will attempt to use these plot classifications in conjunction with LiDAR data to classify forest types throughout the study area during the upcoming fiscal year.

Most other results of forest surveys are reported and discussed in a separate manuscript within this issue of *Summaries of Wildlife Research Findings*. Beyond the scope of this separate manuscript, we observed disproportionate percentages of cavities in some tree species. For example, northern red oak and sugar maple have comparatively greater proportions of stems with suitable cavities, and paper birch and green ash have proportionally fewer (Table 3).

LiDAR Data Collection

Aerial single-photon LiDAR data and associated remote sensing imagery were collected during fall 2017. These data were collected during peak fall color, usually at about 30 return pulses / m^2 (minimum of 12, up to 40–50; J. Corcoran, MNDNR, unpublished data). The quality of green

LiDAR data was not as good as anticipated. Thus, identifying the presence/absence and density of submergent vegetation and depth of water in relatively shallow locations likely will not be discernable.

The preliminary classification tree analysis generated encouraging results, but we anticipate that the structure of the final model and associated predictive capabilities will change when RAP provides updated and improved information for our model inputs. Specifically, RAP is developing a method to classify forest stand type with LiDAR and ancillary remote-sensing data, and we anticipate using resultant stand-type classifications as predictors in our models. The objective of ascertaining the pulse density needed to accurately classify forest and aquatic vegetation characteristics will be addressed after predictive LiDAR models are finalized.

FIA Analysis

Between 1990 and 2014–2018, there were increases in the *population estimates* of live American basswood, red maple, sugar maple species stems ≥22.0 cm DBH but decreases in the estimates of bigtooth aspen, paper birch, and quaking aspen stems (Table 4) in the Laurentian Mixed Forest Province of Minnesota. Further, the *population estimate* of live northern red oak stems peaked during 1990 and generally decreased after that time (Table 4). Interestingly, the *population estimates* of dead stems of these species were more temporally variable than those of live stems (Table 4), but the *overall proportion of stems of these species with a dead status* increased slightly between 1990 and 2014–2018 (Table 5). The *estimated population of aggregated live and dead stems* of these 7 species that were ≥22.0 cm DBH increased substantially between 1990 and 1999–2003, but decreased substantially during later periods through 2014–2018.

DISCUSSION

Wetland Surveys

Initially, we randomly selected wetlands for sampling to obtain an adequate sample size for each NWI class, with special emphasis placed on those classes that are most likely to have diverse vegetation structure. However, these efforts were confounded in-part by limitations of the existing NWI spatial data. Specifically, we observed during field-data collection that NWI classifications of some plots were incorrect, which we attribute to a combination of misclassification of wetland habitats, habitat changes since the original classification, and projection error. Further, the currently available NWI GIS layer often classifies wetlands only to the level of class, which provides little information regarding vegetation type or structure. Thus, it was not possible to select plots based on subclass or vegetation type and structure. Such limitations of available data contributed to an allocation of sampling locations that were not balanced among the 14 types of emergent covers observed. It is likely, however, that the emergent covers sampled were representative of those available in the study area.

Fortunately, we were able to collect data for a substantial number of plots (1) with structurally similar vegetation types that are difficult to distinguish from aerial photographs (i.e., wild rice v rush *spp*.; D. Dustin, MNDNR Fisheries, personal communication), (2) dominated by the types of aquatic vegetation that should begin to subside and thus change structure (e.g., floating-leaf plants, wild rice) approximately when LiDAR imagery was obtained (i.e., late September and October), (3) with vegetation types that may be sparse, and (4) with vegetation types that frequently occur in a mix of other types of vegetation (e.g., floating-leaf plants). We anticipate that a substantial amount of data will be needed to develop reliable LiDAR signatures of such sites. Presumably, wetland habitats with no surface vegetation should have a rather simple and readily identifiable LiDAR signature.

Although identifying potential loafing sites for wood ducks using LiDAR imagery was a secondary objective, we were able to locate 6 types of these structures in randomly selected plots and 3 in non-randomly selected plots. These structures likely are a somewhat important habitat component to wood ducks (McGilvery 1968).

Forest Surveys

Most of our forest-survey results are presented in a separate document within this issue of *Summaries of Wildlife Research Findings*, but there are 3 important points beyond the scope of that report. First, the forest and cavity properties (e.g., species composition, mean DBH, cavity density) we observed on public lands may have been different than those on private land, likely because of ownership-related differences in management practices and site characteristics. We opted not to obtain permission to conduct forest surveys on private lands within the study area to determine whether forest and cavity characteristics are similar to those on public lands, because it probably would have been time consuming to obtain enough data to detect significant differences between the 2 forest ownership classes. The use of other sources of forest-habitat data (e.g., LiDAR, remote sensing imagery, FIA surveys) should permit the discernment of any forest and cavity differences between these ownership groups.

Second, data from our field crews and the databases of natural resource agencies differed in the classification of general forest type of 37% of our plots. This discrepancy may be attributed to misclassification, or changes to these stands caused by natural disturbance, logging, and forest succession that had occurred since the time of classification. Regardless, substantial misclassification of stand type in existing databases could confound our ability to use the variable *forest-stand type* in conjunction with our empirical cavity data to predict the *abundance or occurrence of suitable cavities* across the landscape. Thus, it is likely that FIA data or LiDAR data and associated remote imagery would better predict of the *abundance or occurrence of suitable nesting cavities*.

Third, our results indicate that the *proportion of stems with suitable cavity* varies among *tree species* (Table 3), but that these species-specific proportions appears to vary among study areas (e.g., Soulliere 1990, Bellrose and Holm 1994, Denton et al. 2012*b*). Such differences may be attributable in-part to spatial differences in those variables (e.g., disease, insects, animal populations, soil conditions, weather patterns) that contribute to tree damage and eventually cavity formation (Morin et al. 2016). Thus, forest managers should understand which tree species are most likely to produce suitable nesting cavities for wood ducks in their work area. There also is a need to develop a better understanding of the variables that most influence cavity selection and nest success.

FIA Analysis

Our preliminary results suggest that there have been changes in 2 forest components of the Laurentian Mixed Forest Province of northern Minnesota that are associated with cavities suitable for nesting wood ducks: *species composition* and *proportion of stems with a 'dead' health classification*. Although these preliminary results are interesting, further work needs to be done before we can make inferences regarding temporal changes in suitable nesting cavities in the Laurentian Mixed Forest Province of Minnesota from 1977 until the contemporary period. Specifically, we must examine changes in mean DBH of the 7 target species and further reconcile differences in between our *health status classifications* and those used by FIA before using our empirical findings (i.e., *proportions of suitable cavities in each tree species-DBH-health status class*) and FIA data (i.e., *populations of stems* in each of these classes) to make these inferences. There also is a need to identify the cavity characteristics (e.g., *species*, *source of formation*) selected by wood ducks in in the northern portion of their geographic range, and how cavity availability may change under different scenarios (e.g., changes in climate, disturbance regimes, and timber harvesting).

ACKOWLEDGMENTS

T. Kaebish, S. Hillard, and D. Kepler (MNDNR RAP) provided insight regarding the potential uses of LiDAR data, and facilitated the use of their data in our research. We thank S. Mortenson (Leech Lake D.R.M.), J. Watts (Cass County M.I.S.), S. Dunham, J. Rickers, and T. Tisler (U.S. Department of Agriculture Forest Service) for logistical support. We thank C. Scharenbroich and R. Wright for technical assistance. J. Giudice and V. St.-Louis provided suggestions regarding study design, sampling regime, and data analysis. B. Coleman, T. Duquette, C. McCarty, H. North, T. Roerick, and E. Spry provided valuable assistance with field data collection. J. Lawrence for provided reviews of research proposals and D. Larson reviewed an earlier draft of this report. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

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Table 1. The National Wetland Inventory classification and sample size of plots surveyed in Cass County, Minnesota, USA during 2016–2017.

National Wetland Inventory system, subsystem, class, and subclass of sampled plots [a, b]	Number of plots surveyed
Lacustrine limnetic unconsolidated bottom unknown	1
Lacustrine limnetic unconsolidated bottom sand	3
Lacustrine limnetic aquatic bed rooted vascular	1
Lacustrine littoral aquatic bed unknown	1
Lacustrine littoral aquatic bed rooted vascular	60
Lacustrine littoral aquatic bed floating vascular	5
Lacustrine littoral emergent nonpersistent	233
Lacustrine littoral unconsolidated bottom unknown	12
Lacustrine littoral unconsolidated bottom sand	1
Lacustrine littoral unconsolidated shore unknown	1
Palustrine aquatic bed floating vascular	13
Palustrine aquatic bed rooted vascular	43
Palustrine emergent nonpersistent	130
Palustrine emergent persistent	93
Palustrine emergent *Phragmites australia*	9
Palustrine forested broad-leaved deciduous	1
Palustrine scrub-shrub broad-leaved deciduous	20
Palustrine scrub-shrub broad-leaved evergreen	1
Palustrine unconsolidated bottom sand	3
Palustrine unconsolidated shore organic	1
Palustrine unconsolidated shore sand	5
Riverine lower perennial unconsolidated bottom unknown	2
Riverine lower perennial unconsolidated bottom mud	3
Riverine lower perennial rock bottom unknown	1
Riverine lower perennial emergent nonpersistent	28
Riverine upper perennial aquatic bed rooted vascular	2
Riverine upper perennial emergent nonpersistent	4

[a] Wetlands in the palustrine system are not assigned a subsystem classification in the National Wetland Inventory classification scheme.

[b] The National Wetland Inventory subclasses of some plots were classified as unknown if distinguishing characteristics were not discernable in the field.

Table 2. Crosswalk between the Forest Cover Types of Eyre (1980) and the more general forest types used to classify stands from GIS databases, and sample size of forest plots in each class that were surveyed in Cass County, Minnesota, USA during 2016–2018.

General forest type	Eyre (1980) forest cover type	Number of plots surveyed
Aspen-birch	Aspen (16)	63
	Paper Birch (18)	24
Mixed conifer	Balsam fir (5)	1
	Eastern white pine (21)	3
	Red pine (15)	14
	White pine–northern red oak–red maple (20)	1
Northern hardwood	Sugar maple (27)	4
	Sugar maple–basswood (26)	101
Oak	Bur oak (42)	24
	Northern red oak (55)	54
Lowland hardwood	Black ash–American elm–red maple (39)	24
	Red maple (108)	10

Table 3. The percentage of stems by tree species that were sampled, the percentage of trees of each species with suitable cavities, and the percentage of trees of each species with suitable or marginal cavities that were detected within forest plots located in Cass County, Minnesota, USA during 2016–2018.

Tree species	% of all trees sampled	% of all trees with suitable cavities	% of all trees with suitable or marginal cavities
American basswood *(Tilia americana*)	15.41	18.75	17.65
American elm (*Ulmus americana*)	0.24	–	–
Balsam fir (*Abies balsamea*)	1.60	–	–
Balsam poplar (*Populus balsamifera*)	0.39	–	–
Bigtooth aspen (*Populus grandidentata*)	6.49	8.17	7.19
Black ash (*Fraxinus nigra*)	3.70	–	0.65
Black cherry (*Prunus serotina*)	0.01	–	–
Black spruce (*Picea mariana*)	0.01	–	–
Box elder (*Acer negundo*)	0.06	–	–
Bur oak (*Quercus macrocarpa*)	4.03	1.92	1.96
Eastern cottonwood (*Populus deltoides*)	0.01	–	–
Eastern hophornbeam (*Ostrya virginiana*)	0.01	–	–
Eastern larch (*Larix laricina*)	0.03	–	–
Eastern white pine (*Pinus strobus*)	1.69	0.96	1.31
Green ash (*Fraxinus pennsylvanica*)	2.94	0.48	0.65
Hackberry (*Celtis occidentalis*)	0.05	–	–
Jack pine (*Pinus banksiana*)	0.32	–	–
Northern pin oak (*Quercus ellipsoidalis*)	0.17	–	–
Northern red oak (*Quercus rubra*)	10.60	12.98	11.11
Northern white-cedar (*Thuja occidentalis*)	0.46	–	–
Paper birch (*Betula papyrifera*)	10.28	2.88	3.59
Quaking aspen (*Populus tremuloides*)	16.02	15.38	16.67
Red maple (*Acer rubrum*)	7.23	6.73	9.15
Red pine (*Pinus resinosa*)	5.71	–	–
Sugar maple (*Acer saccharum*)	10.67	27.40	25.49
White spruce (*Picea glauca*)	0.29	–	–
Yellow birch (*Betula alleghaniensis*)	0.80	2.40	2.29
Unidentified ash *spp* (*Fraxinus spp*)	0.08	–	–
Unidentified pine *spp* (*Pinus spp*)	0.05	0.48	0.33
Unidentified aspen *spp.* (*Populus spp*)	0.47	0.96	1.63
Unknown spp	0.17	0.48	0.33

Table 4. The population estimates of American basswood, bigtooth aspen, northern red oak, paper birch, quaking aspen, red maple, and sugar maple stems $\geq$22.0 cm diameter at breast height that were alive or dead within the Laurentian Mixed Forest Province of Minnesota, USA, during 5 survey periods (1990 to 2014–2018). These species were examined because of their importance to nesting wood ducks and their common occurrence in our Cass County, Minnesota, USA study area. Data from the U.S. Forest Service Forest Inventory and Analysis database were used in this summary.

Survey period [a]	Population estimate of live stems	Population estimate of dead stems	Population estimate of live and dead stems aggregated
1990	307,770,110	5,570,692 [b]	313,340,802 [b]
1999–2003	351,543,532	50,882,753 [b]	402,426,285 [b]
2004–2008	291,088,060	52,425,247	343,513,307
2008–2013	270,812,185	48,143,229	318,955,414
2014–2018	261,132,106	49,524,552	310,656,658

[a] All plots in Minnesota were surveyed within approximately 1 year during 1990, but only a subset of 20% of available plots were surveyed during any 1 year thereafter. Therefore, we summarized data for 5-year blocks during 1999–2018.

[b] A subsample of undisturbed plots were modeled (i.e., not remeasured) during 1990, which may have contributed to anomalous estimates of the populations of standing dead trees and aggregated live and dead stems.

Table 5. The proportion of American basswood, bigtooth aspen, northern red oak, paper birch, quaking aspen, red maple, and sugar maple stems $\geq$22.0 cm diameter at breast height with a live or dead health status classification within the Laurentian Mixed Forest Province of Minnesota, USA, during 5 survey periods (1990 to 2014–2018). These species were examined because of their importance to nesting wood ducks and their common occurrence in our Cass County, Minnesota, USA study area. Data from the U.S. Forest Service Forest Inventory and Analysis database were used in this summary.

Survey period [a]	Proportion live	Proportion dead
1990	0.87 [b]	0.13 [b]
1999–2003	0.85	0.15
2004–2008	0.85	0.15
2008–2013	0.84	0.16
2014–2018	0.84	0.16

[a] All plots in Minnesota were surveyed within approximately 1 year during 1990, but only a subset of 20% of available plots were surveyed during any 1 year thereafter. Therefore, we summarized data for 5-year blocks during 1999–2018.

[b] A subsample of undisturbed plots were modeled (i.e., not remeasured) during 1990, which may have contributed to anomalous estimates of the proportions of live and standing dead trees.

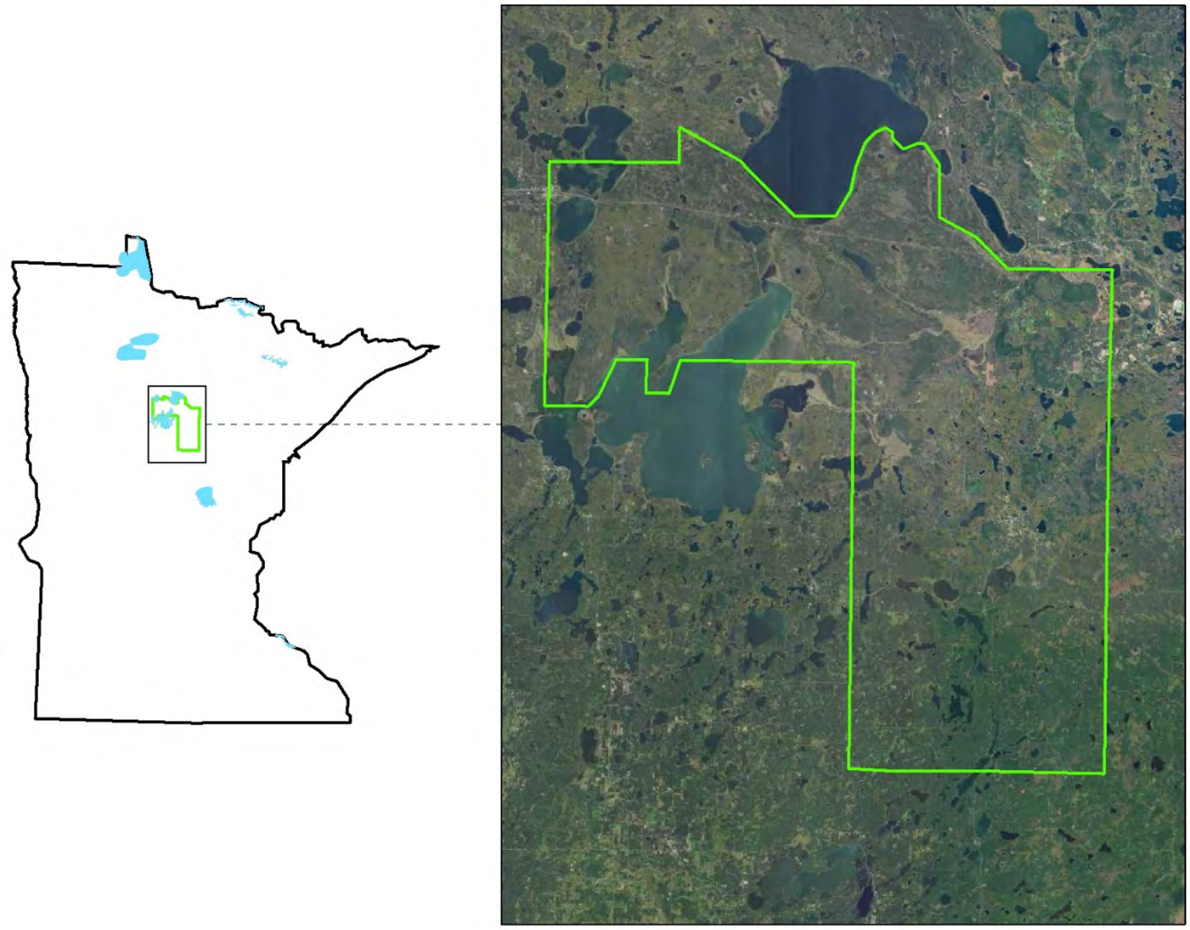

Figure 1. Location of the wood duck-LiDAR project in Cass County, Minnesota, USA 2016-2018.

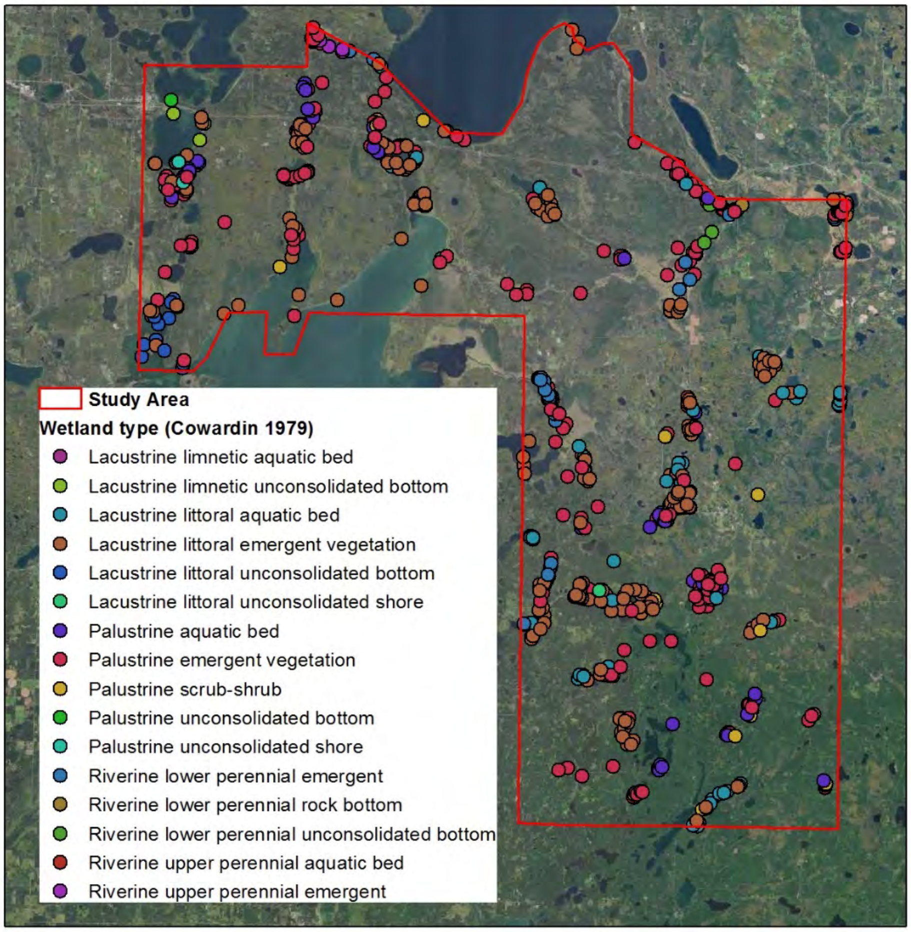

Figure 2. Location of wetland plots of different National Wetland Inventory types (Cowardin et al. 1979) surveyed in in Cass County, Minnesota, USA during Summer and Fall 2016 and 2017.

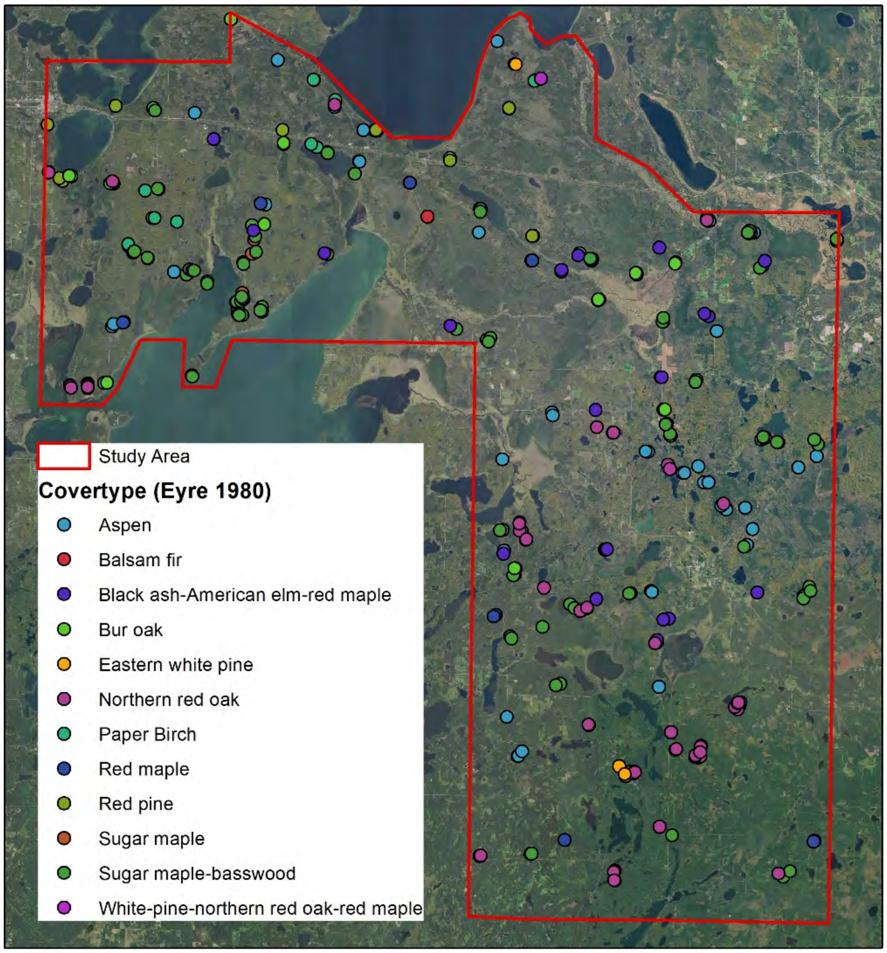

Figure 3. Location of forest plots of different cover types (Eyre 1980) that were surveyed in Cass County, Minnesota, USA during Fall 2016, Spring 2017, Fall 2017, and Spring 2018.

Appendix 1. Numerical codes used in the classification of the health status of trees (from Thomas 1979).

Health status	Description
1	Live tree that has no defects or injuries that will threaten its long-term health.
2	Live tree with defects that contribute to a decline in health. Indicators may include decay on the bole, fungi, large dead limbs, and substantial cracks.
3	Recently dead tree with bark, limbs, and twigs substantially intact.
4	Dead tree that has lost some limbs and almost all twigs.
5	Dead tree that has lost most limbs and all twigs.
6	Dead tree with a broken top and hard bole wood.
7	Dead tree with a broken top and soft bole wood.

FALL MOVEMENTS OF MALLARDS MARKED IN MINNESOTA

Bruce E. Davis

SUMMARY OF FINDINGS

During August-September of 2016, I marked 119 mallards (*Anas platyrhynchos*) with tracking units. I obtained GPS locations from dataloggers recovered by hunters or uploaded through the Argos satellite system to yield 3,506 tracking locations. Locations within the state of Minnesota (*n*=2,848) were used to examine habitat use. Marked birds were retained in Minnesota longer than expected; freeze up dates were later than average in 2016. When marked birds did leave the state, movements upon departure tended to be long with a mean distance 434 km between a bird's last known location in Minnesota and its first known location outside of Minnesota. Marked birds used open water and emergent herbaceous wetland habitats for combined 55-80% of the time. Crop habitats were used most frequently at night (30% of proportional use) and more frequently for birds marked in the south hunting zone than for birds marked in the north hunting zone. Sample sizes were sufficient to detect differences in use of habitats among capture zones and time of day, but I did not detect differences in emigration rates between zones; given the late onset of winter, rates of emigration may have been similar between zones. For birds marked in 2016, biologically relevant differences were detectable when present.

During August-September of 2017, I marked 90 mallards with tracking units. 79 of these units were GPS units that transmitted through the Argos system; the other 11 units were GPS dataloggers. During August-September of 2018, I marked 45 mallards with GPS tracking units that transmitted through the Argos system. We were planning to end field work for the project after the 2018 field season, but we still have 20 tracking units on hand that have not yet been deployed. These units will be deployed in 2019 and tracking will continue through fall and winter 2019-2020.

Herein, I present summaries of preliminary analyses for birds marked in 2016, but have not yet completed analyses for birds marked in 2017 or 2018. Beyond summary statistics, I provide here some example data as well as some comparisons with band recovery data for mallards marked in Minnesota.

INTRODUCTION

Distribution of waterfowl during fall migration and concurrent hunting seasons is affected by numerous factors. Wildlife managers are tasked with setting season dates, bag limits, shooting hours, and further restrictions on harvest. Availability of waterfowl throughout the hunting season (retention) is important to Minnesota waterfowl hunters. Understanding the chronology of immigration and emigration events and the factors affecting those events is imperative.

Many factors may impact emigration rates and use of habitats. Weather plays an important role in the timing of migration by waterfowl during fall; as winter weather severity increases, the probability of southward waterfowl migration also increases (Schummer et al. 2010). Repeated exposures to disturbance associated with hunting have been found to alter the distribution and habitat use and cause increased movements of wintering waterfowl (Dooley et al. 2010, Pease et al. 2005), but the effects of disturbance have not been investigated for waterfowl nearer their breeding habitats. Importantly, the effects of weather and anthropological disturbance are likely confounded; hunting seasons often coincide with changing weather patterns. In the presence of elevated human disturbance to waterfowl habitats that occurs during hunting seasons, it may be difficult to detect causes of temporal or spatial changes to a bird's natural migration pattern. Numerous studies have been implemented to understand aspects of breeding waterfowl and some information is available on wintering waterfowl, but little work has been completed on waterfowl during migration periods. Due to their transient nature, waterfowl are inherently difficult to study during the migration periods. Thus, few studies have been undertaken to investigate patterns of fall migration.

In an effort to provide habitat to local and migrating waterfowl, retain waterfowl on the landscape throughout the duration of the season, provide hunting opportunities for its constituents, and to control waterfowl harvest, the Minnesota Department of Natural Resources has implemented numerous restrictions on duck harvest and disturbance to wetlands. Restrictions include establishment of waterfowl refuges, a 4 PM closure to duck hunting for the earliest portion of the duck season, designation of feeding and resting areas which restrict the use of motorized boats, a statewide ban on motorized decoys for the earliest portion of the season, and a ban on motorized decoys on state owned Wildlife Management Areas (WMA) for the entire season.

The importance of the hunting regulations implemented in Minnesota to provide secure areas for ducks is unclear because fall emigration and factors affecting the chronology of fall migration are poorly understood. Restrictions on afternoon shooting hours unilaterally in Minnesota did result in 3-4% lower recovery rates (a proxy for harvest rates) than when sunset closures occurred, but the researchers were unable to detect a difference in annual survival rates (Kirby et al. 1983). Restrictions on shooting hours that are more restrictive than what is allowed in the federal framework have been in place since 1973 but their importance is unknown. Assessment of the effects of shooting hour restrictions and other hunting regulations on movement patterns warrants investigation. Better understanding of movement patterns gained from this work will allow managers to better set season dates and alter restrictions on harvest.

OBJECTIVES

Overall study objectives were to:

1. Better understand emigration chronology for mallards in Minnesota.
2. Estimate distances and directions moved by mallards in Minnesota.
3. Identify migration stopovers used by mallards in Minnesota.
4. Estimate use of habitats for birds while in Minnesota.

More specifically, during the pilot-year of this study, we sought to inform subsequent years of data collection by addressing these specific objectives:

5. Estimate variability in emigration, movement, and habitat use data within and among hunting zones.
6. Estimate rate of sample size reduction throughout the tracking period.
7. Evaluate alternative tracking units in terms of data quantity and quality.

STUDY AREA

Currently, Minnesota utilizes 3 zones to manage duck hunting seasons (Figure 1). Timing of seasons and restrictions on shooting hours differ among the zones. I attempted to mark equal numbers of birds in each hunting zone, but was unable to mark birds in the central zone in 2016. In 2017, I marked 59, 20, and 11 birds in the north, central, and south zones, respectively.

METHODS

Marking

In 2016, I attached 39 GPS-Argos backpack units (Lotek Wireless Inc., Newmarket, Ontario, Canada) to adult female mallards; these units logged GPS data and then transmitted that data back to the Argos system upon completion of their duty cycle. These units were 15 g and able to record about 100 GPS fixes and transmit those fixes to Argos satellites before exhausting their battery life. In 2016, I also marked 80 hatch year male mallards with GPS-archival backpack units (Lotek Wireless Inc.). These units record GPS location data at a user specified interval, but must be recovered to acquire data. These units weighed 11 g and were configured as backpack type transmitters. I attached these units to hatch-year males because they have the highest recovery rate of any mallard age-sex cohort. Apparent direct (within first hunting year after marking) recovery of hatch year male mallards banded in Minnesota based on band returns was predicted to be 18% and an additional 6% were expected to be recovered in the 2nd hunting season after deployment (USGS, Gamebirds data set).

GPS-logger or GPS-Argos backpack transmitter units receive satellite signals to estimate highly accurate locations; precision of locations is accurate to within a few meters. Of all available options, these units were deemed best suited for estimating detailed parameters associated with habitat use, use of refuge areas, local movements, and migration events. Birds were marked in conjunction with our current banding effort. We paid a $50 incentive for hunters returning tracking units.

After preliminary analyses for birds marked in 2016, we determined that slightly more data per unit cost were attained for birds marked with GPS-Argos units than GPS-logger units. Further, the manufacturer was able to change the firmware and programming of these tracking units for 2017 so that they would consume less battery and collect more data than the 2016 units. Thus, we elected to purchase only GPS-Argos units for use in 2017 and 2018. In 2017, I was also able to reuse 11 GPS-logger units and 8 GPS-Argos units that had been deployed in 2016, recovered by hunters, and refurbished with new harness material.

Tracking

In 2016, GPS-logger units were configured to attain location data every 11.5 hours; GPS-Argos units were configured to attain fixes every 22.5 hours and the units were set to begin this cycle at differing times. This allowed locations throughout the day and locations on each individual bird to shift over days and attain day and night fixes accordingly. In 2017, advances in firmware and programming of the tracking units allowed more frequent tracking; GPS-Argos units were set to attain location data every 11.5 hours in 2017 and 2018.

Movement Data

For birds marked in 2016, estimated point locations were determined to be inside or outside the state of Minnesota. A bird was determined to have emigrated upon its permanent exit from the state. Movement direction was measured as the azimuth between the birds marking location and its first location outside the state upon permanent emigration.

Use of Habitats

For birds marked in 2016, estimated point locations were overlaid on the 2011 National Land Cover Data layer and habitats were determined based on estimated point locations. Similarly, it was determined whether locations were on refuge or non-refuge locations and WMA or non-WMA locations based on appropriate GIS data layers.

Data Analyses - Movement Data

For birds marked in 2016, I determined date of permanent departure from the state (emigration) for each bird based on its location data. I used proportional hazards regression (Allison 1995) to examine variation in emigration rates due to the effects of the bird's age and sex or its location of marking. I present product-limit emigration estimates (Kaplan and Meier 1958) for the marked sample. Further, I present a plot latitude of location data over time and a plot of the array of movement direction upon emigration from Minnesota.

Data Analyses - Use of Habitats

For birds marked in 2016, I divided the tracking period into 3 time periods based on hunting seasons: PREHUNT (the period before regular duck season was opened in Minnesota), HUNT (the period when regular hunting season was open anywhere in Minnesota), and POST (the period after regular duck season had closed anywhere in Minnesota). I divided location data in portions of the day as diurnal (sunrise to sunset) or nocturnal. I collapsed habitats to 5 basic categories for analysis including open water, forested or developed habitats, pasture habitats, crop habitats, or emergent marsh habitats. I determined use of habitats within the state of Minnesota based on 2,848 location estimates from 44 birds using compositional analyses (Aebischer et al.1993). I determined diurnal and nocturnal proportional use of each bird in every habitat during each time period, I replaced zero values with 0.002 (an order of magnitude lower than the lowest nonzero proportion of a habitat used by any bird in a combination of any time period and portion of day. To remove the unit sum constraint, I constructed log ratios by dividing proportional use of each habitat by proportional use of emergent marsh habitat and used Napierian logarithms of these ratios as response variables. I used split-plot, repeated measures multivariate analysis of variance to test for overall effects of season (PREHUNT, HUNT, POST), portion of day (day or night), cohort of marked bird, or zone of capture. I fit a full model containing all 4 of these explanatory factors as well as a term for repeated measures among birds.

Models were fit using backwards-stepwise procedures. I present estimates of proportional use of each habitat, averaged across birds, from the untransformed data within levels of significant ($P \leq 0.050$) explanatory variables from the final fitted model.

Similarly, I examined use of refuge areas (areas closed to waterfowl hunting by statute or regulation) and use of WMAs using analysis of variance after constructing proportions as outlined above. I present proportional use of these habitats below.

RESULTS

Movement Data

For birds marked in 2016, I did not detect differences in rates of emigration among cohorts or zones of capture (Ps>0.018). Retention rates of marked birds in Minnesota remained > 80% until early November then declined to about 45% by mid-November, remaining birds left the state in early-December (Figure 2). Latitudes of marked birds declined throughout the season (Figure 3) and vectors of emigration flights were mostly long and southeasterly (Figure 4). One noteworthy exception to the southeasterly emigration pattern was that in 2016, 6 birds marked in northwestern Minnesota moved north into southern Manitoba (Figure 5). This movement pattern took place in only a small percentage of birds, but represents movement that is not usually seen when examining band recovery data due to low hunting pressure and dilution of the banded sample by large numbers of other mallards.

Use of Habitats

We attained location data for birds marked with GPS-Argos units and assigned location data habitat attributes (Figure 6). For birds marked in 2016, I did not detect differences among proportional use of habitats by seasons or cohorts (Ps >0.090), but proportional use of habitats differed among zones of capture (P=0.018) and portion of day (P < 0.0001). Use of crop habitats were higher for birds marked in the south capture zone than for birds marked in the north capture zone (Figure 7). Use of open water habitats were highest during the day; use of crop habitats were highest during the night (Figure 8).

Use of areas closed to hunting varied by zone of marking and season (Ps<0.0029). Proportional use of areas closed to hunting was 45% in the north hunting zone, whereas use of refuge areas was only 8% in the south hunt zone. Use of refuge areas was highest (44.9%) during the preseason period, but decreased to 22.6% and 27.3% during the hunting season and post-hunting periods, respectively.

Use of WMAs was 56%, 37%, and 27% during the preseason, hunting season, and post-hunting seasons, respectively. Use of WMAs during night was 39%, but 53% during the day.

DISCUSSION

Emigration rates were similar between zones of capture and cohorts; given the late onset of winter that occurred in 2016, this was not surprising. Temperatures were above normal through early December in northern Minnesota. When freeze up did occur in the north hunting zone it also occurred in much of the southern portion of the state shortly thereafter. I speculate that the extended retention time of the marked sample within Minnesota was likely due to the late onset of winter.

Use of open water and emergent wetland habitats was high; these estimates were based on the National Land Cover Database data currently available. More refined analyses of habitat use could be conducted if more informative and accurate GIS data layers are available. Further analyses including data from birds marked in 2017 and 2018 will be conducted in 2019. An additional sample of 25 Mallards are scheduled to be marked in August-September 2019 using GPS-Argos type tracking units.

ACKNOWLEDGMENTS

This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program.

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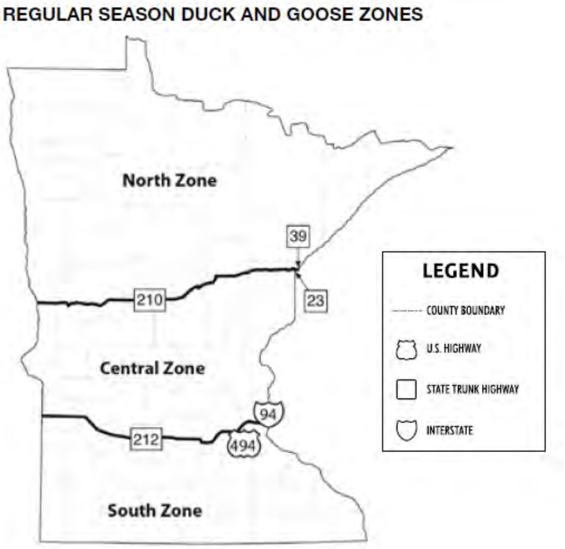

Figure 1. Minnesota waterfowl hunt zones boundaries, 2016.

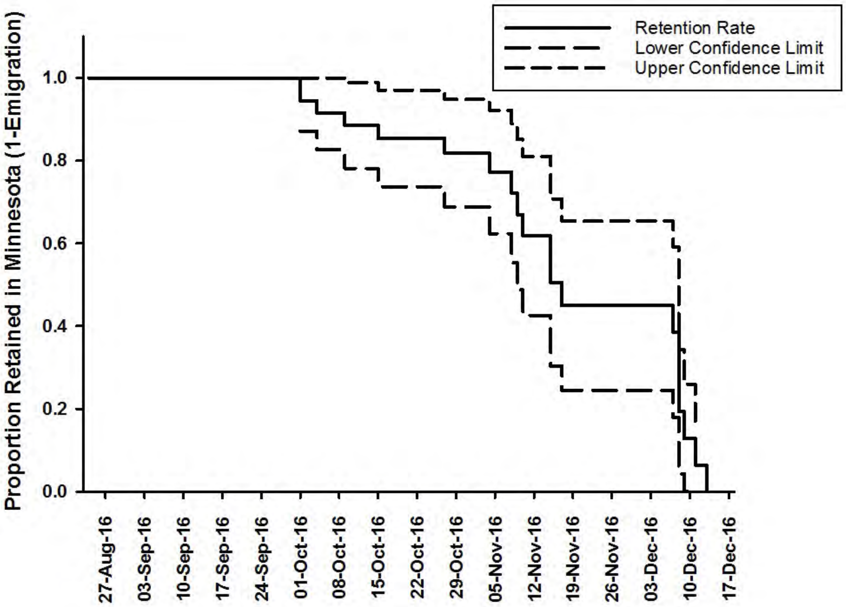

Figure 2. Retention curve for mallards marked with tracking units in Minnesota, 2016.

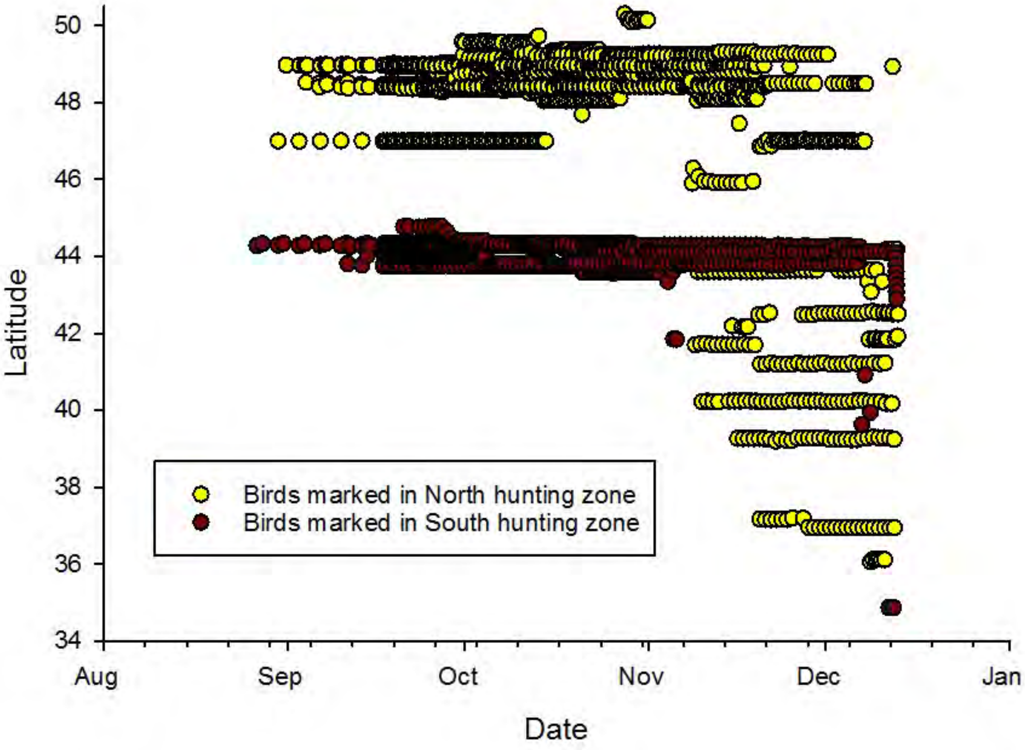

Figure 3. Scatter plot of location latitudes by date for mallards marked with tracking units in Minnesota, 2016.

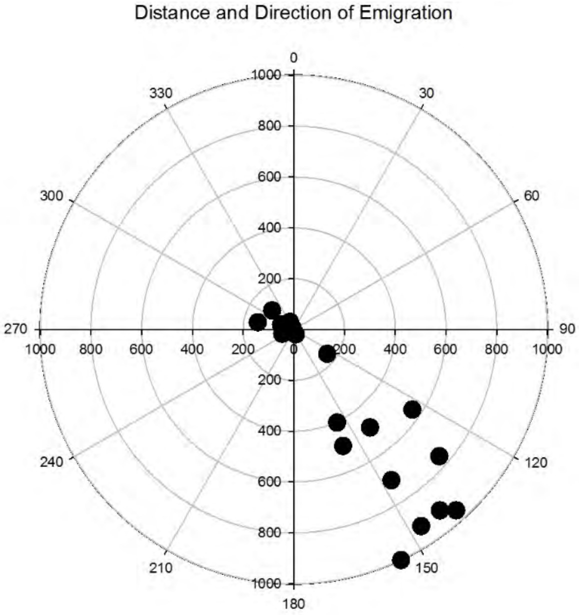

Figure 4. Polar plot of distance and direction of movement on permanent emigration (black dots) for mallards marked with tracking units in Minnesota, 2016. Concentric rings represent distances (km); azimuth (degrees) of movements are indicated on the outermost ring of the plot.

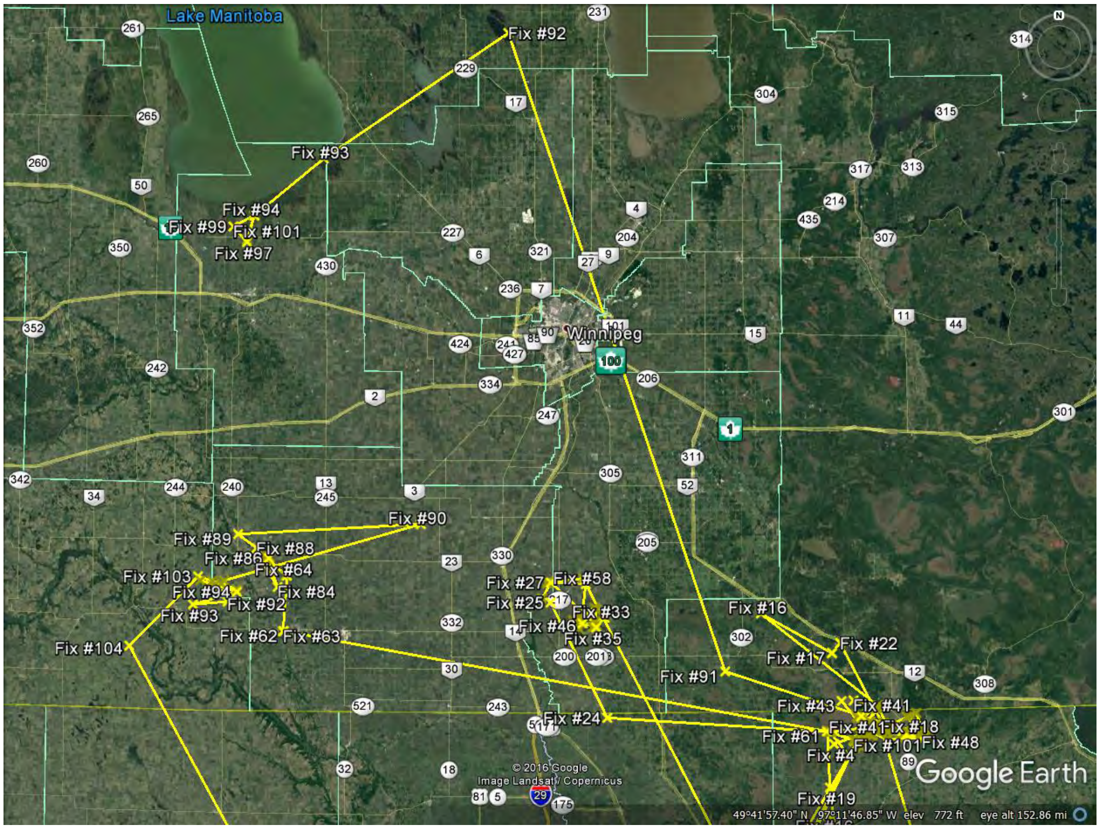

Figure 5. Movements of marked individuals into southern Manitoba from northwest Minnesota in 2016. Yellow Xs on the map represent telemetry fixes and consecutive fixes are connected by yellow lines.

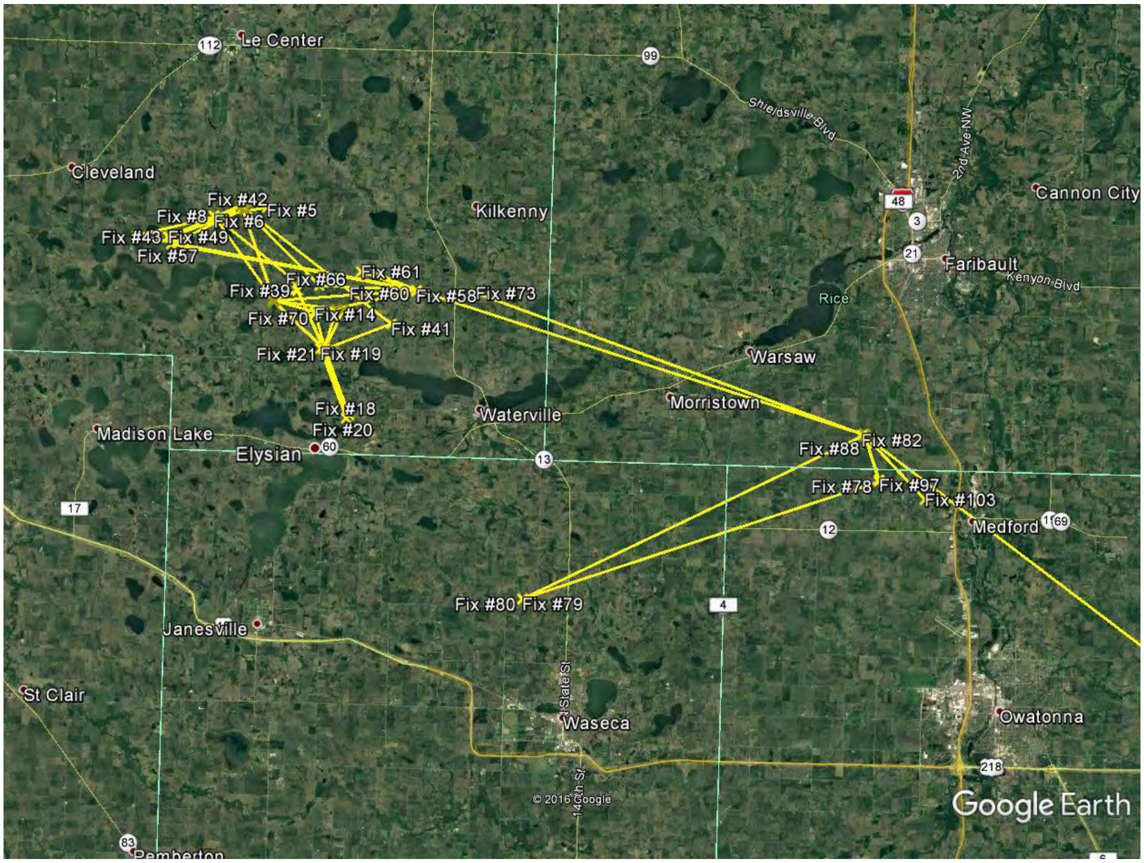

Figure 6. An example of tracking data for a single marked female used for habitat use analyses from Minnesota in 2016-2018. Yellow Xs on the map represent telemetry fixes and consecutive fixes are connected by yellow lines.

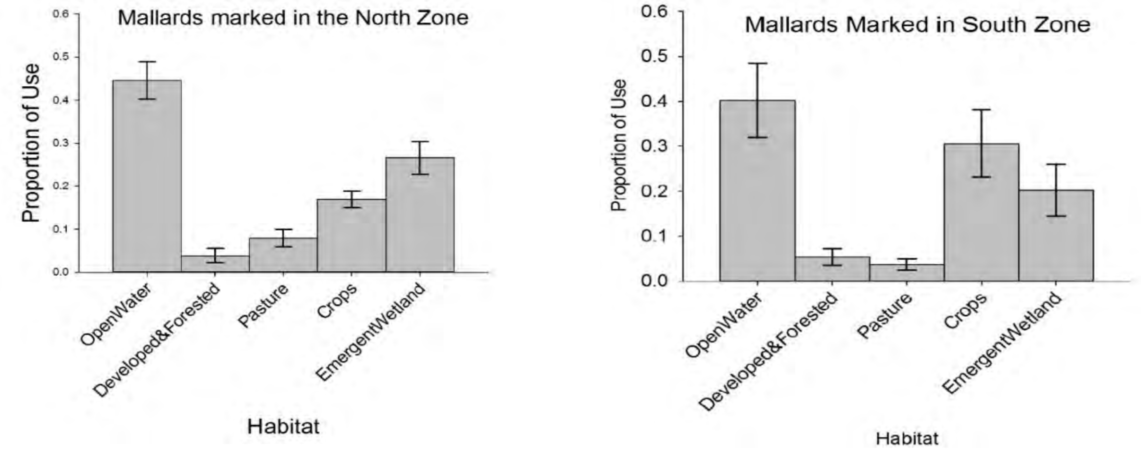

Figure 7. Proportional use of habitats by mallards marked in the Minnesota's north and south hunting zones, 2016. Proportions are expressed as an average across birds.

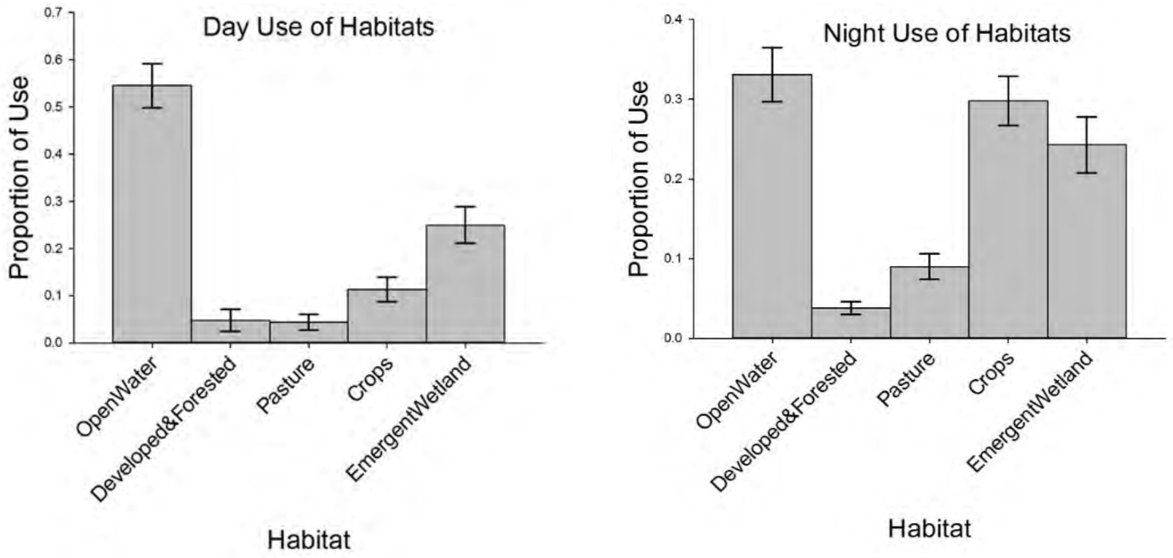

Figure 8. Proportional day or night use of habitats by mallards marked in the Minnesota's north and south hunting zones, 2016. Proportions are expressed as an average across birds.

FOREST INVENTORY ATTRIBUTES PREDICT THE PRESENCE OF CAVITIES SUITABLE FOR NESTING BY WOOD DUCKS

Edmund J. Zlonis and James B. Berdeen

SUMMARY OF FINDINGS

Secondary cavity-nesting birds such as waterfowl and raptors rely on tree cavities developed principally through decay and damage processes or excavation by woodpeckers. Forest and wildlife managers are tasked with maintaining and producing these essential habitat components through forest management practices. Generating predictions about where cavities have developed based on commonly collected forest-inventory data would aid in the conservation of important bird species. Wood ducks (*Aix sponsa*) are a common and well-studied example, though until recently, population-management efforts have primarily focused on artificial nesting structures as opposed to influencing forest-management decisions. We measured and inspected 7,869 trees and 1,186 potential cavities to determine their suitability for use by nesting wood ducks in forests of north-central Minnesota during 2016-2018. Fifteen logistic regression models using tree- and stand-level forest attributes were compared and tested for their utility in predicting whether trees had developed suitable cavities. Our top model was additive and included 3 tree-level predictors: diameter at breast height (DBH), health status, and species. We also found some support for including an interaction between DBH and health status, but it was not in our top model. The top model predicted whether trees had suitable cavities well, with an average area under the receiver-operating-characteristic curve of 0.85. For every 1cm increase in DBH, the odds that a given tree would have a suitable cavity increased by 7.3% (95% CI; 6.0-8.7%). Dead and declining trees were more likely to have suitable cavities than live-healthy trees, with 834% (483-1420%) and 477% (276-807%) higher odds, respectively. When comparing 7 common deciduous species with cavities, sugar maple (*Acer saccharum*) and American basswood (*Tilia americana*) were most likely to have developed cavities. These results can be applied to existing forest-inventory datasets to predict the availability of cavities in the landscape and to maximize conservation benefits for wood ducks and other large-bodied secondary cavity-nesting species.

INTRODUCTION

Conservation of cavity-nesting bird populations depends on diverse cavity excavator communities, but also knowledge of the decay and damage processes associated with both excavated and non-excavated cavities (Wesołowski 2012, van der Hoek et al. 2017, Edworthy et al. 2018). Forest attributes such as tree size and decay class are often linked to these processes (Fan et al. 2003*b*, Gutzat and Dormann 2018). Identifying the forest characteristics associated with cavity formation is particularly important for the conservation of large secondary cavity-nesting species (e.g., waterfowl, raptors), which rely on previously formed cavities that only develop through tree decay and damage or excavation by 1 or 2 woodpecker species (Martin et al. 2004, Cockle et al. 2011).

Wood ducks (*Aix sponsa*) are among the most studied large secondary cavity-nesting species (Bellrose and Holm 1994, Hepp and Bellrose 1995). Although much research and management has focused on artificial nesting structures for this species, recognition that natural cavities are used by most of the population (Bellrose 1990) has led to increased research on use (Robb and Bookhout 1995, Ryan et al. 1998, Yetter et al. 1999, Roy Nielsen and Gates 2007) and availability of natural cavities (Zwicker 1999; Nielsen et al. 2007; Denton et al. 2012*a*, *b*). Most studies have been conducted in east-central USA, often in bottomlands and floodplain forests. However, northern portions of the western Great Lakes states, northern Wisconsin and most of Minnesota, have received relatively little research on natural cavities, despite including portions of the most productive wood duck breeding habitat (Soulliere et al. 2007, Sauer et al. 2017; *but see* Nagel 1969, Gilmer et al. 1978).

Forest attributes associated with the formation of suitable nesting cavities for wood ducks have primarily included tree species and diameter at breast height (DBH) (Bellrose and Holm 1994, Nielsen et al. 2007, Denton et al. 2012*a*). However, broader studies of cavities have also identified tree health status, stand-level variables such as stand age and site productivity, as well as potential interactions between these variables as being important predictors of the occurrence of cavities (Carey 1983; Fan et al. 2003*a*, *b*; Larrieu and Cabanettes 2012). Data on these characteristics are collected during most routine forest inventories and hence can be used to predict the presence or abundance of cavities and provide information to guide forest-management decisions at both stand and regional scales (Fan et al. 2003*b*, Denton et al. 2012*b*, Gutzat and Dormann 2018).

Cavities and associated forest-structural elements like snags are increasingly being considered during forest-management activities. For example, some agencies provide timber-harvest regulations or guidelines specifically targeted at retaining cavities or promoting conditions associated with cavity development (e.g., Minnesota Forest Resources Council 2012). However, specific quantitative measures of forest attributes related to cavity formation are lacking for most secondary cavity-nesting species, including wood ducks. Models that predict cavity occurrence with these standard forest metrics would help managers determine the effects of forest management and target management activities for these species.

In this study, we assessed the utility of selected forest attributes for predicting the occurrence of suitable wood duck nesting cavities in north-central Minnesota. Our primary objectives were to 1) describe the physical characteristics of cavities available for use by wood ducks in this region and 2) compare and validate statistical models based on commonly collected tree and stand-level forest attributes for predicting whether trees have suitable cavities. This information will be useful for forest and wildlife managers tasked with conserving wood ducks and other large-bodied cavity-nesting birds.

METHODS

Study Area

The study was conducted on a 254,000 ha site in northeastern Cass County, Minnesota, USA (47˚N 94˚W; Figure 1) during 2016-2018. The landscape is dominated by forest, with interspersed wetlands, lakes, and small municipalities. Forest types are diverse due to the proximity of the boreal forest to the north and east and the prairie-forest boundary to the south and west (Aaseng et al. 2011). Portions of 3 ecological units occur within the study area: Chippewa Plains, Pine Moraines-Outwash Plains, and St. Louis Moraines (Hanson and Hargrave 1996). The most common forest cover-types are aspen (*Populus* spp), upland pine, northern hardwoods, lowland conifer, lowland hardwoods, and oak (*Quercus* spp). Ownership is largely public, covering 82% of the study area.

Forest Stand and Plot Selection

We focused sampling efforts on publically owned lands with geo-referenced forest-inventory databases. Data from Cass County, State of Minnesota, and United States Department of

Agriculture (USDA) Forest Service were combined by categorizing similar cover types into 6 basic forest-types: aspen-birch, lowland hardwoods, upland conifer, northern hardwoods, oak, and other. ‘Other’ largely consisted of non-forest lands (e.g., brush, grassland, and wetland) and lowland-conifer forests, which likely has few or no cavities that can be used by wood ducks (Soulliere 1990, Clugston 1999, Vaillancourt et al. 2009).

Within the 5 general forest types, we used estimates of stand age to further eliminate stands unlikely to have trees large enough to produce cavities suitable for use by wood ducks. In a nearby study site, Gilmer et al. (1978) indicated that aspen forests >60 years old and northern hardwoods stands >100 years old were most likely to produce cavities used by wood ducks. To ensure we captured the breadth of stands producing trees with potential cavities, we eliminated aspen-birch stands <50 years old and stands of all other types <80 years old. Nearly 7,000 stands met these criteria (22% of public lands). We then randomly selected 60 stands of each forest type for possible cavity sampling.

We randomly placed 1 or 2 0.126-ha (20-m radius) plots in each stand with the stipulation that plots were >50 m apart and >30 m from the nearest stand boundary. Small stands or those with narrow and irregular shapes could often only accommodate 1 plot. Where appropriate, we used ground reconnaissance to adjust the location of plots to be more representative of forest structure (e.g., plots located near ecotones that were not identified in available GIS layers were moved into the stand interior). We attempted to visit all stands and plots, but some were dropped due to accessibility issues, cultural heritage sites, timber harvesting, or improper cover types (e.g., after ground reconnaissance). In addition, we sampled fewer upland conifer and lowland hardwoods stands when compared to other types due to limited numbers of cavities. The random points were placed using ‘genstratrandompnts’ in Geospatial Modelling Environment (Beyer 2012).

Forest Plot Sampling

Tree surveys

Plots were surveyed in leaf-off conditions during late-fall through early spring to ensure adequate detection of cavities in the tree canopy (Denton et al. 2012*a*). At each plot we classified the general forest type based on dominant and codominant trees. In addition, we measured all trees large enough to potentially develop cavities used by nesting wood ducks (≥22cm DBH; Haramis 1975) and tall enough for DBH to be measured (≥1.37 m). For each tree, we recorded species, DBH (0.1cm increments) and health status. Health status codes included 7 categories along a continuum from live-healthy to dead-decomposing trees (Thomas et al. 1979): 1) Healthy live trees with no defects that will threaten its long-term health, 2) live trees with defects that suggest a decline in health (defects include dead limbs, decay on the bole, and the presence of fungi), 3) recently dead trees with bark, limbs, and twigs largely intact, 4) dead trees that have lost some limbs and almost all twigs, 5) dead trees that have lost most limbs and all twigs, 6) dead trees with broken tops and bole wood that is hard, and 7) dead trees with broken tops and bole wood that is soft. Trees with their center beyond the edge of the 20-m radius plots were not measured (e.g., 41-cm DBH tree 19.8 m from plot center). We followed established Forest Inventory and Analysis (FIA) protocols for determining when to delineate an individual stem as a tree to be sampled (e.g., forking trees) and where to measure DBH (e.g., leaning trees; U.S. Department of Agriculture Forest Service 2014).

Cavity surveys

At each tree, 2 to 4 observers used binoculars to conduct a preliminary ground-search for cavities that were potentially suitable for nesting by wood ducks. Depending on the size and height of a given tree, observers circled the tree, stopping frequently to look for cavity entrances and ensuring that all portions of the tree had been examined. During this initial search, we used the minimum entrance dimensions used by a nesting wood duck (6 X 6 cm; Zwicker 1999, Denton et al. 2012*a*) and minimum height of cavity entrance (0.6 m; Strom 1969) to identify all potential cavities to further assess with a camera system. Since observers could not explicitly measure the entrance dimensions at this point in the survey, they were conservative and documented any cavity entrance or similar situation that could potentially meet minimum dimensions and lead to a suitable cavity, including blind spots on tree branches and splits that could not be adequately observed from the ground. We did not formally estimate cavity detectability; with similar minimum entrance dimensions and leaf-off conditions Denton et al. (2012*a*) reported a 98-100% detection rate with ground surveys under similar conditions.

At each potentially suitable cavity, we used a Pyle Model PLCM22IR camera attached via braided wire to a 15.2 m Crain CMR Series telescoping pole (*sensu* Waldstein 2012) to perform a more thorough examination of the entrance and interior of the cavity. We used a handheld tablet to view the camera feed from the ground. We first determined whether cavity-entrance dimensions met minimum criteria by attempting to pass a circular 6 X 6 cm disc attached to the camera through the cavity opening. We then examined cavity interiors with the camera to ascertain whether it was suitable for use by nesting wood ducks using the following criteria: 1) Horizontal depth (from inner edge of the entrance opening toward the back of the cavity) that appeared deep enough for hens to move from the entrance to the interior of the cavity, 2) vertical depth (from the bottom of the cavity to the bottom of the entrance) of ≥10 cm and ≤4.5 m and not hollow to the ground (Bellrose and Holm 1994), 3) nest-platform dimensions of ≥14 x 15 cm (Boyer 1974, Haramis 1975, Denton et al. 2012*a*), and 4) lack of standing water or excess debris in the cavity (Sousa and Farmer 1983).

We classified the suitability of each examined cavity as suitable, marginal, unsuitable, or unknown. We considered a cavity to be ‘suitable’ if all of the above conditions were met. Since we were not able to definitively measure each dimension, a cavity was classified as ‘marginal’ if it was unclear whether all dimensional requirements were met (i.e., ≥1 dimensional measurement appeared to be close to some minimum or maximum value). Cavities were classified as ‘unknown’ if we were unable to completely observe the cavity, either because the location of the cavity or some structural attribute did not permit observation with the camera system. We considered a cavity to be ‘unsuitable’ if any of the dimensional criteria were not met or if there was standing water or excess debris in the cavity. Reasons cavities were unsuitable were recorded and based on the order that structural restrictions would have been encountered as a wood duck entered a cavity (i.e., entrance dimensions, followed by horizontal depth, vertical depth, and finally, dimensions and other characteristics of the platform).

In addition to suitability, we recorded cavity height (0.1-m increments), entrance type (3 classes: opening on the top of the tree, side, or a combination of top and side openings that are joined on the exterior of the tree), the primary source of cavity formation (11 classes: split, broken limb, broken top, woodpecker, fire, lightning, insect, logging wound, decay/rot, other, unknown), and any recent evidence of animal use.

Statistical Analyses

Predictor variables

We identified 3 tree- and 2 stand-level predictor variables expected to influence whether a given tree would develop a cavity suitable for nesting by wood ducks. Tree-level variables were collected as described above and included tree species, health status, and DBH. Stand-level variables included stand age and site index, which were acquired from publically available forest-inventory datasets used in the stand- selection process. Each metric has been shown to influence cavity availability in previous studies (e.g., Carey 1983; Brawn et al. 1984; Allen and

Corn 1990; Fan et al. 2003*a*, *b*; Gutzat and Dormann, 2018) and are collected as part of most standard forest inventories, including FIA.

Health status and species were categorical variables, whereas stand age, site index and DBH were continuous variables. Data were sparse for health status codes 3-5; thus, we collapsed categories into: 1) live-healthy tree, 2) live tree with signs of declining health (e.g., dead limbs, decay), and 3) dead trees (all dead types 3-7). Twenty-seven tree species were sampled (Appendix A), but only 7 species with >500 observations were used in statistical analyses: American basswood (*Tilia americana*), bigtooth aspen (*Populus grandidentata*), paper birch (*Betula papyrifera*), quaking aspen (*Populus tremuloides*), red maple (*Acer rubrum*), red oak (*Quercus rubra*), and sugar maple (*Acer saccharum*).

We defined stand age as the number of years between when a stand originated and when it was sampled for cavities. Site index was recorded as the number of feet a tree would grow in 50 years in a given stand. Site index data were not available for 11 stands on the Chippewa National Forest, so we imputed values from adjacent stands within the same Terrestrial Ecological Unit and of the same cover type (USDA Chippewa National Forest, unpublished data). Trees with at least 1 unknown cavity and no other suitable cavity were removed prior to analysis (n=61) because suitability could not be determined. None of the numeric predictors were highly correlated (r <0.45) and all variance inflation factors (Zuur et al. 2010) were smaller than 2, thus we included all numeric predictors in our analysis.

Model development

We developed 15 candidate logistic regression models to explain the relationship between tree- and stand-level characteristics and the probability that a tree would develop a cavity suitable for nesting by wood ducks (Table 1). Our response metric was the presence-absence of a suitable cavity. DBH was included in each model due to the clear relationship it has with cavity development (Jensen et al. 2002, Fan et al. 2003b). We predicted that stand-level variables were more likely to influence cavity dynamics either in conjunction with or in addition to tree-level factors, thus there were no models with just stand-level predictors (*but see* Fan et al. 2003*a*). We also evaluated potential interactions, primarily between DBH and tree health status or species, as suggested by Fan et al. (2003*b*). Finally, we considered a random effect to account for the clustering of tree data within plots.

We used the ‘glm’ function in R version 3.4.3 (R Core Team 2017) with a logit link function and binomial distribution in all models. Each tree was classified as either having a suitable cavity or not. There was limited support for including plot as a random effect in exploratory models ran using ‘glmer’ (Bates et al. 2014), so all models included fixed effects only. We used odds ratios to compare the relative contribution of each predictor on the outcome that a tree had suitable cavity.

During preliminary modelling, we found a strong, positive, effect of DBH on cavity presence, but confidence intervals were wide at high DBH values. Thus, we collected additional field data targeting only large DBH trees during spring, 2018. Plot selection was similar to the description above, but included aspen-birch stands ≥65 years old and northern hardwoods or oak stands ≥100 years old. In addition to randomly selecting older stands, we also targeted larger trees by only measuring early-successional species >40cm DBH (bigtooth aspen, paper birch, quaking aspen, red maple) and late-successional species >50cm DBH (American basswood, red oak, sugar maple). Up to 5 plots were placed in each stand, using the same criteria as the original plot selection. Other aspects of data collection were unchanged.

Model selection and predictability

We compared the value of candidate models in 2 ways, AIC-based model-selection (Burnham and Anderson 2002) and an evaluation of model predictability using the area under the receiver operating characteristic curve (AUC; Fawcett 2006). We compared AICc values for each model and considered all models within ∆2AICc of the top model as competing models (Arnold 2010). Ultimately, we selected the most parsimonious model (i.e., fewest parameters) from within this group to be the top model for interpretation and recommended application to forest-inventory data (Burnham and Anderson 2002).

Model predictability was measured using 10-fold cross-validation. Tree data were divided into 10 equal subsets; 90% of data were used to train a given model with the remaining 10% used to test the model. The subsets were shuffled 10 times, so each unique set containing 10% of data was used as a test set once. We then bootstrapped this process 1,000 times, averaging the AUC scores of test data calculated in the R package modEVA (Barbosa et al. 2016). In our case, AUC values assessed the combination of the true-positive and false-positive rates when predicting whether a given tree had a suitable cavity. We interpreted the model with highest mean AUC as having the best predictability and compared this to the top model selected based on AICc.

RESULTS

We surveyed 213 forest plots during 2016-2017 (trees ≥22 cm) and an additional 110 plots in 2018 (trees ≥40 cm). Plots were classified as northern-hardwoods (36%), aspen-birch (27%), oak (24%), lowland hardwoods (7%), and upland conifer (6%). A total of 7,869 trees of 27 species were measured and inspected for cavities (Appendix A). We examined 1,186 potential cavities in 880 of these trees with the camera-system (i.e., some trees had multiple cavities). Of these, 223 were suitable for nesting by wood ducks. Eleven tree species had at least 1 suitable cavity.

Cavity Characteristics

Most cavities were classified as unsuitable for nesting by wood ducks (768; 65%), and the remainder were classified as suitable (223; 19%), marginally suitable (111; 9%), or of unknown suitability (84; 7%). The reasons cavities were considered unsuitable were: insufficient vertical depth (44%), entrance dimensions too small (21%), insufficient horizontal depth (18%), insufficient platform dimensions (14%), excessive debris (2%), and too deep or hollow to the ground (1%). For the cavities considered suitable, the primary sources of development included broken limb (38%), split (21%), broken top (18%), woodpecker excavation (16%), decay or rot (2%), other (4%), and unknown (1%). Entrances were primarily on the side of trees (74%), though top (19%) and combination (7%) entrances were also common. Suitable cavities averaged 7.8 m off the ground (0.9-15.2 m).

Thirty-six percent of suitable cavities had evidence of recent animal use. Most signs of use included nesting materials or food caches perceived to be from squirrels (eastern gray squirrel (*Sciurus carolinensis*), northern flying squirrel (*Glaucomys sabrinus*), or American red squirrel (*Tamiasciurus hudsonicus*)) and other small mammals. However, we also found an active wood duck nest, northern saw-whet owl (*Aegolius acadi*cus) nest, 5 additional bird nests containing unknown eggs or eggshell fragments, 2 raccoon (*Procyon lotor*) den sites and a wasp (*Hymenoptera* spp.) nest.

Statistical Model

Our final analysis dataset contained 5,976 trees from 7 species: American basswood, bigtooth aspen, paper birch, quaking aspen, red maple, red oak, and sugar maple. We identified 2 competing models (∆AICc <2) for predicting the probability that a tree would have a suitable cavity; 1) an additive model with DBH, health status and tree species (Mod4), and 2) a similar model but with an interaction between DBH and health status (Mod9; Table 2).

Cross-validation identified a similar subset of models as having the highest predictability (Table 2). The model with highest AUC (0.85) was Mod4, although an additional 6 models had AUC ≥0.83 and all models had relatively good predictability with the univariate DBH model having AUC=0.79. Therefore, our set of competing models was limited to Mod4 and Mod9, where the only difference between the 2 models was an interaction between DBH and health status. Though it has marginally lower AICc, the model with the interaction term included more parameters and had lower overall AUC. Thus, the more parsimonious model (Mod4) was the top model and is what we used for inference. However, we examined the implications of the interaction between DBH and health status (i.e., Mod9).

Mod4 showed a strong positive effect of DBH on the probability that a tree had developed a suitable cavity (Table 3; Figure 2). Holding other predictors at fixed values, for every 1cm increase in DBH the odds that a given tree would have a suitable cavity increased by 7.3% (95% CI; 6.0-8.7%). Dead and declining trees were much more likely to develop suitable cavities than live-healthy trees, with 834% (483-1420%) and 477% (276-807%) higher odds, respectively. Including an interaction between DBH and status (Mod9) resulted in similar conclusions with respect to health status and DBH (Figure 3), although the predicted rates of cavity development were slightly different.

Sugar maple had the highest probability of having a suitable cavity (Figure 2). The odds of finding a suitable cavity in a sugar maple were 26% (95% CI;-21-103%), 79% (-3-246%), 86% (14-211%), 192% (66-439%), 310% (157-566%), and 455%% (149-1381%) higher than in American basswood, red maple, red oak, bigtooth aspen, quaking aspen, and paper birch, respectively.

DISCUSSION

Our results suggest that tree-level attributes collected during most forest inventories can be used to accurately predict the presence of cavities suitable for use by large, secondary cavity-nesting birds like wood ducks. DBH, tree health status, and tree species were good predictors of whether a tree had developed a suitable cavity. Several other studies have found a similar combination of variables when studying cavities and tree-microhabitats available for a broader range of taxa (Fan et al. 2003*b*, Larrieu and Cabanettes 2012, Gutzat and Dormann 2018). We also found support for an interaction between tree health status and DBH. Fan et al. (2003*b*) proposed a similar association, though to our knowledge no studies have explicitly tested for this relationship. With widely available forest-survey data (e.g., FIA) and, increasingly, modelled forest attributes (e.g., via LiDAR; Dubayah and Drake 2000), management agencies can apply these results from local to regional scales for conservation purposes.

Cavity Characteristics

Most cavities that appeared potentially suitable from the ground were not suitable for use by nesting wood ducks when the interior dimensions were inspected. For large species with restrictive dimensional requirements like wood ducks, other studies have found similarly low proportions of suitable cavities (15-33%; Soulliere 1990, Robb and Bookhout 1995, Yetter et al. 1999, Zwicker 1999). However, when a wider range of dimensions were considered suitable, studies have found around 70% of cavities were useable by secondary cavity-nesters (Jensen et al. 2002, Remm and Lõhmus 2011). Studies that do not inspect cavity interiors are likely overestimating cavity availability (Allen and Corn 1990, Fan et al. 2003*b*), especially for species requiring large entrances and interior dimensions.

Similar to other regions, broken tree limbs provided most of the suitable cavities in north-central Minnesota (Soulliere 1990, Denton et al. 2012*a*). Likewise, less than 20% of cavities were excavated by woodpeckers (Soulliere 1990, Yetter et al. 1999, Zwicker 1999, Denton et al. 2012*a*). When assessing cavities available for a broader spectrum of secondary users, woodpeckers appear to excavate higher proportions (Cockle et al. 2011). The relatively low proportion of available wood duck cavities produced by woodpeckers is likely associated with their large dimensional requirements, whereby only pileated woodpeckers (*Dryocopus pileatus*), or, occasionally, enlarged northern flicker (*Colaptes auratus*) cavities can be used (Martin et al. 2004). Yet, several studies have found that wood ducks might actively select woodpecker cavities (Gilmer et al. 1978, Robb and Bookhout 1995, Yetter et al. 1999), indicating that many of the non-excavated cavities in our region, though suitable for nesting, might not be used when abandoned woodpecker cavities exist.

Our cavity-source results differed from more southerly studies of wood duck cavities, with a higher proportion of cavities developed from splits. Frost cracks, which we believe contributed to the majority of the splits we observed, are much more common in trees near their northern range limits (Burton et al. 2008). Most of the cavity-producing trees in our study are in the far northern portions of their ranges in northern Minnesota: sugar maple, American Basswood, red maple, and red oak (Little 1971).

Cavity entrance types were generally similar to those observed in other wood duck studies, with a predominance of side entrances (Soulliere 1990, Denton et al. 2012*a*). Though, broken tree tops and associated bucket-style entrances were somewhat more common [18% in this study vs. 4% in Denton et al. (2012*a*) and 10% in (Zwicker 1999)] and potentially receive proportionally more use by nesting wood ducks in Minnesota (Gilmer et al. 1978). Relatively high density of aspen (*Populus* spp.) in the northern USA might explain this difference. Aspen is commonly infected with heartrot (*Phellinus tremulae*) and other fungal diseases that make the trees more susceptible to windthrow (Hinds 1985), often leaving standing boles with broken tops that can develop useable cavities from the top down (E.Z. and J.B., personal observation).

Our assessment of animal use of cavities was conservative, given sampling only occurred once in fall, winter, or early spring. Many bird species that utilize cavities in our region either had migrated or were not using cavities during sampling. In addition, most evidence of nesting by birds, even large species like wood ducks, deteriorate or are removed after nesting and might not be accurately identified in winter or early spring (Utsey and Hepp 1997). This might explain why we found relatively low use by wood ducks and other secondary cavity-nesting species when compared to studies that actively searched cavities during the primary spring nesting season (<3% this study, 5-13% in Nagel 1969, Robb and Bookhout 1995, Yetter et al. 1999, Zwicker 1999). Yet, results appear to indicate that many suitable cavities are unused and support the finding that a surplus might be available for large-bodied secondary cavity nesting species across much of the Midwestern USA (Denton et al. 2012*b*). Results also provide further evidence that, across the wood duck range, squirrels are likely the primary competitors and users of potential cavities (Bellrose and Holm 1994).

Low use of suitable cavities by wood ducks suggests that cavity-availability is not a major limiting factor of populations in north-central Minnesota and other portions of wood duck range (Zwicker 1999, Denton et al. 2012*b*). However, it might also suggest that the dimensional requirements deemed suitable in these studies are somewhat broad and could include cavities that portions of the nesting population do not select for use. Proposed ideal cavity dimensions and characteristics include entrance dimensions close to minimum requirements, woodpecker cavities, and cavities that are higher above ground level, oriented towards forest openings and close to brood-rearing wetlands (Soulliere 1990, Hepp and Bellrose 1995). However, little is known about nest-site selection by wood ducks and how it relates to cavity dimensions and site-level characteristics (Hepp and Bellrose 1995). Future research should characterize the process whereby cavities are inspected and either rejected or selected for nesting and how this relates to optimum cavity dimensions.

Forest Attributes

We recommend using the more parsimonious additive model that had DBH, health status, and species as predictors (Mod4; Table 3) for application to forest inventory datasets. A strong, positive effect of DBH on the presence of cavities has been repeatedly shown in other studies and our data revealed no exceptions (e.g., Jensen et al. 2002, Fan et al. 2003*b*). The proportion of trees with suitable cavities was generally low for trees <30-cm DBH, but as trees increased beyond 40-cm DBH, the proportion of trees with suitable cavities tended to increased exponentially. Tree size is directly related to the potential size of cavity entrances and interior dimensions and thus is particularly important for large-bodied species like wood ducks (Soulliere 1990). Our data indicate that a reasonable model for predicting the presence of suitable wood duck cavities could be developed solely with DBH. However, the inclusion of tree health status and species significantly improved predictability and model fit. When relating tree-level attributes to cavities, studies have often used live/dead tree status (e.g., live vs. snags; Larrieu and Cabanettes 2012), but our results indicate that including at least 1 additional level distinguishing live-healthy from live-declining trees is important (Fan et al. 2003b, Gutzat and Dormann 2018). Tree health status is acknowledged as an important factor in cavity development for wood ducks (Soulliere 1990), though it has not previously been used to model cavity trees (Nielsen et al. 2007, Denton et al. 2012*b*).

We found declining trees, showing signs of decay through features like fungal growths and dying branches, to be highly associated with the development of cavities. Indeed, decay and related fungal infections of trees are likely the ultimate causes of nearly all cavities, whether they be from woodpecker excavation (Jackson and Jackson 2004, Lorenz et al. 2015) or cavities formed through sources like broken limbs and splits (Wesołowski 2012). With high rates of cavity formation, snags are appropriately thought of as the prototypical cavity tree (e.g., Thomas et al. 1979), though trees in decline are their precursor and likely provide cavities over longer periods of time, resulting in a greater diversity of use by secondary cavity-nesters (Wesołowski 2012, Edworthy et al. 2018).

Results from our competing statistical model (Mod9) support the idea that an interaction between DBH and health status could be important for predicting cavities (Fan et al. 2003*b*). When compared to dead and declining trees, large (>60 cm DBH), live-healthy trees had higher rates of increase in the probability of having suitable cavities. Though in smaller trees, both competing statistical models predicted lower rates of increase for live-healthy trees. The potential synergy between DBH and health status provides further evidence for the utility of using health-impacted or dead trees as a predictor of cavity development, especially in smaller DBH classes.

The propensity of different tree species to produce cavities generally agreed with other studies of wood ducks (Soulliere 1990), as well as more broad taxa (Fan et al. 2003*b*), with sugar maple being a dominant cavity producer. Similar to our study, Denton et al. (2012*a*) found sugar maple and American basswood to be most important on a per tree basis in central Wisconsin. Hard and soft maples (sugar maple, red maple, silver maple; *Acer saccharinum*) were the most important cavity producers in more northern studies of wood ducks and other cavity-nesting waterfowl (McGilvrey 1968, Prince 1968, Gilmer et al. 1978). Conversely, these species were not as important in the southern portions of their ranges (Zwicker 1999), indicating the importance of spatial differences in intraspecific tree damage and disease that eventually lead to cavity formation (Morin et al. 2016).

Though they have a lower number of suitable cavities on a per tree basis, aspen species (*Populus* spp.) are also important cavity sources, given the large number of stems in the region (Minnesota Forest Resource Council 2017). Quaking aspen were the most important species for nesting wood ducks in north-central Minnesota (Gilmer et al. 1978). Many studies in northern temperate and boreal forests have also identified aspen as the dominant producer of wildlife cavities (Martin et al. 2004, Weir et al. 2012) due to their predominance in these regions, but also their attractiveness to woodpeckers for excavation (Jackson and Jackson 2004, Witt 2010). With the exception of yellow birch (*Betula alleghaniensis*), species not included in the analysis appeared to have low rates of suitable cavities, e.g., pines (*Pinus* spp.), ashes (*Fraxinus* spp.), and bur oak (*Quercus macrocarpa*).

Stand-level predictors were not useful in predicting whether trees had developed suitable cavities in our study area. Across the continuum of age classes, stand-age is related to cavity formation (Fan et al. 2003*a*), but in the restricted window of relatively old stands that we selected for sampling, it did not improve model fit. We predicted that site index would be related to cavity formation, as site quality inherently affects growth patterns of trees and associated development of decay processes, with better sites generally growing larger, healthier trees (Carey 1983). However, it is possible that site index was a poor predictor as variations in these processes e.g., tree size and health, were accounted for by tree-level variables, DBH and health status. Additionally, site quality could have more confounding effects than we anticipated, for example, the overall positive effects of DBH in a high quality site might be competing with the negative effects of improved tree health on cavity development.

Forest and Wildlife Management Recommendations

When considering the impacts of forest management decisions on cavity availability for large secondary cavity-nesting species like wood ducks, we recommend retaining large DBH, declining or dead, deciduous trees. In the forests of north-central Minnesota, the most suitable tree species are maples (*Acer* spp.) and American basswood in hardwood forests, northern red oak in oak forests, and quaking or bigtooth aspen in aspen and birch forests. Retention of dead-standing trees is commonly recommended in regards to wildlife and cavity considerations (e.g., Thomas 2002). However, forest and wildlife managers might have the most impact on these resources by identifying declining trees, as most cavities are in live trees and their potential for future and diverse use by secondary cavity-nesters is greater (Fan et al. 2003*b*, Edworthy et al. 2018).

Forest management and harvest techniques including leave trees, selection harvests, and extended rotation forestry could all be used to address these recommendations. For example, Gilmer et al. (1978) found that many of the cavities used by wood ducks were in trees retained after harvest either as leave trees or in uncut patches. Similarly, the tree-level characteristics we found most associated with cavities likely only develop when at least some trees are allowed to grow beyond standard harvest rotations used in temperate deciduous forests.

These characteristics, widely collected as part of forest inventories, are useful for developing practical models of cavities and other habitat components (Fan et al. 2003*a*, *b*; Denton et al. 2012*b*). Managers can use the cavity model and associated quantitative data to predict how harvest and other management decisions might affect cavity availability. Depending on the grain and scale of forest-inventory datasets and intended application, predictions can be made for site-level prescriptions to broader regional strategies. Linking conservation strategies for wood ducks and other secondary cavity-nesting species to forest attributes and forest management decisions will help to consistently provide suitable nesting habitat.

ACKNOWLEDGMENTS

We would like to thank John Giudice and Véronique St-Louis for statistical support and reviewing an earlier draft of this manuscript. Danelle Larson and Katharine Zlonis also reviewed drafts of this manuscript. Steve Mortenson, Jessica Watts, Scott Dunham, John Rickers, and Todd Tisler provided logistical support. Chris Scharenbroich and Robert Wright provided technical assistance. Brielle Coleman, Tony Duquette, Ciara McCarty, Holly North, Tanya Roerick, and Erin Spry helped collect the field data. Jeff Lawrence, Bruce Davis, Mark Hanson, and Steve Cordts provided comments during project development. This project was funded by the Minnesota Department of Natural Resources and the Wildlife Restoration (Pittman-Robertson) Program grant W-71-R-6.

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Table 1 Candidate models for explaining the relationship between tree and stand-level characteristics and the probability that a tree would develop a cavity suitable for nesting by wood ducks in Cass County, MN, 2016-2018. For each model, the predicted effects of covariates are indicated. Models were given an abbreviation for reference between the text and tables.

Model	Abbreviation	Prediction
P(suitable cavity) = DBH	Mod1	Positive effect of DBH
P(suitable cavity) = DBH + Status	Mod2	Positive effect of DBH and differential effects of tree health status levels
P(suitable cavity) = DBH + Species	Mod3	Positive effect of DBH and differential effects of tree species
P(suitable cavity) = DBH + Status + Species	Mod4	Positive effect of DBH and differential effects of levels of tree health status and species
P(suitable cavity) = DBH + Status + Species + StandAge	Mod5	Positive effects of DBH and stand age and differential effects of levels of tree health status and species
P(suitable cavity) = DBH + Status + Species + StandAge + SiteIndex	Mod6	Positive effects of DBH, stand age and site index and differential effects of tree health status levels and species
P(suitable cavity) = DBH * Status	Mod7	Overall positive effect of DBH which varies by levels of tree health status
P(suitable cavity) = DBH * Species	Mod8	Overall positive effect of DBH which varies by tree species
P(suitable cavity) = DBH * Status + Species	Mod9	Overall positive effect of DBH which varies by levels of tree health status and differential effects of tree species
P(suitable cavity) = DBH * Species + Status	Mod10	Overall positive effect of DBH which varies by tree species and differential effects of tree health status levels
P(suitable cavity) = DBH + Status * Species	Mod11	Positive effect of DBH and effect of tree health status that depends on tree species
P(suitable cavity) = DBH * Species * Status	Mod12	Overall positive effect of DBH which varies by both tree species and health status
P(suitable cavity) = DBH + StandAge + SiteIndex	Mod13	Positive effects of DBH, stand age and site index
P(suitable cavity) = DBH + SiteIndex * Species	Mod14	Positive effect of DBH and overall positive effect of site index that varies by tree species
P(suitable cavity) = DBH + StandAge * Species	Mod15	Positive effect of DBH and overall positive effect of stand age that varies by tree species

Table 2 Comparison of candidate statistical models for predicting whether trees have developed suitable cavities for wood ducks in Cass County, MN, 2016-2018. Models were compared based on overall predictability (AUC) and information-theoretic approaches (AICc). The top selected model based on parsimony and predictability is indicated in bold, while the competing model is italicized. Refer to Table 1 for model descriptions.

Model	k	AICc	∆AICc	AICcWt	LL	AUC
Mod9	12	1359.28	0	0.45	-667.62	0.84
Mod4	10	1360.17	0.89	0.29	-670.07	0.85
Mod5	11	1361.85	2.56	0.12	-669.9	0.84
Mod6	12	1362.58	3.3	0.09	-669.26	0.84
Mod11	22	1364.3	5.01	0.04	-660.06	0.83
Mod10	16	1365.34	6.05	0.02	-666.62	0.84
Mod12	42	1384.71	25.42	0	-650.05	0.81
Mod7	6	1397.54	38.26	0	-692.76	0.83
Mod2	4	1397.76	38.47	0	-694.88	0.84
Mod14	15	1450.42	91.14	0	-710.17	0.79
Mod15	15	1462.42	103.14	0	-716.17	0.79
Mod3	8	1463.81	104.53	0	-723.89	0.79
Mod8	14	1469.63	110.35	0	-720.78	0.79
Mod1	2	1483.65	124.37	0	-739.83	0.79
Mod13	4	1484.51	125.23	0	-738.25	0.79

Table 3 Model summary of the top-supported model (Mod4) for predicting suitable cavities for nesting wood ducks in Cass County, MN, 2016-2018. The reference group reflects health status live-healthy and species sugar maple.

Mod4	β	SE
Intercept	-6.72	0.35
DBH	0.07	0.01
Health status		
Declining	1.75	0.22
Dead	2.23	0.24
Species		
American basswood	-0.23	0.24
Red maple	-0.57	0.32
Red oak	-0.62	0.26
Bigtooth aspen	-1.07	0.30
Quaking aspen	-1.41	0.24
Paper birch	-1.71	0.45

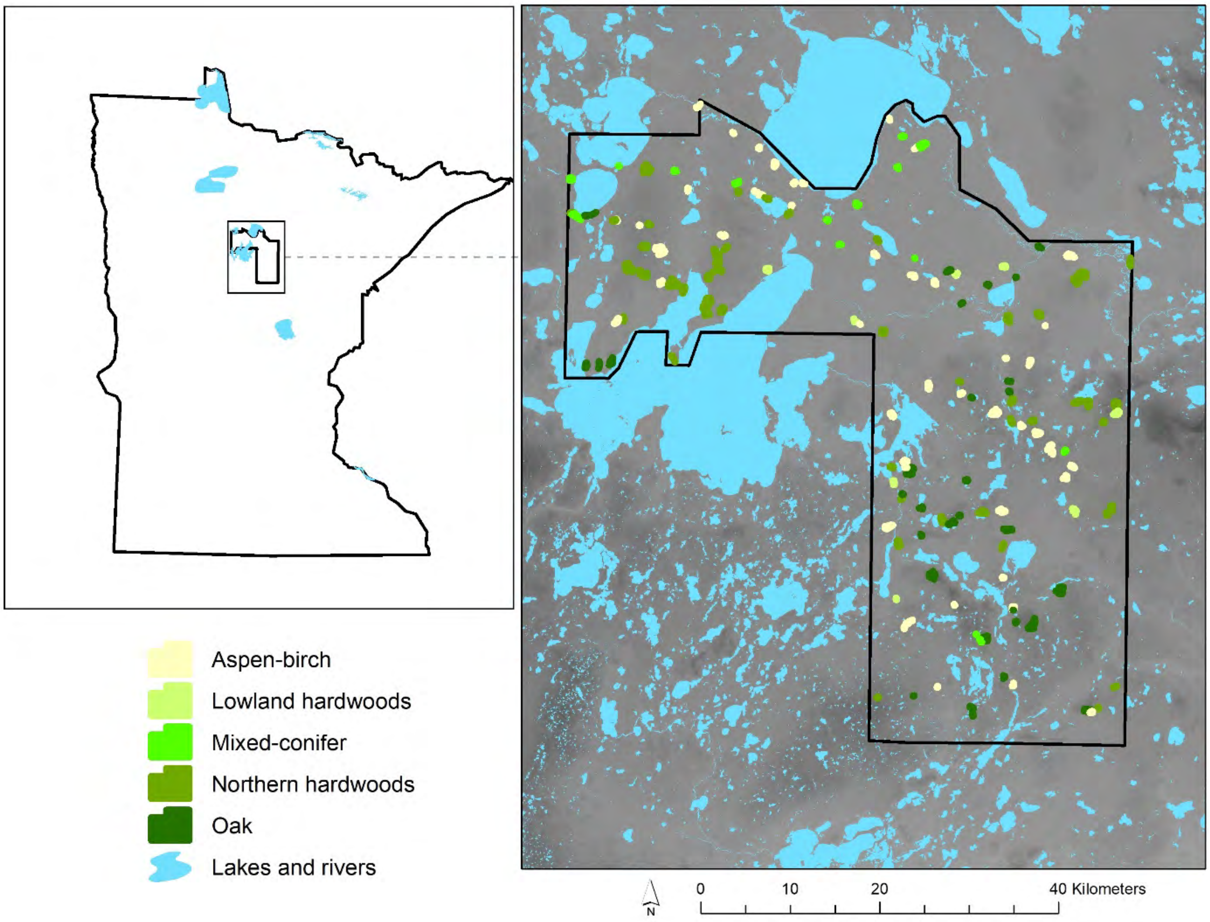

Figure 1. Forest stands sampled for cavities that were suitable for use by nesting wood ducks in Cass County, MN, 2016-2018. Stands were on public lands in county, state, and federal ownerships and were classified into 5 general cover types. Between 1 and 5 20-m radius plots were sampled for cavities in each stand.

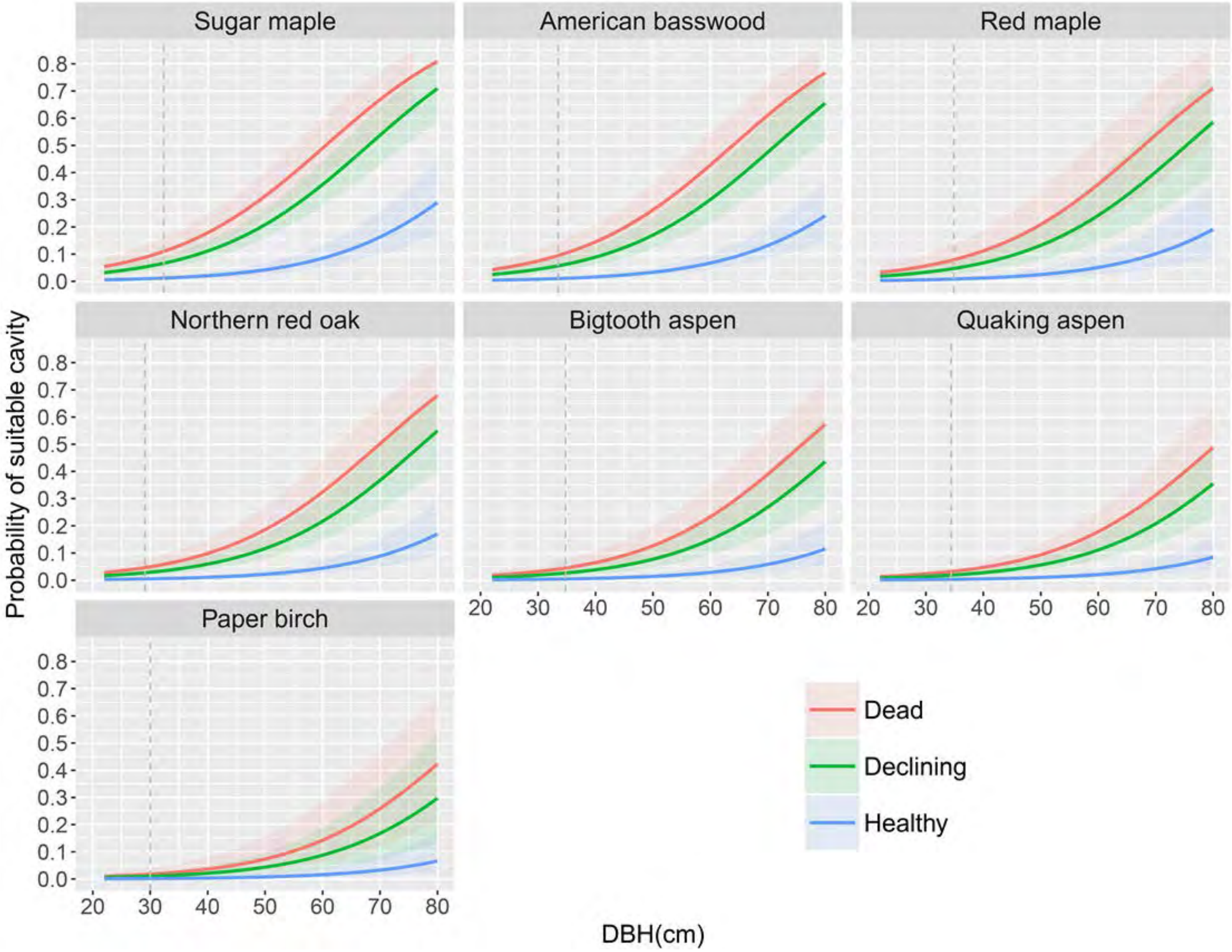

Figure 2. Effect of DBH, health status, and tree species on the probability that trees will have a suitable cavity for nesting by wood ducks in Cass County, MN, 2016-2018. 95% confidence limits are indicated. Dashed lines indicate the mean DBH for a given species.

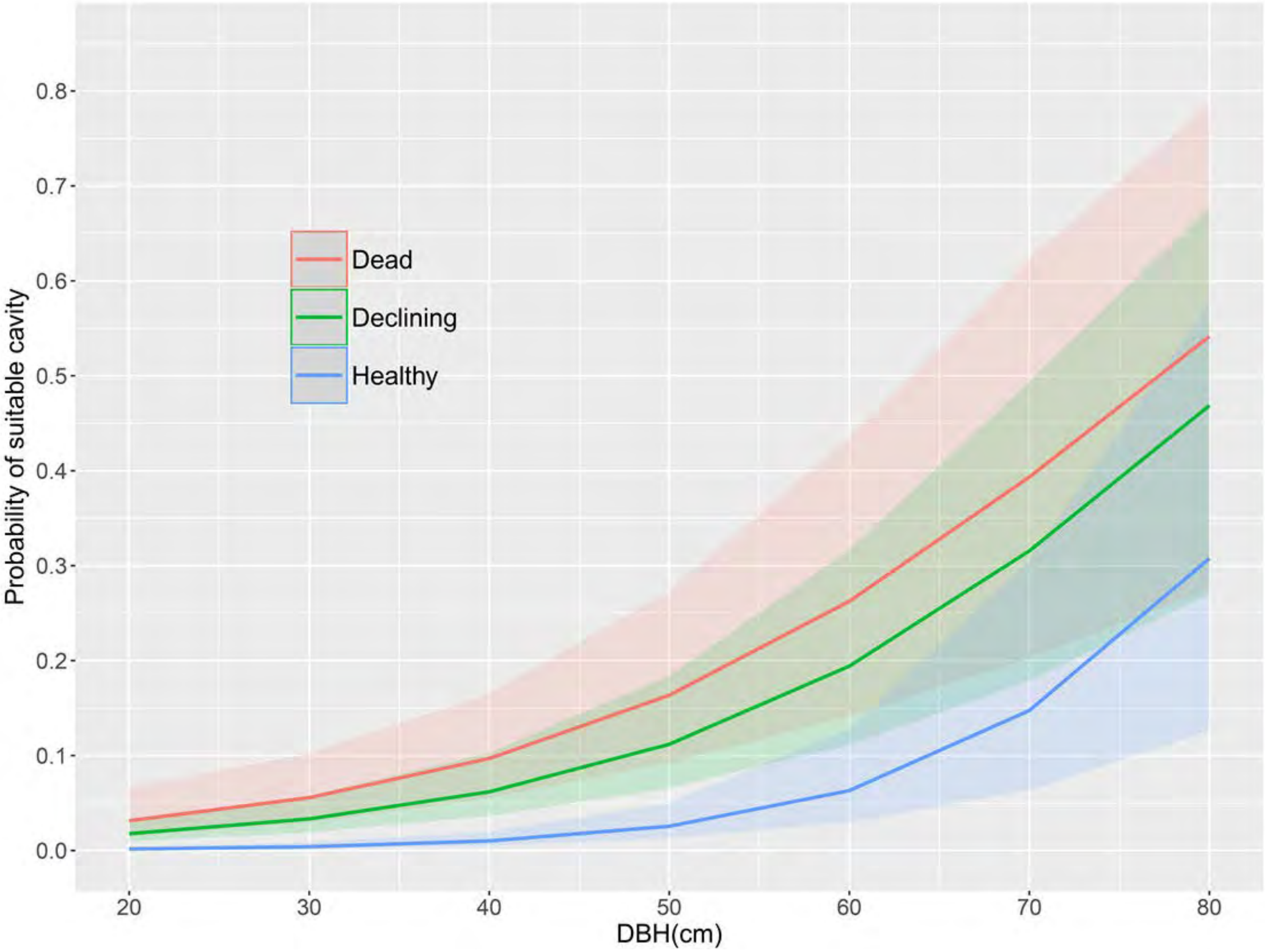

Figure 3. Predicted probability of a suitable wood duck cavity as a function of DBH for dead, declining, and healthy trees in Cass County, MN, 2016-2018. Tree species, the confounding factor, was integrated out to provide a population-level relationship between DBH and health status. Marginal effects were calculated and plotted using package sjPlot in program R (Lüdecke 2018).

Appendix A Total number of trees and suitable cavities counted for each tree species sampled in Cass County, Minnesota, 2016–2018. The proportion of each species within tree health status levels (healthy, declining and dead) is summarized. Some trees had more than 1 suitable cavity, so the number of trees with suitable cavities is also indicated in parentheses. The number of trees sampled in 6 DBH by species bins is also indicated. The proportion of trees with suitable cavities and the associated standard error (calculated from a binomial distribution) are in parentheses. Dashed lines indicate that no trees were sampled or standard errors were not estimable for a tree species-DBH class.

Species	Trees (% total)	Health status	Suitable cavities	22-29cm	30-39cm	40-49cm	50-59cm	60-69cm	70-79cm	≥80cm
Balsam fir (*Abies balsamea*)	126 (1.6)	0.56, 0.17, 0.27	–	105 (0,−)	18 (0,−)	3 (0,−)	–	–	–	–
Box elder (*Acer negundo*)	5 (0.06)	0.4, 0.6, 0	–	3 (0,−)	1 (0,−)	1 (0,−)	–	–	–	–
Red maple (*Acer rubrum*)	569 (7.23)	0.5, 0.4, 0.1	16 (14)	353 (0.008,0.005)	167 (0.024,0.012)	41 (0.146,0.055)	5 (0.2,0.179)	3 (0.667,0.272)	–	–
Sugar maple (*Acer saccharum*)	840 (10.67)	0.57, 0.35, 0.08	61 (57)	393 (0.01,0.005)	218 (0.055,0.015)	103 (0.204,0.04)	93 (0.14,0.036)	26 (0.308,0.091)	5 (0.2,0.179)	2 (1,0)
Yellow birch (*Betula alleghaniensis*)	63 (0.8)	0.52, 0.37, 0.11	5 (5)	20 (0.05,0.049)	12 (0.083,0.08)	18 (0.056,0.054)	11 (0.091,0.087)	–	1 (0,−)	1 (1,0)
Paper birch (*Betula papyrifera*)	809 (10.28)	0.61, 0.2, 0.19	6 (6)	444 (0.005,0.003)	288 (0.01,0.006)	72 (0,−)	4 (0.25,0.217)	1 (0,−)	–	–
Hackberry (*Celtis occidentalis*)	4 (0.05)	1, 0, 0	–	3 (0,−)	1 (0,−)	–	–	–	–	–
Black ash (*Fraxinus nigra*)	291 (3.7)	0.85, 0.12, 0.03	–	214 (0,−)	55 (0,−)	14 (0,−)	5 (0,−)	3 (0,−)	–	–
Green ash (*Fraxinus pennsylvanica*)	231 (2.94)	0.81, 0.15, 0.04	1 (1)	129 (0.008,0.008)	63 (0,−)	27 (0,−)	11 (0,−)	1 (0,−)	–	–
Ash spp (*Fraxinus spp*)	6 (0.08)	0.83, 0.17, 0	–	5 (0,−)	–	1 (0,−)	–	–	–	–
Eastern larch (*Larix laricina*)	2 (0.03)	0.5, 0, 0.5	–	–	1 (0,−)	1 (0,−)	–	–	–	–
Eastern hophornbeam (*Ostrya virginiana*)	1 (0.01)	0, 1, 0	–	1 (0,−)	–	–	–	–	–	–
White spruce (*Picea glauca*)	23 (0.29)	0.83, 0.04, 0.13	–	12 (0,−)	9 (0,−)	2 (0,−)	–	–	–	–

Species	Trees (% total)	Health status	Suitable cavities	22-29cm	30-39cm	40-49cm	50-59cm	60-69cm	70-79cm	≥80cm
Black spruce (*Picea mariana*)	1 (0.01)	0, 1, 0	–	1 (0,–)	–	–	–	–	–	–
Jack pine (*Pinus banksiana*)	25 (0.32)	0.16, 0.04, 0.8	–	12 (0,–)	10 (0,–)	3 (0,–)	–	–	–	–
Red pine (*Pinus resinosa*)	449 (5.71)	0.93, 0.03, 0.04	–	90 (0,–)	181 (0,–)	106 (0,–)	51 (0,–)	16 (0,–)	4 (0,–)	1 (0,–)
Pine spp (*Pinus spp*)	4 (0.05)	0, 0, 1	1 (1)	1 (0,–)	3 (0.333,0.272)	–	–	–	–	–
Eastern white pine (*Pinus strobus*)	133 (1.69)	0.65, 0.17, 0.19	2 (2)	18 (0,–)	31 (0,–)	34 (0,–)	22 (0.045,0.044)	15 (0,–)	6 (0.167,0.152)	7 (0,–)
Balsam poplar (*Populus balsamifera*)	31 (0.39)	0.42, 0.23, 0.35	–	7 (0,–)	17 (0,–)	6 (0,–)	1 (0,–)	–	–	–
Eastern cottonwood (*Populus deltoides*)	1 (0.01)	0, 0, 1	–	1 (0,–)	–	–	–	–	–	–
Bigtooth aspen (*Populus grandidentata*)	511 (6.49)	0.54, 0.28, 0.18	17 (17)	182 (0,–)	154 (0.013,0.009)	109 (0.073,0.025)	49 (0.102,0.043)	14 (0.143,0.094)	3 (0,–)	–
Poplar spp (*Populus spp*)	37 (0.47)	0.05, 0, 0.95	2 (2)	7 (0,–)	16 (0.125,0.083)	10 (0,–)	4 (0,–)	–	–	–
Quaking aspen (*Populus tremuloides*)	1261 (16.02)	0.31, 0.38, 0.3	36 (32)	371 (0,–)	447 (0.018,0.006)	361 (0.05,0.011)	70 (0.129,0.04)	10 (0.1,0.095)	1 (0,–)	1 (0,–)
Black cherry (*Prunus serotina*)	1 (0.01)	0, 0, 1	–	1 (0,–)	–	–	–	–	–	–
Northern pin oak (*Quercus ellipsoidalis*)	13 (0.17)	0.23, 0.77, 0	–	6 (0,–)	7 (0,–)	–	–	–	–	–
Bur oak (*Quercus macrocarpa*)	317 (4.03)	0.83, 0.15, 0.02	5 (4)	163 (0.012,0.009)	90 (0,–)	25 (0,–)	23 (0,–)	15 (0.133,0.088)	–	1 (1,0)
Northern red oak (*Quercus rubra*)	834 (10.6)	0.65, 0.25, 0.1	29 (27)	278 (0.007,0.005)	315 (0.041,0.011)	153 (0.039,0.016)	65 (0.077,0.033)	20 (0.1,0.067)	1 (0,–)	2 (0.5,0.354)

Species	Trees (% total)	Health status	Suitable cavities	22-29cm	30-39cm	40-49cm	50-59cm	60-69cm	70-79cm	≥80cm
Northern white-cedar (*Thuja occidentalis*)	36 (0.46)	0.78, 0.22, 0	−	14 (0,−)	15 (0,−)	4 (0,−)	3 (0,−)	−	−	−
American basswood (*Tilia americana*)	1213 (15.41)	0.84, 0.11, 0.05	40 (39)	522 (0.006,0.003)	381 (0.016,0.006)	168 (0.054,0.017)	95 (0.137,0.035)	37 (0.135,0.056)	5 (0.4,0.219)	5 (0.4,0.219)
American elm (*Ulmus americana*)	19 (0.24)	0.58, 0.26, 0.16	−	16 (0,−)	2 (0,−)	1 (0,−)	−	−	−	−
Unknown spp	13 (0.17)	0.38, 0, 0.62	2 (1)	9 (0,−)	2 (0,−)	−	−	2 (1,0)	−	−

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EVALUATING GRASSLAND WILDLIFE EXPOSURE TO SOYBEAN APHID INSECTICIDES ON PUBLIC LANDS IN MINNESOTA

Katelin Goebel[1] and Nicole M. Davros

SUMMARY OF FINDINGS

Increasing evidence suggests that pesticides may be an important factor explaining declines in grassland-dependent wildlife in agricultural landscapes. Minnesota Department of Natural Resource (MNDNR) wildlife managers and members of the public have reported concerns about foliar-application insecticides in particular. Such insecticides are used on a variety of crops but their use has been especially important for controlling soybean aphid outbreaks in Minnesota. Concerns have been raised about the impacts of chlorpyrifos, a broad-spectrum organophosphate, and other foliar-application insecticides on water quality and human health, prompting the Minnesota Department of Agriculture (MDA) to release guidelines for voluntary best management practices for their use. Although lab studies have shown chlorpyrifos and other insecticides used to target aphids are highly toxic to non-target organisms, including economically important game species and pollinators, few studies have investigated the environmentally-relevant exposure of free-ranging wildlife to these chemicals. Our objective was to assess the direct and indirect exposure of grassland wildlife to the 3 most common soybean aphid insecticides (i.e., chlorpyrifos, lambda-cyhalothrin, and bifenthrin) along a gradient from soybean field edge to grassland interior. During summer 2017 and 2018, we sampled 5 treatment and 4 control sites across western and southern Minnesota. We detected chlorpyrifos at all distances examined (0-400 m) at both treatment and control sites, suggesting that some background level of chlorpyrifos exposure is occurring in the environment regardless of landowner activities in the adjacent row crop field. Our preliminary analyses of filter paper samples (used to quantify direct exposure) showed that insecticide deposition tended to be greater at the field edge than the grassland interior at treatment sites. Furthermore, we detected chlorpyrifos deposition amounts above levels known to cause mortality or morbidity in lab tests for some bird and pollinator species. Our future analyses will use a model-selection approach to determine the effects of weather, vegetation, distance from field edge, and spray application method (i.e., airplane or ground boom) on direct and indirect exposure of wildlife and their invertebrate food resources to these insecticides. Our results will be used to help natural resource managers and private landowners better design habitats set aside for grassland wildlife in Minnesota's farmland region.

INTRODUCTION

Grassland loss and fragmentation is a major concern for grassland-dependent wildlife throughout the Midwestern United States (U.S.). In particular, habitat loss due to agricultural intensification has been implicated as a primary reason for the declines of many grassland nesting birds (Sampson and Knopf 1994, Vickery et al. 1999). However, concerns are

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increasingly being raised about the impacts of pesticides on birds and other wildlife in agriculturally-dominated landscapes (e.g., Hopwood et al. 2013, Hallmann et al. 2014, Main et al. 2014, Gibbons et al. 2015), and some evidence exists that acute toxicity to pesticides may be more important than agricultural intensity in explaining grassland bird declines in the U.S. (Mineau and Whiteside 2013).

Soybean aphids were first discovered in southeastern Minnesota during 2000 and subsequently spread throughout the farmland zone by 2001 (Venette and Ragsdale 2004). Although these aphids pose significant risks to agriculture, their presence does not automatically translate to reduced yield or income (Vennette and Ragsdale 2004). In response to concerns over yield loss, the University of Minnesota Extension Office (hereafter, UM Extension) released guidelines on how to scout for aphids and when to consider treatment for infested fields (UM Extension 2014). Foliar applications of insecticides using ground sprayers or airplanes are common treatment methods when chemical control of aphids is necessary. The 2 most common insecticides used are chlorpyrifos and lambda-cyhalothrin (MDA 2005, MDA 2007, MDA 2009, MDA 2012, MDA 2014a) but bifenthrin is also frequently used (N. Davros, unpublished data; E. Runquist, unpublished data). Withholding times vary by chemical (chlorpyrifos: 28 d; lambda-cyhalothrin: 45 d; bifenthrin: up to 14 d); thus, the timing of product use within the growing season needs to be considered. If retreatment is necessary due to a continued infestation, landowners are encouraged to use an insecticide with a different mode of action to prevent resistance (UM Extension 2014) or reduce the impact of insecticide-resistant aphids (UM Extension 2017, UM Extension 2018). Therefore, multiple chemicals may be used on the same field at different times of the year in some situations. Alternatively, landowners may choose to use a product that combines 2 or more chemicals together (e.g., chlorpyrifos + lambda-cyhalothrin), and such products are readily available on the market.

Chlorpyrifos (common trade names include Dursban, Govern, Lorsban, Pilot, Warhawk, and Yuma) is a broad-spectrum organophosphate insecticide that disrupts the normal nervous system functioning of target- and non-target organisms through direct contact, ingestion, and inhalation (Christensen et al. 2009). Although first registered for use in the U.S. in 1965, its use as an ingredient in residential, pet, and indoor insecticides was removed in 1997 (except for containerized baits) due to human health concerns (Christensen et al. 2009, Alvarez et al. 2013 and references therein, MDA 2014b). Furthermore, MDA released guidelines for best management practices for the use of chlorpyrifos due to water quality concerns (MDA 2014b). Lab studies have shown chlorpyrifos to be toxic to a variety of aquatic and terrestrial organisms (reviewed in Barron and Woodburn 1995), and some bird and beneficial insect species are especially susceptible to acute toxicity from chlorpyrifos exposure (Christensen et al. 2009, MDA 2014a). Chlorpyrifos is very highly toxic to gallinaceous bird species such as the ring-necked pheasant (*Phasianus colchicus*) and domesticated chickens (*Gallus gallus domesticus*), with a lethal oral dose causing death in 50% of treated animals (LD_{50}) of 8.41 mg/kg and 32-102 mg/kg, respectively (Tucker and Haegele 1971, Christensen et al. 2009). Several other bird species are also particularly susceptible to chlorpyrifos, including American robins (*Turdus migratorius*), common grackles (*Quiscalus quiscula*), and mallards (*Anas platyrhynchos*; Tucker and Haegele 1971, Christensen et al. 2009). Yet few field studies have been able to document direct mortality of birds from chlorpyrifos exposure (e.g., Buck et al. 1996, Martin et al. 1996, Booth et al. 2005), and an ecotoxological risk assessment conducted by Solomon et al. (2001) concluded that the available evidence did not support the presumption that chlorpyrifos use in agroecosystems will result in extensive mortality of wildlife. However, chlorpyrifos exposure leading to morbidity (e.g., altered brain cholinesterase activity, altered behaviors, reduced weight gain, impaired migratory orientation) has been documented in both lab and field studies of several avian species (McEwen et al. 1986, Richards et al. 2000, Al-Badrany and Mohammad 2007, Moye 2008, Eng et al. 2017). Thus, sub-lethal effects leading to indirect mortality (e.g., via increased predation rates) or lost breeding opportunities may be a concern for wildlife, especially birds, exposed to chlorpyrifos.

Lambda-cyhalothrin (common trade names include Charge, Demand, Excaliber, Grenade, Hallmark, Icon, Karate, Kung-fu, Matador, Samurai, and Warrior) is a broad-spectrum pyrethroid insecticide that affects the nervous systems of target- and non-target organisms through direct contact, ingestion, and inhalation [National Pesticide Information Center (NPIC) 2001]. Although lambda-cyhalothrin is considered low in toxicity to birds, it is highly toxic to pollinators such as bees (NPIC 2001). Furthermore, field studies have shown lower insect diversity and abundance in fields exposed to lambda-cyhalothrin (Galvan et al. 2005, Langhof et al. 2005, Devotto et al. 2006). Because insects are an especially important source of protein for birds during the breeding season, fewer insects could mean reduced food availability for fast-growing chicks.

Bifenthrin (common trade names include Bifenture, Brigade, Discipline, Empower, Tundra, and Xpedient) is a broad-spectrum pyrethroid insecticide that affects the central and peripheral nervous systems of organisms by contact or ingestion (Johnson et al. 2010). Bifenthrin is low in toxicity to birds, including game species such as northern bobwhite (*Colinus virginianus*) and mallards (oral LD_{50} values of 1800 mg/kg and <2150 mg/kg, respectively; Johnson et al. 2010). However, there are exposure risks for birds that feed on fish and aquatic insects because bifenthrin is very highly toxic to aquatic organisms (Siegfried 1993, Johnson et al. 2010). Some non-target terrestrial insects are also susceptible to bifenthrin (Siegfried 1993). For example, bifenthrin is very highly toxic to bumblebees, with one study showing 100% mortality by contact (Besard et al. 2010).

Minnesota DNR wildlife managers and members of the public have reported concerns about the effects of soybean aphid insecticides on non-target wildlife, including economically important game bird and pollinator species. Although perhaps unfounded, a frequent public concern is that indiscriminate spraying without first scouting for aphid outbreaks has become the norm and fewer birds and insects are observed after spraying has occurred. Yet little is known about the true exposure of birds and terrestrial invertebrates to these insecticides in Minnesota's grasslands. Distances reported for drift from application of foliar insecticides vary widely in the literature (5-75 m; Davis and Williams 1990, Holland et al. 1997, Vischetti et al. 2008, Harris and Thompson 2012), and a recent butterfly study in Minnesota found insecticide drift on plants located up to 1,600 m away from potential sources (E. Runquist, personal communication). The distance of travel for spray drift is dependent on several factors including droplet size, boom height or width, and weather conditions (e.g., humidity, wind speed, dew point) at the time of application. Guidelines for pesticide application are readily available to landowners and licensed applicators (MDA 2014b, MDA 2014c) so that the likelihood of spray drift can be minimized but there is likely large variation in typical application practices.

OBJECTIVES

Our goal was to assess the environmentally-relevant exposure of grassland wildlife to the 3 most commonly-used soybean aphid insecticides (i.e., chlorpyrifos, lambda-cyhalothrin, and bifenthrin; hereafter, target chemicals) in Minnesota's farmland region. Specific objectives included:

1. *Direct and Indirect Exposure:* Quantified the concentration of target chemicals along a gradient from soybean field edge to grassland interior to assess the potential for grassland wildlife (particularly nesting birds and their young, and beneficial insects) to be exposed to these chemicals: 1a) directly via contact with spray drift, and 1b) indirectly through consumption of insect prey items exposed to the insecticides.

2. *Indirect Effects:* Quantified and compared the relative abundance, richness, diversity, and

biomass of invertebrate prey items along a gradient from soybean field edge to grassland interior prior to and post-application to assess the indirect impact of the target chemicals on food availability for grassland nesting birds and other wildlife.

STUDY AREA

We conducted our study within the southwest (SW), west central (WC), and central (C) regions of Minnesota's farmland zone (Figure 1). Corn and soybeans combined account for approximately 90%, 67%, and 71% of the landscape across these three regions, respectively [U.S. Department of Agriculture (USDA) 2019a, USDA 2019b]. Area set aside as grassland cover on public and private land accounted for 6.9%, 10.0%, and 5.6% of the landscape in these regions, respectively (Messinger and Davros 2018). Since 2003, these regions have also experienced some of the highest estimated use of chlorpyrifos and lambda-cyhalothrin (MDA 2005, MDA 2007, MDA 2009, MDA 2012, MDA 2014a).

METHODS

Experimental Design

A treatment study site consisted of a MNDNR Wildlife Management Area (WMA) immediately adjacent to a soybean field that was sprayed to control for aphids. We worked closely with wildlife managers and private landowner cooperators to select treatment sites. We used sites dominated by a diverse mesic prairie mix containing warm-season grasses and forbs because this mix is commonly used by MNDNR managers and agency partners in the farmland zone to restore habitats for the benefit of grassland birds and beneficial insect species. We also selected control study sites with similar site characteristics except that control sites had corn as the adjacent crop and they were not sprayed with any chemicals to control aphids. We chose sites that were predicted to be downwind (typically east or north) from cooperators' agricultural fields based on typical wind direction patterns determined from archived daily summaries of National Weather Service data.

We sampled 5 treatment sites and 4 control sites across 2 field seasons (summer 2017 and summer 2018; Table 1). Within each treatment site prior to spraying, we established sampling stations at distances of <1 m, 5 m, 25 m, 50 m, 100 m, and 200 m along each of 3 transects. If the site was large enough, we also established a station at a distance of 400 m along each transect. This design gave us a total of 18-21 stations per site. We established transects and stations the same way within control sites. At all sites, transects ran perpendicular to the edge of the cooperator's field and were spaced 90-100 m apart to reduce the likelihood of duplicate insecticide exposure from the spraying event.

Data Collection

To assess the potential for direct exposure of birds and other wildlife to our target chemicals *(Objective 1a),* we deployed passive sampling devices (PSDs) to absorb any chemical drift that occurred. We placed PSDs in treatment fields on the day of but prior to spraying of soybeans. The PSDs were 14 cm tall by 7 cm in diameter and consisted of Whatman^{TM} Qualitative Filter Paper (grade 2; GE Healthcare U.K. Ltd, Little Chalfont, United Kingdom) attached to 0.5-in^2 hardware cloth formed to a cylinder shape to approximate the size and shape of a large songbird or a gamebird chick. We placed the PSDs at 2 heights (ground and 0.5 m high [hereafter, mid-canopy]) at each of the 18-21 sampling stations per site for a total of 36-42 PSDs/site. Ground-level sampling represented ground-nesting birds and other wildlife that spend the majority of their time on the ground (e.g., gamebirds, small mammals, many species of invertebrates). Mid-canopy sampling represented above-ground nesting birds, songbirds, and many species of spiders and insects. We retrieved the PSDs from the field ≤2.25 h after spraying and properly stored them for later chemical analysis. At control sites, we placed PSDs at both ground and mid-canopy levels at each of the stations. We left the PSDs on site for the same amount of time as PSDs at treatment sites before we collected and stored them for later analysis.

During 2017 only, we used water-sensitive cards (Syngenta Global, Basel, Switzerland) to collect spray droplets from chemical drift. These cards changed from yellow to dark blue when they encountered liquid. We attached 4 cards next to each PSD (2 cards on the vertical plane and 2 cards on the horizontal plane) at each canopy layer (ground, mid) of each sampling station. We used these cards to determine if they could be used as a quicker and cheaper method for qualitatively detecting spray drift in grasslands.

During 2018 only, we deployed PSDs during the pre-spraying period (i.e., 1-3 d prior to spraying) at each <1 m sampling station at 3 treatment and 2 control sites. These samples provided us with a secondary field-based control to determine if our target chemicals could be detected within a site prior to known sampling events (treatment sites only).

To assess the potential for birds and other insectivorous wildlife to be exposed to the target chemicals indirectly via consumption of prey items (hereafter, indirect exposure; *Objective 1b*), we sampled invertebrates ≤4 h post-spraying at the <1 m, 5 m, and 25 m stations along each transect (total = 9 stations/site). We sampled ground-dwelling invertebrates using a hand-held suction vacuum and canopy-dwelling invertebrates using a sweepnet. We collected vacuum and sweepnet samples along a 30-m doubled transect (30 m x 2 = 60 m total length sampled) to the right side of the sampling stations and parallel to the soybean field. We combined vacuum and sweepnet samples taken from the same station into 1 sample and properly stored them for later chemical analysis. We sampled control sites using the same methods and timing, with the timing based on when we deployed the PSDs at these sites.

To quantify and compare the effects of target chemicals on the abundance, richness, diversity, and biomass of invertebrate prey items (hereafter, indirect effects; *Objective 2*), we collected vacuum and sweepnet samples from the <1 m, 25 m, and 100 m distances along the 3 transects at each site (total = 9 stations/site). We collected these samples 1-3 d prior to spraying and between 3-5 d and 19-21 d post-spraying at treatment sites. We collected samples along a 20-m doubled transect (20 m x 2 = 40 m total length sampled) but on the left side of the sampling stations and parallel to the soybean field. We combined vacuum and sweepnet samples into 1 sample per station per sampling period and stored them in ethanol for later sorting, identification, counting, and measuring. Each time we returned to the site, we started sampling from the endpoint of the previous 20-m sampling transect. During the 3-5 d and 19-21 d sampling efforts, we also collected invertebrate samples at the same 3 distances along 1 additional transect established >60 m away from but parallel to our 3 main transects. This additional transect provided us with post-spraying control samples to address any concerns about whether our repeat disturbance of the main transects impacted our estimates of indirect effects. We used the same methods and timing to collect our indirect effect samples at each of our control sites. During identification in the lab, we placed emphasis on 4 invertebrate orders important in the diets of grassland nesting birds: Araneae (spiders), Orthoptera (grasshoppers, crickets, and katydids), Coleoptera (beetles), and Hemiptera (true bugs). We sorted all individuals from these orders and identified them to at least the family level for analysis. Quantifying the spider community also allowed us to examine potential impacts on an additional trophic level because spiders are an important predator of insects.

We used portable weather meters (Kestrel 5500AG Agricultural Weather Meters) mounted on tripods and equipped with weather vanes to measure relevant weather data (e.g., temperature, wind speed, wind direction, humidity, dew point) along the center transect at the <1 m, 100 m, and 200 m stations during the deployment of PSDs and at the <1 m, 25 m, and 100 m stations during pre- and post-spraying insect sampling at each site.

At each site, we collected vegetation data 1-3 d prior to spraying at all stations and again at 3-5 d and 19-21 d post-spraying at the reduced subset of stations (i.e., those that coincided with the indirect effects sampling efforts for invertebrates). We sampled multiple vegetation plots at each station: 1 plot at each PSD station and 1 plot at each end of the 20-m and 30-m insect sampling transects. Data collected at each plot included percent ground cover, percent canopy cover, maximum height of live and dead vegetation, litter depth, vertical density, and species richness. Using a modified point-intercept method, we categorized ground cover into bare ground, litter, or other [i.e., woody debris, rock, or gopher mound; Bureau of Land Management (BLM) 1996]. To determine canopy cover, we took a nadir digital photograph of a 30 cm x 55 cm quadrat at a height of 1.5 m above the ground and used the program SamplePoint to estimate percent canopy cover (Booth et al. 2006). Canopy cover categories included grass, forb, standing dead vegetation, woody vegetation, and other. We recorded the maximum height of live and dead vegetation within each plot to the nearest 0.5 dm. We measured litter depth to the nearest 0.1 cm at 1 point within the plot that represented the average condition of the plot. We measured vertical density by placing a Robel pole in the center of each plot and estimating the visual obstruction reading (VOR) from 4 m away and 1 m above the ground in each of the 4 cardinal directions (Robel et al. 1970). We counted the unique number of grass and forb species in each plot to estimate species richness. Finally, we recorded the dominant grass and forb species (up to 3 species in each category) at each PSD station to obtain a qualitative assessment of the vegetation present at each site.

We sent the PSD samples and invertebrate samples (i.e., the direct and indirect exposure samples, respectively) to the USDA Agricultural Marketing Service's National Science Lab (USDA/AMS-NSL) in Gastonia, NC for chemical residue analysis. Samples were analyzed using a solvent-based extraction method. Extracts were concentrated by evaporation and then analyzed using a gas chromatography/mass spectrometry-negative chemical ionization (GC/MS-NCI) technique or other appropriate method. The USDA/AMS-NSL equipment was capable of an extremely high degree of sensitivity in the limit of detection (LOD) and reported all results to us in parts per billion (ppb). Additionally, although our experimental design focused on soybean fields sprayed with foliar insecticides to control aphids, the chemical analyses allowed us to quantify residue of additional pesticides (e.g., neonicotinoids, fungicides) at minimal extra cost. Obtaining information about other pesticide residues provided us with valuable supplementary information that can be used to support other Section of Wildlife research and management goals.

As an additional control, we sent 5 filter paper samples to the USDA/AMS-NSL lab for chemical residue analysis. These samples were not deployed in the field but had been attached to PSD wire frames and held in a storage bin in the back of a field truck prior to shipment to the lab.

Data Analyses

Data analyses are ongoing at the time of this report. Preliminary analyses related to Objective 1a (direct exposure) are discussed below, and we report means and standard deviations unless otherwise noted. Analyses related to Objective 1b (indirect exposure) and Objective 2 (indirect effects) are too preliminary and are not included here. Results may be subject to change by our final reporting.

RESULTS AND DISCUSSION

During fall 2016, we surveyed 12 farmer cooperatives in 12 counties to gather more specific, localized information about chemical spraying (e.g., type of insecticide, spray application method) in southern Minnesota. Congruent with MDA's pesticide usage reports (MDA 2007, MDA 2009, MDA 2012, MDA 2014a), the cooperatives reported that chlorpyrifos, lambda-cyhalothrin, and bifenthrin were the most commonly-used foliar soybean insecticides in recent years. Additionally, we learned that neonicotinoids are also present in the chemical mixes used as foliar treatment of crop pests. This information is contrary to the widespread belief that neonicotinoids are only used as a prophylactic seed treatment to protect plants systemically. Based on estimates provided by 8 of the 12 farmer cooperatives, an average of 63% of fields were sprayed by airplane (range: 40-85%) whereas 37% of fields (range: 15-60%) were sprayed from the ground in 2016. Fields are less accessible by tractor when conditions are wet or soybeans are too tall; thus, these factors can influence a landowner's decision on the type of application method (airplane or ground) to use.

In late winter and early spring 2017, we also mailed surveys to landowners adjacent to potential WMA study sites to learn about their soybean aphid spraying practices and to ask for their cooperation with our study (see Appendix 1 in Davros and Goebel 2016 for details). Although our mail surveys helped us identify willing cooperators, we ultimately found that soliciting landowner cooperation by visiting their residences or calling them was more effective. Therefore, we abandoned the mail survey in 2018. Once we secured landowner cooperation, we kept in contact with them throughout the growing season to determine if and when they would be spraying their soybeans for aphids. After they sprayed, we followed up with them to obtain additional relevant data (e.g., insecticide product used, application rate, tank pressure); however, some landowners declined to provide some of the information (Table 1).

We sampled sites between 28 July – 14 September 2017 and 18 July – 5 September 2018, coinciding with peak activity for aphid spraying in the farmland zone (Table 2). We collected a total of 368 direct exposure PSD samples (*Objective 1a*), 81 indirect exposure invertebrate samples (*Objective 1b*), and 297 indirect effects invertebrate samples (*Objective 2*) across both years. Additionally, we collected 30 pre-spraying PSD samples as our secondary field-based controls in 2018.

Our preliminary analyses of direct exposure to drift *(Objective 1a)* indicated that target chemicals were detected on PSDs at all distances examined (0-400 m) at both treatment and control sites (Table 3; Figure 2). These results suggest that some background level of deposition is occurring in the environment at the time of our sampling regardless of spraying status of the adjacent cooperator's field. Although our control sites did not have target chemicals sprayed during our sampling timeframe, our experimental design did not control for nearby fields, including other row crop fields that were adjacent to our WMA sites but not included in our landowner coordination efforts. If other landowners sprayed for aphids near the time of our sampling and drift occurred, then our PSDs would have detected any drift that traveled onto the WMA site. Although shorter distances of 5-75 m for drift from application of foliar insecticides are reported in the literature (e.g., Davis and Williams 1990, Holland et al. 1997, Vischetti et al. 2008, Harris and Thompson 2012), a recent butterfly study in western Minnesota found insecticide drift on plants located up to 1,600 m away from potential sources (E. Runquist, personal communication).

Our preliminary analyses of our secondary field-based controls (i.e., the PSDs deployed during the pre-spraying period in 2018) found that target chemicals were present at very low levels within control and treatment sites (6 ± 2 ppb and 7 ± 4 ppb, respectively) prior to spraying. These results further support our conclusion that chemical deposition is occurring from elsewhere in the environment besides our cooperating landowners. Similar to our other samples, chlorpyrifos was the main chemical detected in these field-based control samples.

Our preliminary analyses also found that target chemical deposition on PSDs tended to be greater and more variable at the field edge than the grassland interior at treatment sites, particularly for sites sprayed by airplane (Table 3; Figure 2). Although an edge effect of drift from adjacent treatment fields might be expected, further analyses are underway to examine this pattern and determine if spray method may be an important factor.

Our cooperating landowners used chlorpyrifos more often than lambda-cyhalothrin or bifenthrin. Thus, we also conducted a preliminary examination of chlorpyrifos deposition levels independent of these other chemicals. Similar to all target chemicals combined, chlorpyrifos deposition tended to be greater along the field edge and treatment fields sprayed by airplane also showed more variable deposition levels out to 50 m compared to treatment fields sprayed from the ground (Table 4; Figure 3). Additionally, the levels of chlorpyrifos we detected as drift onto WMAs (Table 4) were sometimes above the LD_{50} values reported by Christensen et al. 2009 and Corbin et al. 2009 (Table 5). Chlorpyrifos is highly toxic to honey bees (*Apis* sp.) and can poison non-target insects for up to 24 h after spraying (Christensen et al. 2009). Chlorpyrifos is also very highly toxic to several common farmland bird species, including ring-necked pheasants, American robins, and common grackles. Our preliminary results suggest that birds, pollinators, and other grassland wildlife are being directly exposed to chlorpyrifos drift in Minnesota’s farmland regions. However, we did not collect field data to determine if grassland wildlife species were experiencing lethal or sublethal (e.g., impaired movement, reduced foraging, lethargy, reduced body condition) effects from this exposure, and further research would be needed to address these potential impacts.

Our objective with using the water-sensitive cards was to obtain an immediate, qualitative visual assessment of insecticide deposition. However, even moderately high humidity levels produced a color change in the absence of insecticide deposition (Figure 4a and 4b). The cards also picked up dew droplets from the surrounding vegetation that caused discoloration. Thus, we were unable to reliably detect insecticide deposition and quantify drift using these cards. We discontinued their use in 2018.

We will be finalizing our analyses over the next 2-3 months. This fall, we will begin sharing our findings with multiple constituent groups. Our first step will be to share individual, field-level results with each cooperating landowner to engage them, make them aware of how their participation benefited our research efforts, and show them how the aggregated data will be shared with other groups. Subsequently, we will invite these landowners, other agricultural groups (e.g., University of Minnesota’s Southwest Agricultural Experiment Station personnel; Soybean Growers Association), and various natural resource professionals to a seminar where we will present our overall findings and public land management recommendations. Our proximate goal with these agricultural community outreach events is multifold: 1) bring awareness to the issue of soybean aphid insecticide drift onto grasslands, 2) engage agricultural partners in coming up with solutions to reduce the potential for drift to occur on these grasslands, and 3) promote good will and communication that could be beneficial if MNDNR conducts further pesticide research in the future. However, our ultimate goal is to provide natural resource managers with information on patterns of soybean aphid insecticide drift onto grassland cover in the agricultural matrix of Minnesota. Understanding these patterns will help us improve management of public lands and better design private lands conservation programs to aid grassland wildlife conservation.

ACKNOWLEDGMENTS

We would like to thank the many MNDNR wildlife managers who provided discussions on the topic and/or information on potential study sites. We are especially grateful to the landowners, agronomists, and pilots who cooperated with us by returning our mail survey, calling us to provide information, and/or coordinated with us on the timing of spraying and field sampling. T. Klinkner contacted the farmer cooperatives to ask about soybean spraying practices. T. Klinkner, H. Saloka, and K. Steffl each provided administrative assistance. V. St-Louis and J. Giudice provided valuable feedback on initial study design, and D. Andersen, P. Rice, T. Kissane Johnston, and J. Petersen provided further feedback on study design and methodologies. S. Crosby, C. Fortier, G. Gehring, J. Letlebo, C. Reep, L. Welch, and H. Witt provided assistance with field data collection. J. Alvarado, B. Bessingpas, M. Fox, K. Friedrichs, S. Johnson, E. Oechsle, B. Parajuli, L. Spann, F. Teramayi, M. Vipond, L. Welch, and J. Westfield processed invertebrate samples and/or provided assistance with data entry. L. Christianson and J. Miller identified invertebrate samples. This project has been funded by the Environment and Natural Resources Trust Fund (ENRTF) as recommended by the Legislative-Citizen Commission on Minnesota Resources (LCCMR). The MNDNR Section of Wildlife contributed funding through the Federal Aid in Wildlife Restoration (Pittman-Robertson) Act and the U.S. Geological Survey's Cooperative Fish and Wildlife Research Unit at the University of Minnesota has provided in-kind support.

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Table 1. Spray method and application data for soybean aphid spraying events by cooperating landowners adjacent to Wildlife Management Areas (WMA) that were sampled for insecticide spray drift between 28 Jul - 14 Sep 2017 and 18 Jul - 5 Sep 2018 in Minnesota's farmland zone.

Site ID[a]	Spray method	Insecticide trade name	Insecticide active ingredients	Insecticide application rate (L/ha)	Sprayer application rate (L/ha)	Application speed (m/s)	Boom height (m)	Tank pressure (kPa)
tA	Ground	Endigo	lambda-cyhalothrin + thiamethoxam	0.26	140.3	4.0	0.2-0.3	275.8
tB	Airplane	Bolton	chlorpyrifos + gamma-cyhalothrin	0.88	18.7	67.9	1.5	275.8
tC	Ground	Lorsban 4E	chlorpyrifos	NA[b]	93.5	NA	NA	137.9-206.8
tD	Airplane	Lorsban Advanced	chlorpyrifos	1.17	18.7	55.9	2.7-4.0	275.8
tE[c]	Airplane	Lorsban Advanced; Warrior II	chlorpyrifos; lambda-cyhalothrin	0.44; 0.22	NA	NA	NA	NA

[a] WMA names are not provided to protect private landowner cooperators.

[b] Data is not available because cooperator declined to provide this information.

[c] This cooperating landowner combined two different trade name insecticides during the spraying event.

Table 2. Location, site type, year sampled, and timing of sampling for Wildlife Management Areas (WMA) sampled for insecticide drift from adjacent row crop fields sprayed for soybean aphids during summers 2017 and 2018 in Minnesota's farmland zone.

Site ID[a]	Region[b]	County	Site type[c]	Year sampled	Range of dates when field sampling occurred[d]
tA	SW	Jackson	Treatment	2017	28 July - 18 Aug
tB	SW	Murray	Treatment	2017	9 Aug - 30 Aug
cA	SW	Jackson	Control	2017	21 Aug - 14 Sept
cB	SW	Lyon	Control	2017	7 Aug - 31 Aug
tC	WC	Lac qui Parle	Treatment	2018	10 Aug - 29 Aug
tD	C	Stearns	Treatment	2018	28 July - 16 Aug
tE	WC	Yellow Medicine	Treatment	2018	7 Aug - 28 Aug
cC	C	Kandiyohi	Control	2018	17 Aug - 5 Sept
cD	WC	Lac qui Parle	Control	2018	18 Jul - 8 Aug

[a] WMA names are not provided to protect private landowner cooperators.

[b] Regions sampled in this study include the southwest (SW), west central (WC), and central (C) regions. The boundaries for these regions follow the same boundaries as outlined in the Minnesota Department of Natural Resources' annual August Roadside Survey.

[c] Treatment sites had adjacent soybean fields that were sprayed for aphids; control sites had adjacent corn fields that were not sprayed for aphids.

[d] Includes first day of pre-spray sampling through last day of post-spray sampling for data collection activities.

Table 3. Mean (± SD) values of target chemicals detected on passive sampling devices (PSDs) by distance from soybean field edge to grassland interior on Wildlife Management Areas (WMAs) between 28 Jul - 14 Sep 2017 and 18 Jul - 5 Sep 2018 in Minnesota's farmland zone. Target chemicals included chlorpyrifos, lambdacyhalothrin, and bifenthrin. Values are reported in parts per billion (ppb).

	Distance from soybean field edge (m)						
Site type[a]	0 m	5 m	25 m	50 m	100 m	200 m	400 m
Treatment[b]	35,322 (±145,015)	16,260 (±64,298)	26,712 (±92,827)	385 (±906)	40 (±68)	14 (±20)	699 (±3,508)
Airplane	57,198 (±185,976)	27,080 (±82,113)	44,504 (±117,734)	629 (±1,115)	50 (±84)	7 (±9)	8 (±8)
Ground	2,510 (±5,538)	30 (±30)	25 (±27)	19 (±21)	24 (±30)	23 (±26)	2,254 (±6,322)
Control	41 (±76)	21 (±20)	21 (±19)	21 (±20)	22 (±23)	19 (±18)	30 (±30)

[a] Treatment sites had adjacent soybean fields that were sprayed for aphids; control sites had adjacent corn fields that were not sprayed for aphids.

[b] Cooperating landowners at treatment sites sprayed for aphids using either airplane or ground booms.

Table 4. Mean (± SD) values of chlorpyrifos detected on passive sampling devices (PSDs) by distance from soybean field edge to grassland interior on Wildlife Management Areas (WMAs) between 28 Jul - 14 Sep 2017 and 18 Jul - 5 Sep 2018 in Minnesota's farmland zone. Values are reported in parts per billion (ppb).

	Distance from soybean field edge (m)						
Site type[a]	0 m	5 m	25 m	50 m	100 m	200 m	400 m
Treatment[b]	34,875 (±144,686)	16,049 (±63,954)	26,489 (±92,626)	373 (±879)	38 (±65)	14 (±20)	699 (±3,508)
Airplane	56,451 (±185,631)	26,729 (±81,703)	44,132 (±117,524)	608 (±1,082)	48 (±80)	7 (±9)	8 (±8)
Ground	2,509 (±5,538)	30 (±30)	25 (±27)	19 (±21)	24 (±30)	23 (±26)	2,254 (±6,322)
Control	38 (±72)	20 (±20)	19 (±20)	21 (±20)	21 (±23)	18 (±19)	24 (±22)

[a] Treatment sites had adjacent soybean fields that were sprayed for aphids; control sites had adjacent corn fields that were not sprayed for aphids.

[b] Cooperating landowners at treatment sites sprayed for aphids using either airplane or ground booms.

Table 5. Acute toxicity (lethal dose [LD_{50}] values[a]) of chlorpyrifos for various species as reported in Christensen et al. 2009[b] and Corbin et al. 2009[b].

Species	Scientific name	Oral LD_{50}	Overall toxicity
Pollinator species			
Honey bees	*Apis* sp.	59-360 ng/bee	Toxic
Avian species			
Ring-necked pheasants	*Phasianus colchicus*	8.41 mg/kg	Very highly toxic
Mallards - adults	*Anas platyrhynchos*	76-490 mg/kg	Moderately toxic
Mallards - ducklings	*Anas platyrhynchos*	112 mg/kg	Moderately toxic
Northern bobwhite	*Colinus virginianus*	32 mg/kg	Highly toxic
Canada geese	*Branta canadensis*	40-80 mg/kg	Highly toxic
Sandhill cranes	*Grus canadensis*	25-50 mg/kg	Highly toxic
Common grackles	*Quiscalus quiscula*	5.62 mg/kg	Very highly toxic
Red-winged blackbirds	*Agelaius phoeniceus*	13.1 mg/kg	Highly toxic
American robins[c]	*Turdus migratorius*	NA	
Rock doves	*Columba livia*	10-26.9 mg/kg	Highly toxic
House sparrows	*Passer domesticus*	10 mg/kg	Highly toxic
European starlings	*Sturnus vulgaris*	75 mg/kg	Moderately toxic
Lab species			
Domestic chickens		32-102 mg/kg	Highly toxic
Mice		60 mg/kg	Moderately toxic
Rats		95-270 mg/kg	Moderately toxic
Rabbits		1,000-2,000 mg/kg	Slightly toxic

[a] The LD_{50} value is one common measure of acute toxicity and represents the lethal dose that causes death in 50% of treated animals from a single or limited exposure. The LD_{50} does not reflect any effects from chronic exposure that may occur at levels below those that cause death.

[b] See Literature Cited at end of report for full citations.

[c] American robins are the most frequently reported avian species killed in field incidents; however, the LD_{50} values are unknown.

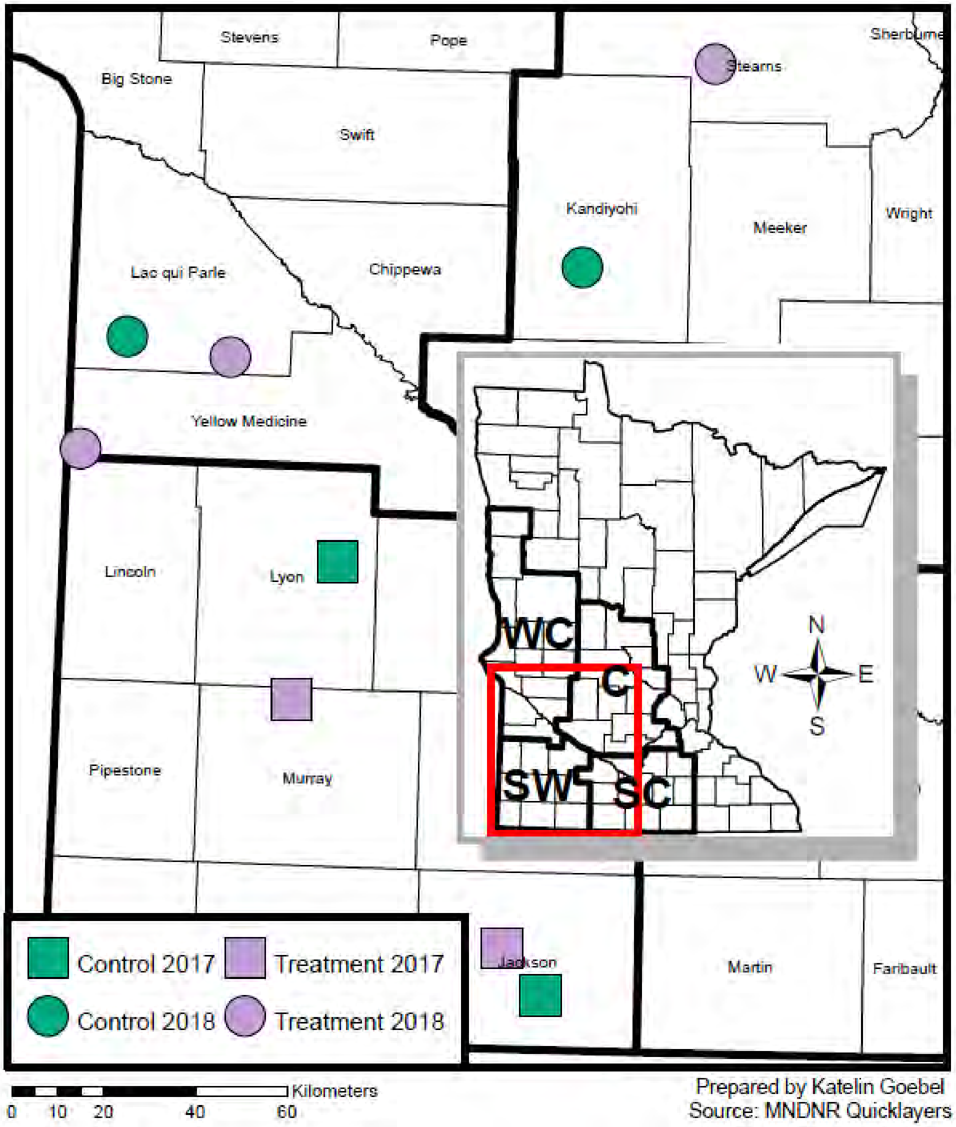

Figure 1. Location of treatment (purple symbols) and control (green symbols) sites during 2017 (square symbols) and 2018 (round symbols) field sampling efforts. Treatment sites were Wildlife Management Areas (WMA) adjacent to soybean fields sprayed for aphids; control sites were WMAs adjacent to corn fields that were not sprayed with insecticides to control soybean aphids. Regions shown are the same as those outlined in Minnesota Department of Natural Resource's annual August Roadside Survey reports and include: SW = southwest, SC = south central, WC = west central, and C = central.

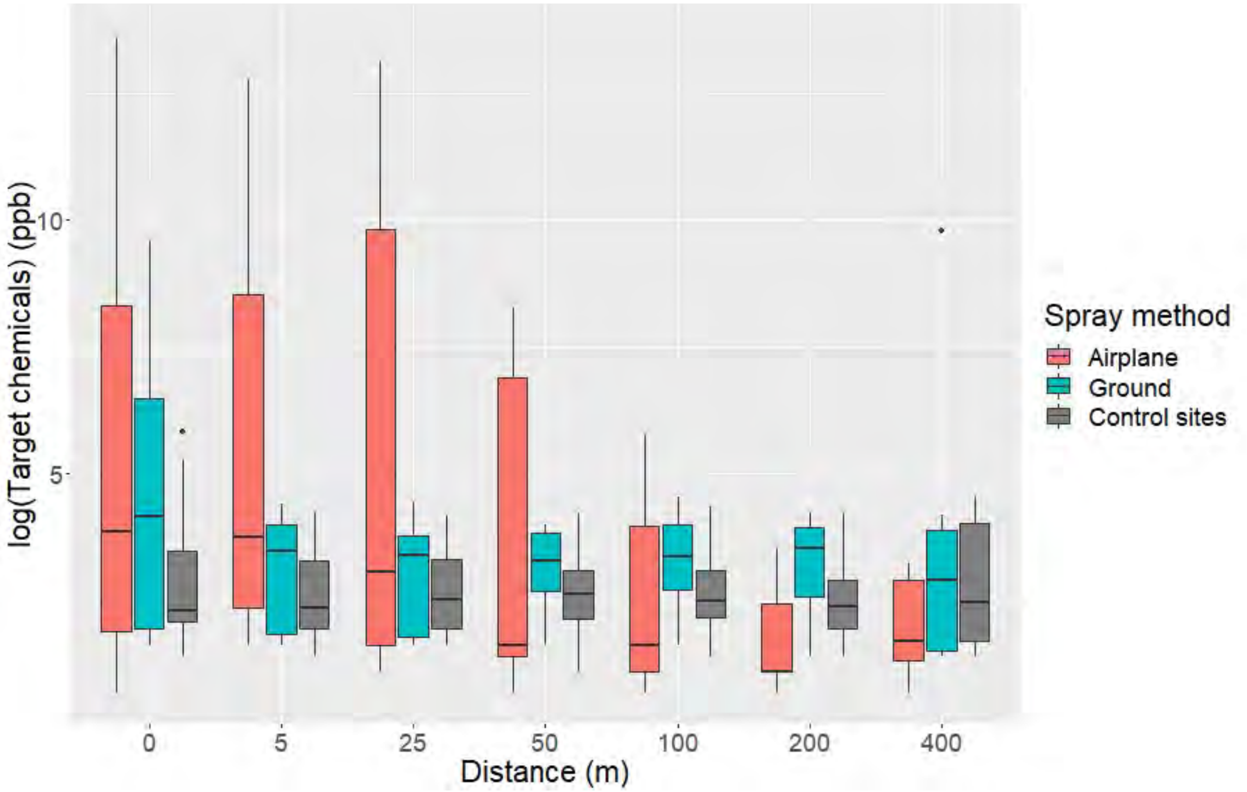

Figure 2. Box plot summaries of target chemical deposition on passive sampling devices (PSDs; *n* = 368) by distance from field edge to grassland interior for treatment sites sprayed by airplane (orange) or ground boom (blue) and control sites (gray), July-September 2017 and 2018 in Minnesota's farmland zone. The PSDs were used to quantify the potential for grassland wildlife to be exposed to chlorpyrifos, lambda-cyhalothrin, and bifenthrin directly through spray drift. Spraying at treatment sites occurred on soybean fields adjacent to grasslands; control sites were grasslands adjacent to unsprayed corn fields. The 0 m distance represents the grassland/row crop edge. Note that distances shown on the x-axis are not graphed to scale.

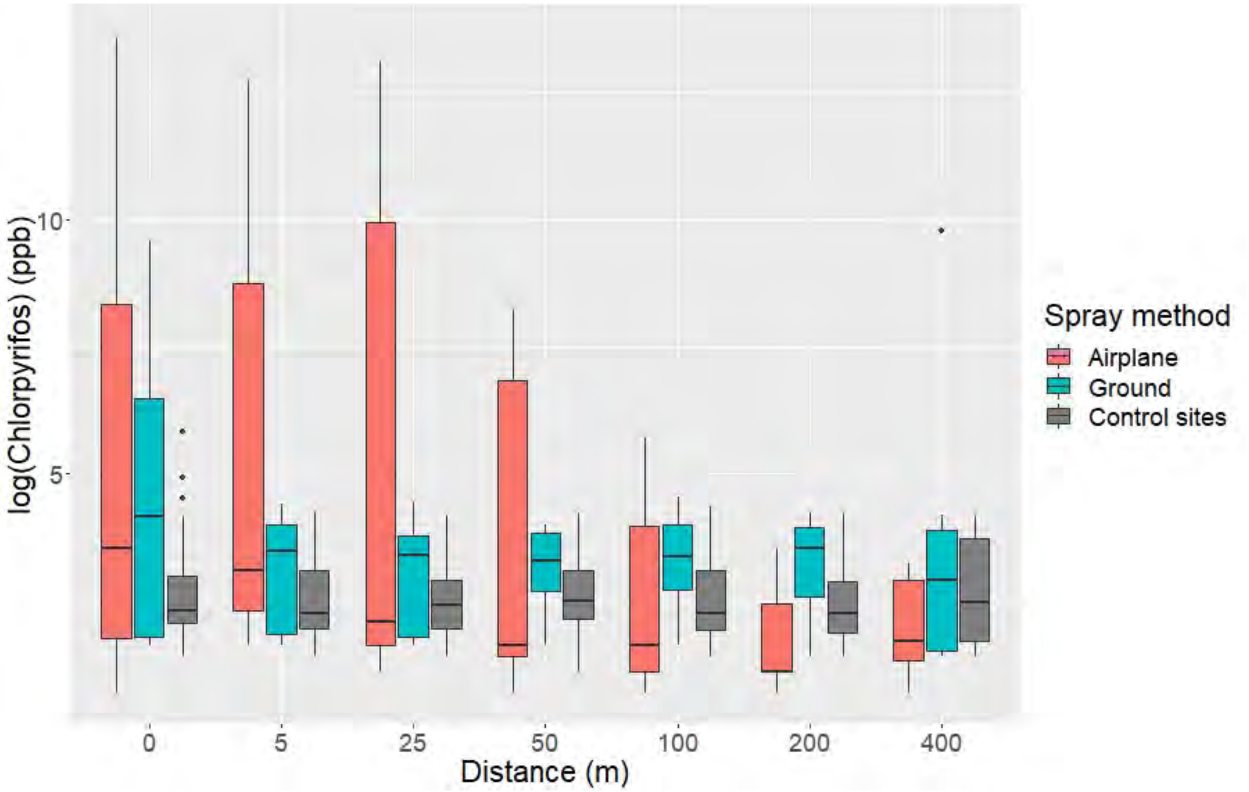

Figure 3. Box plot summaries of chlorpyrifos deposition on passive sampling devices (PSDs; *n* = 368) by distance from field edge to grassland interior for treatment sites sprayed by plane (orange) or ground boom (blue) and control sites (NA; gray), July-September 2017 and 2018 in Minnesota's farmland zone. The PSDs were used to quantify the potential for grassland wildlife to be exposed to chlorpyrifos directly through spray drift. Spraying at treatment sites occurred on soybean fields adjacent to grasslands; control sites were grasslands adjacent to unsprayed corn fields. The 0 m distance represents the grassland/row crop edge. Note that distances shown on the x-axis are not graphed to scale.

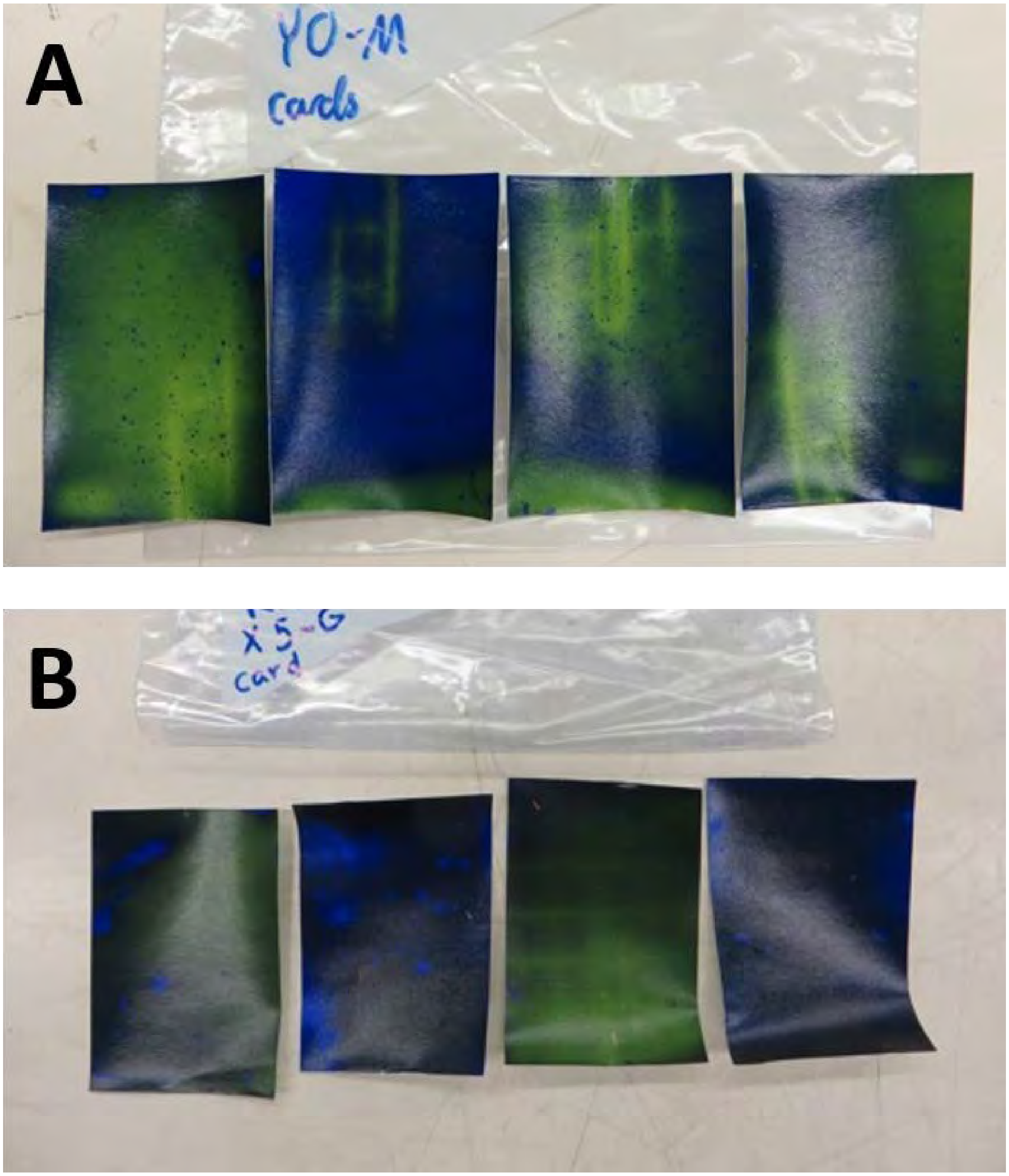

Figure 4. Water-sensitive cards were used during July-September 2017 in an attempt to qualitatively assess insecticide deposition along a gradient from soybean field edges to grassland interiors in Minnesota. Cards turned from yellow to blue when exposed to liquid but relative humidity (RH) levels above 60% also caused the cards to discolor significantly. A) Spray droplets are visible on cards placed at the mid-canopy height a distance of 0 m from the soybean edge at a treatment site; RH was 91% at the time of sampling and also caused major discoloration. B) No evidence of spray droplets is visible on cards placed on the ground at a distance of 5 m from the corn field edge at a control site; RH was 60% and caused the cards to be almost completely discolored; droplets from dew are also visible.

2018 ROADSIDE DISTANCE-SAMPLING SURVEYS OF WHITE-TAILED DEER IN SOUTHERN MINNESOTA

John H. Giudice, Brian S. Haroldson, Tyler R. Obermoller, and Andrew S. Norton

SUMMARY OF FINDINGS

This project was the first year of a 2-year pilot study designed to evaluate the feasibility of using roadside distance-sampling (DS) surveys to generate a reliable and cost-effective population monitoring metric for white-tailed deer (*Odocoileus virginianus*) in Minnesota’s farmland and transition zones. In spring 2018, we surveyed 15 primary sampling units (PSUs) ≥3 times to assess temporal variation in deer population estimates; we observed a similar number of deer across replicates 1–3 (total deer/replicate for all PSUs = 1,038, 1,002, and 1,082, respectively). PSUs included high- and low-density road segments based upon juxtaposition to deer cover. Mean perpendicular sighting distance was greater in the low-density stratum (135 m) compared to the high-density stratum (108 m). As expected in convenience sampling from roadways, deer detections spiked away from the road, which likely reflected road avoidance rather than animal movement. Among-plot variation accounted for approximately 89% of total variation in raw deer counts. Thus, variation due to survey day (run) was relatively small compared to variation in counts among PSUs. Among the 8 DS models fit to the survey data, the 2 best-supported models included a covariate for relative visual obstruction (RVO). Models with strata as a covariate did not fit the data well, which suggests that the detection function [g(x)] did not vary significantly among the 2 strata. The deer density estimate from the top model was 8.6 deer/mi^2 (95% CI = 6.1–12.2). Estimates from the other models were similar. Likewise, the density estimate when data from each stratum were analyzed separately was nearly identical ($\widehat{D}$ = 8.5, ~95% CI = 5.5–11.3), which supports the decision to use a stratified DS estimator where data are pooled across strata to estimate g(x). The density estimate from a winter aerial survey ($\bar{x}$ = 6.4, 95% CI = 5.1–7.7) was comparable. Precision of the density estimate from our top model was reasonable (CV = 17.1%), but likely optimistic because it may not adequately reflect variation due to survey date. Precision was much lower (mean CV = 24.8%) when we bootstrapped distance data using PSU and run (surrogate for survey date). Overall, density estimates seem reasonable and precision was better than expected. We have identified and resolved several data collection and survey-design challenges and have developed detailed field protocols to ensure consistency in data collection. Another year of data collection will be helpful for evaluating the ultimate question of whether a DS metric can be effectively and reliably used to help monitor white-tailed deer populations in Minnesota’s farmland and transition areas.

INTRODUCTION

White-tailed deer (*Odocoileus virginianus*) hunting season recommendations should use the most reliable information available to determine the status of the deer population relative to goal, while prioritizing objectivity. Because hunting season recommendations are made annually, this information needs to be collected on an annual basis. Currently, the only objective annual data the Minnesota Department of Natural Resources (MNDNR) collects at the deer permit area (DPA) scale is winter severity, hunter-reported harvest, and hunter effort. Although these data can provide inference about population trends, they require subjective user inputs that can result in incorrect inference. Objectivity and reliability of harvest-based models can be improved by collecting annual information to model variation of non-harvest vital rate parameters or any other model parameters that vary annually (e.g., harvest reporting rates). A potentially more cost-efficient and alternative approach would be to collect annually recurrent information to independently estimate population trends. Winter aerial surveys can provide this index, but financial and environmental (e.g., snow cover, conifer cover) constraints limit their use to every 5- to 10-years for each DPA; moreover, they are not considered reliable across western Minnesota where seasonal migration is suspected to violate DPA closure assumptions between winter surveys and fall hunting seasons. Several Midwestern states have explored the use of recurrent roadside observation surveys for monitoring deer population trends (Rolley et al. 2016). Variation in the observation process, possibly as a function of annual variation in deer distribution and resource use, has limited the reliability of these indices. DS methods can be used to statistically model the detection probability and calibrate annual variation in the observation process. However, problems have been identified with sampling deer from roadside surveys. Further research is needed to identify an optimal sampling design and evaluate robustness of roadside observation surveys to assumption violations.

Our objective was to evaluate the feasibility of using roadside DS surveys to generate a reliable (potentially biased but reasonably precise and repeatable) and cost-effective population monitoring metric for white-tailed deer in Minnesota’s farmland and transition zones.

METHODS

Sampling Design

The 2,787-mi^2 sampling frame consisted of 4 DPAs (252, 253, 296, and 299) in southern Minnesota (Figure 1). We used a geographic information system (GIS; ArcGIS v. 10.4, Environmental Systems Research Institute, Inc., Redlands, CA) to stratify land-cover within the sampling frame into “high” and “low” strata based upon expected deer density. We defined high-density polygons as being within a 250-m buffer of woodland, grassland (permanent to semi-permanent, excluding pasture), and wetland cover classes. Low-density polygons were the remaining areas (e.g., agricultural land, open water, and urban/developed areas). Data sources for deer-density polygons included Minnesota Land Cover Classification and Impervious Surface Area by Landsat and Lidar: 2013 update – Version 2 (woodlands), a compilation of public/private grassland layers (e.g., Waterfowl Production Areas, Wildlife Management Areas, conservation easements, etc.), and the National Wetlands Inventory for Minnesota (wetlands). We then overlaid the sampling frame with a hexagonal grid, with township-sized hexagons (size = 36.1 mi^2) having >50% of their area inside the sampling frame serving as PSUs. We chose this size because it represented the approximate area that could be surveyed within a 4–6 hr period each night. We randomly selected a spatially balanced sample (Stevens and Olson 2004) of 16 PSUs, but discarded 1 PSU that was on the edge of the sampling frame and contained the city of Mankato. We then used a GIS to identify all secondary (e.g., county and township) roads within each PSU, defined by juxtaposition to deer-density strata (high, low). Finally, we randomly selected road segments (pooling roads $\geq$0.25 mi from all PSUs) using an equal allocation of effort by stratum (~200 mi per stratum). Thus, each PSU contained a combination of high- and low-strata road segments. We derived road data from the Roads of Minnesota, 2012 database. For the purposes of the pilot study, we were interested in obtaining sufficient observations in the low stratum to make informed decisions about the detection process and the potential to modify the stratification and allocation scheme; however, we envision putting more sampling effort into the high-density stratum in an operational survey.

Field Protocols

We surveyed each PSU 3–4 times, with repeated survey dates being close in time within a PSU (i.e., variation in survey dates was greater among than within PSUs). We did this to evaluate daily variation in counts while minimizing the confounding effect of among-PSU differences in counts. We based the start of the survey season on anecdotal information on spring dispersal of deer (from wintering areas to spring-summer-fall range). To be consistent among years and to match the “modeled population”, it was important that deer were on their spring-summer-fall range. We began surveys approximately 1 hr after sunset and we surveyed 1–2 PSUs per night. We conducted surveys with 2-member crews (driver and observer) using extended-cab pickup trucks. We detected deer using FLIR Scout III (FLIR Systems, Inc., Wilsonville, OR) hand-held infrared (IR) sensors attached to the rear windows of the vehicle with window mounts. We viewed images on dual computer monitors attached to the front passenger seat using customized mounts. Monitor power was supplied via the vehicle’s electrical system. The observer searched for deer along the survey route within each PSU. We initially oriented sensors at 45- and 315-degree angles from the direction of travel, but we adjusted them as needed to account for visual obstruction due to variable terrain, woody cover, buildings, etc. Survey speed ranged from 5–30 mph depending upon vegetative cover density. When a deer group (≥1 animal) was identified, the observer directed the driver to an approximate perpendicular angle (i.e., 90 or 270 degrees) from the group to minimize sighting distance and counted group size. Then, while the observer shined the animal(s) with a spotlight, the driver measured distance and angle to the group using a laser rangefinder and digital protractor, respectively. We used a real-time, moving-map software program (DNRSurvey; Haroldson et al. 2015), coupled to a global positioning system receiver and convertible tablet computer, to guide route navigation and record survey metrics (e.g., PSU, run [replicate], deer and vehicle location, distance, bearing, count, cover type) to GIS shapefiles. Cover type designations included woodland, wetland, grassland, pasture, standing crop, harvested crop, other, and unknown classes. We recorded weather data (temperature, wind speed, cloud cover, precipitation) at the beginning, middle, and end of each survey route.

We also conducted a winter helicopter survey of the DS study area using a quadrat-based design, where quadrats were delineated by Public Land Survey section (640 ac) boundaries. We stratified quadrats into 3 density categories (high, medium, low) using the local wildlife manager’s knowledge of deer abundance and distribution. Using optimal allocation, we randomly selected a spatially balanced sample (Stevens and Olson 2004) of 162 plots to survey. Within each plot, a pilot and 2 observers searched for deer along transects spaced at 270-m intervals until they were confident all “available” deer were observed. To maximize sightability, we completed surveys when snow cover measured ≥6 in and we varied survey intensity as a function of cover and deer numbers (Gasaway et al. 1986).

Data Analysis Objectives

1. Perform an exploratory data analysis (EDA) on the 2018 survey dataset (year 1).
2. Fit, evaluate, and compare DS models for estimating deer abundance and density in the sampling frame.
3. Decompose variation in counts due to among-plot (PSU) and within-plot (run or survey date) sources of variation. Also, compare DS models and population estimates from different runs (replicated surveys within PSUs). Temporal variation is especially important in this application because if counts and resulting population estimates are highly variable over time, then a single-effort operational survey (non-replicated counts) may not be reliable.
4. Conduct a power analysis to help evaluate the feasibility of using roadside DS surveys to estimate deer density in Minnesota’s farmland and transition zones. More

specifically, determine how many PSUs would be required to obtain a target level of precision given the current stratification and allocation scheme and observed among-plot (PSU) and within-plot (survey date) sources of variation in roadside counts.

5. Evaluate an alternative stratification scheme (using a 500-m buffer vs. the current 250-m buffer around deer-cover polygons) by re-stratifying road segments (sample only) and deer observations. This is an exploratory post-stratification analysis to determine whether the precision of the population estimate might be improved by modifying the stratification scheme to identify more uniform strata (both in terms of the detection process and relative deer densities).

RESULTS AND DISCUSSION

Summary Statistics and EDA

We completed 48 surveys during 23 nights from 1 April to 6 May 2018. Mean start time was 2055 hours (0.8 hr post-sunset) and mean survey duration was 4.1 hours. All 15 PSU were surveyed 3 times and 3 PSU were surveyed 4 times. Within each PSU, we completed 3 replicates within 8 days and all replicates within 35 days. In total, we detected 931 deer groups (clusters) consisting of 3,194 individual deer (596 deer along low-density road segments and 2,598 deer along high-density road segments). We observed a similar number of deer in replicate surveys 1–3 (total deer/replicate for all PSUs = 1,038, 1,002, and 1,082, respectively). Mean group size (observed) was 4.1 in the low-density stratum (range = 1–41, median = 3), 3.3 in the high-density stratum (range =1–42, median = 2), and 3.4 overall. Group size was not correlated with distance (r = 0.025, 95% CI = -0.039–0.089), which suggests we may not need an adjustment for group-size bias in our DS estimator (a common issue in DS). In the low stratum, 62% of group detections were located in harvested crop fields. Conversely, only 42% of detections were in harvested crop fields in the high stratum, with relatively more detections in grasslands (24% vs. 13%) and woodlands (12% vs. 8%). As expected, mean perpendicular sighting distance was greater in the low stratum (135 m; range = 0–679) compared to the high stratum (108 m; range = 0–503). Additionally, there was a spike in deer detections away from the road (Figure 2). We observed a similar pattern in both strata, although the peak was shifted right in the low stratum, likely because road segments in the low stratum had less deer cover adjacent to roads. As Stainbrook (2001) noted, this could result in a negatively biased population estimate in DS because the mean probability of detection will be overestimated based on the assumptions that g(0) = 1 and objects are distributed randomly with respect to transect lines. This is a common and valid criticism of convenience sampling from roadways. However, if the bias is consistent over space and time, then the DS estimator might still generate a useful long-term and large-scale monitoring metric.

Fit and Evaluate DS Models

Data truncation

A useful rule of thumb in DS is to right truncate at least 5% of the data for robust estimation of the detection function (Buckland et al. 1993:106). The 95th percentile of our distance data was 289 m; therefore, we set w = 300 m which resulted in 4.3% of the data being truncated. We also considered left truncation because the peak in observations was away from the road (Figure 2). However, the peak likely reflects road avoidance rather than animal movement (e.g., due to disturbance, which is unlikely in this case because crews used IR cameras for initial detection). Thus, left-truncation methods would not resolve the underlying issue that animals are not randomly distributed with respect to the transect line. Left-truncation with rescaling would (and did) improve the fit of the model(s) to the data because we now have a shoulder at g(0). However, one would then need to generate a separate ad hoc estimate of abundance for the sampling space that is within some distance *x* of the road transect. Thus, for this pilot-study application, it seemed prudent to set left truncation = 0 and focus on evaluating the consistency of the detection function (i.e., recognizing that the resulting density estimate is likely biased [Stainbrook 2001, Marques et al. 2013], but it may still serve as a useful monitoring index if the bias is reasonably consistent over space and time).

Model structure

The half-normal and hazard-rate key functions are robust estimating functions and allow the inclusion of covariates (Buckland et al. 1993, 2004). Therefore, we focused on these 2 key functions for this analysis. Our base models included no adjustments or covariates. We then added a cosine adjustment to each base model. Finally, we evaluated 2 covariates (with adjustment = NULL). The first covariate, strata, was used to test whether the detection function varied by strata. The second covariate, RVO, was a surrogate for relative visual obstruction (low vs. high) based on mean detection distance by cover type. The "high" visual-obstruction category included woodland, grassland, standing crop, and "other" cover types and contained 487 deer clusters with a mean detection distance of 90 m (range = 0–412). The "low" visual-obstruction category included harvested crop, pasture, wetland, and "unknown" cover types and contained 444 deer clusters with a mean detection distance of 136 m (range = 14–679). The goal here was to determine if RVO could help explain some uncertainty in the detection function, including why g(x) might vary among strata. If RVO could accomplish the latter, then distance data could be pooled over strata to generate a more precise detection function while still generating separate density estimates for each stratum (i.e., a stratified DS estimator; Buckland et al. 2013:99–103, Miller et al. 2016). Conversely, if g(x) varied significantly by stratum (after accounting for RVO), then we would need stratum-specific distance functions. We tested this by comparing density estimates from our top model (where distance data were pooled to compute one detection function) to estimates from a similar model structure but where strata were analyzed separately.

Among-plot and within-plot variation

We decomposed the sampling variance of raw deer counts by PSU and run to determine if "run" (survey date) was a significant source of variation. This is an important consideration because large variation or uncertainty due to "run" would be difficult to control statistically or through survey design, whereas variation due to PSUs could, in theory, be reduced by increasing the sample size. Among-plot variation accounted for approximately 89% of total variation in raw deer counts. Thus, variation by survey day was relatively small compared to variation in counts among PSUs. Consequently, we restricted subsequent DS analyses, including model comparisons, to run #1. Next, we used a bootstrap procedure (with replacement where samples were drawn from both PSU and run) to obtain a more realistic estimate of Var(D) that included among-plot (PSUs) and within-plot (survey date or runs) sources of variation. This should be more reflective of how an operational survey would likely be conducted (i.e., using a single, non-replicated survey). We also used the bootstrap routine to examine precision of the population estimate as a function of sample size (PSUs | allocation is approximately 50:50). This will be useful for evaluating the "feasibility" of conducting an operational survey given some target level of precision. The true expected precision of the estimate is likely somewhere between our top DS model and the bootstrap routine because we cannot completely separate allocation from stratification in either case. To properly estimate total sampling uncertainty, we would need to replicate the entire sampling process, which includes selection of PSUs, road segments within PSUs, and survey dates.

Model comparisons and parameter estimates

We fit 8 DS models to survey data from both strata but restricted to run #1, which provided 281 deer-cluster observations after right truncation (Table 1). We fit all models using the "ds" function in the R library “Distance” (Miller et al. 2016, Miller 2017; R Core Team 2018). The top-supported model (lowest AIC; model 7), was based on the hazard-rate key function and included the RVO covariate. The next-best model (ΔAIC= 9.5; model 3) also included the RVO covariate but was based on the half-normal key function. Models with strata as a covariate did not fit the data well, which suggests that g(x) did not vary significantly among the 2 strata. On the other hand, RVO was useful for describing variation in g(x) associated with cover type (Figure 3), with the underlying mechanism likely being the relative amount of visual obstruction between the observer and the first deer detected. Because relatively more deer were located in harvested cropland in the “low” stratum, RVO also described differences in g(x) between the 2 strata (i.e., 64% of deer groups in the “low” stratum were located in the “low” RVO class, whereas 55% of deer in the “high” stratum were located in the “high” RVO class). The hazard-rate detection function is described by the following equation:

$$g(x) = 1 - \exp[-(x/\sigma)^{-b}]$$

where the parameter *b* is a shape parameter, *σ* is a scale parameter, and *x* is the perpendicular sighting distance (which may be standardized). Covariates enter the detection function via the scale parameter (e.g., $\sigma = \beta_0 + \beta_1 RVO$). The detection function parameters from our top model were $\hat{b}$ = 1.154 (SE = 0.174), $\hat{\beta}_0$ = 5.302 (SE = 0.100), and $\hat{\beta}_1$ = -0.544 (SE = 0.128). Given these parameters, mean detection probability was 0.594 (SE = 0.033, CV = 5.6%), which describes the area under the detection curve. When adjusted for the covariate RVO, the predicted mean probability of detection was 0.491 for deer located in cover types with relatively “high” levels of visual obstruction versus 0.764 for animals with relatively “low” levels of visual obstruction.

The density estimate from model 7 was 8.6 deer/mi^2 (95% CI = 6.1–12.2). Estimates from the other models were similar (Table 1). Likewise, the density estimate when data from each stratum were analyzed separately was nearly identical ($\widehat{D}$ = 8.5, ~95% CI = 5.5–11.3), which supports the decision to use a stratified DS estimator where data are pooled across strata to estimate g(x). Finally, the deer density estimate from the Jan 2019 aerial survey was 6.4 deer/mi^2 (95% CI = 5.1–7.7; MNDNR, unpublished data), which is reasonably similar to the DS estimates given the time lag (spring vs. winter) in surveys.

Precision vs. Sample Size

Precision of the density estimate from our top model was reasonably good (CV = 17.1%), but this is likely optimistic because it may not adequately reflect variation due to survey date. Not surprisingly, precision was much lower (mean CV = 24.8%) when we bootstrapped distance data using PSU and run (surrogate for survey date). This is probably a more realistic expectation of precision for an operational survey with *n* = 15 PSUs and approximately equal allocation of survey effort in each stratum. A common target level of desired precision for management surveys is CV ≈ 15%. To achieve this level of precision with our current design (stratification scheme and allocation) and assuming a single non-replicated operational survey would likely require increasing the number of PSUs from 15 to approximately 30 (Figure 4). Whether this is a feasible option is unknown at this point in time, and an additional year of data is needed to better inform these types of questions.

Post-Stratification Analysis

The above estimates of precision are based on the current stratification and allocation scheme. We anticipate putting more effort into the high-density stratum in an operational survey, which is consistent with DS design recommendations (Buckland et al. 1993). We elected to use an equal allocation of effort in the pilot study to ensure we collected sufficient distance data to evaluate g(x) in the low-density stratum. Finally, our initial stratification scheme, based on a

250-m buffer around deer-cover patches $\geq$2 ac, was exploratory and we have since developed some alternative stratification schemes based on modifying minimum patch size and buffer distance. Unfortunately, it is challenging to evaluate these new schemes using existing distance data (i.e., post-stratification analysis). For example, reclassifying the 2018 distance data using a minimum patch size of 2 ac but with a larger buffer (500 m) did not appreciably change relative precision (17.6%), although it unexplainably generated a larger density estimate (9.2 deer/mi^2; 95% CI = 6.4–13.2). In theory, stratification should improve precision if the stratification scheme is effective, whereas the point estimate should be similar among sampling designs (i.e., it should be design unbiased). However, it is more complicated in DS because we are also dealing with the detection function. And in a post-stratification analysis, the number of observations for estimating g(x) is fixed and sample allocation is confounded with the stratification scheme. Thus, a post-stratification analysis has limited utility for answering the primary question of interest: "which stratification scheme and allocation of effort will produce the most precise estimate?" Obtaining a reliable answer to this question will require a more sophisticated analysis that will likely involve simulating the distribution and DS of deer in a computer-generated landscape (*sensu* Buckland et al. 2004:226–228). Again, another year of data collection would be helpful for constructing such an analysis/simulation.

CONCLUSIONS

The results from the first year of the pilot study are encouraging. Density estimates seem reasonable and precision was better than expected. We identified and resolved several data collection and survey-design challenges and developed detailed field protocols to ensure consistency in data collection. Another year of data collection will be helpful for evaluating the ultimate question of whether a DS metric can be effectively and reliably used to help monitor white-tailed deer populations in Minnesota's farmland and transition areas. More specifically, we will be evaluating whether: (1) g(x) and the distribution of deer relative to roads and cover is relatively consistent over time and space; (2) the effect of variation in spring dispersal can be minimized by using observational cues to inform the start of the survey; and (3) can we afford (staff time and cost) to collect a sufficient sample of distance data in an operational survey to generate a reasonably precise density index for monitoring purposes.

ACKNOWLEDGMENTS

We thank G. Gehring, M. Diamond, J. Westfield, T. Klinkner, M. Orr, and R. Tebo for conducting the surveys. E. Michel and N. Davros reviewed an earlier draft of this report. This project was funded in part by the Wildlife Restoration (Pittman-Robinson) Program.

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Table 1. Distance sampling models based on Akaike's Information Criterion (AIC) used to evaluate roadside surveys of white-tailed deer (*Odocoileus virginianus*) in southern Minnesota, spring 2018. For all models, we restricted survey data to the initial run (replicate) of the 15 primary sampling units, after right truncation. Analysis was restricted to the half-normal and hazard-rate key functions. Covariates included deer density strata (Strata) and a relative measure of visual obstruction (RVO). Mean detection probability, deer density estimates, and summary statistics (CI, CV) are also presented. Confidence intervals for deer density estimates were based on α = 0.05.

Model	Key function	Covariates	AIC	ΔAIC	Detection probability ($\overline{x}$)	Density (deer/mi^2)	95% CI	CV (%)
7	Hazard-rate	RVO	3097	0.0	0.594	8.6	6.1–12.2	17.1
3	Half-normal	RVO	3107	9.5	0.555	9.2	6.5–13.0	17.1
8	Hazard-rate	Strata	3116	18.4	0.606	8.5	5.9–12.1	17.6
5	Hazard-rate	Null	3117	20.2	0.614	8.6	6.0–12.5	18.3
6	Hazard-rate + cosine	Null	3117	20.2	0.614	8.6	6.0–12.5	18.3
2	Half-normal + cosine	Null	3123	25.5	0.630	8.4	5.6–12.7	20.6
1	Half-normal	Null	3124	26.8	0.575	9.2	6.4–13.3	18.0
4	Half-normal	Strata	3124	26.8	0.573	9.0	6.3–12.8	17.7

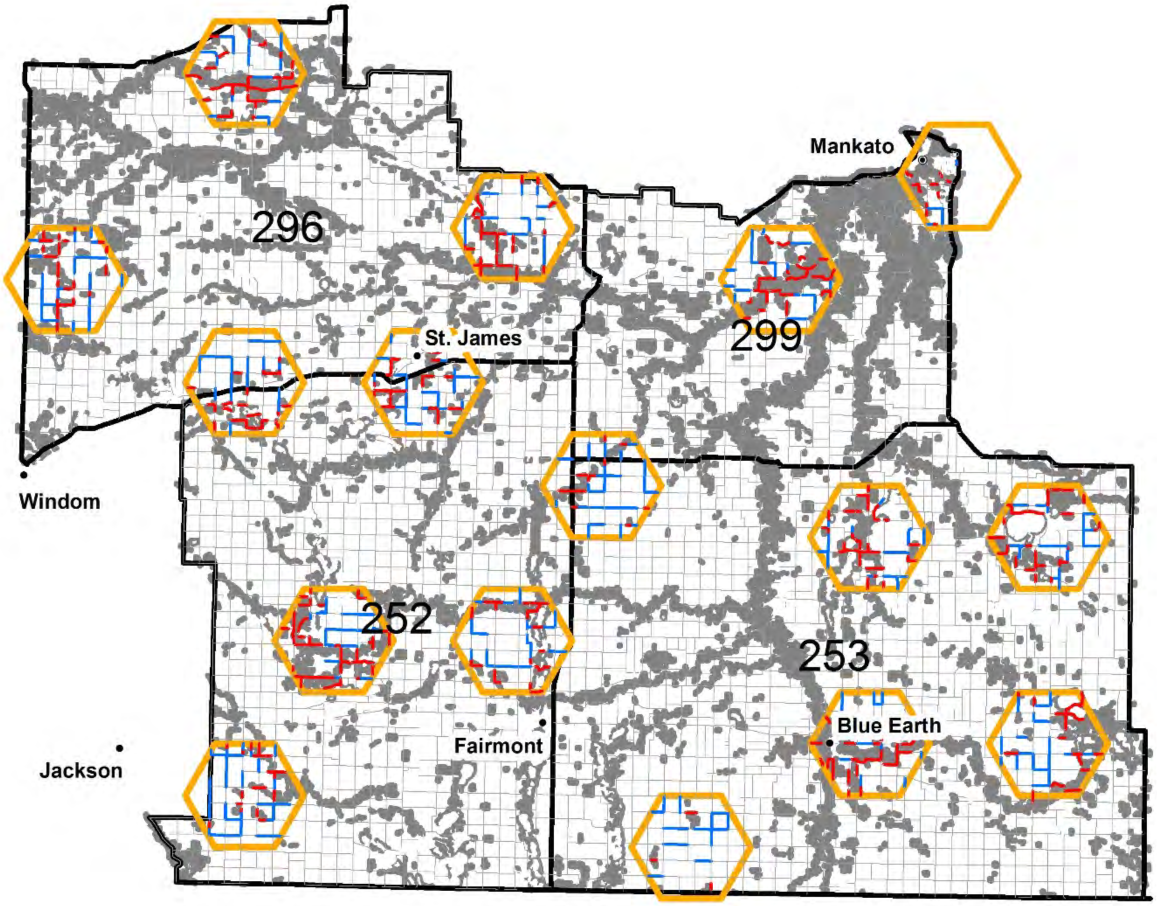

Figure 1. Sampling frame (deer permit areas 252, 253, 296, 299), primary sampling units (PSU; hexagons), and secondary sampling units (road segments; red = high-density stratum, blue = low-density stratum) for roadside distance-sampling surveys of white-tailed deer (*Odocoileus virginianus*) in southern Minnesota, spring 2018. Grey areas denote deer-cover polygons ($\geq$2 ac) consisting of woodland, grassland, and wetland cover types with a 250-m buffer. The northeast PSU was dropped prior to beginning surveys because it was on the edge of the sampling frame, contained the city of Mankato, and included few rural roads.

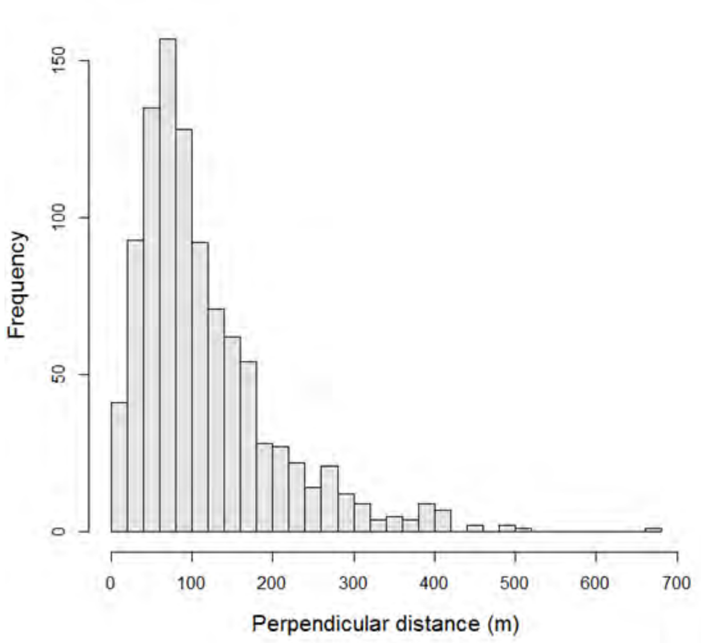

Figure 2. Histogram of deer-cluster observations as a function of perpendicular sighting distance from roadside distance-sampling surveys of white-tailed deer (*Odocoileus virginianus*) in southern Minnesota, spring 2018. Data include distance measurements collected during replicate surveys of 15 primary sampling units.

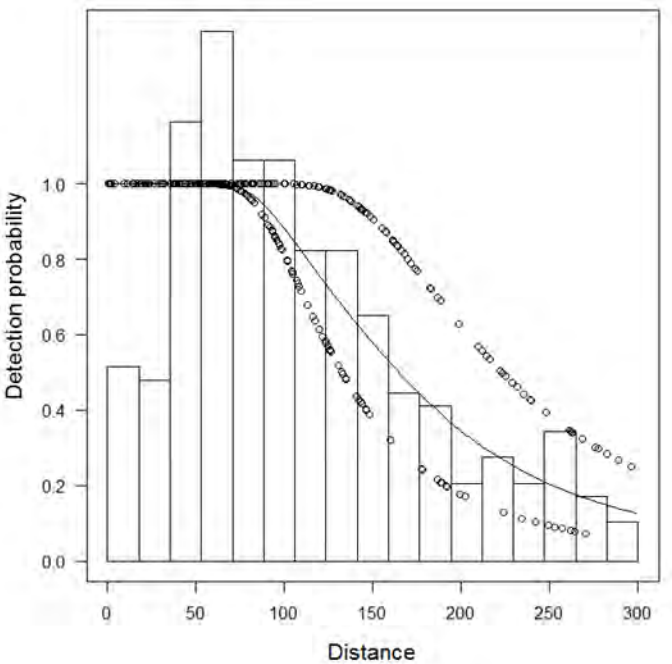

Figure 3. Estimated detection function g(x) from the best-fit model (model 7; based on Akaike's Information Criterion) overlaid on a histogram of deer-cluster observations as a function of perpendicular sighting distance from roadside distance-sampling surveys of white-tailed deer (*Odocoileus virginianus*) in southern Minnesota, spring 2018. We restricted survey data to the initial replicate of the 15 primary sampling units, after right truncation. The solid curved line

denotes the average detection function. The open circles describe the effect of the covariate RVO, which was a binary indicator variable for cover classes where visual obstruction was relatively high (lower line of circles; e.g., woodland and grassland cover) versus where visual obstruction was relatively low (upper line of circles; e.g., harvested cropland and pasture).

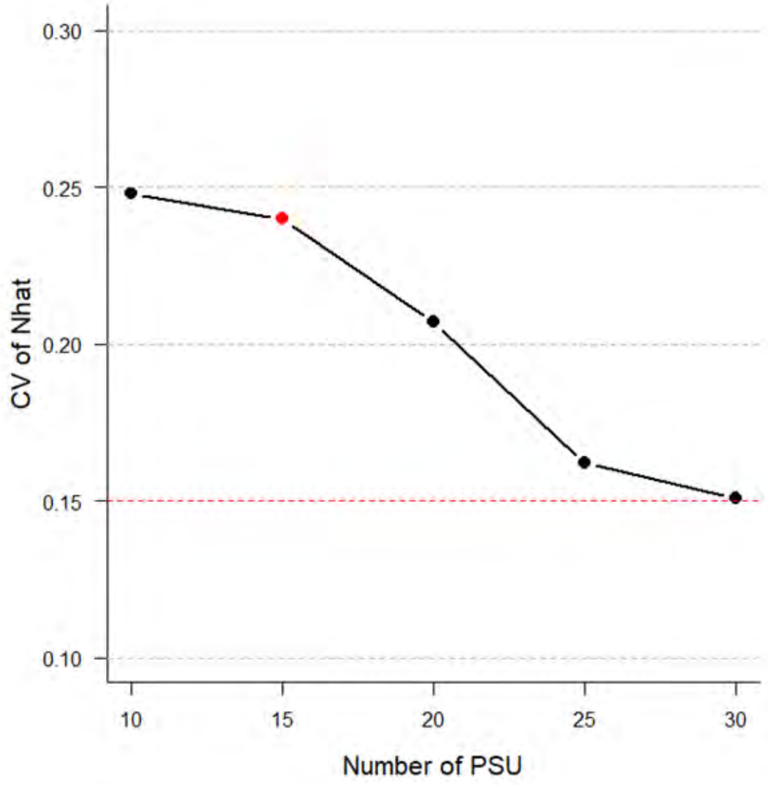

Figure 4. Precision of population estimates as a function of sample size (number of primary sampling units; PSU) from roadside distance-sampling surveys of white-tailed deer (*Odocoileus virginianus*) in southern Minnesota, spring 2018. Estimates are based on bootstrapping of PSUs and replicate surveys (survey date) where land cover was stratified (high, low) according to expected deer density and survey effort was allocated approximately equally within each stratum. The red circle denotes the current sample size. The red dashed horizontal line denotes a common target level of precision for management surveys.

EVALUATION OF GPS-SIZED EXPANDABLE RADIOCOLLARS DESIGNED FOR WHITE-TAILED DEER FAWNS

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SUMMARY OF FINDINGS

During May 2018–July 2019, we tested fit and function of 3 Global Positioning System (GPS)-sized expandable collar mock-up designs on newborn white-tailed deer (*Odocoileus virginianus*) fawns. We fitted 26 captive fawns with ear tags and collars (20 Vectronic Vertex, 3 Telonics TGW, 3 Telonics Recon) and ear-tagged 5 captive control fawns without collars. We collected neck measurements from fawns at birth and at approximately 6, 9, and 12 months of age. Additionally, we conducted observations of fawns to evaluate the potential effects of collars on behavior. The folds of all 6 Telonics mock-ups expanded prematurely by 75.8 ± 27.9 (SD) days resulting in extremely loose collars. Upon expansion, fawns were able to step through collars with their forelimbs resulting in collars positioned around the chest or waist. We observed one example in which premature expansion led to a dropped collar at just 80 days. On fawns ≥11 months of age, only 3 Vectronic Vertex mock-up collars dropped and none exhibited premature expansion. Neck measurements indicated newborn fawns would benefit from a smaller band circumference. For fawns through ≥10 months of age, we did not observe any collars being restrictive. Notable effects on fawn behavior included high-stepping during locomotion and erratic jumping, particularly when fawns were ≤1 month of age. Our results suggest that the GPS-sized expandable collars tested in this study would benefit from modifications before being deployed in the field. We recommend modifications to each design, such as an improved stitching patterns, alternative thread and elastic materials that facilitate a more gradual elastic expansion, decreased battery housing size and weight, and improved weight distribution of the electronic components.

INTRODUCTION

Knowledge of population parameters (e.g., sex ratio, age structure, survival, recruitment) informs decision-making for management of white-tailed deer (*Odocoileus virginianus*) populations (Jacobson et al. 1997, Keyser et al. 2005). Survival of white-tailed deer fawns is one of the most important factors influencing population growth. However, estimating survival of fawns to recruitment is logistically challenging using current very high frequency (VHF) collar technology (Moen et al. 1996, Rodgers et al. 1996, Bowman et al. 2000, Pusateri-Burroughs et al. 2006, Severud et al. 2015). Accurate estimation of survival requires capturing and collaring fawns soon after birth and intensively monitoring them for the first few months of life, as most mortalities occur during this time (e.g., predation, starvation, disease; Pusateri-Burroughs et al. 2006). The ideal design of radiocollars should ensure the welfare of the animal, minimize impacts on behavior, and

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maximize collar retention (Grovenburg et al. 2014). Expandable radiocollars are designed to stretch, open at folds, deteriorate, and drop off of animals to accommodate rapid neck growth throughout the first 12 months of life (Smith et al. 1998, Grovenburg et al. 2014). Multiple field studies have reported premature loss or failure of expandable radiocollars for deer fawns (Vreeland et al. 2004, Pusateri-Burroughs et al. 2006, Rohm et al. 2007, Hiller et al. 2008, Grovenburg et al. 2014, Obermoller et al. 2018). Ultimately, premature loss of collars reduces the sample size of studies, decreasing the power of inference.

Integration of GPS technology with expandable collar designs would allow researchers to more efficiently and effectively investigate survival and movements of white-tailed deer fawns (Bowman et al. 2000, McCance and Baydack 2017). The primary factor limiting use of GPS technology is the size and weight of the batteries required to support GPS transmitters (McCance and Baydack 2017). Additional modifications to expandable collar designs have been proposed to improve retention and facilitate a more gradual collar expansion (Diefenbach et al. 2003, Cherry et al. 2014, Grovenburg et al. 2014, Obermoller et al. 2018). Some of these modifications to expandable collar designs have been deployed in the field (Diefenbach et al. 2003, Bowman et al. 2014, Cherry et al. 2014, Grovenburg et al. 2014). However, testing of GPS-sized expandable collars in controlled settings is warranted before extensive deployment in field studies.

To date, GPS-sized expandable radio-collar designs have not been fitted to white-tailed deer fawns and have been deployed only within the last decade on other neonatal ungulates in the wild (moose [*Alces alces*], Severud et al. 2015; fallow deer [*Dama dama*], Kjellander et al. 2012) or in captivity (domestic horse [*Equus caballus*], Hampson et al. 2010). Using animals in a captive facility allows researchers to evaluate the efficacy of GPS-sized collars over time and observe collar fit and function, the overall health and well-being of animals, and the impact GPS-sized units may have on the behavior of fawns. Also, collars that become overly restrictive on captive fawns may be safely removed. Because most prior studies have deployed GPS-sized collars on species which give birth to considerably larger young than white-tailed deer (Hampson et al. 2010, Kjellander et al. 2012, Severud et al. 2015), it is logical to assume that the relatively larger offspring would better support the weight of a GPS collar. Therefore, testing of GPS-sized collars on white-tailed deer fawns in a controlled setting is warranted to ensure animal welfare for the duration of collar evaluations.

OBJECTIVES

1) Evaluate the efficacy of GPS-sized expandable radiocollars designed for white-tailed deer fawns
2) Determine the effects of GPS-sized radiocollars on the behavior of white-tailed deer fawns

METHODS

Study Site

We conducted our study at the Whitehall Deer Research Facility on the University of Georgia campus in Athens, GA. We held captive deer in 1-2-acre outdoor paddocks, each containing 12-14 adult does and their fawns. We provided all deer with pelleted feed, hay, and water *ad libitum.* The University of Georgia Institutional Animal Care and Use Committee approved all methods under Animal Use Proposal A2018 03-019-Y2-A0.

Animal Capture and Handling

We captured 31 fawns during May–July 2018. We captured, handled, and released each fawn within the first 24 hours after birth. We collected morphometric measurements of fawns (i.e., total body length, chest girth, hindfoot length, and neck circumference at upper, middle, and lower neck), affixed individually identifying ear tags in both ears (Allflex USA Inc., DFW Airport, Texas, USA), and fitted 26 fawns with GPS-sized expandable radio-collars. We fitted 20 fawns with Vectronic Vertex collars (Vectronic Aerospace GmbH, Berlin, Germany), 3 fawns with Telonics TGW collars (Telonics, Inc., Mesa, Arizona, USA) and 3 fawns with Telonics Recon collars (Telonics, Inc., Mesa, Arizona, USA). Five uncollared fawns served as experimental controls for our behavioral assessments. After handling, we immediately returned fawns to the outdoor paddocks where they were housed with their mothers until weaning.

Collar Fit and Function

We conducted assessments of collar fit and function 3 times per week on each collared fawn throughout the first 12 months of life. Using binoculars, we remotely observed fawns in outdoor paddocks, recorded scores of collar fit and body condition, and examined necks of fawns for signs of hair loss or lesions (Table 1). Additionally, we examined the expandable folds of each collar, recording the date at which each fold opened. To calculate collar retention, we recorded the date at which collars failed, dropped, or required removal to ensure animal welfare. At approximately 6, 9, and 12 months of age, we manually restrained all fawns to inspect the integrity of collars, evaluate the condition of fawns, and collect neck circumference measurements.

Vectronic Vertex

Vectronic Vertex (Vectronic GmbH; Figure 1) collars weighed about 138 g with the battery, VHF transmitter, and GPS transmitter located within a single large housing at the front of the collar (dimensions = 6.2 cm x 3.9 cm x 4.4 cm). The housing was attached to the collar using high-performance glue and 2 plastic cable ties. The antenna was coated with a thin protective layer of plastic and measured 26.5 cm with 20.5 cm of its full length exposed. The neck band was 4 cm wide, composed of nylon and rubber materials with an initial circumference of 22.3 cm. The neck band included 6 expansion folds (3 sections of 2 folds each) which were each 2 cm long. The section of folds furthest from the housing had a single stitch running through the middle of its folds, the middle section had 2 stitches (2.4 cm apart) through its folds, and the section closest to the housing had 2 stitches (3.1 cm apart) through its folds. Fully expanded, the circumference of the neck band was approximately 34 cm, not including stretch of the elastic band material.

Telonics TGW and Recon

Telonics TGW (Telonics, Inc.; Figure 2) and Telonics Recon (Telonics, Inc.; Figure 3) collars weighed about 140 g and 150 g, respectively. The primary differences between the TGW and the Recon designs were the battery housing material and the distribution of electronics. On the Telonics TGW, the battery, VHF transmitter, and GPS transmitter were located in 3 housings: a polymeric housing contained the battery (dimensions = 5.5 cm x 2.9 cm x 3.3 cm) and 2 plastic housings contained the VHF (dimensions = 1.8 cm x 0.5 cm x 1.8 cm) and GPS (dimensions = 2.8 cm x 0.9 cm x 2.8 cm) transmitters. On the Telonics Recon collar, the battery, VHF and GPS transmitters were distributed between 2 housings: an aluminum housing contained both the battery and the VHF transmitter (dimensions = 4.3 cm x 2.6 cm x 3.5 cm) and a plastic housing contained the GPS transmitter (dimensions = 2.8 cm x 0.9 cm x 2.8 cm). The battery housings of both the TGW and the Recon designs were attached to the collar using 4 screws and glue. The antenna of each Telonics model measured 26.8 cm with 20.9 cm of its full length exposed. The collar bands of both Telonics models were 3.8 cm-wide and composed of nylon and rubber (i.e., elastic portion) sewn to a 3.7 cm-wide strip of static polymer. The length of the static polymer material for each model was 15 cm and initial length of the elastic portion of each was 7.9 cm. Therefore, the initial band circumference (i.e., pre-expansion) of each Telonics model was 22.9 cm. The bands of both designs included 6 expansion folds (3 sections of 2 folds each) which were each 2 cm long. The section of folds closest to the housing had a single stitch running through the middle of its folds, the middle section had 2 stitches (1 cm apart) through its folds, and the section furthest from the housing had 4 stitches (0.5 cm apart) through its folds. Fully expanded, the circumference of the band was 34.9 cm, not including stretch of the elastic band material.

Fawn Behavior

We conducted focal observation sessions of each fawn to evaluate effects of collars on their behavior during the first 12 months of life. The frequency of our sessions decreased as fawns aged: ≤30 days of age, we aimed to conduct ≥1 morning and ≥1 evening session every week for each fawn; during 30-60 days of age, we conducted 1 morning session per week; during 60-200 days of age, we conducted 1 morning session every other week; and during 200-365 days of age, we observed each fawn during 1 morning focal session every 4 weeks. We conducted focal sessions from 4.5-m stationary observation platforms within paddocks during crepuscular sampling periods: in the evening from 06:00 to 10:00 EST or in the evening from 17:00 to 21:00 EST. Before conducting the first focal session of a sampling period, the observer sat quietly for 15 minutes to minimize impacts of human activity on the behavior of deer in the paddocks. During a focal session, we recorded the body orientation and behavior of the focal fawn each minute for 30 minutes. We recorded the overall body position, neck position, head position, and head tilt of the fawn based on a pre-determined scoring system (Figure 4). We coded all behaviors (e.g., vigilant, sleeping, foraging, suckling) in a preconstructed ethogram (Table 2). If the focal fawn moved out of sight for >5 minutes, we terminated the focal session and censored the data. We kept a running tally of fawn vocalizations, mother vocalizations, and the number of times a fawn exhibited any attention to its collar. We monitored fawn-mother proximity throughout focal sessions using a laser rangefinder and a compass. Every 5 minutes, the observer recorded a distance (m) and compass azimuth for the fawn and mother, then solved for the Euclidean distance using the Law of Cosines. We averaged all fawn-mother distances to obtain a mean fawn-doe proximity for each session. If the mother of the focal fawn was not visible, the observer did not record this information during the session.

RESULTS

For fawns ≥342 days old, no Vectronic collars exhibited premature expansion. Three Vectronic collars dropped from fawns in the outdoor paddocks, but none due to compromised stitching or elastic material. In one case, a fawn shed its collar at 256 days of age by snagging and tearing its collar (at the expandable material) on a perimeter fence while being moved through the facility. In the other 2 cases, fawns shed their collars at 265 and 276 days of age due to a large tear in the expandable material of the collars, likely caused by collars catching on fencing in outdoor paddocks. Overall, Vectronic collars accommodated the neck growth of fawns during the first year of life. As the necks of fawns grew larger, collars initially became tight (but not restrictive) around the lower neck. Added pressure to the expandable materials of the collar eventually caused 1-2 expansion folds to open, increasing the band circumference of the collar. The initial expansion, in most cases, resulted in some hair loss from the neck as loosened collars moved more freely around and along necks of fawns. This likely caused some minor discomfort; however, fawns quickly grew into the expanded collars.

The collar folds of all 6 Telonics mock-ups expanded prematurely by 75.8 ± 27.9 (mean ± SD) days. The ill-fitting collars moved freely along and around the necks of fawns, causing significant hair loss on the necks of all 6 fawns. Once the stitching of folds was compromised and the elastic material began to degrade, all 6 fawns were able to step through collars with their forelimbs. This displacement resulted in collars positioned around the chest or waist of fawns. For 2 fawns, premature collar drop occurred ≤20 days after the last fold expanded at approximately 80 days of age. For the other 4 fawns with Telonics mock-ups, we removed fully-expanded collars from the chest or waist at approximately 6 months of age when we restrained fawns to collect neck measurements.

We collected >200 hours of behavioral observations to date. Notable effects of collars on fawn behavior included high-stepping with forelimbs during locomotion, erratic jumping behavior, and several instances of forelimbs getting caught in ill-fitting collars. Each of these atypical behaviors were most prevalent in younger collared fawns, from newborn to approximately 3-4 weeks of age. High-stepping and erratic jumping behavior occurred most frequently in fawns fitted with Vectronic collars. We observed several instances of young fawns (<4 weeks old), fitted with both Telonics and Vectronic collars, getting their forelimbs caught in loose-fitting collars (pre-expansion). In these cases, a fawn's leg was restrained in the collar for 1-6 minutes.

DISCUSSION

Based on our preliminary results, we developed several recommendations for Telonics, Inc. and Vectronic Aerospace GmbH to improve their GPS-sized expandable radiocollars for white-tailed deer fawns. At this time, we cannot recommend the collar designs tested in our study for use in field studies. However, with modifications to each collar design and further testing in controlled settings, researchers may have access to viable GPS fawn collar options in the foreseeable future.

We recommended that Vectronic decrease the initial band circumference of their collar, improve weight distribution, and reduce size and weight of the battery housing in order to minimize effects on behavior. Poor weight distribution, paired with an initial collar band circumference that was larger than the necks of newborn fawns (Table 3), caused the battery housing to swing side to side as fawns moved forward. The high-stepping behavior appeared to be the fawns' attempts to step around the housing while it swung, to minimize contact with their forelimbs. The erratic jumping behavior observed in young collared fawns appeared to be a display of discomfort and frustration with cumbersome, loose-fitting collars. Decreasing the initial band circumference may alleviate some of these behavioral issues and lessen the chance of a fawn getting a forelimb caught in a loose-fitting collar. Weight of the Vectronic Vertex collar was focused at the front where a single large housing held all of the electronics. We believe that distributing electronics more evenly around the collar, perhaps in multiple smaller housings, would reduce the effects of collars on fawn behavior.

For the Telonics TGW and Recon collars, we recommended a slightly smaller initial band circumference to accommodate the smaller necks of newborn fawns (Table 3). A better-fitting collar may minimize issues with high-stepping and decrease the chance of a fawn getting a forelimb caught in a loose-fitting collar. The primary issues with the Telonics collar designs focused around the expandable material intended to accommodate rapid growth of fawns during the first year of life. Weak thread and elastic, as well as a potentially flawed stitching pattern, caused collars to expand and deteriorate at an accelerated rate. Exposure to environmental elements (e.g. sunlight, temperature, humidity, precipitation) likely played a role in the rapid expansion and degradation of collar materials. To increase collar retention and promote a more gradual elastic expansion, we recommended incorporating an improved stitching pattern and more durable thread and elastic material. Ideally, Telonics would utilize materials more similar to those on the expandable band of the Vectronic Vertex collars. We recommended the use of the polymeric-style housing (TGW) rather than the aluminum housing (Recon) because of lighter weight. When designing collars intended for newborn fawns, minimizing weight wherever possible is important. Therefore, we recommended that Telonics decrease collar weight to improve fit, reduce pressure on expandable materials, and prevent premature expansion.

The VHF technology of fawn collars currently used in field studies limits the abilities of researchers to efficiently estimate fawn survival, recruitment, movements, and habitat use. Enhancing our understanding of these factors would improve management of white-tailed deer populations (Moen et al. 1996, Rodgers et al. 1996, Bowman et al. 2000, Severud et al. 2015). Integrating GPS technology with expandable collar designs would provide researchers with more accurate information regarding the behavior of white-tailed deer (Bowman et al. 2000, McCance and Baydack 2017). With the primary limiting factor being the size and weight of batteries required to support GPS transmitters, we believe that further testing of GPS-sized collars in controlled settings is warranted before extensive deployment in field studies. The results of this study will provide important information to telemetry technology companies seeking to improve collar performance and produce less invasive collar designs.

Telonics, Inc. and Vectronic GmbH applied modifications to collar designs based on the preliminary results of this study and the recommendations we provided to each company. We will conduct additional testing of modified GPS-sized expandable collar designs during 2019-2020.

ACKNOWLEDGMENTS

We thank Bill Burger and Telonics, Inc., as well as Chris Kochanny and Vectronic Aerospace GmbH for their collaboration to alter collar designs. We thank volunteers Andrew Bray, Courtney Bunch, Kathy Clark, Randall Clark, Sarah Clark, Allison Colter, Tripp Colter, Seth Cook, Ryan Darsey, Jordan Dyal, Adam Edge, Matthew Hill, Kayla Reeves, Jacalyn Rosenberger, Mischa Schultz, Nikki Shellong, Eryn Watson, A. J. Wesner, Gina Wesner, and Jordan Youngmann for their assistance with fawn handling. Nicole Davros and Tonya Klinkner of Minnesota Department of Natural Resources provided administrative support. This project was funded by Minnesota Department of Natural Resources Section of Wildlife, Daniel B. Warnell School of Forestry and Natural Resources, and the Wildlife Restoration (Pittman-Robertson) Program.

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Table 1. System used for scoring collar fit and body condition of white-tailed deer (*Odocoileus virginianus*) fawns at Whitehall Deer Research Facility in Athens, GA, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

Collar fit score	Body condition score	Neck hair loss score	Neck lesions score
1 = Very loose	1 = Emaciated	0 = No hair loss	0 = No lesions
2 = Little loose	2 = Thin	1 = Coat thinning	1 = Single lesion ≤1cm
3 = Good fit	3 = Prime	2 = Single bald patch ≤1cm	2 = Multiple lesions ≤1cm
4 = Little tight	4 = Heavy	3 = Multiple bald patch(es) ≤1cm	3 = Single lesion >1cm
5 = Very tight	5 = Obese	4 = Bald patch(es) >1cm	4 = Multiple lesions >1cm

Table 2. Ethogram used for recording behavior of white-tailed deer (*Odocoileus virginianus*) fawns during focal sessions at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

Behavior	Code	Definition
Locomotion	L	Focal animal is moving forward (e.g., walking, running, jumping)
Foraging	F	Focal animal is eating or drinking (not suckling)
Suckling	S	Focal animal is actively suckling at adult doe
Grooming	GG	Focal animal is grooming another individual
Groomed	GD	Focal animal is being groomed by another individual
Grooming self	GS	Focal animal is grooming itself
Urogenital grooming	UG	Focal animal is being groomed by another at the urogenital region
Vigilant	V	Focal animal has eyes open and appears to be alert
Sleeping	SL	Focal animal has eyes closed and appears to be asleep
Undefined	U	Focal animal is exhibiting an undefined behavior
Out of sight	OS	Focal animal has moved out of sight

Table 3. Neck measurements (mean ± SD) collected from white-tailed deer (*Odocoileus virginianus*) fawns at 4 different ages at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

Fawns measured	Age (months)	Mean upper neck (cm)	Mean middle neck (cm)	Mean lower neck (cm)
51	0	16.4 ± 1.4	16.6 ± 1.4	18.5 ± 1.5
22	6	26.0 ± 2.1	26.7 ± 2.4	30.7 ± 3.2
20	9	29.7 ± 2.9	30.8 ± 2.8	38.4 ± 4.4
18	12	31.4 ± 2.7	31.0 ± 2.6	34.1 ± 3.2

Figure 1. Vectronic Vertex collar deployed on white-tailed deer (*Odocoileus virginianus*) fawns at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

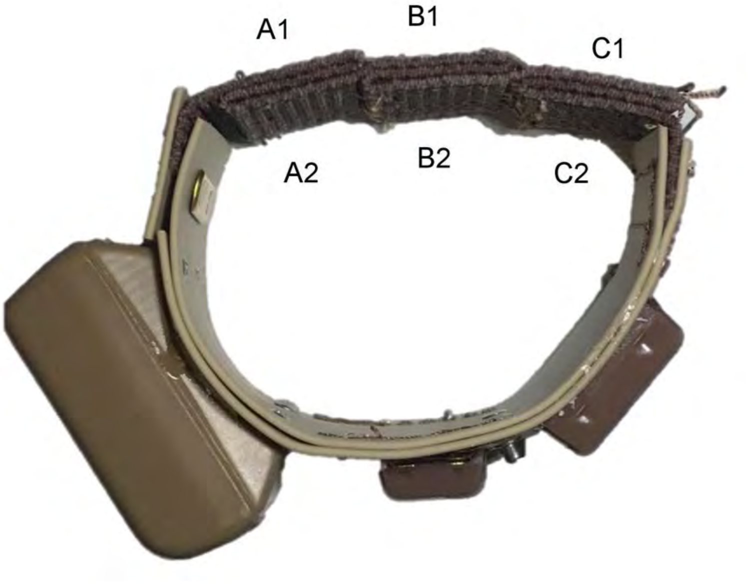

Figure 2. Telonics TGW collar deployed on white-tailed deer (*Odocoileus virginianus*) fawns at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

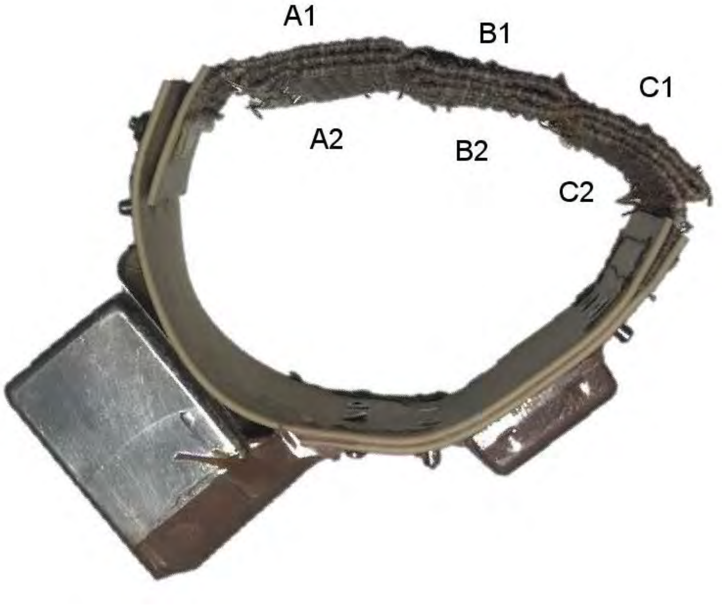

Figure 3. Telonics Recon collar deployed on white-tailed deer (*Odocoileus virginianus*) fawns at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

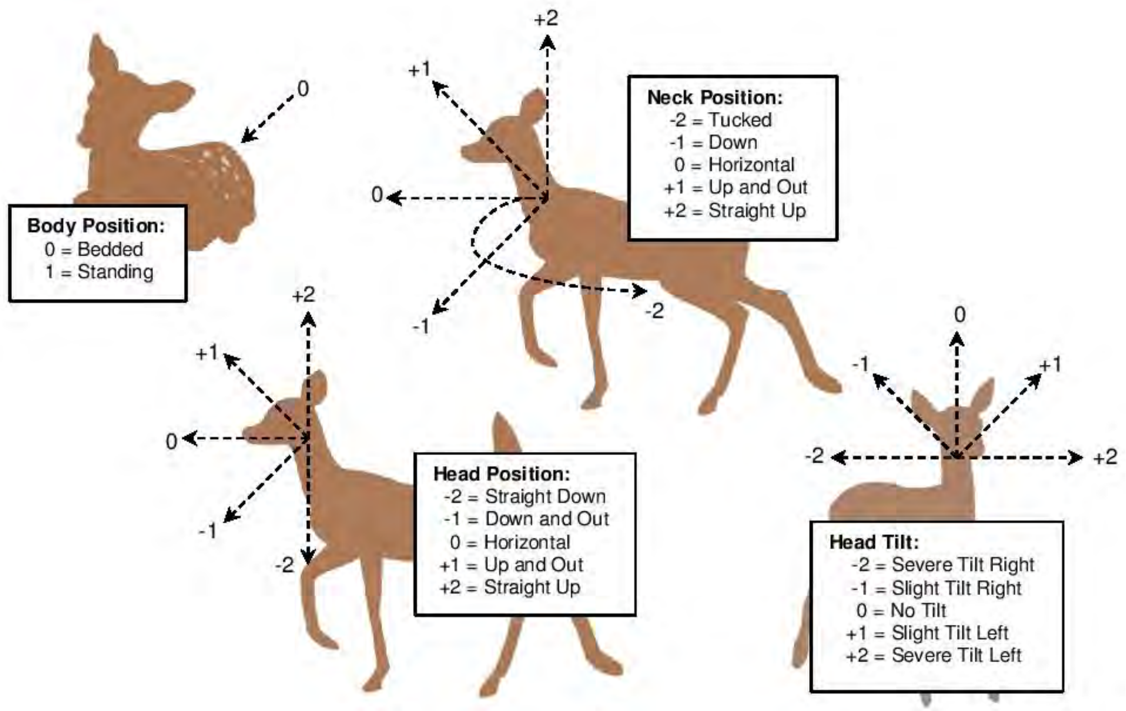

Figure 4. System for scoring body orientation during focal observation sessions of white-tailed deer (*Odocoileus virginianus*) fawns at Whitehall Deer Research Facility in Athens, Georgia, USA, during 2018-2019 for testing of Global Positioning System (GPS)-sized expandable radiocollar designs.

NESTING AND BROOD-REARING HABITAT SELECTION AND SURVIVAL RATES OF RING-NECKED PHEASANTS IN PRAIRIE RECONSTRUCTIONS IN SOUTHWEST MINNESOTA

Nicole M. Davros and Lindsey N. Messinger

SUMMARY OF FINDINGS

Ring-necked pheasant (*Phasianus colchicus*) responses to the amount of grassland on the landscape have been well documented, but we lack current information on the individual components of reproductive success (e.g., nest success, brood success, chick survival) that are driving pheasant population dynamics in Minnesota. From early spring 2015 through summer 2018, we radiocollared 164 hens on 2 study sites in southwest Minnesota and monitored them during nesting and brood-rearing each year. We collected vegetation data on nest site selection and survival data on hens, nests, broods, and individual chicks. In 2016 and 2017, we also collected data on brood-rearing habitat selection within grasslands. Video cameras were used to document nest predation events in 2015 and 2016. We describe preliminary findings within this report; final results are pending. Ultimately, our results will help us better understand the factors that limit reproductive success of pheasants so that natural resource managers can prioritize their grassland management and land acquisition strategies.

INTRODUCTION

Ring-necked pheasant (*Phasianus colchicus*) population dynamics are driven largely by variation in survival rates. Although adverse weather can impact survival rates in some years, predation is the primary cause of mortality for hens and their young (Peterson et al. 1988, Riley et al. 1998). Predator control efforts can help improve reproductive output over short time periods, but such efforts are economically and ecologically inappropriate over the long-term and at the landscape scale (Chesness et al. 1968, Riley and Schulz 2001). Management aimed at increasing pheasant populations has instead focused primarily on providing abundant nesting cover to minimize the effects of predation and maximize reproductive success. As acres enrolled in the Conservation Reserve Program (CRP) and similar cropland retirement programs decline in Minnesota, providing suitable habitat on public lands to sustain populations will become more critical for mediating the effects of predation and weather on pheasant population dynamics. However, the interaction between habitat and predation will no doubt remain. Thus, gaining new insights into the relationship between pheasant habitat selection and subsequent survival rates will be important for improving wildlife management strategies on publicly-owned lands.

Predation during the nesting season is a major factor affecting pheasant population dynamics. Nest predation is the leading cause of nest failure for many grassland-nesting birds, including pheasants (Chesness et al. 1968, Clark et al. 1999), and can limit productivity. Additionally, hens take only short recesses from incubating which puts them at greater risk to predation during nesting (Giudice and Ratti 2001, Riley and Schulz 2001). Management efforts aimed at increasing patch size and reducing edge effects are assumed to alleviate rates of predation on birds and their nests (e.g., Johnson and Temple 1990, Sample and Mossman 1997, Winter et al. 2000); however, the composition of the landscape surrounding a patch (Clark et al. 1999, Heske et al. 2001) and the vegetation within a patch (Klug et al. 2009, Lyons 2013, Fogarty et al. 2017) also play important roles in determining susceptibility to nest predation.

Advances in video camera technology have allowed better monitoring of bird nests and provided evidence that nest predator communities are more complex than previously thought (Pietz et al. 2012). In particular, the predators associated with nest depredation events can vary with the structure and diversity of nesting cover (e.g., percent cover of litter, forbs, or cool-season grasses; Klug et al. 2009, Lyons 2013). Thus, management actions attempting to mitigate the impact of predators may not necessarily reduce rates of nest predation but rather create a spatial or temporal shift in the nest predator community and susceptibility to nest predation (Benson et al. 2010, Thompson and Ribic 2012). Nest predator communities also vary across regions and habitats and results from studies of other species or in other states may not be entirely applicable to Minnesota’s pheasant population (Thompson and Ribic 2012, Benson et al. 2013). Understanding how management at the site level (e.g., vegetation structure, composition, and diversity) impacts the dynamics of nest predation is an important but as of yet unintegrated step in our ability to manage habitat for increased productivity of pheasants and other grassland birds (Jiménez and Conover 2001).

Chick survival is also a vital component of pheasant population dynamics but it remains poorly understood (Riley et al. 1998, Giudice and Ratti 2001). Assessing the causes of pheasant chick mortality has been difficult because many previous studies have relied on estimates of brood survival (e.g., the proportion of broods in which ≥1 chick survived to a certain age) rather than survival of individual chicks within a brood (e.g., Meyers et al. 1988, Matthews et al. 2012; but see Riley et al. 1998). Using brood survival estimates is likely unreliable because brood mixing can occur (Meyers et al. 1988; N. Davros, personal observations). Further, lack of data on individual chicks (e.g., body condition, cause of death) prevents us from understanding the role of different factors (e.g., exposure, food limitation, predation) that lead to variation in recruitment. Evidence that predation is the leading cause of chick mortality for grassland gamebirds in North America is well-established (e.g., Riley et al. 1998, Schole et al. 2011). Food availability has been implicated as an important factor explaining chick survival for many gamebird species in Europe (Green 1984, Hill 1985, Potts 2012); however, strong evidence that food is a major limiting factor for survival of chicks in North America is still lacking. Moreover, food availability and rates of predation likely interact in relation to vegetation structure and composition and confound conclusions from chick survival and food resource studies (Hill 1985). Finally, death from exposure has been shown to decrease chick survival rates, especially after periods with increased precipitation when chicks are still very young and unable to fully thermoregulate (Riley et al. 1998, Schole et al. 2011). Risk of exposure and starvation may interact to decrease chick survival, but few studies have been able to directly address this question (but see Riley et al. 1998). Therefore, additional data are needed to understand the interplay between these potential limiting factors on brood habitat selection and chick survival in different grasslands within Minnesota’s pheasant range.

Minnesota Department of Natural Resources (MNDNR) wildlife managers in the farmland region have indicated a need for more information on pheasant nesting, brood habitat suitability, and chick survival in relation to prairie reconstruction and management activities. Indeed, better understanding the factors that limit brood production and chick survival will help natural resource agencies prioritize their management strategies at both the local (e.g., forb interseeding) and landscape (e.g., acquisition priorities) levels in the face of reduced CRP acreages. Additionally, obtaining data on individual components of pheasant population dynamics will aid in future assessment of MNDNR management activities [e.g., Prairie Plan implementation (Minnesota Prairie Plan Working Group 2011), conservation grazing] and agricultural land use practices (e.g., cover crops, pesticide use) on Minnesota's pheasant population.

OBJECTIVES

Our overall objective was to evaluate the relative importance of within-patch diversity (e.g., sites dominated by cool-season grasses, warm-season grasses, and high diversity grass-forb mixtures) within Wildlife Management Area (WMA) project areas on pheasant productivity. Specifically, we:

1. Evaluated pheasant nest site selection and nest, brood, chick, and adult hen (hereafter, hen) survival in relation to grassland vegetation cover and composition.
2. Evaluated pheasant brood-rearing habitat selection in relation to grassland vegetation cover and composition.
3. Evaluated the relative importance of different factors (e.g., vegetation, predation, weather) on nest, brood, chick, and hen survival.

STUDY AREA

We conducted our study in the southwest region of Minnesota, a core region of the state's pheasant range. Topography ranged from flat to gently rolling. This region was intensively farmed, and corn and soybeans combined accounted for approximately 75% of the landscape (U.S. Department of Agriculture 2013a, U.S. Department of Agriculture 2013b). Grasslands, including those on private land [CRP, Reinvest in Minnesota (RIM), Conservation Reserve Enhancement Program (CREP), and Wetlands Reserve Program (WRP)] and public land [MNDNR Wildlife Management Areas (WMA) and U.S. Fish & Wildlife Service (USFWS) Waterfowl Production Areas (WPA)] accounted for 6.3% of the landscape in this region (Davros 2016).

For our study sites, we selected 2 WMA project areas representative of the grassland/wetland habitat complexes that are a goal of the Prairie Plan (Minnesota Prairie Plan Working Group 2011) and the Pheasant Summit Action Plan (MNDNR 2015). Each study site was about 9 mi^2 in size and contained extensive amounts of permanently protected grasslands. The Lamberton WMA study site (Redwood County) was a large, nearly contiguous WMA complex with >1,100 acres of permanently protected upland and wetland habitats. The Worthington Wells study site (Nobles County) had >1,500 acres of permanently protected habitat that spanned multiple WMAs, the Okabena-Ocheda Watershed District, and USFWS lands.

METHODS

Data Collection

We conducted our research during the 2015-2018 breeding seasons. Our 2015 pilot season allowed us to refine methods and protocols for the study's expansion during 2016-2018.

We captured hen pheasants in each study site during 6 time periods: 2 February – 15 April 2015, 7 October – 11 November 2015, 11 January – 29 April 2016, 26 September – 15 November 2016, 18 March – 14 April 2017, and 18 September – 11 October 2017 (hereafter referred to as spring 2015, fall 2015, spring 2016, fall 2016, spring 2017, and fall 2017, respectively). We used 2 capture techniques: baited walk-in traps and netting via nighttime spotlighting from a 6-wheel utility-task vehicle (UTV). We weighed each hen to the nearest 5.0 g, measured her right tarsus to the nearest 0.5 mm, and placed a uniquely numbered aluminum leg band on her right leg. We then fitted her with a 16.0-g necklace-style very high frequency (VHF) radiotransmitter with integrated mortality switch (Advanced Telemetry Systems (ATS), Isanti, MN) before releasing her at the site of capture.

We began radiotracking hens 3-5 times per week in late April each year to determine the onset of incubation. We assumed incubation had begun when a hen's radio signal was projected from the same location for several consecutive days. We flushed each hen from her nest once between incubation day 5-20 to determine clutch size and floated 3-5 eggs to estimate hatch date (Westerskov 1950, Carroll 1988). We marked the location of nests using a global positioning system (GPS) receiver. We also placed flagging ≤5 m from nests to aid relocation efforts. If a hen began making large daily movements prior to being flushed, we assumed nest failure and waited for the hen to resettle and begin incubating again before attempting another flush. We used the homing technique on radiocollars emitting a mortality signal to retrieve the collars and determine a fate. We used the presence and condition of any bodily remains and the condition of the radiocollar (e.g., teeth marks, feathers plucked, body intact but frozen, frayed collar, missing crimp) and nearby evidence (e.g., predator scat, den site) to determine survival status (e.g., mortality vs. unknown) and assign a potential cause of death (e.g., predation, human/machinery, weather), if applicable.

During 2015 and 2016 only, we placed miniature color video cameras (GE 45231 MicroCam Wired Color Camera, Louisville, KY) at a random subset of nests in an attempt to document nest predation events (Cox et al. 2012). We placed cameras during the same hen flushing event in which we floated eggs, and our total time near the nest was ≤20 min. We placed cameras 1-5 m away from the nest bowl at a height of approximately 0.3 m. Cameras had infrared light-emitting diodes (LEDs) to allow recording at night and were connected via a >20 m cable to digital video recorders (DVRs; Model MDVR14H, Super Circuits, Austin, TX) with SD memory cards for video storage and deep-cycle marine batteries for remote power. We stored the DVRs and battery equipment in waterproof containers located >20 m from nests. We later reviewed video footage and archived relevant video clips.

Near the estimated hatch date of known nests, we monitored hen activity 2-3 times daily to pinpoint a hatching event. We assumed hatching was occurring when a hen's signal fluctuated in intensity (Riley et al. 1998). We captured 1-3 chicks by hand between day 0-2 (day 0 = hatch day) once the hen and her brood had moved away from the nest. We used 2 techniques to capture chicks. The first technique involved flushing the hen from her brood and using a decoy and playback to call chicks to us while we hid in the grass. The second technique involved flushing the hen from her brood just before sunrise during brooding and capturing chicks by hand as they scattered. We never captured more than 50% of the brood at one time. We also never kept the hen away from her remaining brood for >30 minutes to minimize risk of hypothermia for the uncaptured chicks. We discontinued chick capture attempts for a particular brood if we were unsuccessful at capturing any chicks by the end of day 2.

We transported captured chicks in a small cooler or waist belt heated with hand-warmers to a nearby field truck for processing. We weighed each chick to the nearest 0.1 g and we measured tarsus length to the nearest 0.5 mm before suturing a 0.65-g backpack-style VHF radiotransmitter without mortality switch (ATS, Isanti, MN) to the chick's back (Burkepile et al. 2002, Dahlgren et al. 2010). Handling time lasted <5 min per chick and we returned all captured chicks to the hen within 30-60 min of capture. We followed the methods of Riley et al (1998) to return chicks to the hen.

We triangulated hens and their broods 2-3 times daily ≥3 times per week. Each bearing-coordinate pair was taken ≥100 m from target hens and their broods to reduce disturbance. We used specialized computer software (LOAS, Ecological Software Solutions LLC) to generate estimated locations from bearing-coordinate pairings. We monitored hens and their broods via triangulation through the first 4 weeks post-hatching. On day 14 and day 30, we flushed the hen just before sunrise to determine brood survival status and size.

To estimate individual chick survival, we listened for the signal of each radiomarked chick every 1-3 days in conjunction with monitoring the hen. We relied primarily on fluctuation in the chick's signal to determine survival status as backpack transmitters were too small to accommodate a mortality sensor. If the signal indicated that the chick was not moving, we used the homing technique to locate the transmitter and we searched the area for a carcass and any evidence for a cause of death.

We collected vegetation data at the nest site within 7 d of hatching for successful nests. For nests that failed, we also collected vegetation data at the nest site ≤7 d after the estimated hatch date. At each nest site, we visually estimated percent cover (Daubenmire 1959) of the upper canopy (i.e., grasses, forbs, standing dead vegetation, woody vegetation) using a 0.5 m^2 sampling quadrat. We estimated percent cover using 8 classes: 0%, 1-10%, 11-25%, 26-50%, 51-75%, 76-90%, 91-99%, and 100%. We estimated litter depth to the nearest cm and we counted the number of grass and forb species to determine species richness within the quadrat. We also recorded visual obstruction readings from 4 m away at a height of 1 m (VOR; Robel et al. 1970) in each of the 4 cardinal directions to determine the vertical density of vegetation to the nearest 0.5 dm around the nest. Finally, we recorded the maximum height of live and standing dead vegetation within 0.5 m of the Robel pole. We repeated these sampling efforts at 2 random locations within 15 m of the nest site.

To evaluate brood habitat selection within grasslands, we collected vegetation data at 5 brood locations estimated via triangulation and 10 random points outside but within 400 m of each brood's biweekly home range. We generated biweekly home ranges twice for each brood: one home range for the first 2 wk of age and a second home range for age 3-4 wk. For broods not surviving a 2-wk observation period, we generated home ranges and sampled vegetation if at least 1 chick from the brood survived for the first 7 d of the observation period. We used a Geographic Information System (GIS; ArcMap 10.2, ESRI, Redlands, CA) to estimate each biweekly home range using the minimum convex polygon tool. We also used GIS to generate the random points for sampling within 400 m of the biweekly home range. We restricted the selection of random points so that they were within the same habitat type (i.e., grassland). Any estimated brood locations or generated random points falling in habitat types other than grassland (e.g., row crop, wetlands, woodlots) were not sampled. We considered road right-of-ways in the grasslands category and included them in sampling efforts. We collected vegetation data within 7 d of the end of each biweekly interval. At each estimated brood point, we sampled 1 center point and 3 equidistant points (10 m north, 10 m southwest, and 10 m southeast) to capture the spatial variation associated with a brood location (i.e., hens and their broods were mobile and thus distributed around a point). We estimated percent upper canopy cover, litter depth, species richness, VOR, and maximum height of live and dead vegetation using the same methods described above for nest site selection. We separately estimated the cover of the ground layer (i.e., litter and bare ground) using the same 8 cover classes described above. We repeated this sampling scheme at each of the 10 random points associated with each brood's biweekly home range. If more than 50% of our brood triangulations were in habitat types other than grassland, we did not sample vegetation for that home range and associated random points. If a hen with a similar-aged brood had a home range that overlapped with another hen, we only sampled 5 additional random points associated with that second brood's home range.

Data Analyses

To date, we have conducted preliminary analyses on hen and nest survival. We also calculated basic descriptive statistics for nest site selection and brood habitat selection. Preliminary and final data analyses were ongoing at the time of this report; thus, not all analyses have been included here and not all research objectives are addressed below.

We conducted a preliminary survival analysis to evaluate hen survival during the nesting and brood-rearing phases (15 April – 15 October; hereafter, breeding season) only. We estimated cumulative survival using a Kaplan-Meier analysis approach in R v3.3.2 (R Core Team, 2016). The Kaplan-Meier approach assumes a known fate for each individual. As such, some individuals were censored at various intervals during the analysis period because they were reported missing and not relocated or their fate was otherwise unknown (e.g., slipped or failed radiocollar). We also excluded individuals with mortality or censor events occurring outside of the analysis period from this particular analysis.

Using 2015-2017 data, we conducted a preliminary nest survival analysis using the logistic-exposure method (Shaffer 2004) to estimate daily survival rate (DSR) of nests. We used a constant survival model (PROC GENMOD; SAS v9.3; SAS Institute, Cary, NC) which assumes that survival is constant across time and does not include any nest-specific explanatory variables.

RESULTS AND DISCUSSION

We radiocollared 164 hen pheasants during the 6 trapping periods across our 2 study sites from spring 2015 through fall 2017. We used baited walk-in traps during spring 2015 and spring 2016 only as they were not a productive capture technique. In 2016, we set cameras at the traps and found that pheasants were not motivated to use the bait when winter conditions were mild. Only 3 hens were captured using the walk-in traps (2% of total hen captures) during those 2 seasons whereas 161 hens (98%) were captured by spotlighting across all trapping seasons. We ended spotlighting capture efforts at the onset of the nesting season which limited our ability to increase sample sizes. In the future, we would consider using baited walk-in traps in late winter if weather conditions were severe enough to warrant this method. Winter conditions are considered severe for pheasants when snow is ≥6 inches deep and temperatures reach ≤0° F.

From 2015-2018, cumulative survival for hens during the breeding season (183-day period pooled across years) across study areas was 0.73 (*n* = 133; CI: 0.66-0.81; Figure 1). During the 4 breeding seasons, 59% of marked individuals (*n* = 79) suffered a known mortality event. Of these mortality events, 86% were attributed to predation events, 5% to human causes (specifically, vehicle collision and agricultural equipment), and <1% to research-related marking. Although the Kaplan-Meier survival method provides a quick estimate of hen survival, the strict assumptions of this model are likely inappropriate given our dataset. Because nearly 32% of individuals were censored during this analysis due to unknown fates (in particular, slipped radiocollars), our subsequent survival analyses may use methods that include expert knowledge to incorporate uncertainty in fate to refine survival estimates (Walsh et al. 2018).

We monitored 132 potential nesting attempts during the 2015-2018 seasons. Using data from 2015-2017 only, we evaluated 99 potential nesting attempts for nest survival analyses. We excluded 12 nests from analysis due to various reasons (e.g., hen abandoned after initial flushing event, nest area flooded prior to researcher's visit, and actual nest bowl was never found). The remaining 87 nests provided 657 observation intervals and 1,244 exposure days for analysis (Shaffer 2004). Overall, we found DSR was 0.9777 ± 0.20 (range: 0.9674-0.9848) which results in a 59.5% overall nest success rate when extrapolated to a 23-day incubation cycle. Our nest success rate is comparable to the rate found by Clark et al. (2008) in their study of a large, contiguous grassland landscape of northern Iowa. Although the relationship between landscape fragmentation and nest success cannot be automatically inferred across study areas or regions (Benson et al. 2013), our results lend further support to the idea that landscapes with large amounts of grassland cover can benefit pheasant nest productivity in the upper Midwest. We used a constant survival model for our initial analyses; however, our future analyses will examine the role of vegetation, spatial (e.g., distance to edge), and temporal (e.g. nest age, date, year) covariates on nest DSRs. In particular, time-specific patterns of nest survival have been documented in several duck and passerine species (Grant et al. 2005, Grant and Shaffer 2012) and such analyses are likely more appropriate for pheasants given their long nesting cycle and extended breeding seasons.

We placed video cameras on approximately 40% of nests in 2015 and 2016. Most hens were tolerant of cameras but a few hens did abandon their nests. However, these hens may have abandoned due to our other research activities (specifically, being flushed to float eggs too early in incubation) rather than the presence of the camera itself. We potentially captured 2 predation events on camera in 2016 but the video qualities were poor due to obstruction from vegetation and windy conditions which greatly reduced our ability to clearly view activity at the nests. One notable observation included a rooster visiting a hen at her nest almost daily during late incubation. Although we could clearly view all nests when cameras were first placed, the rapid growth of vegetation during the nesting cycle later impacted our ability to view nest contents or activities in the immediate area. Windy conditions often compounded our inability to review camera footage by causing vegetation to blow in front of the camera. In the future, we would consider using cameras to document nest predation events but we would alter our camera set-up (e.g., distance to nest, height of camera) to reduce the impact of vegetation and wind on the quality of the footage.

During 2015-2018, we collected vegetation data from 112 nest sites. Using data from 2015-2017 only, we calculated means and standard errors (SE) for 2 groups of comparisons: nest sites versus random points (*n* = 90; Table 1), and successful versus depredated nests (*n* = 52 and *n* = 17, respectively; Table 2). We included all nests regardless of nest fate (e.g., successful, depredated, abandoned, other failure) for the comparison of nest sites versus random points. Percent cover of forbs was similar between nest sites and random points ($\bar{x}$ ± SE = 16.4 ± 2.10% vs. 20.8 ± 2.40%, respectively; Table 1, Figure 2a); however, successful nests had more than twice as much forb cover compared to depredated nests (18.8 ± 3.01% vs. 8.3 ± 2.97%, respectively; Table 2, Figure 2b). The percent cover of standing dead vegetation was greater at nest sites compared to random points nearby (15.0 ± 1.36% vs. 9.9 ± 1.06%, respectively) but hatched nests and depredated nests had similar amounts of standing dead cover (15.1 ± 1.54% and 13.2 ± 4.20%, respectively). Vertical density of the vegetation, as measured by VOR, was similar between nest sites and random points (5.2 ± 0.21 dm vs. 5.3 ± 0.22 dm, respectively; Figure 3a) yet successful nests had greater vertical density compared to depredated nests (5.4 ± 0.26 dm vs. 4.5 ± 0.60 dm, respectively; Figure 3b). Although preliminary, these findings have practical implications for wildlife management activities. For example, previous MNDNR research found that neither burning nor mowing were successful management techniques for increasing forb diversity in warm-season grass-dominated stands (Davros et al. 2014), yet our results show that forb cover is positively associated with pheasant nest success. Therefore, managers should consider other options for increasing forb cover in prairie reconstructions to benefit pheasants. Additionally, we found that hens selected nest sites with more standing dead canopy cover, which is especially important for visual concealment early in the nesting season, and they were more successful when nests had greater vertical density, which is also assumed to provide visual concealment. However, recent studies suggest that increased structural heterogeneity of the vegetation may better conceal nests from olfactory-based nest predators (Fogarty et al. 2017, Fogarty et al. 2018). Management actions that create vegetation that provides both visual and olfactory concealment may be key to creating more productive nesting habitat.

We monitored 62 broods for survival during the 4 years of our study. We documented at least 1 hen who re-nested after losing her brood between 1-2 wk of age. Although rare, other researchers have also documented second brood attempts after early loses of first broods

(Dumke and Pils 1979; A. Annis and T.J. Fontaine, personal communication; T. Bogenschutz, personal communication; T. Lyons, personal communication).

We captured and radiomarked 84 chicks between day 0-2 during the 2015-2017 breeding seasons. During the 2016 and 2017 field seasons only, we recaptured 7 chicks between day 12-15 and replaced their 0.65 g transmitters with sutured,1.1-g or 1.3 g backpack-style transmitters (*n* = 6; ATS, Isanti, MN) or another 0.65 g transmitter (*n* = 1). Recapturing radiomarked chicks at this age was relatively easy and may be a viable option to replace lighter transmitters with heavier ones that have a longer battery life, thereby allowing monitoring of chicks beyond 4 wk of age in future work. We attributed known fates (*n* = 38) as follows: 32% died due to predation, 26% died from exposure, 24% of chicks died due to unknown causes, 11% died due to human-induced causes (specifically, agricultural operations, vehicle collisions, and researcher activities), and 8% survived beyond 30 days. Similar to our hen survival analyses, our future chick survival analyses may use methods that incorporate expert knowledge about uncertain fates to refine estimates (Walsh et al. 2018).

During 2016 and 2017 only, we collected vegetation data related to brood habitat selection for 40 broods. Our preliminary descriptive analyses suggest brood-rearing and nearby random locations were similar in vegetative structure and composition (Table 3, Figure 4). Brood and random locations were composed primarily of litter at the ground level ($\bar{x}$ ± SE = 80.2 ± 3.05% vs. 82.9 ± 2.40%, respectively; Table 3, Figure 4). Grasses were the predominant canopy cover at brood and random locations (55.3 ± 2.89% vs. 52.1 ± 2.32%, respectively) followed by forb cover (15.7 ± 1.57% vs. 17.0 ± 1.34%, respectively). Our future analyses will focus on a hierarchical modeling process to investigate brood-rearing habitat selection during each biweekly period (i.e., weeks 1-2 and weeks 3-4). If sample sizes allow, we will also evaluate differences in habitat selection between successful and unsuccessful broods.

The final results from this study will relate survival rates to nesting and brood-rearing habitat selection within prairie reconstructions. Ultimately, the data gathered will help managers better understand factors that may limit pheasant productivity so that they can prioritize their within-field management activities (e.g., grassland reconstruction efforts, forb interseeding) in an era of reduced grassland habitat acres on the landscape.

ACKNOWLEDGMENTS

We thank area wildlife staff, especially K. Kotts, W. Krueger, J. Markl, C. Netland, B. Schuna, D. Trauba, C. Vacek, and J. Zajac for their valuable discussions on issues and management efforts related to pheasant brood habitat. J. Giudice, V. St-Louis, M. Grund, M. Larson, J. Lawrence, L. Cornicelli, and G. Hoch reviewed earlier drafts of the research proposal and provided valuable input on the design and methodology of this study. T. Fontaine, D. Hoffman, T. Lyons, and S. Chiavacci provided great discussions and feedback on field methods and equipment. We also thank M. Adamek, B. Bermel, J. Bushman, S. Crosby, C. Fortier, K. Goebel, G. Gehring, M. Howell, Q. Huber-Heidorn, J. Johnson, J. Letlebo, N. Smetana, C. Reep, M. Rice, N. Schmidt, M. Rice, L. Welch, H. Witt, and J. Youngmann for their dedication to field work and data collection, and E. Andstedt, S. Buck, K. Deweese, S. Endres, B. Nosbush, R. Tebo, and M. Tuma for their volunteer efforts during hen trapping. Staff at the Nicollet wildlife office, the Windom wildlife office, the Talcot Lake wildlife office, and Blue Mounds State Park loaned fleet equipment during our hen capture efforts. This study would not be possible without the permission of the many landowners who have graciously allowed us onto their land as part of our monitoring efforts. This project was funded by the Federal Aid in Wildlife Restoration Act (Pittman-Robertson) and the MNDNR Section of Wildlife.

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Table 1. Descriptive statistics for vegetation surveys at sites used for nesting by ring-necked pheasant hens and nearby random points (≤15 m away) as a comparison in southwest Minnesota during the 2015-2017 breeding seasons.

	Nests sites (*n* = 90)		Random points (*n* = 90)	
	Mean	SE	Mean	SE
% Canopy cover				
Grasses	40.3	2.80	42.9	2.49
Forbs	16.4	2.10	20.8	2.40
Standing dead	15.0	1.36	9.9	1.06
Species richness				
Total	4.0	0.27	4.5	0.28
Grasses	1.8	0.10	1.9	0.09
Forbs	2.2	0.23	2.5	0.23
Maximum height (cm)				
Live vegetation	53.1	4.74	57.2	5.21
Dead vegetation	56.9	6.64	44.8	4.60
Litter depth (cm)	2.9	0.31	4.2	0.29
Vertical density (dm)[a]	5.2	0.21	5.3	0.22

[a] Vertical density is the average visual obstruction reading (VOR) as determined by using a Robel pole.

Table 2. Descriptive statistics for vegetation surveys at successful versus depredated nest sites of ring-necked pheasants in southwest Minnesota during the 2015-2017 breeding seasons.

	Successful nests (*n* = 52)		Depredated nests (*n* = 17)	
	Mean	SE	Mean	SE
% Canopy cover				
Grasses	41.1	3.42	43.0	8.35
Forbs	18.8	3.01	8.3	2.97
Standing dead	15.1	1.54	13.2	4.20
Species richness				
Total	4.3	0.37	3.8	0.63
Grasses	1.9	0.13	1.8	0.22
Forbs	2.4	0.30	1.9	0.62
Maximum height (cm)				
Live vegetation	53.1	6.56	63.7	7.73
Dead vegetation	55.2	8.42	50.2	12.80
Litter depth (cm)	2.5	0.28	3.0	0.47
Vertical density (dm)[a]	5.4	0.26	4.5	0.60

[a] Vertical density is the average visual obstruction reading (VOR) as determined by using a Robel pole.

Table 3. Descriptive statistics for vegetation surveys at locations used by ring-necked pheasant broods and nearby paired random locations[a] in southwest Minnesota during the 2016 and 2017 breeding seasons. Vegetation data were collected biweekly up to the first 4 weeks of brood rearing and was constrained to grassland habitats [e.g., Wildlife Management Areas (WMA), roadsides, Conservation Reserve Program (CRP) fields] only.

	Brood locations (*n* = 48)		Random locations (*n* = 52)	
	Mean	SE	Mean	SE
% Ground cover				
Litter	80.2	3.05	82.9	2.40
Bare ground	14.6	2.91	12.3	2.27
% Canopy cover				
Grasses	55.3	2.89	52.1	2.32
Forbs	15.7	1.57	17.0	1.34
Standing dead	6.7	0.59	7.4	0.53
Woody	0.2	0.09	0.2	0.07
Other	1.9	0.76	0.3	0.10
Species richness				
Grasses	2.2	0.10	2.3	0.08
Forbs	2.0	0.17	2.0	0.16
Maximum height (cm)				
Live vegetation	61.0	6.04	57.1	5.72
Dead vegetation	36.0	4.06	36.7	3.94
Litter depth (cm)	3.6	0.26	4.1	0.21
Vertical density (dm)[b]	5.5	0.24	5.7	0.19

[a] Paired random locations were outside of but within 400 m of a brood's biweekly home range [determined by the minimum convex polygon (MCP)].

[b] Vertical density is the average visual obstruction reading (VOR) as determined by using a Robel pole.

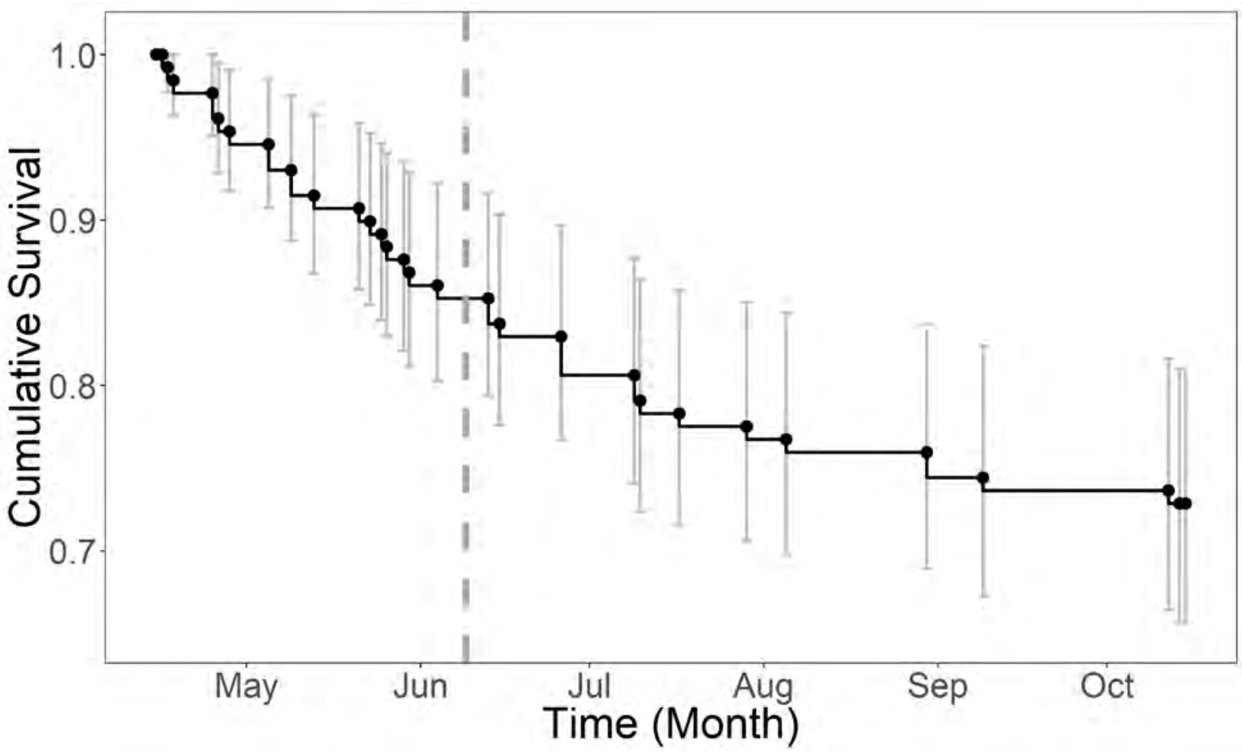

Figure 1. Cumulative survival of radiocollared ring-necked pheasant hens during the 2015-2018 breeding seasons (15 April – 15 October) in southwest Minnesota. Points represent survival estimates at intervals where mortality events took place. Error bars (vertical gray lines extending from each point) represent the upper and lower 95% confidence interval for each survival estimate. The 10-year average (2007-2016) for peak hatch of pheasant nests in Minnesota, as estimated by MNDNR's annual August roadside count surveys, is 12 June and is shown with the vertical gray dashed line.

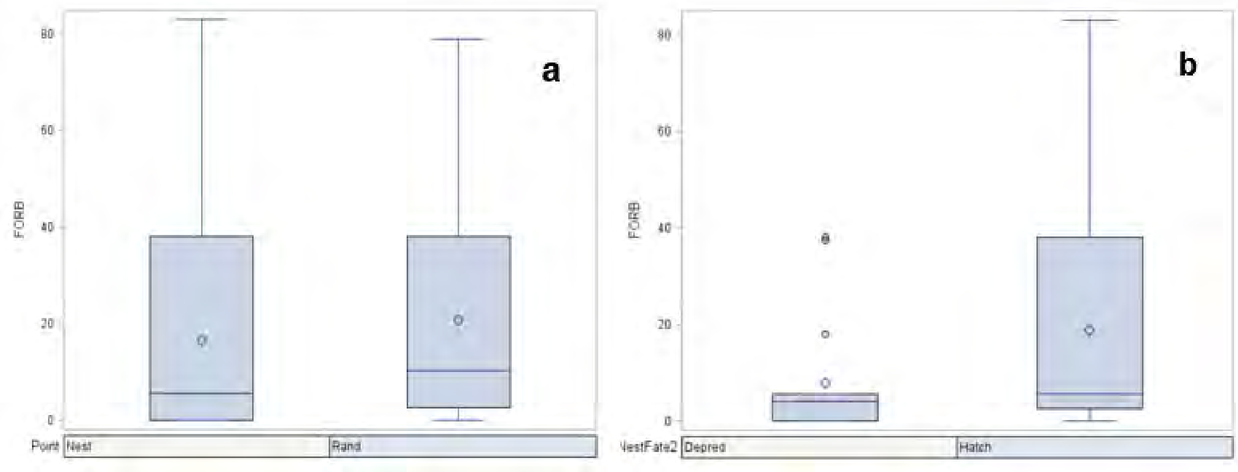

Figure 2. Box plot comparisons of the estimated percent cover of forbs at (a) nest sites versus nearby random points (<15 m away) and (b) depredated versus hatched nests of ring-necked pheasants in southwest Minnesota, 2015-2017.

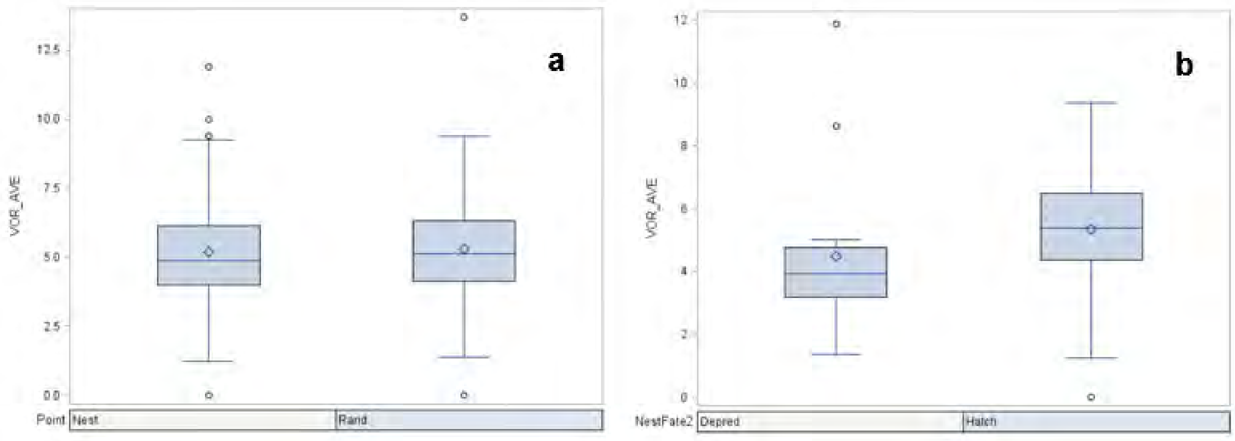

Figure 3. Box plot comparisons of vegetation vertical density, as measured by a visual obstruction reading (VOR) using a Robel pole, at (a) nest sites versus nearby random points (<15 m away) and (b) depredated versus hatched nests of ring-necked pheasants in southwest Minnesota, 2015-2017.

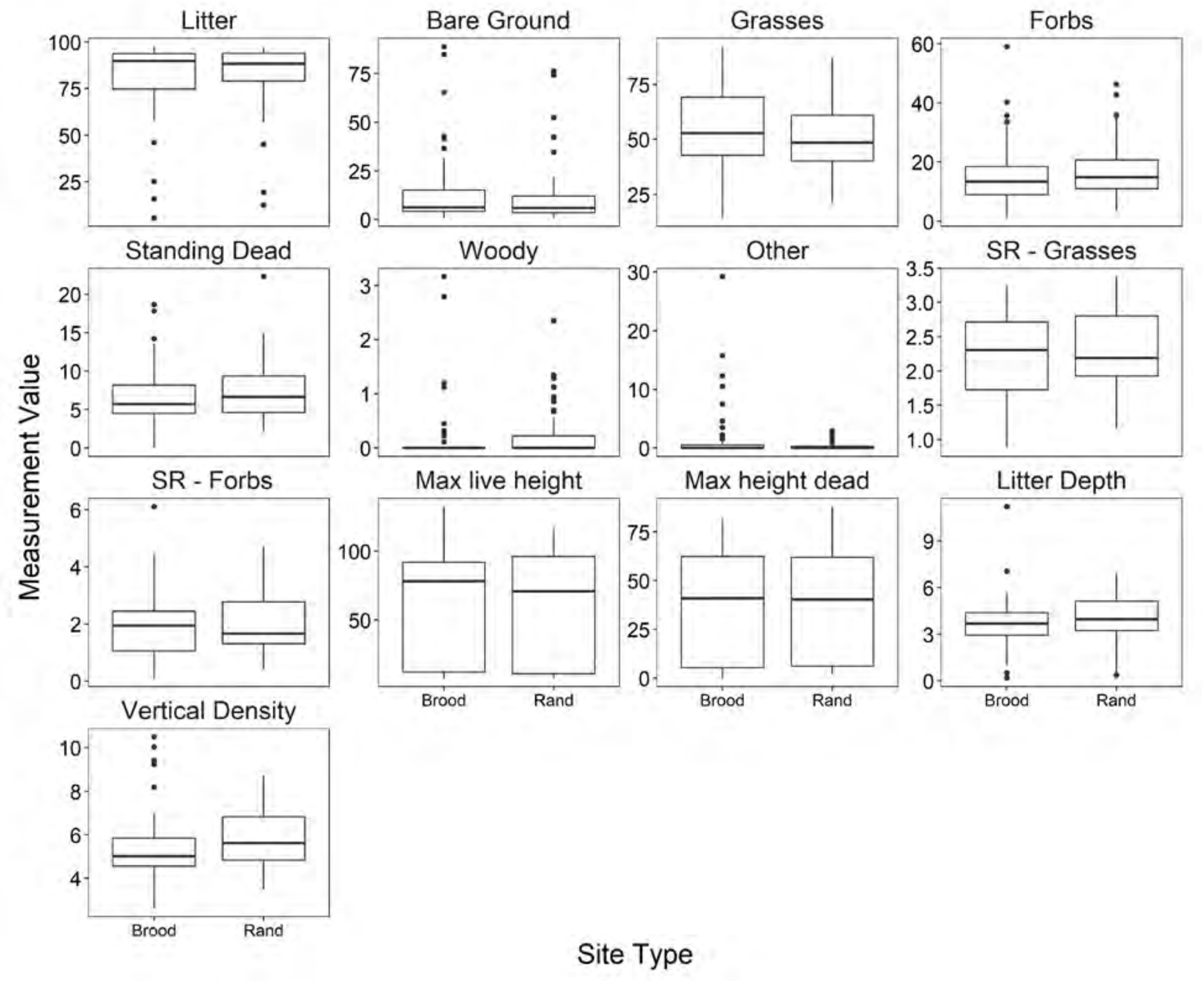

Figure 4. Box plot comparisons of vegetation structure and composition at brood-rearing versus nearby random locations in grasslands in southwest Minnesota, 2016-2017. Random locations were outside of but within 400 m of a brood's biweekly home range [determined by the minimum convex polygon (MCP)]. Vegetation data were collected biweekly up to the first 4 weeks of brood-rearing.

Human Dimensions Summaries

THE 2016 RUFFED GROUSE HUNTING SEASON IN MINNESOTA

Susan A. Schroeder

SUMMARY OF FINDINGS

The Minnesota Cooperative Fish and Wildlife Research Unit, at the University of Minnesota, in collaboration with the Minnesota Department of Natural Resources (MNDNR) conducted a survey of ruffed grouse (*Bonasa umbellus*) hunters following the 2016 season. A statewide survey of small game hunters pre-screened for grouse hunting participation was conducted to understand: grouse hunting behavior and intensity of participation, motivations for grouse hunting, and preferences for regulations and hunting experiences.

INTRODUCTION

Ruffed grouse populations undergo a periodic cycle (Zimmerman et al., 2008). During the 20th century grouse hunter participation in Minnesota has fluctuated with the population cycle. This variation has implications for managing grouse hunting experience, especially given the potential for wide differences in expectations and real experiences, and ultimately satisfaction (Schroeder et al., 2019). This study was undertaken with these concerns in mind.

Objectives

1. Evaluate grouse hunters' patterns and rates of participation and harvest
2. Understand grouse hunters' satisfaction with the activity
3. Determine factors that motivate participation and influence satisfaction
4. Understand differences in metro-county, and non-metro county hunters' beliefs, attitudes and behaviors about grouse hunting

METHODS

Sampling

The sample frame for this study was Minnesota residents aged 18 and over that possessed a small game license for the 2016 hunting seasons. A random sample of 8,000 hunters was selected from this population for participation; stratified by place of residence (4,000 residents of the 7-county area surrounding Minneapolis/Saint Paul, and 4,000 non-metropolitan county residents). Hunters were mailed a screening postcard to assess participation in grouse hunting. 1,332 postcards (668 metropolitan and 644 non-metropolitan) were returned from grouse hunters willing to participate.

Data Collection

Data were collected following the recommendations of Dillman (2000). Participants were sent a survey packet containing a personalized cover letter, questionnaire, and a self-addressed business-reply envelope. Three full survey packets were distributed to non-respondents at roughly 3-week intervals. Following distribution of the third survey packet, a shortened survey was sent to non-respondents to assess non-response bias.

Analysis

Descriptive statistics were generated for all variables, and mean and proportion differences in outcomes were estimated between metro and non-metro hunters on variables of interest using paired sample *t*-tests and chi-squared tests respectively.

RESULTS

Response Rates

Of the 668 surveys sent to metropolitan county hunters, 461 were returned. This represents a response rate of 69% Of the 644 surveys sent to non-metropolitan county hunters, 426 were returned for a response rate of 74%. Thirty-eight and 52 short surveys used to gauge non-response bias were returned from metro and non-metro hunters respectively.

Grouse Hunting Participation and Harvest

Grouse hunters spent on average 9.3 days afield during the 2016 season. Over the course of the season, hunters harvested roughly 4 grouse, or 0.43 grouse per person/per day. Grouse hunters, on average, had been hunting grouse for 36 years. Respondents took an average of 6.4 trips during the 2016 season to hunt grouse. Differences were observed between metro and non-metro hunters in trip length, with metro hunters traveling an average of 151 miles per trip and non-metro hunters traveling 88 miles per trip.

Satisfaction, Crowding and Change in Quality of Hunting over Time

Overall, hunters were satisfied with grouse hunting in Minnesota with a majority indicating that they were either “slightly,” “moderately,” or “very” satisfied. However, differences were observed when examining various aspects of satisfaction with grouse hunting access, the number of grouse flushed, and quality of grouse habitat. For instance, a smaller proportion of respondents indicated that they were either “very” or “moderately” satisfied with the number of grouse they flushed during the 2016 season.

The majority of respondents did not experience crowding while hunting grouse in 2016, with 80% indicating either “not at all” or “somewhat” when asked to evaluate their perceptions of the number of other hunters or people they encountered while hunting. Among those hunters that did experience interference, ATVs and other off-highway vehicles were the most commonly reported source.

When asked to compare the quality of grouse hunting in Minnesota in 2016 to 5 and 10 years ago, most hunters felt that grouse hunting in 2016 was about the same. However, on average, hunters felt that the number of grouse they flushed and the amount of crowding they experienced was slightly worse today when compared to the past.

Grouse Management and Trust in MNDNR

Hunters were asked about their perceptions of bag limits for grouse management. Items measured hunters’ beliefs that bag limits should be set based on grouse biology (biology), what is fair (fairness), what is desirable (social desirability), what is right (injunctive), and what other hunters commonly think bag limits should be (descriptive). On average, hunters agreed with items measuring these beliefs about grouse bag limits, with the exception of social desirability. Differences were also found between the ratings of metro and non-metro hunters, where metro hunters more strongly agreed that grouse bag limits should be set based on the species biology and what is right.

On average, hunters agreed with statements expressing aspects of trust in MNDNR including a belief that the MNDNR “does a good job managing ruffed grouse,” “can be trusted to make decisions about ruffed grouse,” and that the MNDNR “listens to grouse hunters’ concerns,” among others. However, agreement on these items was only slightly positive. (~3 on a 5 point scale).

Importance of Grouse Hunting, Motivations, and Preferences

Very few hunters reported that grouse hunting was either their most important recreational activities, or their least important recreational activity. The vast majority indicated (>80%) that grouse hunting was either less important than their other recreational activities or no more important than their other recreational activities. Slightly greater than 10% of respondents reported that grouse hunting was one of their most important recreational activities.

Respondents were asked how important a variety of experiences were to them as determinants of their satisfaction with grouse hunting. On average, the most important experiences for grouse hunters were; enjoying nature and the outdoors, reducing tension and stress, and seeing grouse in the field. Hunters placed less importance on harvest related experiences including; bagging enough grouse to share, bagging enough grouse to freeze for the future, and bagging a daily limit. Hunters reported enjoying nature and stress reduction. Harvest orientation was further examined, with around 85% of respondents agreeing strongly that “a grouse hunting trip can be enjoyable even if no grouse are bagged” and one-third agreeing that “the more grouse I bag the happier I am.”

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SURVEY OF SOUTHEAST MINNESOTA DEER HUNTERS

Susan A. Schroeder[1]; Adam C. Landon[2]; Louis Cornicelli[2]; Leslie McInenly[2]; Todd Froberg[2]

INTRODUCTION

The Section of Wildlife in the Minnesota Department of Natural Resources, in cooperation with the Minnesota Cooperative Fish and Wildlife Research Unit at the University of Minnesota, conducted a survey of southeast Minnesota deer hunters' values, beliefs, attitudes and behaviors associated with deer hunting, chronic wasting disease (CWD), and CWD management during the fall and winter of 2018-19. The primary objectives of this study were to 1) assess the acceptability of current and potential management actions and programs proposed to address the threat of CWD in the region, 2) evaluate support for financial and non-financial incentives to maintain participation and improve hunting access in the CWD management zone, and 3) determine hunters' perceived risks from CWD.

METHODS

Sampling

Two strata were identified as the sample frame for the study: 1) adult firearms deer hunters who indicated at the time of license purchase that the southeast CWD management zone (deer permit area 603) was their primary deer hunting area, and 2) adult firearms deer hunters who indicated that one of the 300-series deer permit areas surrounding the southeast CWD management zone was their primary deer hunting area. A census of 603 hunters was conducted (*n*=2,195), and a random sample of 2,800 firearms deer hunters was drawn from the hunters who indicated that they would hunt in 300-series permit areas.

Data Collection

Data were collected following the recommendations of Dillman (2000). Participants were sent a survey packet containing a personalized cover letter, questionnaire and a self-addressed business-reply envelope. Three full survey packets were distributed to non-respondents at roughly 3-week intervals. Following distribution of the third survey packet, a shortened survey was sent to individuals who had not yet responded to assess non-response bias.

RESULTS

Response Rates

A total of 2,086 usable questionnaires were returned (DPA 603 = 880, and 300 series = 1,206). This represents an effective response rate of 42% after correcting for non-deliverable addresses and invalid responses. Results presented here are from a non-weighted combined sample.

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[1] Minnesota Cooperative Fish and Wildlife Research Unit

[2] Minnesota Department of Natural Resources

Support for Potential and Existing Strategies to Manage CWD

Hunter support for existing and potential regulatory options, season structures, and bag limits was quite varied. The current use of professional culling as a tool to manage CWD was opposed by slightly more than 60% of hunters. Alternatively, over 80% of hunters were either neutral or supportive of allowing the take of one buck per season in the CWD management zone. There was also strong support for expanding the venison donation program, banning recreational deer feeding, and prohibiting carcass movement as CWD management tools. So-called earn-a-buck strategies where hunters would be required to harvest an antlerless deer before taking a buck were opposed by most hunters surveyed. Other strategies saw a more even split between support and opposition (Table 1).

Support for Financial and Non-Financial Incentives for Harvest and Access

Hunters, on average, did not support the use of financial incentives to motivate participation, access or harvest in the CWD management zone. Roughly 50% of hunters opposed paying landowners for hunting access, paying landowners for CWD positive deer shot on their property, or paying landowners for every deer shot on their property. Similarly, a near majority of hunters opposed offering financial payments to hunters that shoot CWD positive deer. Hunters were more evenly split between opposition and support on the use of non-financial incentives that motivate deer harvest in the CWD management zone including offering free deer licenses or extra tags to hunters that shoot CWD positive deer (Table 2).

Perceived Risks from CWD

Hunters perceived a variety of risks from CWD. Of note, hunters were particularly sensitive to the potential for deer and elk farms to spread CWD, with 33.5% indicating that they are “extremely concerned.” Hunters, on average, were not particularly concerned about the risks that CWD poses to land values. Roughly 21% of hunters reported that they were either not at all concerned or extremely concerned that CWD will cause disease in them personally (Table 3).

LITERATURE CITED

Dillman, D. (2000). Mail and internet surveys: The tailored design method. New York: Wiley & Sons, Inc.

Table 1. SE Minnesota hunters' support for current and potential CWD regulations and strategies per the survey on the fall and winter of 2018-19.

	Oppose (%)	Neutral (%)	Support (%)
Professional culling of deer in local areas after the deer season.	61.4	20.7	17.9
Allow hunters to take unlimited bucks in the disease management zone.	46.7	19.6	33.6
Allow hunters to take a buck per season (archery, firearms, muzzleloader).	19.0	20.2	60.8
Implement a ban on recreational deer feeding.	13.3	22.2	64.5
Expanded venison donation program.	5.3	29.0	65.6
Expand the size of the disease management zone.	27.4	39.6	33.1
Hunters must take one antlerless deer before taking a buck (earn-a-buck).	55.0	20.9	24.1
Hunters must take more than one antlerless deer before taking a buck.	74.1	18.2	7.8
Series of short (3-day) post-muzzleloader season hunts.	31.3	39.4	29.3
Longer early antlerless season (currently 4 days).	29.0	40.6	30.3
Longer youth season.	21.7	35.4	42.9
Muzzleloader weekend in October.	37.1	34.3	28.5
Free permits to landowners to use before the regular deer season.	28.6	24.6	46.8
Prohibit export of all carcasses from the CWD management zone, including fawns.	21.4	28.6	50.0
Hunters having the ability to earn extra buck tags by taking multiple does (2 or 3 does earns an extra buck).	46.6	25.4	28.0
Reduce free landowner license acreage requirements (currently 80 acres).	22.0	38.4	39.7
Create hunter/landowner database to connect hunters to landowners who allow hunting access.	18.5	33.9	47.5

Table 2. SE Minnesota hunters' support for financial and non-financial incentives per the survey on the fall and winter of 2018-19.

	Oppose (%)	Neutral (%)	Support (%)
Pay landowners to allow people to hunt their property.	48.4	26.3	25.3
Pay landowners for CWD positive deer killed on their property.	47.4	27.3	25.3
Pay hunters for CWD positive deer they kill.	48.6	25.9	25.5
Expanded venison donation program where meat is distributed to local communities.	8.0	26.8	65.2
Lifetime deer hunting license for killing a CWD positive deer.	51.4	25.0	23.6
For each deer killed in the CWD zone, receive one lottery ticket for entry into a drawing to win equipment.	41.8	25.0	33.3
Pay landowners for every deer shot on their property.	57.2	25.1	17.8
Work with Legislature to develop program to give tax breaks to landowners who allow public hunting.	32.3	25.2	42.5
Financial contribution to a charity of your choice for killing a CWD positive deer.	42.5	35.6	21.9
Provide hunters with an extra buck tag for killing a CWD positive buck.	31.6	22.2	46.3
Provide hunters with an extra buck tag for killing any CWD positive deer.	35.3	24.7	40.0
Provide free deer license for the following year if hunter provides a CWD positive deer.	31.6	24.4	44.1

Table 3. SE Minnesota hunters perceived risks from CWD per the survey on the fall and winter of 2018-19.

How concerned are you that CWD will…	Percent response Not at all concerned 1	2	3	4	5	6	Extremely concerned 7
…spread throughout the deer population <u>where you hunt</u>.	8.7	11.1	10.5	15.4	18.2	15.9	20.3
…dramatically reduce the deer population <u>where you hunt</u>.	10.3	10.6	12.0	14.9	15.1	17.2	19.9
…affect the health of the deer population <u>where you hunt</u>.	7.8	10.1	10.2	14.9	17.7	18.8	20.6
…have the potential to kill the entire deer population <u>where you hunt</u>.	19.8	13.3	11.7	11.8	11.0	11.7	20.6
…threaten your deer hunting opportunity.	10.8	9.3	10.1	14.0	14.8	16.8	24.2
…threaten the future of deer hunting for your children and grandchildren.	10.7	8.6	8.6	11.4	12.9	18.4	29.4
…affect the future existence of deer on the Minnesota landscape.	13.5	12.5	9.6	12.9	14.3	15.0	22.3
…spread to livestock.	18.9	15.7	10.8	15.0	11.2	11.8	16.6
…have economic impacts on businesses that depend on deer hunting.	13.2	12.0	11.8	18.7	16.3	15.1	13.1
…lead to declining land values.	31.0	17.0	12.0	15.3	8.1	7.9	8.8
…spread because of deer and elk farms.	10.0	8.9	7.3	12.5	11.6	16.2	33.5
…cause your family to stop eating deer meat.	18.4	14.3	11.1	17.4	12.3	12.6	14.0
…cause you to have concerns about eating deer meat.	14.5	12.3	9.4	16.4	14.6	16.4	16.4
…threaten your personal health or the health of my family.	16.8	14.7	9.1	13.7	12.4	12.3	21.1
…cause disease in humans.	17.6	15.8	8.3	12.7	11.2	11.6	22.9
…cause disease in you personally.	21.7	15.8	8.5	11.7	9.6	11.4	21.4

THE 2017 WATERFOWL HUNTING SEASON IN MINNESOTA: A STUDY OF HUNTERS' OPINIONS AND ACTIVITIES

Susan A. Schroeder and Steven D. Cordts

SUMMARY OF FINDINGS

The Section of Wildlife in the Minnesota Department of Natural Resources in collaboration with the Cooperative Fish and Wildlife Research Unit at the University of Minnesota conducted a survey of waterfowl hunters' participation and harvest, motivations, involvement, satisfaction, and opinions about bag limits, season dates, and youth waterfowl opportunities. This study was conducted in an effort to obtain information on hunters' preferences relative to management alternatives associated with season dates and bag limits for waterfowl in the state. Variation in satisfaction with waterfowl hunting opportunity by geographic region and species was also of substantive interest in the study; as were preferences for potential changes to sandhill crane hunting regulations.

Ducks were revealed to be the most commonly sought waterfowl among Minnesota hunters. More hunters hunted the Central zone than the North or South zones during the 2017 season. The majority of duck hunters (71%) and goose hunters (63%) were satisfied with their hunting experience. Hunters were supportive of the Youth Waterfowl Hunting Day. Roughly two-thirds of hunters indicated that the 6-duck overall, 2-hen mallard, and 3-wood duck bag limits were "about right." A minority in each instance believed that the limits were either too high, too low, or did not have an opinion. Around 10% of respondents reported that waterfowl hunting was their "most important recreational activity." Social aspects of waterfowling were found to be most important attributes of involvement in the activity. Hunters were supportive of allowing open water hunting on large water bodies, but opposed restricting the use of motorized decoys. North zone duck hunters preferred a straight 60 day season, compared to Central and South. While South zone hunters preferred a split season with a later end date, compared to Central and North. Hunters, on average, supported amendments to Sandhill crane hunting in Minnesota, including an expanded hunt boundary.

INTRODUCTION

Obtaining information on hunters' preferences for aspects of hunting activities can inform season setting, regulations, and resource management decisions that optimize opportunities for hunters to have preferred experiences (Schroeder et al., 2019; Schroeder et al., 2006; Vaske, Fedler, and Graefe, 1986). The Minnesota Department of Natural Resources (MNDNR) regularly conducts assessments of the attitudes, opinions, and behaviors of hunters and other stakeholders to achieve these goals. The number of registered waterfowl hunters in the state has fallen by one-third since 2000. Waterfowl hunting generates substantive economic (Grado et al., 2011) and social benefits (Arnett and Southwick, 2015), in addition to contributing to wildlife conservation (Vrtiska et al., 2013). Therefore, declining waterfowl hunter numbers are a cause for concern for MNDNR as the trustee of waterfowl in the state, and the agent responsible for facilitating citizens' connection to wildlife resources.

Objectives

This study was undertaken with the following objectives:

1. Describe hunter effort in Minnesota in 2017 including: species and seasons hunted; number of days hunted; effort during weekdays, weekends, and opening weekend; and management regions hunted.
2. Describe hunting satisfaction with waterfowl (duck and goose) hunting in Minnesota in 2017.
3. Examine the importance of various experiences preferences (related to bagging waterfowl) and actualization of those experiences in waterfowl hunting during 2017.
4. Examine importance of and involvement in waterfowl hunting to Minnesota and intentions to participate in the future.
5. Determine Minnesota waterfowl hunters' opinions concerning bag limits and other management strategies for maintaining waterfowl numbers;
6. Determine Minnesota waterfowl hunters' opinions on season dates and split seasons.
7. Determine Minnesota waterfowl hunters' opinions on changes to sandhill crane hunting.
8. Determine Minnesota waterfowl hunters' support for and participation in Youth Waterfowl Hunting Day.
9. Determine demographics of waterfowl hunters in Minnesota.
10. Examine trends in waterfowl hunters' characteristics and opinions over time.

METHODS

Sampling

The primary sample frame consisted of Minnesota residents aged 18 and older that purchased a state waterfowl stamp for the 2017 season. 3,600 individuals were randomly selected to participate, stratified by 4 geographic regions corresponding to waterfowl hunt zones (North, Central, South) and the Minneapolis-Saint Paul metropolitan region, by zip code of residence (900 hunters per strata). Additional samples of 900 pictorial waterfowl stamp purchasers, and 900 Sandhill crane permit holders were randomly selected for participation.

Data Collection

Data were collected using a repeat-mail design. Participants were sent a survey packet with a cover letter, questionnaire, and self-addressed, postage-paid return envelope. Mailing lists were adjusted between rounds. Non-respondents were sent 3 additional survey packets. In the fourth mailing, participants were also provided a $1 incentive in an effort to increase response rate. A 1 page survey was sent to non-respondents after the fourth round in order to quantify non-response bias. Data collection occurred between January and May, 2018.

Data Entry and Analysis

Data were entered and analysis was conducted using the Statistical Program for the Social Sciences (SPSS for Windows 21). Basic descriptive statistics and frequencies were calculated for the statewide, pictorial, and crane permit holder samples. The 3 research strata and regional results were compared using one-way analysis of variance (ANOVA) and cross-tabulations. Frequency weights were applied to calculate statewide estimates based on the proportion of hunters that reside in each of the geographic strata in the sample.

RESULTS

Response Rate and Respondent Demographic Characteristics

Collection efforts yielded an overall response rate of 48.6% 134 of the original 3,600 addresses were undeliverable or otherwise invalid resulting in an adjusted sample of 3,466 individuals. Response rates ranged from 50.3% to 45.8% by strata. A non-response bias was detected when respondents were compared to the population of hunters; where respondents tended to be older. However, when analytical weights were applied to the data no meaningful differences in estimates were detected when compared to the unweighted results. Therefore, the data were not weighted by age.

Respondents were on average 44.5 years old, and had lived in Minnesota for an average of 41 years. On average, respondents possessed an annual household income of approximately $113,000. Roughly 39% of respondents had completed a 4-year college degree or greater.

Participation and Waterfowl Hunting Activities

The vast majority (87%) of waterfowl permit holders reported that they went waterfowl hunting during the 2017 season. Of those that reported hunting, 94% hunted ducks and a majority of goose hunters hunted Canada Geese during the regular season. Early season goose hunting, other geese, and sandhill crane were less commonly sought targets. Average season bags for ducks, Canada geese, and other geese were 11.0, 6.4, and 1.6 respectively.

Satisfaction

Overall, (71%) respondents were satisfied with their duck hunting experience. Forty-nine percent of duck hunters reported that they were satisfied with harvest, and more than 50% were satisfied with duck hunting regulations. A slightly lower plurality of goose hunters (63%) reported that they were satisfied overall, while a similar pattern to duck hunters was observed for satisfaction with harvest and regulations.

Importance, Motivation, and Involvement in Waterfowl Hunting

Around half of respondents said that waterfowl hunting was one of their most important recreational activities, while 28% reported that waterfowl hunting was no more important than others. Ten-percent indicated that waterfowl hunting was their most important recreational activity, and 11% indicated that it was less important than other activities. Respondents agreed that waterfowl hunting provided an opportunity to be with family and friends, and that it is one the most enjoyable things that they do. Fewer respondents indicated that waterfowl hunting was a central part of their life. Four dimensions were identified among factors that influence hunters' satisfaction with waterfowl hunting; seeing ducks and geese, attracting waterfowl with decoys and calls, bagging a lot of waterfowl, and other specialized aspects of bagging waterfowl. On average, respondents rated seeing ducks and geese as the most important determinant of their satisfaction, and bagging a lot of waterfowl as the least.

Opinions about Youth Waterfowl Hunting Day

A majority (73%) of respondents supported the Youth Waterfowl Hunting Day, and 46% strongly supported it. Twelve-percent of respondents reported participating in the Youth Waterfowl Hunting Day, escorting an estimated 18,027 youth hunters to the field in 2017.

Opinions about Waterfowl Regulations

Respondents' opinions about current duck bag limits were queried. The majority (~two-thirds) indicated that the 6-duck, 2-hen mallard, and 3-wood duck bag limits were “about right.” Minorities in each instance reported that they felt the bag limits were either too high, too low, or did not have an opinion on the matter. Respondents indicated preference for the current 3 zone (North, Central, and South) configuration. Support was also expressed for the potential to allow open water hunting on a select number of large water bodies. Respondents, on average, expressed the least support for restricting the use of motorized decoys.

Opinions about Season Dates and Splits

Respondents were asked to indicate their preference for alternative configurations of season dates and splits. An option for a continuous 60 day season, and 2 different split seasons were presented, as was a “no preference” alternative. Fifty-eight percent of respondents in the North zone preferred a continuous season, while 43% of South zone hunters preferred the split season with the latest end date. A similar pattern was observed with respect to season opening dates, with North zone hunters preferring an early start date and Central and South zone hunters preferring a later one.

Opinions about Sandhill Crane Hunting in Minnesota

Five potential changes to sandhill crane hunting regulations were presented to participants for evaluation. Respondents, on average, supported a) expanding the crane hunting area, b) moving the opening day to an earlier date, c) expanding the size of the current zone, d) increasing the season length from 37 to 58 days, and e) increasing the daily bag limit from 1 to 2 birds. Mean values ranged from 3.8 (a) to 3.3 (e), where 1 = strongly oppose, 3 = neutral, and 5 = strongly support.

Comparisons to Previous Assessments

Satisfaction with waterfowl hunting in 2017 was higher than in several previously studied years including 2000, 2005, 2007, 2010, 2011, and 2014 (but not 2002). Support for the Youth Waterfowl Hunting Day was also higher than in years where it was examined.

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WEST NILE VIRUS EXPOSURE AND INFECTION RATES IN MINNESOTA RUFFED GROUSE

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SUMMARY OF FINDINGS

Cooperating hunters (n = 117) voluntarily collected 273 samples from ruffed grouse (*Bonasa umbellus*) harvested during the 2018 hunting season as part of a multi-state, collaborative West Nile virus (WNV) study. Hunters collected biological samples (blood and heart) and information on the age, sex, and location where the bird was harvested. Blood and heart samples were submitted to the Southeastern Cooperative Wildlife Disease Study (SCWDS) at the University of Georgia to assess both exposure to the virus and if the bird was undergoing an active infection. Laboratory results will be returned summer 2019 and will be shared with participating hunters. These findings will further understanding of the role that WNV plays in ruffed grouse in Minnesota and allow comparisons with other states in the Great Lakes Region.

INTRODUCTION

West Nile virus (WNV) is a mosquito-borne virus that causes encephalitis and myocarditis in individuals with active infections. West Nile virus has historically been found in Europe and Africa, but was first detected in North America in 1999 when an outbreak of encephalitis was reported in humans in New York City (Eidson et al 2001). Since its arrival, WNV has become established in all of the lower 48 US states and has been reported in over 300 bird species (Center for Disease Control, 2017), including ruffed grouse. Although some species, like American crows (*Corvus brachyrhynchos*) and blue jays (*Cyanocitta cristata*), readily die of WNV infection; most infected birds survive. Interestingly, since the arrival of WNV to the United States, mortality events due to WNV in some bird species have never been documented, [e.g. American robins (*Turdus migratorius*), chickadees (*Poecile spp.*), house wrens (*Troglodytes aedon*)] whereas others had an initial period of reduced survival for several years until they gained immunity to the virus, and yet some still continue to die from WNV annually (LaDeau et al. 2007, George et al. 2015).

The first documented WNV case in ruffed grouse in Minnesota occurred in 2005 (Ruffed Grouse Society, unpublished data). The suspected mosquito vector for ruffed grouse in Minnesota, *Culex restuans,* feeds almost exclusively on birds and is abundant in forests (D. Neitzel, Minnesota Department of Health, personal communication). We suspect that ruffed grouse in northern Minnesota are exposed to WNV annually but do not know if the virus causes active infections in juvenile or adult birds. Experimental infection of ruffed grouse and sage grouse (*Centrocercus urophasianus*) with WNV has indicated high susceptibility of these species to the disease (Naugle et al. 2004, Clark et al. 2006, Nemeth et al. 2017). Furthermore, recent study of the potential impact of WNV on ruffed grouse suggested chick survival was negatively affected by the virus (Nemeth et al. 2017). Past avian WNV outbreaks have occurred at the beginning of summer (late-June through mid-July; George et al. 2015) when grouse chicks may be most vulnerable to mortality.

Concern for WNV in ruffed grouse in Minnesota was heightened after the 2017 hunting season failed to meet harvest expectations, following a spring drumming count increase of 57% from the previous spring (Roy 2017). Hunters expected better-than-average hunting experiences, as has historically been the case when the 10-year cycle is nearing its peak (Amman and Ryel 1963, Stoll 1980). However, the drumming count is an index to the adult breeding population and these surveys occur before annual production, meaning drumming counts do not necessarily forecast the juvenile contribution to the fall population. Poor grouse production can adversely impact hunter experiences because juveniles comprise much of the fall harvest (Dorney 1963). Despite 10-year cycles around a stable population average for decades in the core of Minnesota ruffed grouse range, some hunters indicated that hunting experiences have been less rewarding over that time period, leading many to speculate that something has been affecting juvenile production.

In an effort to understand the effects of WNV on ruffed grouse populations, Pennsylvania researchers conducted statewide serosurveys for WNV from hunter-harvested birds in 2016 and 2017 and found apparent prevalence rates of 14% (n = 202) and 22% (n = 217), respectively (J. Brown, Pennsylvania Game Commission, unpublished data). A recent study in Pennsylvania indicated that ruffed grouse population recovery may be impaired in areas with poorer habitat and WNV (Stauffer et al. 2018). The interaction between invasive diseases and land-use can result in complex effects on survival of wild birds (George et al. 2015). In 2017, the Michigan Department of Natural Resources (MIDNR) also confirmed WNV in wild ruffed grouse in Michigan for the first time. Two grouse were found dead, and 3 hunter-harvested grouse were submitted for testing because they were malnourished and acting strangely; heart lesions caused by WNV were observed in all 5 cases (MIDNR, 2017). Recently, WNV was also confirmed for the first time in Wisconsin ruffed grouse, as 3 birds tested positive for the virus, with 2 also being co-infected with Eastern equine encephalitis (Wisconsin DNR, unpublished data).

In this pilot study we are assessing WNV exposure and infection rates in Minnesota ruffed grouse during the fall by partnering with grouse hunters to obtain samples from their harvested birds. We are estimating serological exposure to WNV and also examining hearts for lesions consistent with the disease in both juvenile and adult grouse. Juvenile birds may represent recent population exposure to WNV, for a direct correlation to current viral load on the landscape; whereas adult birds represent either recent or maintained exposure, given the magnitude of their titer levels and presence or absence of associated lesions.

Importantly, this study is a multi-state collaborative effort with other natural resource agencies in the Great Lakes Region, including Wisconsin and Michigan. This concerted effort will provide a more comprehensive view of the role of WNV in the region than any individual state could execute alone and demonstrates the interest of regional biologists in responding to hunter concerns.

OBJECTIVES

1. Assess the feasibility of working with grouse hunters to obtain biological samples from wild ruffed grouse for disease screening and to collect relevant metadata.
2. Estimate exposure to WNV in ruffed grouse populations in northwest and north-central MN.
3. Determine prevalence of active infections of WNV in ruffed grouse populations in northwest and north-central MN by age class (juvenile and adult).
4. Correlate exposure to WNV with active infection using paired samples from the same bird.

METHODS

Our study area focused on a 60-mile radius around Grand Rapids, Longville, and Bemidji, MN (Figure 1), with a sample goal of 400 birds during the fall 2018 hunting season. This area was chosen in an attempt to sample along a moisture gradient from west to east, based on rainfall received the previous year, and to simplify logistics of sampling kit dissemination to a few pick-up/drop-off locations. Regional MNDNR headquarters are located in Bemidji and Grand Rapids and provided a location for distribution of sampling kits. Pineridge Grouse Camp, which is located in Longville, was committed to assisting with our sampling effort and provided a third location for distribution of kits. Numerous organized hunts are also conducted annually in the study area (e.g., Ruffed Grouse Society National Hunt, Northwoods Bird Dogs/Bowen Lake Lodge, Akeley Grouse Hunt), which further facilitated sample collection.

Hunter Outreach

Multiple press releases were shared with the public with the first on 21 May 2018 announcing the multi-state collaboration between Wisconsin, Michigan, and Minnesota. Another press release came out on 23 Aug 2018 to provide more details for hunters interested in voluntarily participating in sampling efforts. Progress about the sampling efforts were shared by multiple media outlets throughout the hunting season to encourage public engagement (e.g., Duluth News Tribune, Outdoor News). We gave presentations at local universities (e.g., Itasca Community College, Bemidji State University), hunting camps (e.g., Pineridge Grouse Camp), regional DNR staff meetings, and distributed kits during these visits. We also attended organized hunts to distribute and collect kits (e.g., Akeley Grouse Hunt, Ruffed Grouse Society National Hunt, Northwoods Bird Dogs/Bowen Lake Lodge). A short paragraph about the study was added to the 2018 Minnesota Hunting Regulations and a contact was provided for more information. Information about the study was also posted on the Minnesota Department of Natural Resources website DNR Grouse Hunting Page.

Field Sample Collection

Each WNV sampling kit contained the following: 1 Nuboto filter strip (Advantec) for blood collection, 1 snack-sized zipper-top plastic bag (e.g., Ziploc brand) for storage of the filter strip in the field, 1 3-inch coin envelope for storage of filter strip once blood had dried, 1 4-oz whirlpak to collect the heart, a quart-sized zippered plastic bag for collection of feathers to confirm sex and age, and a datasheet to record hunter contact information and sampling location. A protocol was provided with each sampling kit along with *A Grouse in the Hand* pamphlets, courtesy of the Ruffed Grouse Society, to allow the hunter to determine the sex and age of their harvested bird based on feather characteristics. Ruffed grouse sex can be determined through tail length and rump feather dot patterns. Likewise, juvenile (<1 year) and adult (>1 year) age classes can be determined via fall feather wear of primary feathers collected from the wing. The instructions stressed the importance of collecting the blood on the filter strip within 30 min of harvest but also indicated samples collected after 30 min had value. Hunters were instructed to thoroughly coat the filter strip with blood until uniformly red and to allow the strip to air dry following the hunt. We asked hunters to record date and time of harvest and blood collection, location of harvest (GPS coordinates or distance and direction from nearest town), county of collection, hunter determined age class (juvenile, adult, or unsure) and sex (male, female, or unsure), any relevant comments, and hunter contact information (address, phone, and/or email address) if communication of results was desired. Samples collected through organized hunts or through local hunting camps were stored at room temperature (Nuboto strips) or frozen (heart samples) until submitted. Otherwise, hunters were provided with mailing kits with pre-paid UPS shipping labels, along with freezer packs and thermal bubble mailers to keep samples cold during shipment the following business day.

Laboratory Analysis

We confirmed age and sex of harvested birds before sending blood and heart samples to the Southeastern Cooperative Wildlife Disease Study (SCWDS) at the University of Georgia (Athens, GA) for diagnostic testing after the end of hunting season. Sample results had not been received from the lab at the writing of this report but are expected to arrive in summer 2019. Nuboto strips will be reconstituted to test for exposure to WNV, and viruses will be isolated. Hearts will be checked for virus by the polymerase chain reaction (PCR) and if virus is present, histological examination of the tissue will be performed.

Data Analysis

Apparent prevalence rates of WNV will be calculated using the number of positive detections from serum collected with Nuboto strips relative to the total number of blood samples collected. Seroprevalence by age class will be estimated by calculated apparent prevalence rates for juveniles and adults separately. Active infection rates will be calculated using the number of PCR-positive tests of heart tissue divided by the total number of heart samples submitted. Both seroprevalance and active infection data will be mapped using harvest location information and compared between the 2 sampling sites.

RESULTS AND DISCUSSION

Hunters (n = 117) collected 273 samples from ruffed grouse harvested during the hunting season 15 Sep 2018 – 1 Jan 2019 (Figure 2), of which 213 were collected from within the 60-mile sampling foci. Most of the samples (71%) were collected in October, 21% were collected in September, and submissions from November and December were 4.3% and 3.3%, respectively. Most of the returned kits contained all components requested, but 22 samples did not contain hearts, 40 samples were missing some or all feathers for sex or age determination, and 4 samples were missing location information.

Overall, the returned samples were from 160 juveniles (65%) and 87 adults (35%); however, age could not be confirmed for 26 birds due to missing primary feathers (Figure 3). The preponderance of juveniles in the sample was within the range reported by other studies (53% in Ohio, 75% in Wisconsin; Davis and Stoll 1973, Dorney 1963, respectively) and was expected given that juveniles typically make up the majority of birds harvested in the fall and in the fall population in general (Dorney and Kabat 1960, Dorney 1963). The sex of sampled birds was fairly evenly split between males (54%) and females (46%), but sex could not be confirmed for 14 birds due to missing feathers. Adult sex ratios for hunter harvested birds usually favor males, but juvenile sex ratios are usually closer to 50:50 (Dorney 1963, Davis and Stoll 1973). When our sample was split among age classes, males comprised 60% and females comprised 40% of the adult sample; whereas the sex ratio for juveniles was 50:50 as expected from other studies.

Verification of sex and age indicated that hunters were fairly accurate using feather characteristics for age and sex determination. However, the hunter-determined age needed to be corrected 51 of 212 times (24%), and sex was corrected 16 of 246 times (6.5%), not including cases where hunters indicated that they were unsure, or when feathers were not provided for verification.

We plan to share lab results with hunters about the birds they submitted and the overall findings of the study when we receive lab results. Given that we fell short of our sample goal by 32% for this pilot study, we will continue this study for 1 more year of data collection. We plan to expand the sampling area in fall 2019 to include a larger portion of ruffed grouse range in Minnesota and provide opportunities to other hunters interested in participating outside the original sampling area. To accomplish this, we will make sampling kits available at Wildlife Area Offices throughout ruffed grouse range in Minnesota. Sampling kits will be available on a first-come first-serve basis until depleted. We hope to collect 400 samples in the upcoming season to gain insights into year-to-year variability and inform regional comparisons.

ACKNOWLEDGMENTS

We would like to thank all the hunters that submitted samples for this study, with a special thanks to Pineridge Grouse Camp, Bowen Lodge and Northwoods Bird Dogs, the Akeley Grouse Hunt, the Ruffed Grouse Society, contributing students from Itasca Community College, DNR staff and retirees, as well as friends and family, with a special thanks to Charlie Tucker and Matt Breuer for their large sample contributions. We couldn't do this study without your help. We would also like to thank the Ruffed Grouse Society for providing partial funding of this study, as well as Federal Aid in Wildlife Restoration (Pittman_Robertson) for funding. We are grateful to Erik Hildebrand, Patrick Hagen, Margaret Dexter, and Lindsey Peterson for assistance with kit building. Mike Larson provided helpful comments on an early version of this report.

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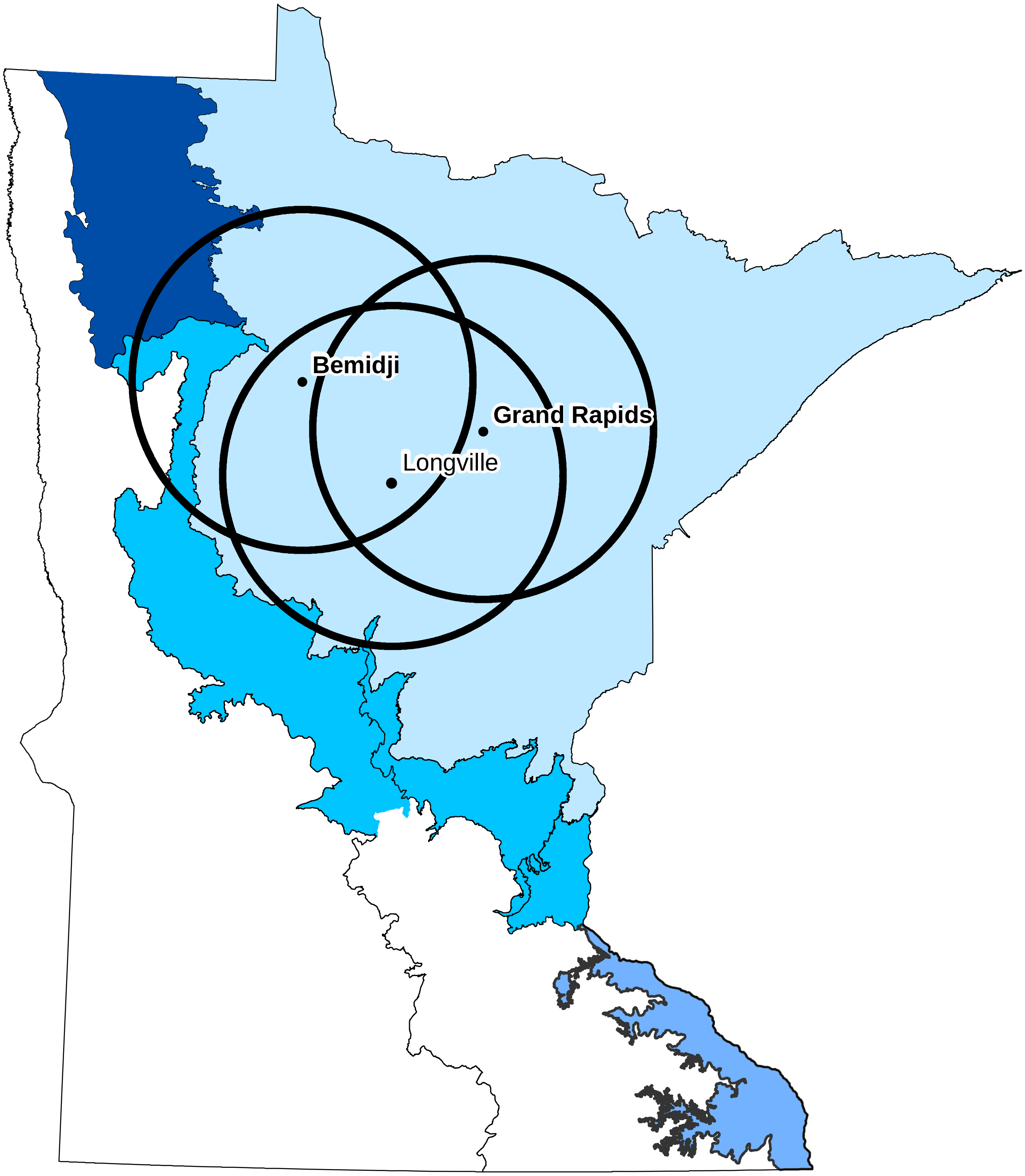

Figure 1. The study area in 2018 was comprised of 60-mile radii centered on Bemidji, Grand Rapids, and Longville in Minnesota. The ruffed grouse drumming survey regions are indicated in blue for reference.

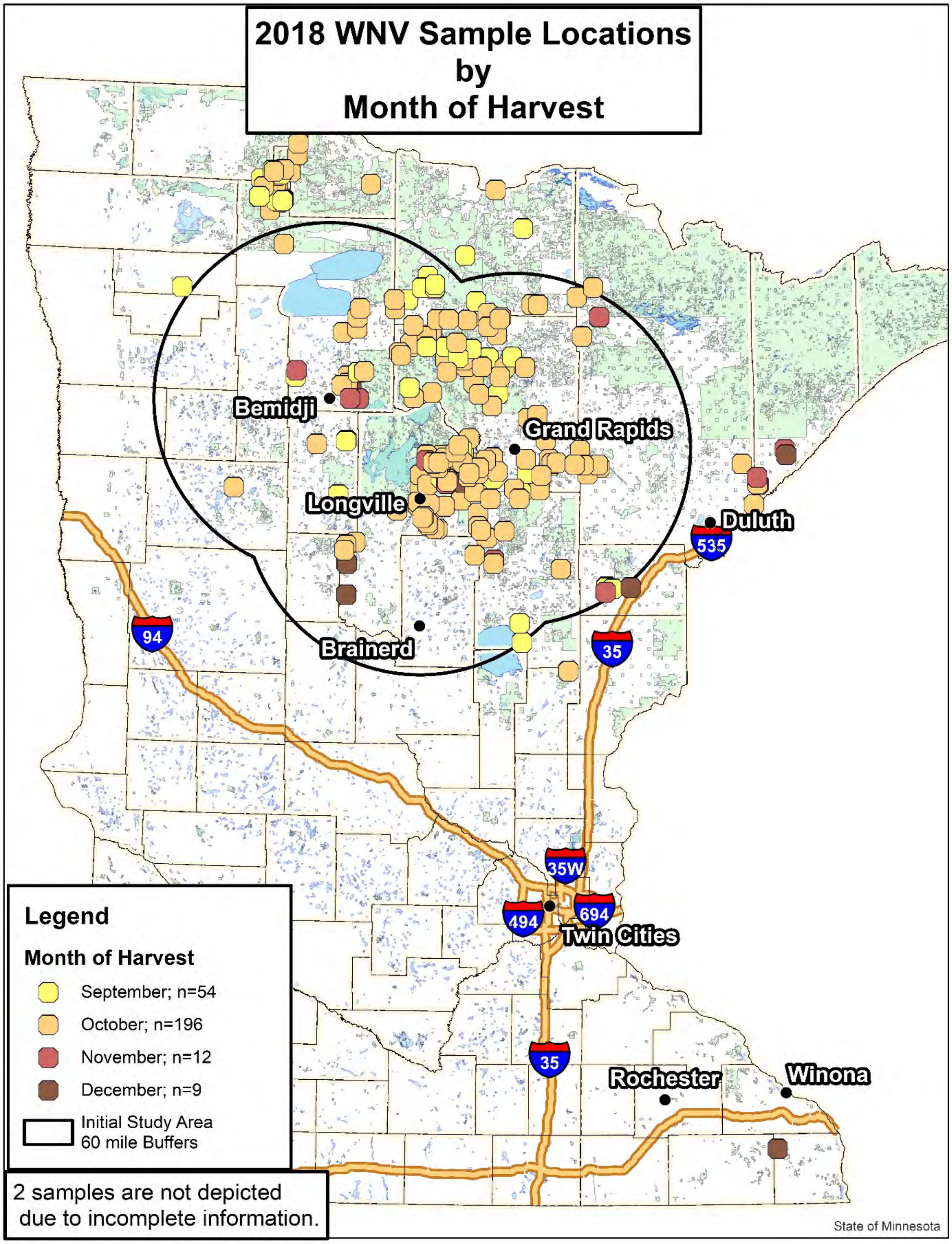

Figure 2. The distribution of hunter-harvested ruffed grouse samples both inside and outside the planned study area in Minnesota during hunting season in 2018.

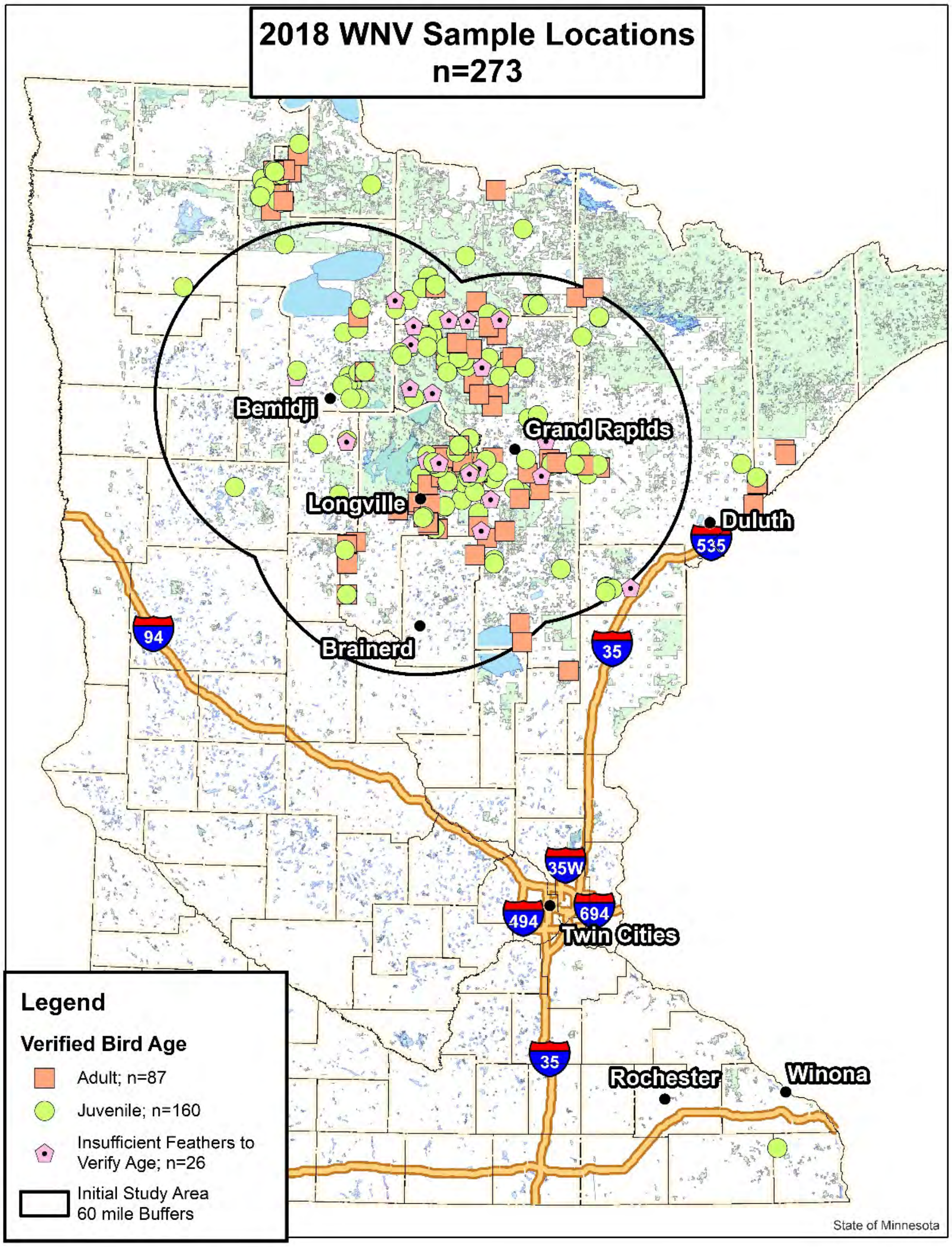

Figure 3. The age distribution of hunter-harvested ruffed grouse samples collected within and outside our study buffers in Minnesota during hunting season in 2018.

ECOLOGY AND POPULATION DYNAMICS OF BLACK BEARS IN MINNESOTA

David L. Garshelis, Andrew Tri, Spencer J. Rettler[1], and Brian J. Dirks[2]

SUMMARY OF FINDINGS

During April 2018–March 2019, we monitored 31 American black bears (*Ursus americanus)* previously radiocollared (mostly with GPS collars) at 4 study sites representing contrasting portions of the bear's geographic range in Minnesota: Voyageurs National Park (VNP, northern extreme, poorest food), Chippewa National Forest (CNF; central), Camp Ripley Training Center (southern fringe), and a site at the northwestern (NW) edge of the range. During summer, we captured and collared 6 more bears in the CNF, and in winter collared 5 female yearlings in this area. Hunting has been the primary source of mortality in all areas; even though VNP and Camp Ripley are unhunted, bears may wander off to other areas in fall where they are vulnerable to hunters. Hunters were asked not to shoot collared bears, all of which are conspicuously marked with large colorful eartags. Two were shot and not reported to us this year. Reproduction was strongly affected by food supply. Bears grew fastest and matured earliest in the NW and Camp Ripley. Age of primiparity was most delayed in VNP. Litter size, though, did not correspond with weight. Bears in the CNF are tending to use more nest dens and fewer underground dens than they did in the 1980s and early 1990s. We posit that this behavioral shift may be an effort to use dens that do not flood with early spring snowmelt.

INTRODUCTION

Telemetry-based research on American black bears (*Ursus americanus)* was initiated by the Minnesota Department of Natural Resources (MNDNR) in 1981, and has been ongoing continuously since then. For the first 10 years, the bear study was limited to a site near the geographic center of the Minnesota bear range, in an area mainly within or abutting the Chippewa National Forest (CNF; Figure 1). The CNF is one of the most heavily hunted areas of the state, with large, easily-accessible tracts of public (national, state, and county) forests. A chief goal at that time was to assess causes of mortality, specifically the relative impacts of hunting. This study began just as the MDNR switched from an unrestricted bear hunt to a quota on bear hunting license sales in 1982, in reaction to a concern that the population was being over-hunted. We used black radiocollars and inconspicuous eartags, and requested that hunters treat study bears as they would any other bear.

Commensurately, we studied the reproductive rate, and factors influencing that rate (Noyce and Garshelis 1994). The view among bear biologists at that time was that black bears had one of the lowest reproductive rates of any large mammal in North America. But, we found higher rates of reproduction in Minnesota than studies in western states.

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Using physical mark–recapture, where collared bears were the marked population, we tracked an increase in the study population on the CNF through the 1980s, despite heavy hunting, with harvest rates often exceeding 20% (Garshelis 1991, Garshelis et al. 1988). We found no other factor that significantly affected the population size or trend. A striking finding was that bears rarely died of natural causes. For example, despite (or because of) being in winter dens for 6 months per year, winter weather conditions had no effect on their survival.

We also gained considerable new information about the ecology of bears. Notable findings included: a flexible diet, but with conspicuous effects of food supply on rates of mortality (from hunting: Noyce and Garshelis 1997) and reproduction; varied use of habitat that reflected the changing availability of foods during the course of the year and among years, as well as bears apparently striving for dietary diversity (Garshelis and Noyce 2008); extensive seasonal movements (migrations) to take advantage of richer food sources outside their normal home ranges (Noyce and Garshelis 2011, 2014); and diverse sites and structures used as den sites, which showed no relation to reproduction or survival.

To better understand the dynamics of bear populations across Minnesota, we added more study sites, including 2 sites where bear hunting was prohibited, and 1 site at the front of a recent geographic expansion, where the habitat was dominated by agriculture. We hypothesized that causes of mortality and rates of reproduction would differ among these sites.

OBJECTIVES

1. Compare sources of bear mortality in different parts of the bear range.
2. Quantify temporal and spatial variation in cub production and survival.
3. Assess whether bears are using different types of dens now, compared to the 1980s.

STUDY SITES

During the 1980s, the CNF study site was heavily logged, resulting in a matrix of stands of various ages, with many small unpaved logging roads. A unique feature of this study site is that the east side is dominated by upland forests, largely aspen (*Populus tremuloides, P. grandidentata*), whereas a large swath of the western part of this area is a forested bog. In recent years, more roads have been paved and/or widened; also, forest cutting on federal lands has diminished, so forests have aged and the amount of aspen on the landscape has declined. Nearly all bear foods are found in the understory. Oaks (*Quercus* sp.) are not common anywhere in this area, and commercial agriculture is nonexistent.

In 1991 our study expanded to include Camp Ripley Training Center, a National Guard facility at the southern periphery of the primary bear range (Figure 1). Bear hunting is prohibited on Camp Ripley, but bears may be hunted if they range outside. Oaks are plentiful. The 210-km^2 area is long and narrow (6–10 km wide), and bordered by highways and cornfields.

In 1997 we added Voyageurs National Park (VNP), located along the northern edge of the Minnesota bear range (bordering Canada; Figure 1). The study site is a 300-km^2 roadless peninsula bounded by 3 large lakes. VNP had the poorest and shallowest soils, no timber cutting, and a largely coniferous forest, so provided the least food for bears. Hunting is prohibited, but bears are exposed to hunting if they leave the park.

In 2007, we initiated work at the northwestern edge of Minnesota's bear range (NW; Figure 1). This area is largely agricultural, although only 2% of the land area is planted with crops consumed by bears. Forested land, which comprises less than 20% of the area, is patchily distributed in small, privately-owned woodlots and state Wildlife Management Areas, which are open to hunting. The density of roads is high. A low density of bears occupied this area until 1995, when poor natural food in neighboring areas spurred an influx of bears attracted to corn and sunflowers. Since then the density of bears in this area has remained higher.

METHODS

During May−July, 2018, we captured bears in the CNF with barrel traps or Aldrich foot snares, and immobilized them with ketamine-xylazine. During December–March, we visited all radiocollared bears once or twice at their den site and immobilized them with Telazol. For all handling, we measured and weighed bears, assessed body condition, took blood and hair samples, and extracted a vestigial first premolar to estimate age on all bears whose age was unknown (i.e., first handling of bears older than cubs). We changed or refit the collar, as necessary. We collared bears that we thought would not disperse from the study area. We used GPS-Iridium collars (Telonics Inc., Mesa, AZ) on all but yearlings in the CNF, and mainly very high frequency (VHF) collars on the other study sites. All collared bears had brightly-colored, cattle-size ear tags (7x6 cm; Dalton Ltd., UK) that would be plainly visible to hunters. Licensed hunters could legally shoot collared bears, although they were asked not to. Prior to the hunting season (1 September–mid-October), hunters were mailed a letter requesting that they not shoot collared bears with large ear tags, and this request was also made through news releases. Requests to hunters to voluntarily not shoot collared bears have been made through the news media and MNDNR hunting regulations and website since 2001.

We assessed reproduction by observing cubs in March dens. We sexed and weighed cubs without drugging them. We quantified cub mortality by examining dens of radiocollared mothers the following year; cubs that were not present as yearlings with their mother were presumed to have died.

When visiting bears in dens, we categorized the type of den (open nest, brush, under roots, excavated, or other). These are the same categories used since the 1980s, enabling a comparison across time.

RESULTS AND DISCUSSION

Radiocollaring and Monitoring

As of April 2018, the start of the current year's work, we were monitoring 31 radiocollared bears: 17 in the CNF, 9 at Camp Ripley, 2 in VNP, and 3 in the NW (Table 1). During May–July we captured and collared 6 bears (all males) in the CNF. One of these had been initially collared in 2016, but the collar had failed.

The sex ratio of our sample in the CNF is heavily skewed toward males (13M, 5F), reflecting, it seems, a skewed sex ratio in this population. To increase the number of females in our sample at this site, we collared 5 CNF female yearlings in their dens. At all other study sites, we are monitoring only females.

Mortality

Since 1981 we have recorded the cause of death for 387 radiocollared bears, 76% of which died (or likely died) from legal hunting (Table 2). In all 4 study sites, legal hunting was the primary cause of mortality (Figure 2), despite (a) Camp Ripley and VNP being unhunted (but bears wander outside during fall on foraging trips), and (b) hunters being asked to not shoot collared bears with large ear tags for the past 18 years (spanning the full period of the NW study).

During the 2018 bear hunting season, 2 collared CNF bears were shot (Table 1). Neither of these were reported to us. We found the collar of one of these under the snow when we were going to what we thought was the bear's den site. For the other, we found the collar in a U.S. Forest Service truck (the employee previously found it in the woods during the bear hunting season, so we suppose it was shot by a hunter).

Vehicle collisions are another significant source of mortality. One large (507 lbs in December) CNF male was struck and killed by a logging truck in June 2018. Vehicle collisions are most common at Camp Ripley (Figure 2), which is flanked by 2 highways with fast-moving traffic.

The only other mortality this year was a yearling female in the NW who seemed to have been killed by wolves. She did not have a GPS collar and so we do not know when she died; we found her collar in March 2019, when searching for her den. We did not find her remains, but the collar was severely chewed by wolves, and was not near any roads or dwellings. However, it was near a private landowner's deer-feeding area, where wolves were common. This was the first natural mortality in the NW study site. Only 17 of 387 (4%) of bears died of natural mortality among all sites since this study began (Table 2).

Reproduction

Since 1982, within the 4 study areas, we handled 315 litters with 819 cubs. We observed spatial differences in reproduction, but no consistent temporal trends have been evident.

Litter size averaged 2.6 cubs (range of study site means = 2.2–2.8; Tables 3–6). In CNF and NW, 3-cub litters were most common (Figure 3). In VNP, 2-cub litters were most common. In Camp Ripley, 3-year-old first-time mothers tended to have 2-cub litters; excluding these, an equal number had 2- and 3-cub litters (Figure 3). Since 2005, we have not observed a litter of <3 cubs in the CNF (Table 3). We cannot explain why CNF bears, which are generally not as heavy as Camp Ripley bears, tended to produce larger litters. This year, the only collared female in CNF that produced cubs was just 158 lbs in the den yet had a litter of 4.

Sex ratio of cubs in March litters was 50.5% male (Tables 3–6). The sex ratio among yearlings in winter dens was more female-skewed (48.0% male), due to a higher mortality among male cubs (23.4%) than female cubs (15.2%). We were unable to check 3 of the dens with yearlings in Camp Ripley this year because the females denned in road culverts or in the impact area (where we were not allowed). Denning in road culverts is common in Camp Ripley. One male in the CNF denned in a culvert, which was the first time a bear selected a culvert den in the CNF since our study began in 1982.

Cubs remained with their mother for about 17 months, so the normal reproductive interval is 2 years. All bears that were expected to have cubs this year, based on this normal cycle, did so. Overall, bears at Camp Ripley, despite being relatively large, have had a higher rate of missed litters (3-year litter intervals) than bears in the other study sites (Table 7).

No collared bears produced a first litter this year. Two 5-year-old females (sisters) in VNP that could have done so, did not; an advanced age of primiparity is not unusual for this area. Previously, 3 bears in VNP produced first litters (with at least 1 surviving cub) at 5 years old, 3 at 6, and 1 at 9. By contrast, 38% of females in the CNF and 86% in Camp Ripley and the NW produced a first surviving litter by age 4. The differing ages of primiparity reflect the differing growth rates of bears at these study sites.

Types of Dens

During the 1980s through early-1990s, underground dens (either fully excavated or under tree roots) were the most common den type in the CNF, for both females (73%) and males (66%). Brush piles were used as dens by ~20% of both sexes, and above-ground open nests were used by 6% of females and 14% of males. Although underground dens are still most common, a significant shift toward greater use of nests occurred for both sexes in the CNF during 2016–2019 (Figure 4; sexes combined X^2=12.8, P=0.0004). Remote cameras at dens showed some bears being flooded out of underground dens during spring thawing, forcing them to abandon the dens or emerge to collect more bedding material in an effort to stay dry. Although we have no data on this behavior before 2016 (when we first began using cameras at dens), it seems logical that earlier spring temperatures would result in earlier flooded dens, and bears may be responding to this by switching away from underground dens, which collect water from melting snow dripping in from the ceiling and entrance. In choosing a nest den, bears tradeoff less protection from cold and snow (which accumulates on their back) in mid-winter for drier dens in spring (Figure 5).

ACKNOWLEDGMENTS

We thank Drs. Paul Iaizzo, Tim Laske, and Tinen Iles (University of Minnesota), who greatly assisted with fieldwork and led the associated work on heart monitoring (not covered in this report). Dave Rettler, Michael McMahon, and Brent Hemly also provided valuable field assistance. Agassiz National Wildlife Refuge kindly provided use of their bunkhouse. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program grant W-68-D-15.

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Table 1. Fates of radiocollared black bears in Chippewa National Forest (CNF), Camp Ripley, Voyageurs National Park (VNP), and northwestern Minnesota (NW) study sites, April 2018−March 2019.

	CNF	Camp Ripley	VNP	NW
Collared sample April 2018	17	9	2	3
Trapped and collared	6			
Collared in den	5			
Killed in vehicle collision	1			
Killed by Minnesota hunter[a]	2			
Natural mortality				1
Removed radiocollar	1			
Dropped radiocollar	1			
Collared sample April 2019	23	9	2	2

[a] Hunters were asked not to shoot collared bears (although it was still legal). Neither of these collars were turned in.

Table 2. Causes of mortality of radiocollared black bears ≥1 year old in 4 Minnesota study sites, 1981–2019. Bears did not necessarily die in the area where they usually lived (e.g., hunting was not permitted within Camp Ripley or VNP, but bears were killed by hunters when they traveled outside these areas).

	CNF	Camp Ripley	VNP	NW	All combined
Shot by hunter[a]	237	13	16	14	280
Likely shot by hunter[b]	9	1	0	4	14
Shot as nuisance	22	2	1	3	28
Vehicle collision	14	10	1	3	28
Other human-caused death	9	1	0	0	10
Natural mortality	8[c]	3	5	1	17[c]
Died from unknown causes	5	2	0	3	10
Total deaths	304	32	23	28	387

[a] Since 2001, the MNDNR has asked hunters not to shoot collared bears, so the proportion killed due to this cause is no longer representative of the population at large.

[b] Lost track of during the bear hunting season, or collar seemingly removed by a hunter.

[c] Only 1 bear died of “old age”.

Table 3. Black bear cubs examined in dens of radiocollared mothers in or near the Chippewa National Forest, Minnesota, during March, 1982–2019. High hunting mortality of radiocollared bears severely reduced the sample size in recent years.

Year	Litters checked	Number of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 year[a]
1982	4	12	3.0	67%	25%
1983	7	17	2.4	65%	15%
1984	6	16	2.7	80%	0%
1985	9	22	2.4	38%	31%
1986	11	27	2.5	48%	17%
1987	5	15	3.0	40%	8%
1988	15	37	2.5	65%	10%
1989	9	22	2.4	59%	0%
1990	10	23	2.3	52%	20%
1991	8	20	2.5	45%	25%
1992	10	25	2.5	48%	25%
1993	9	23	2.6	57%	19%
1994	7	17	2.4	41%	29%
1995	13	38	2.9	47%	14%
1996	5	12	2.4	25%	25%
1997	9	27	3.0	48%	23%
1998	2	6	3.0	67%	0%
1999	7	15	2.1	47%	9%
2000	2	6	3.0	50%	17%
2001	5	17	3.4	76%	15%
2002	0	0	—	—	—
2003	4	9	2.3	22%	0%
2004	5	13	2.6	46%	33%
2005	6	18	3.0	33%	28%
2006	2	6	3.0	83%	33%
2007	2	6	3.0	67%	17%
2008	1	3	3.0	100%	33%
2009	1	3	3.0	33%	33%
2010	1	4	4.0	100%	50%
2011	1	4	4.0	25%	50%
2012	1	3	3.0	67%	33%
2013	1	3	3.0	67%	0%
2014	1	3	3.0	67%	—[b]
2015	0	0	—	—	—
2016	0	0	—	—	—
2017	1	3	3.0	—	0%
2018	4	12	3.0	42%	0%
2019	1	4	4.0	50%	—
Overall	185	491	2.7	53%	18%

[a] Cubs that were absent from their mother’s den as yearlings were considered dead.

[b] Mother was killed by a hunter so status of cubs unknown.

Table 4. Black bear cubs examined in dens in northwestern Minnesota during March, 2007–2019.

Year	Litters checked	Number of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 year
2007	2	6	3.0	33%	100%
2008	5	15	3.0	67%	22%
2009	1	3	3.0	33%	33%
2010	6	17	2.8	41%	13%
2011	2	4	2.0	75%	25%
2012	4	10	2.5	60%	10%
2013	3	9	3.0	67%	18%
2014	3	8	2.7	0%	33%
2015	2	5	2.5	60%	0%
2016	2	6	3.0	50%	0%
2017	1	3	3.0	0%	0%
2018	1	4	4.0	50%	25%
2019	1	2	2.0	50%	
Overall	33	92	2.8	44%	17%[a]

[a] Excludes the total loss of a 5-cub litter in 2007 (which was not within the designated study area).

Table 5. Black bear cubs examined in dens in or near Camp Ripley Training Center, Minnesota, during March, 1992–2019.

Year	Litters checked	Number of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 year[a]
1992	1	3	3.0	67%	0%
1993	3	7	2.3	57%	43%
1994	1	1	1.0	100%	—
1995	1	2	2.0	50%	0%
1996	0	0	—	—	—
1997	1	3	3.0	100%	33%
1998	0	0	—	—	—
1999	2	5	2.5	60%	20%
2000	1	2	2.0	0%	0%
2001	1	3	3.0	0%	33%
2002	0	0	—	—	—
2003	3	8	2.7	63%	33%
2004	1	2	2.0	50%	—
2005	3	6	2.0	33%	33%
2006	2	5	2.5	60%	—
2007	3	7	2.3	43%	0%
2008	2	5	2.5	60%	0%
2009	3	7	2.3	29%	29%
2010	2	4	2.0	75%	25%
2011	3	8	2.7	50%	25%
2012	1	2	2.0	100%	0%
2013	6	14	2.3	50%	21%
2014	1[b]	—[b]	—	—	—
2015	6	15	2.5	20%	10%
2016	0	0	—	—	—
2017	4	10	2.5	60%	0%
2018	2	5	2.5		—[c]
2019	3	7	2.3		
Overall	52	124	2.4	49%	18%

[a] Blanks indicate no cubs were born to collared females or collared mothers with cubs died before the subsequent den visit to assess cub survival.

[b] Cubs heard, litter not handled. Camera set outside den indicated that all cubs died. This litter not included in total.

[c] No yearling dens could be checked: all were in culverts or in impact area.

Table 6. Black bear cubs examined in dens in Voyageurs National Park, Minnesota, during March, 1999–2019. All adult collared females were killed by hunters in fall 2007, so sample sizes greatly diminished afterward.

Year	Litters checked	Number of cubs	Mean cubs/litter	% Male cubs	Mortality after 1 year[a]
1999	5	8	1.6	63%	20%
2000	2	5	2.5	60%	80%
2001	3	4	1.3	50%	75%
2002	0		—	—	—
2003	5	13	2.6	54%	8%
2004	0		—	—	—
2005	5	13	2.6	46%	20%
2006	1	2	2.0	50%	0%
2007	3	9	3.0	44%	—
2008	0		—		—
2009	0		—		—
2010	1	2	2.0	50%	0%
2011	1	2	2.0	0%	0%
2012	1	2	2.0	0%	50%
2013	1	2	2.0	50%	—
2014	1	3	3.0	33%	0%
2015	0	0	—	—	—
2016	0[b]	0	—	—	—
2017	0	0	—	—	—
2018	0	0	—	—	—
2019	0	0			
Overall	29	65	2.2	48%	25%

[a] Blanks indicate no cub mortality data because no cubs were born to collared females, or collared mothers were lost from study (died or lost collar) before denning with yearlings.

[b] One bear that likely had cubs was not checked because access to her den was precluded by poor ice conditions.

Table 7. Intervals between litters (where at least 1 cub survived) for black bears within 4 study sites in Minnesota through March 2019 (CNF since 1981, Camp Ripley since 1991, VNP since 1997, NW since 2007). Cubs typically remain with their mother for about 17 months, so the normal reproductive interval is 2 years.

Study area	2-year reproductive intervals	≥3-year reproductive intervals	% intervals ≥3 years
CNF	112	8	7%
Camp Ripley	35	5	13%
VNP	15	1	6%
NW	19	0[a]	0%

[a] Excluding 1 missed litter (3-year interval) that was due to the bear leaving the den after disturbance and aborting the litter.

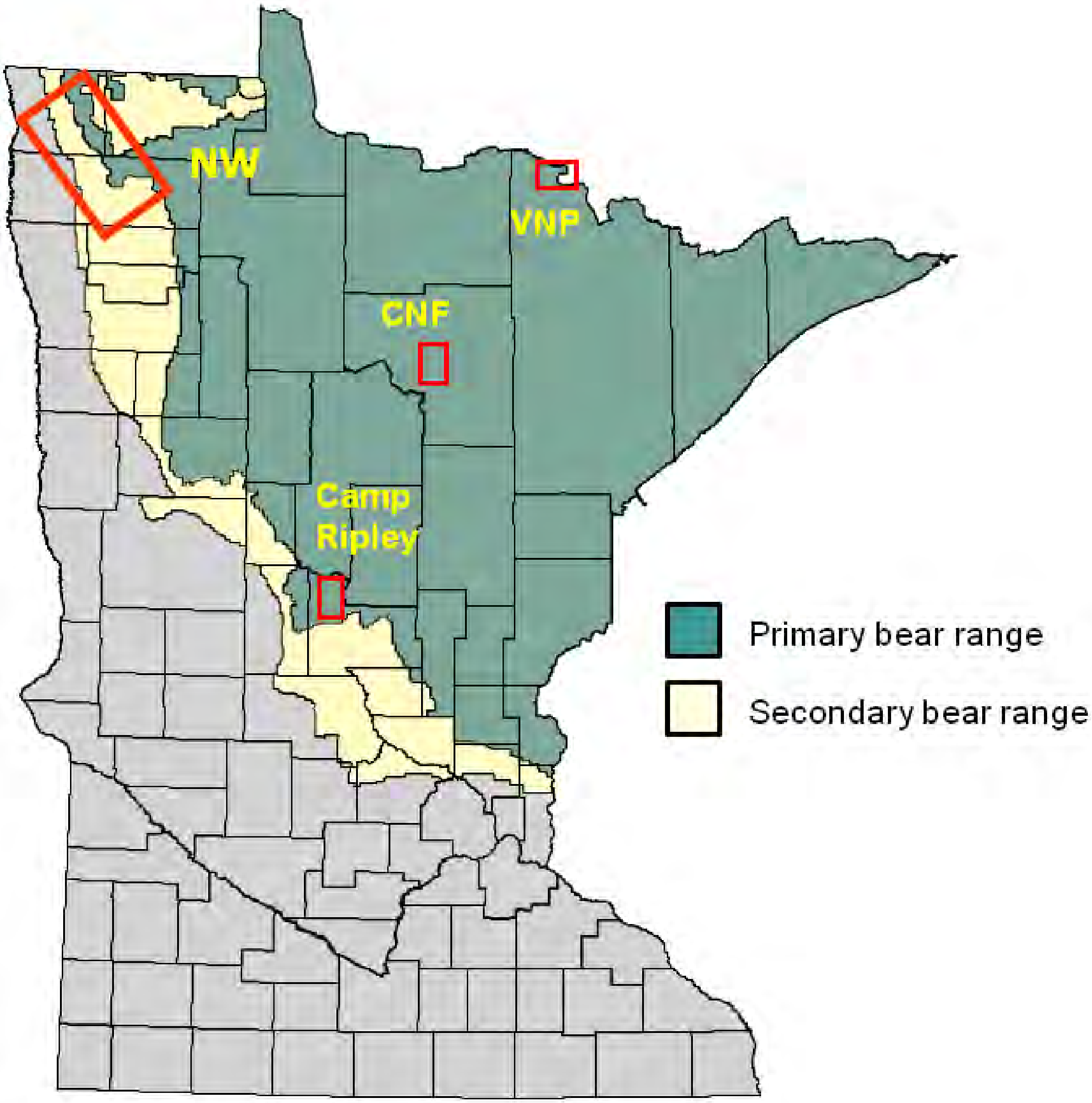

Figure 1. Location of 4 study sites within Minnesota's bear range: CNF (Chippewa National Forest, central bear range; 1981–2019); VNP (Voyageurs National Park, northern fringe of range; 1997–2019); Camp Ripley Military Reserve (near southern edge of range; 1991–2019); NW (northwestern fringe of range; 2007–2019).

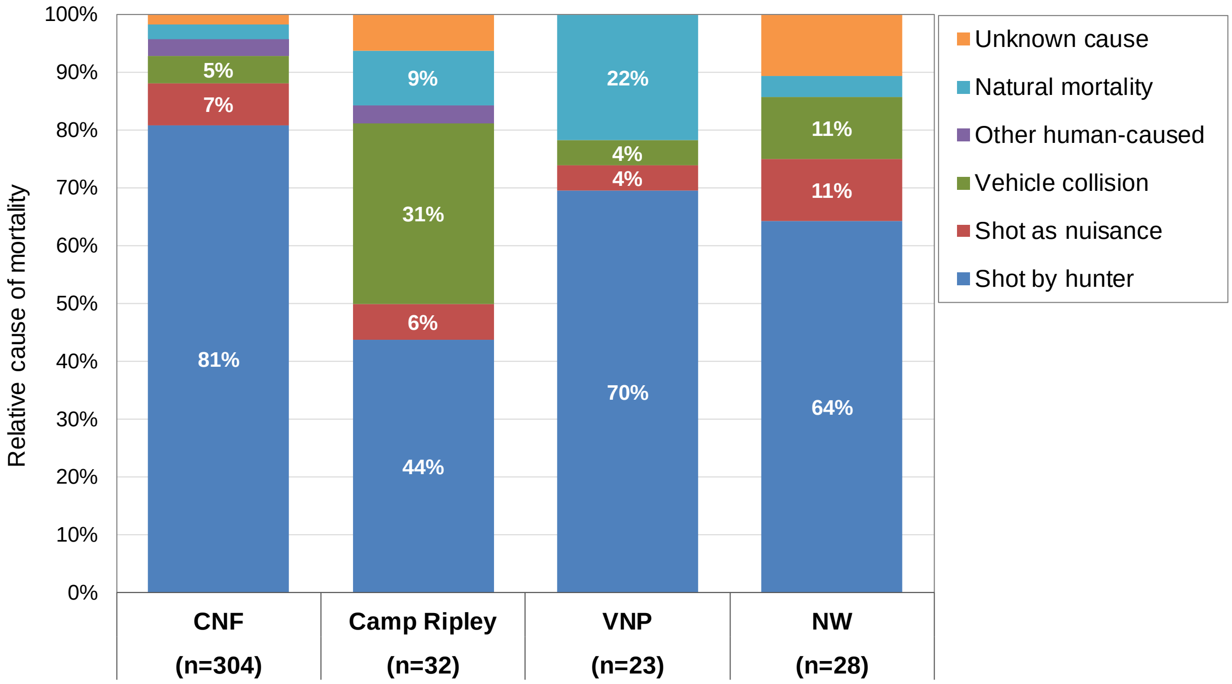

Figure 2. Proportional causes of death of radiocollared bears in each of 4 study sites in Minnesota, since the beginning of the study in each area through the 2018 hunting season (see map and dates for each study site in Figure 1). CNF expectedly had the highest proportion of bears killed by hunters because this is primarily public land that is heavily hunted. Camp Ripley and VNP are unhunted but bears are vulnerable when they leave on foraging forays. Hunters were asked not to shoot collared bears during the entire span of the NW study, so the proportion killed by hunters there is an underestimate.

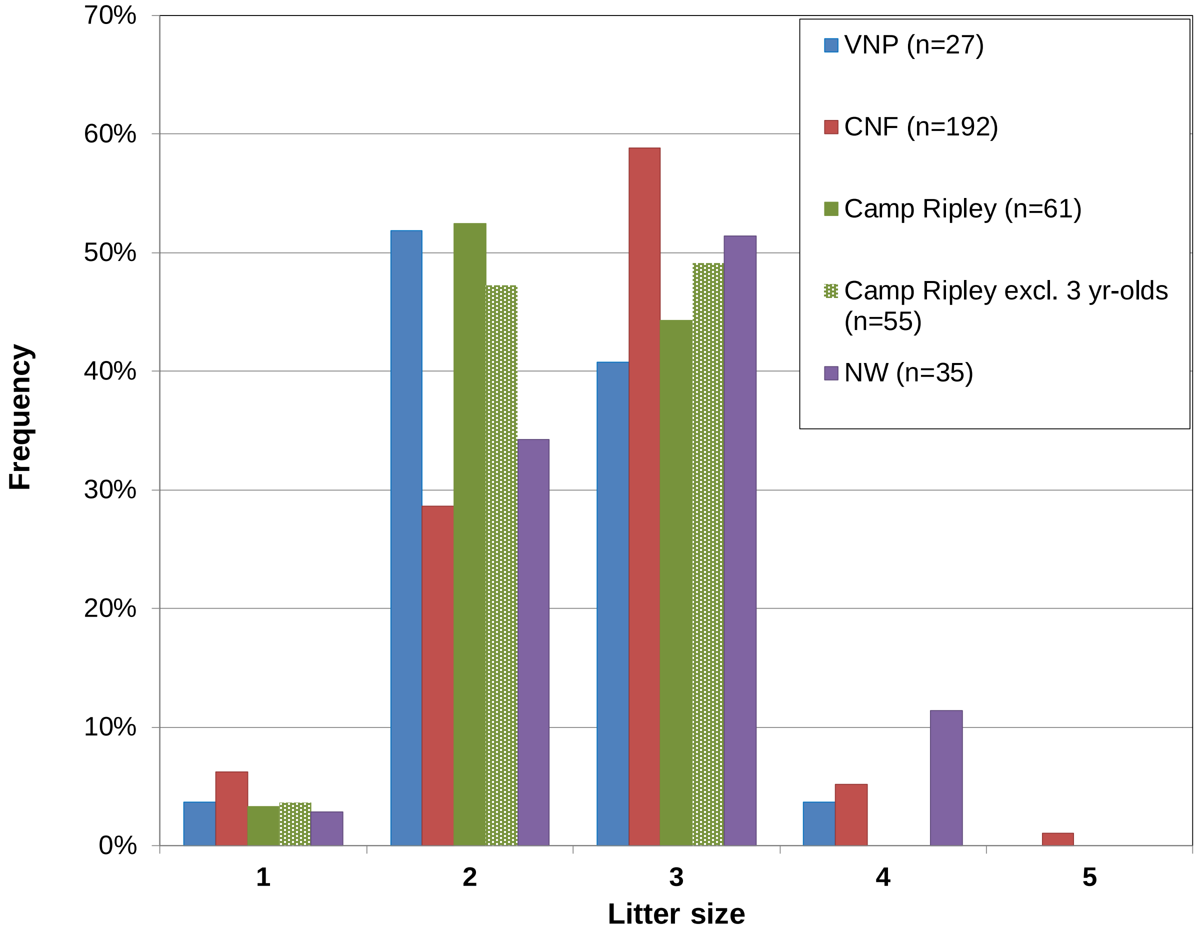

Figure 3. Frequency of cub litter sizes (examined in natal dens in March) within 4 Minnesota study sites (see Figure 1) through March 2019. Data include only litters that survived 1 year (even if some cubs in the litter died). Camp Ripley data are shown for mothers of all ages, as well as excluding 3-year-old mothers. For the other sites, elimination of 3-year-olds did not make a difference.

Figure 4. Types of dens used by black bears in the CNF study site in Minnesota during the early years of this study (1981–1993) versus recently (2016–2019).

Figure 5. Nest dens (left) are being used more frequently by bears now than they were 30 years ago in Minnesota, possibly because bears remain drier in nests during early spring thaws than underground dens (right), where snowmelt seeps in from the ceiling and entrance.

BEHAVIORAL RESPONSES OF AMERICAN BLACK BEARS TO REDUCED NATURAL FOODS: MIGRATION PATTERNS AND DIET

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SUMMARY OF FINDINGS

Our previous work found that the fleshy fruits and nuts that constitute the main natural foods for American black bears (*Ursus americanus*) declined greatly in the Chippewa National Forest (CNF) study site in north-central Minnesota, from the 1980s to mid-2010s. So far, we have not been able to detect differences in sex-age-specific winter weights of bears between these 2 time periods, nor any difference in reproduction. Here we explored some potential reasons for why weights have not changed despite the lower availability of natural foods. We investigated whether bears more frequently left the CNF to find richer foraging areas during the 2010s than they had previously. During the 1980s, a portion of the radiocollared bears made fall migrations, mainly southward to better foraging areas, each year. Data from GPS-collared bears during 2016–2018 suggest that such migrations may be more frequent now. We used stable isotope analysis of segmented hair samples to investigate whether bears that migrated enhanced their diet with corn, which has an amplified δ^{13}C signature. One male bear that fed in a cornfield had greatly enhanced δ^{13}C values for the hair segments representing the fall diet. Other male bears, whether they migrated or not, showed a less dramatic increase in dietary corn in the fall, possibly from hunters' baits. Many male bears also apparently found corn products around people's houses during the summer. Females, on average, had lower δ^{13}C values, but also showed evidence of feeding on corn products during the summer. Bears appear to be using human-related foods to make up for reduced natural foods.

INTRODUCTION

American black bears (*Ursus americanus*) in Minnesota forage on carbohydrate-rich fleshy fruits in summer, and then seek out fat-rich nuts (oaks [*Quercus* sp.] and hazelnuts [*Corylus* sp.]) in fall to gain necessary fat reserves for hibernation. However, within the Chippewa National Forest (CNF) study area, near the center of Minnesota's bear range, there has been a significant decline in availability of natural bear foods since we started studying bears there in the early 1980s (Rettler et al. 2018). The aim of this study is to understand how bears are responding to this reduced availability of food. Does it affect their weight, body condition, growth rate, reproduction, and survival? Or have they found adaptive ways to compensate? Our data so far indicate that bears in winter dens have similar sex- and age-specific weights and reproduction as in the 1980s, suggesting that they may have found alternate food sources to compensate for the reduction in fruits and nuts. Here we investigate whether migration out of the CNF to better food-producing areas could explain the normal winter weights and reproduction, despite the decline in the CNF food base.

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During the 1980s, a portion of bears (averaging 44% of males and 39% of females) from the CNF migrated each year in late summer and fall, generally southward, to better food-producing areas, specifically areas richer in oaks or agricultural crops (Noyce and Garshelis 2011). Counter to expectations, that research found that migrations were more common during years with high food production, and less frequent during poor years, suggesting that bears perceived that in poor food years, their travel might not yield a compensatory food reward (in other words, that poor food conditions would occur across the broad landscape, and not just local to the CNF). Here we examined whether such migrations have become more or less frequent in response to diminished food on the CNF and whether bears that made migrations during the recent period (2010s) benefited in terms of one particular food reward—corn. This dietary item is easily discernible from stable isotope analysis of hair, which assimilates the different dietary components as it grows, so some aspects of the diet can be reconstructed through time by dividing the hair in segments (Ben-David and Flaherty 2012, Ditmer et al. 2016).

OBJECTIVES

1. Compare the proportion of bears on the CNF that migrated to fall feeding areas during the mid-2010s to that during the 1980s.
2. Determine whether bears make use of human-related foods as a dietary supplement.

STUDY AREA

Located in northcentral Minnesota, the CNF study area falls in the transition zone between the boreal forests to the northeast and the temperate forests in the central part of the state. Our 620-km^2 study area was dominated (42% of area) by the eastern extent of the CNF; the remainder included part of the George Washington State Forest and other state-owned land (11%), county land (6%), private land (18%), commercial timber industry (8%), and open water (15%). The eastern two-thirds of the study area was upland forests dominated by various combinations of aspen (*Populus tremuloides, P. grandidentata*), maple (*Acer spp.*), red pine (*Pinus resinosa*), paper birch (*Betula papyfiera*), and balsam fir (*Abies balsamea*). The western third was largely lowlands dominated by speckled alder (*Alnus incana*), black spruce (*Picea marina),* tamarack (*Larix laricina*), black ash (*Fraxinus nigra*), northern white-cedar (*Thuja occidentalis*), quaking aspen, and balsam poplar (*Populus balsamifera*). Many lakes, forest roads, and recreational trails occur throughout the public land. This area was heavily hunted for bears due to the large extent of easily-accessible public land. Larger oak stands and agricultural landscapes do not exist on the study area but can be found to the south and west.

METHODS

We fit bears with GPS radiocollars and monitored them during 2016–2018. GPS locations were obtained at 2-hour intervals during the non-denning period (April–November). We classified migrations as significant movements (>5 km) outside the summer home range during mid-July to October. We excluded bears shot during the September hunting season because we could not discern whether they would have migrated.

To quantify the amount of corn in the diet, we used stable isotope analysis of bear hair samples collected from GPS-collared bears in winter dens. Bears molt and grow new hair each spring, so hair samples represented a bear's diet from mid-June/early-July until hibernation (when hair growth ceases). For each bear, we cut approximately 30 hairs into 8 equal-length segments (Figure 1), making the assumption that differing hair lengths among bears was due to differing growth rates; thus, each segment of each bear would correspond to diet assimilation during approximately the same 2-week period over the 4-month span from start of hair growth to hibernation (e.g., segment 1 at the distal end represents the diet in early July, and segments 7–8, closest to the root, represent the fall diet).

Hair samples were analyzed in a Thermo Delta V isotope ratio mass spectrometer interfaced to a NC2500 elemental analyzer at the Cornell Isotope Laboratory. Isotope results are expressed with a "δ" notation to indicate the change in isotope ratios relative to an international standard. Higher δ^{13}C values indicate a larger contribution of C_4 plants (corn or sugarcane) in a bear's diet, whereas lower values indicate more natural vegetation (Ditmer et al. 2016). Higher δ^{15}N values identified food from a higher trophic level or contained higher amounts of protein (not discussed in this report).

RESULTS

We observed fall migrations for 57% (*n*=27) of 47 bear-years during 2016–2018. Migrations occurred for 53%, 50% and 61% of bears in 2016, 2017, and 2018, respectively (Figures 2–4). Among all 3 years, migrations occurred for 59% of female (*n*=17) and 57% of male (*n*=30) bear-years. Most migrations were directed to the south or southwest of the study area. Males traveled up to 130 km and spent up to 8–9 months away from their summer home ranges (because some overwintered in the area where they had migrated to).

Hair sample segments representing the fall showed a pulse of enriched carbon for males, especially those that did not migrate (Figure 5). We excluded 1 migratory bear which, based on GPS locations, spent most of the fall (in all years) in a cornfield and had a very carbon-enriched hair signature in fall (Figure 6). In contrast, for females, whether migrating or not, peak δ^{13}C values occurred in hair segments representing the summer (Figure 5). Carbon values for non-migrating males were higher than for females for all hair segments.

DISCUSSION

Preliminary results suggest that the direction and extent of fall migrations from the CNF were on par with those observed in the 1980s. The most migrations occurred in 2018, when food conditions on the CNF were much higher than in 2016 and 2017, a pattern consistent with the 1980s (Noyce and Garshelis 2011). However, the proportion of bears migrating during the years with low food (2016 and 2017) was higher than observed during the 1980s, possibly suggesting that more frequent migrations to areas with richer foods was one way that CNF bears were able to increase body weights in fall.

Another way that bears may have filled in the gap of low natural fall foods was through greater reliance on hunters' baits, which are available to bears from mid-August to mid-October. These baits generally include sugar or corn-based products (e.g., high-fructose corn syrup), so should show up as enriched δ^{13}C in fall hair segments. We will explore the possibility that bears increased their use of baits over the decades by comparing δ^{13}C values for bear hair samples collected during 2016–2020 versus those collected in the 1980s–early 1990s.

We have records (GPS points or camera trap photos from hunters) of some of the current GPS-collared bears visiting hunters' baits, and their carbon signatures are enhanced in fall hair segments. However, sometimes the δ^{13}C signature is not distinct, possibly because the bear did not consume a significant amount of bait or because a large proportion of the consumed bait was not sugar- or corn-based. We surveyed hunters in the CNF area to find out what mixture of bait products they used, and we will be examining the carbon and nitrogen signature for these (Kirby et al. 2017).

Stable isotope signatures also indicated that some bears obtained corn-based products at other times of year. The δ^{13}C signature for males is generally higher than for females throughout the year (Figure 5), and some of the males in the sample were known to consume human-related sources of food, near people's homes. These foods likely include birdseed, which we should be able to detect with a nitrogen signature (Ditmer et al. 2016). We also have direct evidence, from scat samples, of bears obtaining corn in summer (Figure 7), presumably from deer (*Odocoileus virginianus*) feeders. It appears that one reason that bears are able to maintain weights comparable to the 1980s in the face of reduced natural foods is through the use of human-related foods.

ACKNOWLEDGMENTS

We thank Danielle Brown, Andrew Michel, Dalton Mouritsen, Mac Mullins, Kendra Pahnke, Suzy Tupy, Nathan Wallace, and the many other University of Minnesota undergraduate students who assisted with the tedious job of cutting bear hairs. This project was funded in part by the Wildlife Restoration (Pittman-Robertson) Program grant W-68-D-15.

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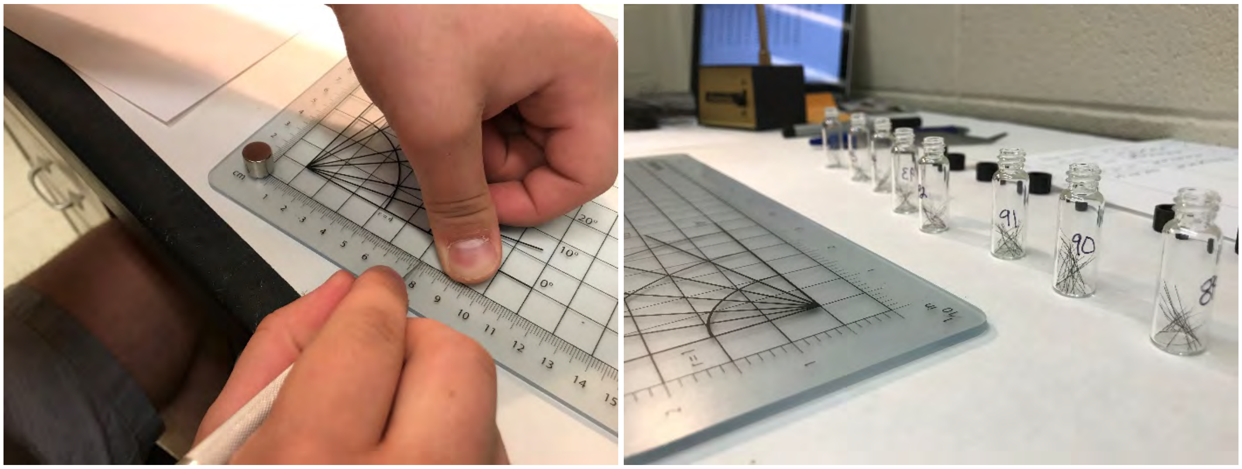

Figure 1. Hairs were collected from GPS-collared bears in winter dens, then cut in 8 equal lengths, representing approximately 2 weeks of growth. Approximately 30 hairs from each bear were segmentally cut to obtain a sufficient sample of each portion of hair for stable isotope analysis.

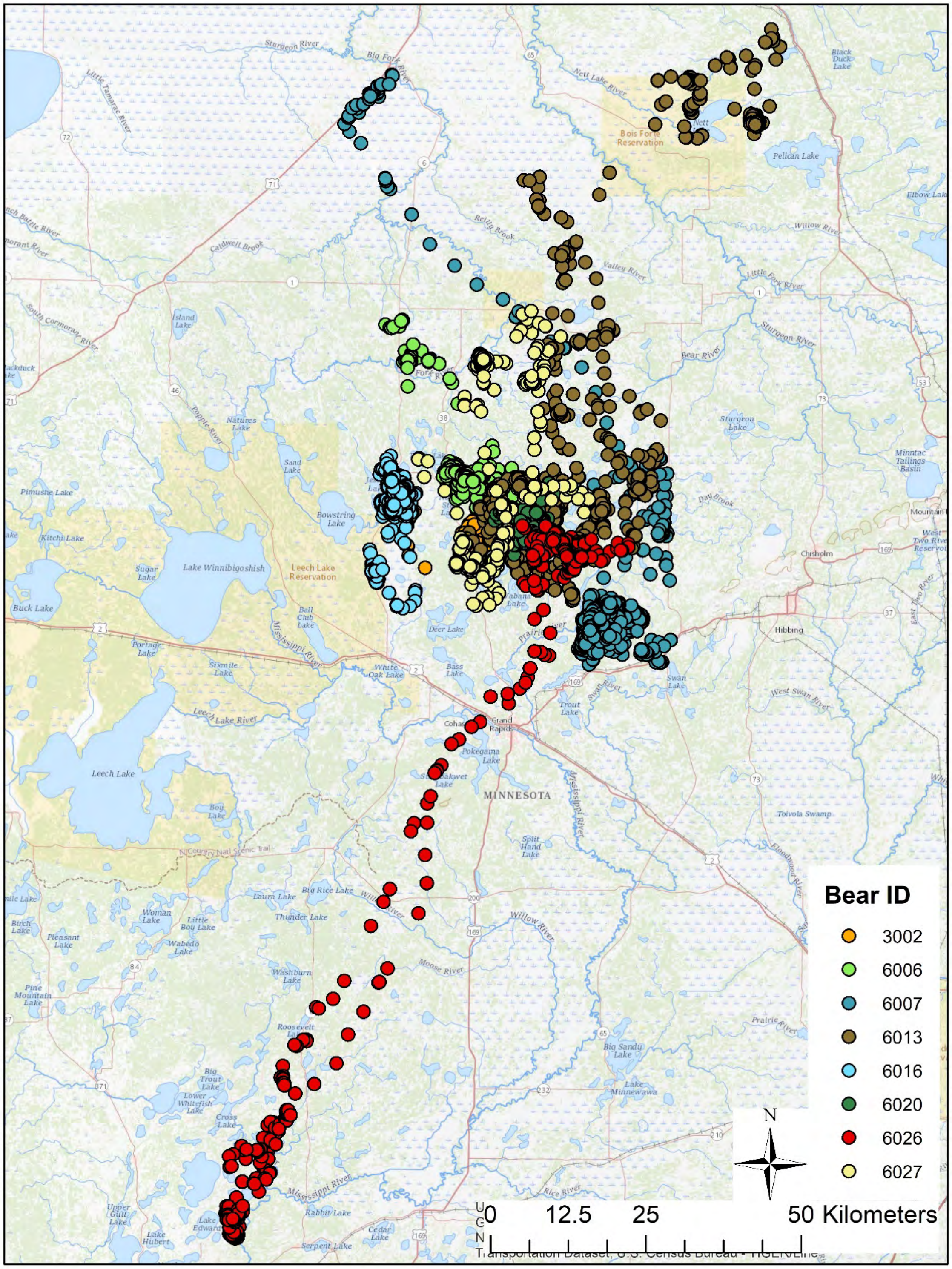

Figure 2. Fall migration movements of 8 GPS-collared bears (3 male, 5 female) from the CNF in 2016 in Minnesota.

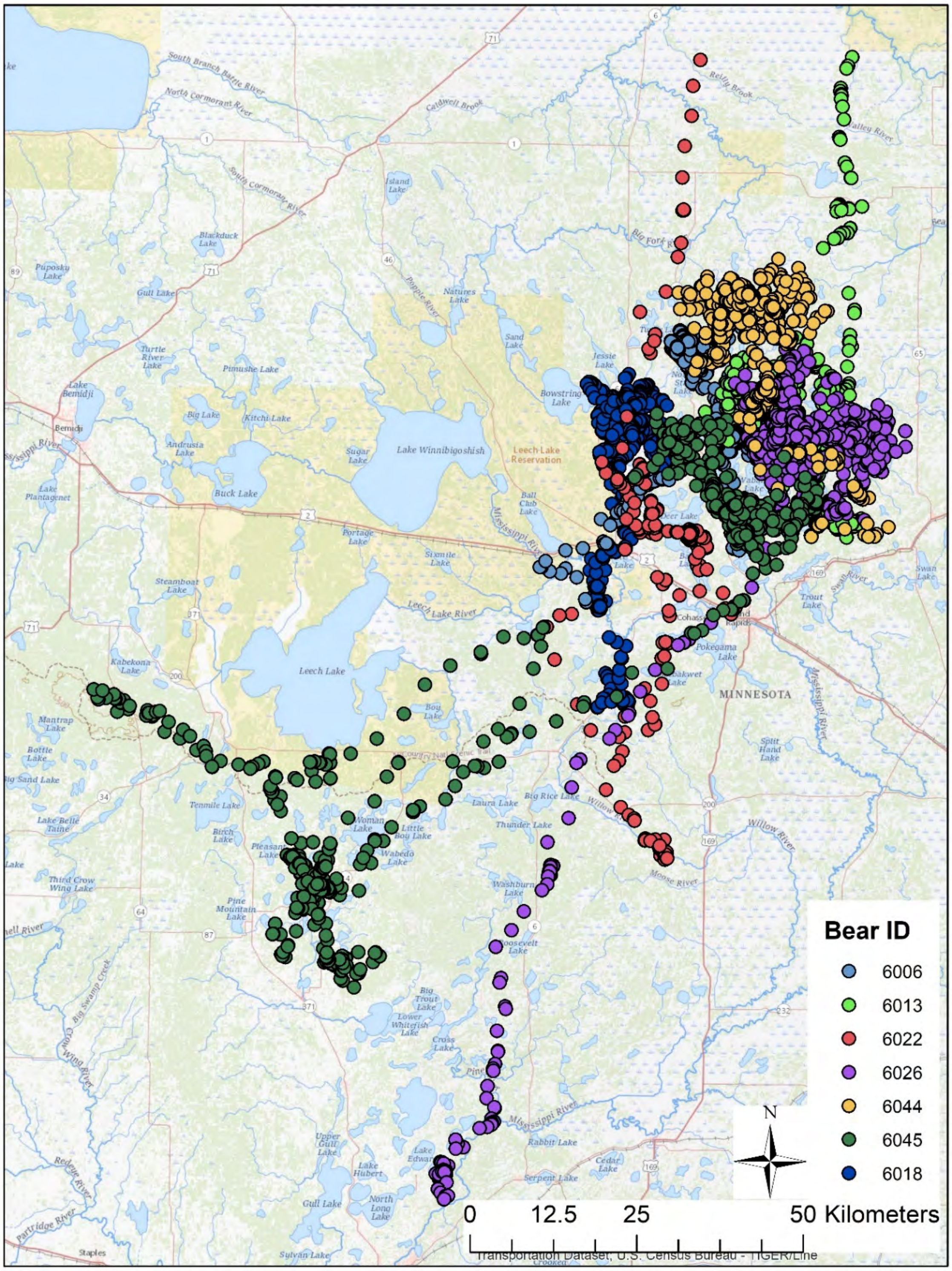

Figure 3. Fall migration movements of 7 GPS-collared bears (6 male, 1 female) from the CNF in 2017 in Minnesota.

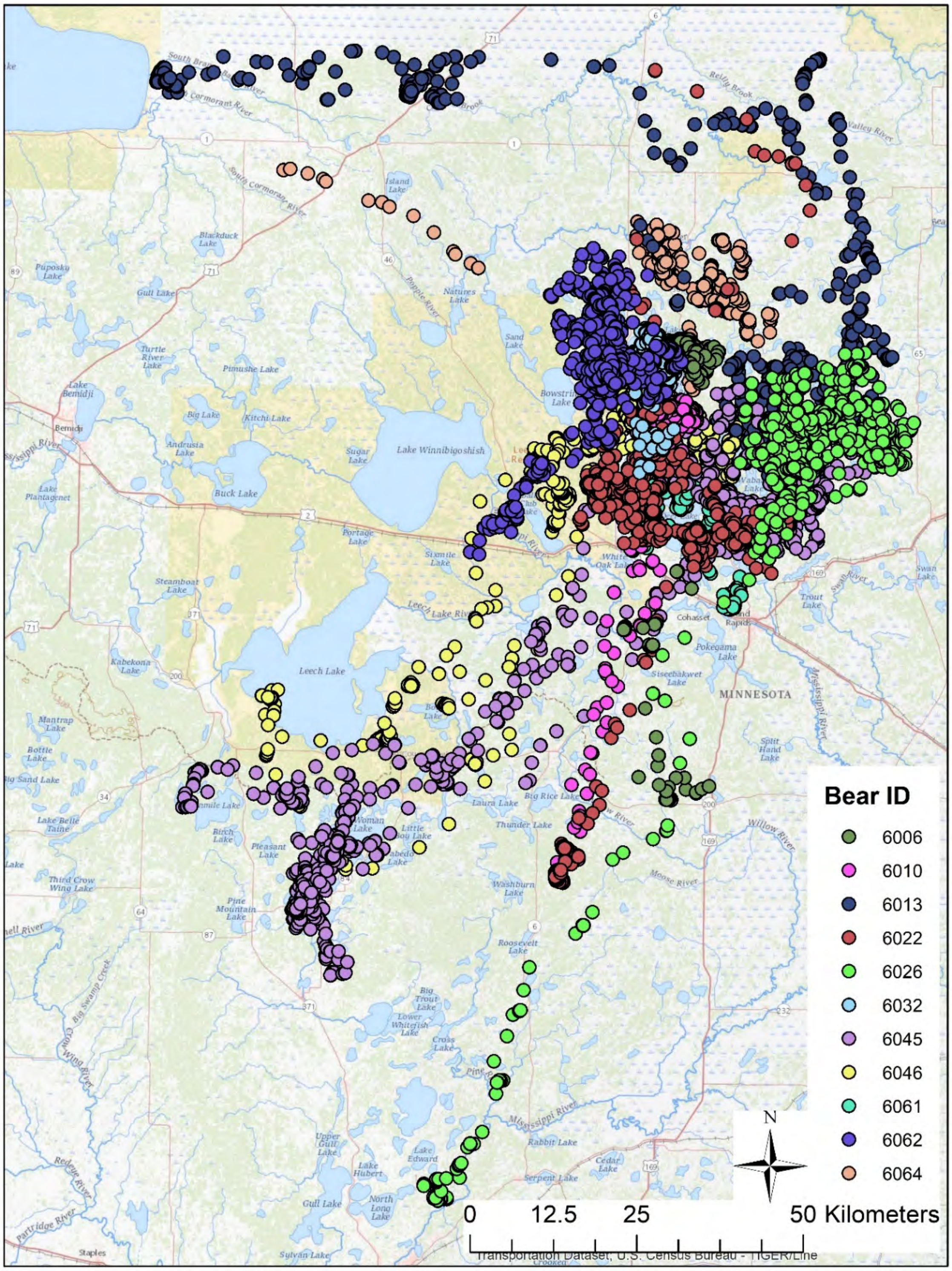

Figure 4. Fall migration movements of 11 GPS-collared bears (8 male, 3 female) from the CNF in 2018 in Minnesota.

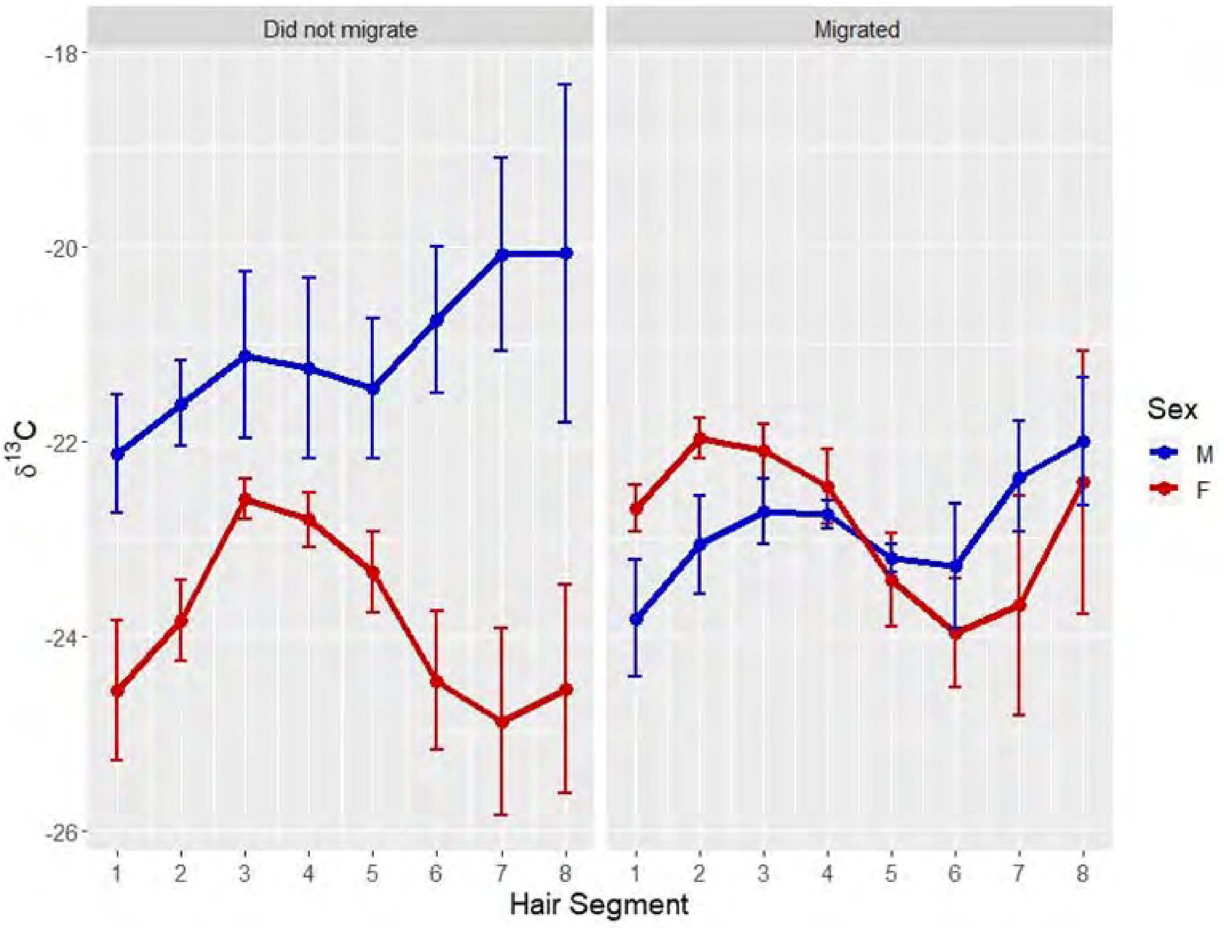

Figure 5. Isotopic mean values for $\delta^{13}C$ (±1 SE) for 8 equal-length hair segments (collected during hibernation) for male and female GPS-collared bears that did not (left panel) versus did (right) migrate to a fall feeding area outside their summer home range on the Chippewa National Forest study site during 2016 and 2017 in Minnesota. Hair segment 1 represents the oldest growth, which corresponds with the summer molt in late June to early July. Segments 7 and 8 are the newest growth, representing foods eaten in fall, prior to hibernation. The panel of migrating males excludes 1 bear that visited a cornfield in both 2016 and 2017 because his $\delta^{13}C$ values are so much higher than all other bears, which would greatly skew the mean. The data for this excluded bear are shown in Figure 6.

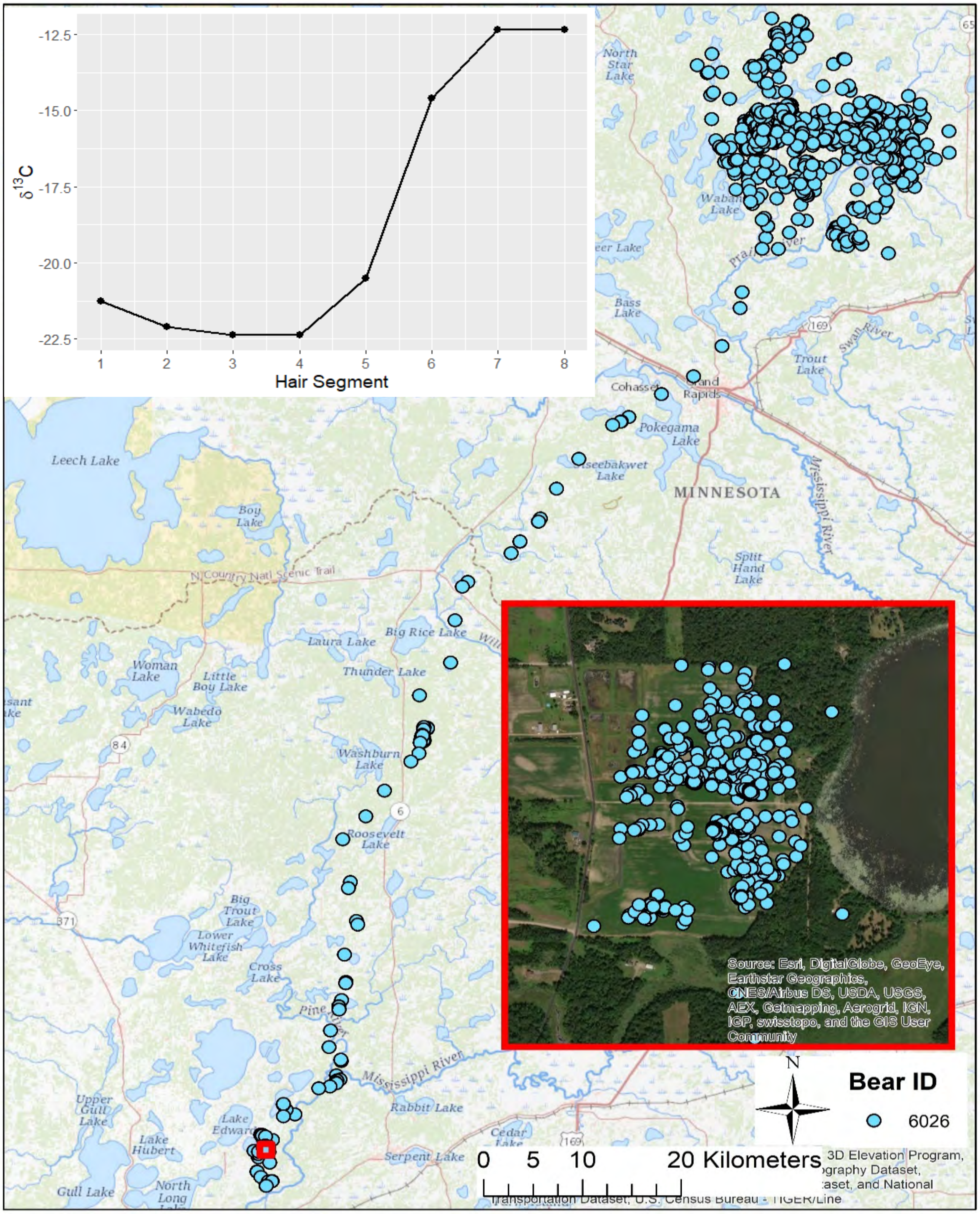

Figure 6. Fall migration of an adult male bear (#6026) from the CNF study site in 2017 in Minnesota. GPS locations and carbon isotope values (insets on right side and top, respectively) suggest that this bear primarily fed largely on corn in the fall. This bear migrated to the same area in 2016 and 2018 (data not shown). The $\delta^{13}C$ values of this bear in fall (hair segments 6–8) are much more extreme than all other bears (Figure 5).

Figure 7. A scat filled with corn, collected from a bear in the CNF during June 2018, indicating that although there are no large cornfields in the area (or ripe corn anywhere in Minnesota in June), the bear found corn, likely from a deer feeder at someone's house.

ESTABLISHING THE FEASIBILITY OF MAKING FINE-SCALE MEASUREMENTS OF HABITAT USE BY WHITE-TAILED DEER IN NORTHERN MINNESOTA, WINTERS 2017–2018 AND 2018–2019

Bradley D. Smith,[1] Glenn D. DelGiudice, and William J. Severud[2]

SUMMARY OF FINDINGS

The Minnesota Department of Natural Resources (MNDNR) began a 2-year pilot study of white-tailed deer (*Odocoileus virginianus*) habitat in northcentral and northeastern Minnesota during winters 2017–2018 and 2018–2019. This study is using cutting-edge global positioning system (GPS)-collar, remote sensing, and geographical information system (GIS) technologies to monitor and assess deer habitat use on 2 winter ranges. During March 2018−May 2019, we recovered 30 of 60 collars that had been fitted to free-ranging deer. These collars stored 34,758 locations on-board (100% fix-success) and successfully transmitted 27,177 (88%) GPS locations. The mean horizontal error was 16 m (± 0.07) and median error was 10 m. We classified a total of 604 and 1,012 cover type polygons at the stand level within the Inguadona Lake and Elephant Lake study sites, respectively. Spatially, dense conifer stands accounted for 12% and 23% and forage openings for 12% and 11% of the 2 study sites. During winter 2017−2018, collared deer using dense conifer stands were a mean of 146 m (± 8) and 240 m (± 5) from the nearest forage opening at the Inguadona and Elephant Lake sites, whereas they were a mean of 136 m (± 5) and 190 m (± 4) from the center of the stand they were using. Deer using forage openings were a mean of 247 m (± 7) and 179 m (± 7) to the nearest dense conifer stand at the 2 sites and 206 m (± 5) and 146 m (± 3) from the center of the opening in use. The mean area of dense conifer stands being used was 8 ha (± 0.2) and 47 ha (± 2) at Inguadona Lake and Elephant Lake, respectively. The ability to make fine-scale measurements of available habitat and how it is being used by deer will allow us to characterize the area, shape, juxtaposition, and arrangement of cover types and assess their value on winter ranges in a way that can be incorporated into integrated habitat and forest management prescriptions.

INTRODUCTION

Based on recommendations from the Office of the Legislative Auditor, the Minnesota Department of Natural Resources (MNDNR) developed a statewide white-tailed deer (*Odocoileus virginianus*) management plan to maintain deer numbers within management units near population goals for improved hunting and wildlife viewing (MNDNR 2018). Habitat management is a key component of this plan. Because winter is the nutritional bottleneck for northern deer, has the greatest impact on their natural survival rates, and may have a pronounced impact on spring fawning, wildlife managers focus most of their efforts on improving

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winter habitat as a means of positively influencing population performance (DelGiudice et al. 2002, 2006, 2013*a*). During winter 2017–2018, the MNDNR initiated a deer habitat study in northcentral and northeastern Minnesota, which is using a combination of global positioning system (GPS) collar, remote sensing, and geographic information system (GIS) technologies to better understand deer use of cover types and how area, shape, juxtaposition, and arrangement of conifer stands, forage openings, and other cover types influence their use and well-being (DelGiudice et al. 2017).

Previous studies of deer use of winter habitat and requirements in northern Minnesota, Canada, and the Great Lakes region that relied primarily on very high frequency (VHF) telemetry collars were restricted by inherent constraints compared to the use of more advanced GPS collars (Morrison et al. 2003; Potvin et al. 2003; DelGiudice et al. 2013*a*,*b*, 2017). Constraints included lower location-fix accuracy (≥95 m), limited temporal distribution of location-fixes (i.e., daytime locations only), fair weather flying only (i.e., safe flying conditions), relatively infrequent location-fixes (i.e., small numbers of seasonal locations per individual deer), and greater costs (time and monetary) required to collect the data (Pellerin et al. 2008, Kochanny et al. 2009). Because of these limitations, more precise information regarding winter habitat use by deer is essential to a more thorough understanding of their seasonal requirements and improved habitat management prescriptions.

Advancements in technology have allowed for notable enhancements in performance of GPS collars. With improved accuracy and precision of location-fixes and higher fix- and transmission-success rates, GPS collars facilitate collection of a plethora of near real time data, including habitat use and selection, movement rates, and interspecific interactions. Before collar deployment and assessing winter habitat use by deer, the influence of canopy closure and cover type on their performance required testing (Rempel et al. 1995, Dussault et al. 1999). Studies have shown that different habitats have diverse, adverse effects on GPS collar performance (e.g., accuracy, fix-success) associated specifically with varied canopy cover, stem density, basal area, and topography (Moen et al. 1996, Rempel and Rogers 1997, Dussault et al. 1999). However, recently, Telonics, Inc., a GPS collar manufacturer in Mesa, Arizona, incorporated programming for Quick Fix Pseudoranging (QFP) into their Globalstar Recon collars, which enhances their ability to obtain accurate location-fixes with as little as a 3−5-second view of a satellite constellation, compared to the 30–90 seconds required for a typical GPS location-fix. This is particularly valuable to studies of habitat use by deer and other ungulates.

Use of improved GPS collar technology has the potential to maximize collection of accurate location data not obtainable in studies using VHF telemetry or less sophisticated GPS collars, and to facilitate fine-scale measurements of habitat use. These data permit 24-hour monitoring of habitat use to better understand (1) individual variability associated with selection of forest cover types, and (2) how structure, size, shape, arrangement, interspersion, and perimeter (edge):area influence habitat use at the stand level (DelGiudice et al. 2017).

OBJECTIVES

1. To assess performance of GPS collars recovered from free-ranging deer, including horizontal error, fix-success rates, and reliance on QFP locations
2. To classify and inventory cover types on the Inguadona Lake (IN) and Elephant Lake (EL) study sites

3. To provide examples of fine-scale measurements of winter habitat use by deer

STUDY AREA

The study includes 2 deer winter range sites located in northern Minnesota's forest zone (Figure 1). The IN site is located in the northcentral part of the state in Cass county, 2 km south of the Chippewa National Forest border. This site is 46 km^2 and is a mosaic of state, county, and private land, with most of the latter occurring along lake shores. Reported pre-fawning deer densities in this area were 7–9 deer/km^2 (D'Angelo and Giudice 2016), and included both residential deer (year-round) and seasonal migrators (Fieberg et al. 2008). Topography is undulant with elevations of 400–425 m above sea level. The area is classified as part of the Pine Moraines region (MNDNR 2015), and includes uplands dominated by deciduous and mixed deciduous-conifer stands and lowlands dominated by mixed conifers. The uplands included red (*Pinus resinosa*), white (*P. strobus*) and jack pine (*P. banksiana*); paper birch (*Betula papyrifera*); black ash (*Fraxinus nigra*); red maple (*Acer rubrum*); balsam fir (*Abies balsamea*); and trembling aspen (*Populus tremuloides*; DelGiudice 2013*a*.). Lowlands included northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), balsam fir, and tamarack (*Larix laricina*).

The MNDNR calculates an annual Winter Severity Index (WSI) by accumulating 1 point for each day with an ambient temperature ≤ –17.7° C and an additional point for each day with a snow depth ≥38 cm during November–May. During 1981−2010, mean January temperature was –13° C and mean annual snowfall was 110 cm (MNDNR Climatology 2018). Over the past 8 years, WSI in the IN site indicated moderately severe or severe conditions in just 1 winter (2013–2014; WSI ≥140; MNDNR Climatology 2018).

The EL site, located in St. Louis county, is representative of the forest zone in northeastern Minnesota. The EL site is 76 km^2 and includes state, federal, county, and private land. Pre-fawning deer densities are lower than at the IN site and remain below management's goal of 3–5 deer/km^2 since the 2 severe winters of 2010–2011 and 2013–2014 (D'Angelo and Giudice 2016). Topography is undulant with elevations ranging from 400 to 450 m above sea level. The area is part of the Northern Superior Upland region (MNDNR 2015) with lowland conifer stands and upland conifer and mixed deciduous-conifer stands. The lowlands included northern white cedar, black spruce, and tamarack. The uplands included northern white cedar; balsam fir; red, white and jack pine; aspen; and paper birch (MNDNR 2015). Mean January temperature was –15° C and mean annual snowfall was 165 cm during 1981−2010 (MNDNR Climatology 2018). Since 2011, WSI reflected moderately severe to severe winters in 3 years over the past 8-year period (2010–2011, 2012–2013, 2013–2014; MNDNR Climatology 2018).

The primary source of natural mortality of adult deer at both study sites was wolf (*Canis lupus*) predation (DelGiudice et al. 2002). The most recent wolf population estimate (2017) in northern Minnesota was 2,856, or 4 wolves/km^2 (Erb et al. 2017). Black bear (*Ursus americanus*) and wolf predation have been major causes of fawn mortality (Kunkel and Mech 1994, Carstensen et al. 2009). As of 2014, the bear population of northern Minnesota was estimated at about 15,000 (Garshelis and Tri 2017).

METHODS

During winter 2017–2018, 10 adult (≥1.5 years) female deer were captured at each study site (Figure 1). A total of 19 deer were captured via net-gunning from helicopter (Hells Canyon Helicopters, Clarkston, Washington), and 1 deer at the IN site was ground-captured using a

Clover trap (DelGiudice et al. 2001). An additional 20 adult female deer were net-gunned at each of the 2 sites during 5–8 February 2019 (Figure 1; Quicksilver Air, Inc., Fairbanks, Alaska). Handling of animals consisted of blind-folding, hobbling, recording a rectal temperature (° C), measuring chest girth and hind leg length (cm), affixing an ear-tag to each ear, fitting a GPS collar, and administering a broad-spectrum antibiotic. Collars were programmed to obtain 1 location-fix every 2 hours during December–June and 1 location-fix every 4 hours during July–November. Location data were transmitted to a base station every 10 hours (maximum 6 locations per transmission). These collars included QFP programming, which will obtain a QFP location only when a GPS-fix is unsuccessful (see Introduction for more information on QFP); they are stored-on-board along with activity data collected every 5 minutes using an accelerometer. These data are retrieved and downloaded once collars are recovered.

We classified cover types at the forest stand level on the 2 study sites using a mirror stereoscope (Model MS27, Sokkia Co., Ltd., Tokyo) and 9”x 9” color infrared aerial photographs (1:15,840 scale) taken during October 2010 and 2012, to capture the color contrast of peak autumn foliage. We used National Agriculture Imagery Program (NAIP) coverage from 2013, 2015, and 2017 to adjust for changes over time (Smith et al. 2019). We also relied on Light Detection and Ranging (LiDAR), collected during May 2011 and April 2012 at EL and IN, respectively, at a resolution of 1 pulse per m^2. Derived products from the LiDAR point cloud were used to extract accurate tree heights and calculate percent forest canopy closure at the stand level, and to assist with delineating stand boundaries while digitizing the photointerpretation.

We delineated forest stands according to a classification system developed to assign dominant and co-dominant tree species, height class, and canopy closure class (for conifer stands, Figure 1; Smith et al. 2019). Forage sites—defined as open areas with regeneration <2 m in height—swamps and lakes were also delineated. We interpreted forest stands to a minimum size of 0.5 ha (DelGiudice et al. 2013*a*). Habitat *training sites* (i.e., for ground-truthing) were established at locations of fresh deer snow-urine (i.e., urine in snow) collection. The snow-urine samples were being collected and analyzed to assess the nutritional status of deer (DelGiudice et al. 1989, 2017), but these locations also allowed documentation of vegetation information relevant to the habitat classification system and aerial photointerpretation being conducted during winter 2017−2018 and 2018–2019.

We conducted a preliminary assessment of the feasibility of making fine-scale habitat measurements for a better understanding of the variability of individual use of cover types. We examined habitat use based on pooled location-fixes from winter 2017–2018 home ranges (Figure 3). We characterized cover types by structure (forest stands only), area, and arrangement of conifer forest cover and forage openings. Specifically, we analyzed 4,775 and 5,255 winter location-fixes at the IN and EL sites, respectively, and assigned the following characteristics: cover type being used; dominant and co-dominant tree species; stand height and canopy closure classes; distance (m) from fix to center of stand being used; distance (m) to nearest conifer cover class, if not in use; distance (m) to nearest opening/foraging site, if not in use; area of cover type in use; and edge:area ratio of cover type in use. We made measurements efficiently and accurately using the tool “Near” in the most recent version of ArcGIS (ArcGIS Pro 2.2.2, ESRI 2018). We calculated a 95% Kernel Density Estimate (KDE) of each deer’s home range during winters 2017−2018 and 2018−2019 using adehabitat (Calenge

2006) in R (R Core Team), which will facilitate comparisons of habitat composition within home ranges and between the 2 study sites.

RESULTS AND DISCUSSION

We recovered GPS collars from 30 deer (10 March 2018–31 May 2019), downloaded and analyzed the data as we had done during the pre-deployment collar-testing (Smith et al. 2019). The GPS transmission-success rate was 88% and fix-success of the 34,758 expected locations was 100%, with 3,903 (11%) being QFP fixes (Table 1). Overall mean horizontal error estimated by Telonics was 16 m (± 0.07) and median horizontal error was 10 m. The Telonics horizontal error estimate was slightly higher than the actual location error we calculated during our pre-deployment testing (Smith et al. 2019). Fix-success rates from recovered collars were consistent with rates from collars used in the pre-deployment test. Higher horizontal error estimates and lower transmission rates may be due to the increased frequency of dense conifer use on winter ranges (Morrison et al 2003; DelGiudice et al 2013*a, b*; Smith et al. 2019). The mean location error estimates are far superior (smaller) to those reported from previous GPS-collar studies (32−100 m; Rempel et al. 1995, Moen et al. 1996, Dussault et al. 1999). The addition of QFP locations is critical to our habitat study, providing 100% fix-success rates in important cover types that typically hinder location-fix success.

Overall, 95% KDEs of winter home ranges were highly variable on both sites during both winters (Tables 2 and 3). Home ranges tended to be larger for deer at IN than at EL during both winters, and were greater at both sites during winter 2018−2019 (Table 3) compared to 2017−2018 (Table 2); however, as assessed by 95% confidence intervals (mean ± 1.96 × SE), none of these differences were statistically significant. Home ranges for deer during winter 2017−2018 are depicted in Figure 3.

A total of 604 and 1,012 cover type stands were classified for the IN and EL sites, respectively (Figure 2). Dense conifer stands comprised 23% of the EL site compared to 12% of the IN site (Table 4). Northern white cedar dominated the dense conifer cover at the EL site. The IN site consists of more red pine plantations, which usually provide moderate canopy cover. The proportion of forage openings was similar at both sites, 11% and 12% at IN and EL (Table 4). When deer were using dense conifer stands, they were a mean of 146 m (± 8) and 240 m (± 5) from forage openings at IN and EL, respectively (Table 5). Mean distance to the center of the dense conifer stand in use was 136 m (± 5) and 190 m (± 4) (Table 5), and mean area of those stands was 8 ha (± 0.2) and 47 ha (± 2.0). Similarly, when deer were using forage openings, they were a mean of 247 m (± 7) and 179 m (± 7) from dense conifer cover at IN and EL, respectively, and 206 m (±5) and 146 m (±3) to the center of the opening in use (Table 5). The mean area of forage openings being used was 19 ha (± 0.4) and 8 ha (± 0.2). Deer were a mean of 35 m (± 0.5) and 38 m (± 0.5) to the nearest edge at the 2 sites. We did not find a difference in edge:area ratios for the different stand types being used (Table 5). Other landscape metrics also will be explored to better describe the shape, juxtaposition, and interspersion of cover types being used. Our preliminary measurements are consistent with findings of previous research suggesting dense conifer cover should be arranged within 355 m of forage openings (Morrison et al. 2003, Potvin et al. 2003, Beyer et al. 2010). Data represent only late-winter 2017−2018, but these preliminary analyses demonstrate the feasibility of making these fine-scale habitat measurements using our combined GPS collar, remote sensing, and GIS technologies. Analyses of habitat use data sets from winter 2018−2019 are in progress and will help capture additional individual and winter variability relative to varying snow depths and ambient temperatures.

Future work will include expanding the habitat classification and inventory of the 2 sites relative to the additional ~33,000 deer locations from winter 2018−2019, and we will continue analyses to further assess and understand winter habitat use and requirements as snow depth and temperature change during the progression of winter. Prior to winter 2020−2021, we plan to select a third study site, this one well within northeastern Minnesota's moose range. Habitat will be similarly classified and inventoried to allow examination and comparison of deer use at the stand level. Ultimately, the ability to make these fine-scale habitat measurements using GPS collars, remote sensing, and GIS as winters progress and vary annually will allow us to assess the area, shape, juxtaposition, and arrangement of dense conifer cover, forage openings, and other cover types to assist managers in formulating prescriptions that effectively integrate forest and habitat management strategies and practices.

ACKNOWLEDGMENTS

We thank B. Matykiewicz, B. Wagner, M. Pike and C. Marvet for their technical assistance with all aspects of the fieldwork. We gratefully acknowledge P. Backman and P. Coy for their skills and efforts applied to deer mortality investigations and collar recovery. P. Backman also shared valuable background knowledge of the Elephant Lake site. C. Humpal skillfully conducted laboratory analyses of our deer snow-urine and bone marrow samples. We also acknowledge the USDA Forest Service's LaCroix and Deer River Ranger Districts for providing housing during the winter field seasons. This project is supported by the Minnesota Department of Natural Resources Section of Wildlife and the Wildlife Restoration (Pittman-Robertson) Program. The Minnesota Deer Hunters Association provided supplemental funding for post-doctoral research assistance.

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Table 1. Summary statistics of location-fix data downloaded from global positioning system (GPS) collars[a] recovered from 30 adult (≥1.5 yr), female white-tailed deer during March 2018–May 2019, and associated performance metrics. Collars were deployed at the Inguadona Lake (IN) and Elephant Lake (EL) study sites, northcentral and northeastern Minnesota.

Collar ID	Study site	Mean horizontal error[b] (m)	Median horizontal error (m)	Overall fix-success rate (%)	Percent QFP locations	GPS-fix transmission-success rate[c] (%)
697084A	IN	17	10	100	18	71
697085A	IN	16	10	100	13	77
697086A	IN	15	10	100	10	80
697092A	IN	13	10	100	9	86
697095A	IN	14	10	100	0	91
697096A	IN	17	10	100	12	81
697098A	IN	15	10	100	18	74
699964A	IN	17	10	100	15	77
699966A	IN	16	10	100	13	61
706038A	IN	15	10	100	1	96
706039A	IN	18	10	100	0	99
706040A	IN	13	10	100	0	100
706057A	IN	14	10	100	0	99
706059A	IN	14	10	100	1	98
706070A	IN	12	10	100	1	96
697087A	EL	17	10	100	12	59
697090A	EL	17	10	100	15	77
697091A	EL	16	10	100	7	85
697093A	EL	18	10	100	15	72
697094A	EL	15	10	100	9	84
697097A	EL	17	11	100	17	67
699965A	EL	16	10	100	8	66
699967A	EL	17	10	100	14	73
706030A	EL	16	10	100	6	88
706036A	EL	18	12	100	10	86
706048A	EL	15	10	100	2	97
706052A	EL	12	10	100	0	100
706055A	EL	13	10	100	2	96
706062A	EL	24	14	100	2	96
706064A	EL	15	10	100	6	94
Overall		16	10	100	11	88

[a] Globalstar Recon GPS units (Model IGW-4660-4; Telonics, Inc., Mesa, Arizona).

[b] Horizontal error was calculated by Telonics and downloaded with the location data. Quick Fix Pseudoranging (QFP) locations were recorded only when a GPS-fix was unsuccessful.

[c] Transmission-success rate is calculated from the GPS locations only (i.e., QFP locations excluded).

Table 2. Overall mean (± SE) size (ha) of winter home ranges (95% Kernel Density Estimator) of adult (≥1.5 yr), female white-tailed deer at the Inguadona Lake and Elephant Lake study sites, northcentral and northeastern Minnesota, 12 March−1 May 2018.

Study site	*n*	Mean	SE	Range
Inguadona Lake	9	289	108	53−1,020
Elephant Lake	10	157	51	33− 584

Table 3. Overall mean (± SE) size (ha) of winter home ranges (95% Kernel Density Estimator) of adult (≥1.5 yr), female white-tailed deer at the Inguadona Lake and Elephant Lake study sites, northcentral and northeastern Minnesota, 1 November 2018−1 May 2019.

Study site	*N*	Mean	SE	Range
Inguadona Lake	24	358	77	60−1,209
Elephant Lake	26	267	73	5−1,473

Table 4. Cover type composition (% of study sites) of winter range of adult (≥1.5 yr), female white-tailed deer at the Inguadona Lake and Elephant Lake study sites, northcentral and northeastern Minnesota, winters 2017−2018 and 2018−2019.

Stand type	Elephant Lake	Inguadona Lake
Open conifer	4	3
Moderate conifer	5	10
Dense conifer	23	12
Hardwood	32	31
Mixed hardwood/conifer	12	3
Forage	12	11
Total area (km^2)	76	46

Table 5. Mean fine-scale measurements of winter habitat use by adult (≥1.5 yr), female white-tailed deer at the stand level at the Inguadona Lake and Elephant Lake sites, northcentral and northeastern Minnesota, 12 March−1 May 2018.[a]

Stand type[b]	Distance[c] to forage (m)	Distance to dense conifer (m)	Distance to edge (m)	Distance to center (m)	Area (ha)	Edge:area (m:m^2)
Inguadona Lake						
Open conifer	291	68	34	108	8	0.026
Moderate conifer	119	205	28	158	14	0.020
Dense conifer	146	NA	21	136	8	0.031
Hardwood	280	345	34	180	16	0.022
Mixed hardwood/conifer	367	557	28	84	4	0.025
Forage	NA	247	44	206	19	0.023
Elephant Lake						
Open conifer	282	150	41	204	18	0.018
Moderate conifer	74	180	22	100	5	0.025
Dense conifer	240	NA	48	190	47	0.020
Hardwood	312	278	41	277	30	0.016
Mixed hardwood/conifer	239	242	39	174	15	0.022
Forage	NA	179	23	146	8	0.029

[a] Locations are from winter 2017−2018 home ranges calculated using the 95% Kernel Density Estimator.

[b] Stand type indicates the stand being used. Open, moderate, and dense conifer represent the 3 canopy closure classes used for conifer stands only; open = 0−39%, moderate = 40−69%, and dense = 70−100%. Mixed hardwood stands are stands with hardwood as the dominant species and conifer as the co-dominant.

[c] Distances were measured using the Near tool in ArcGIS Pro 2.2.2. Mean calculations are based on all of the winter (12 Mar−1 May 2018) locations of all GPS-collared deer using these stand types within the respective study sites.

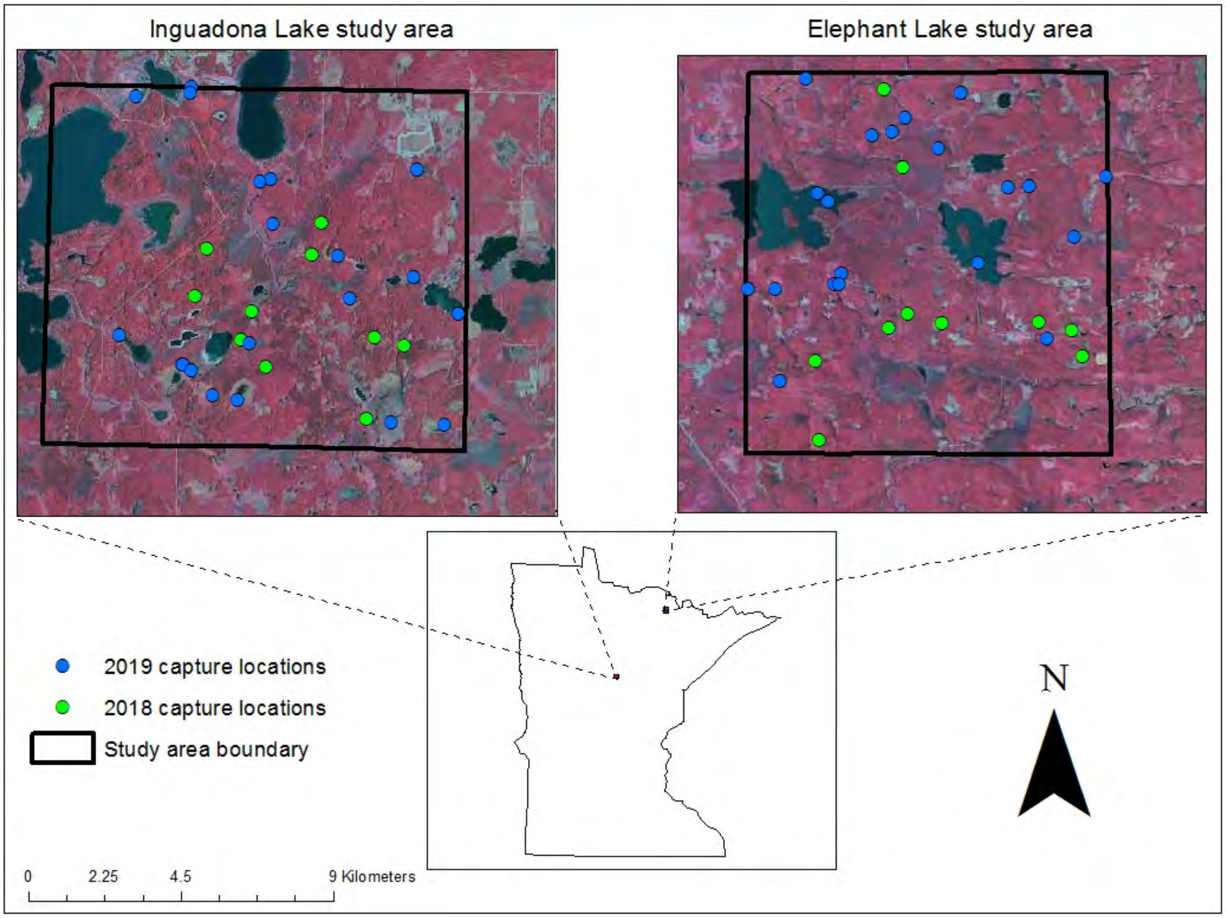
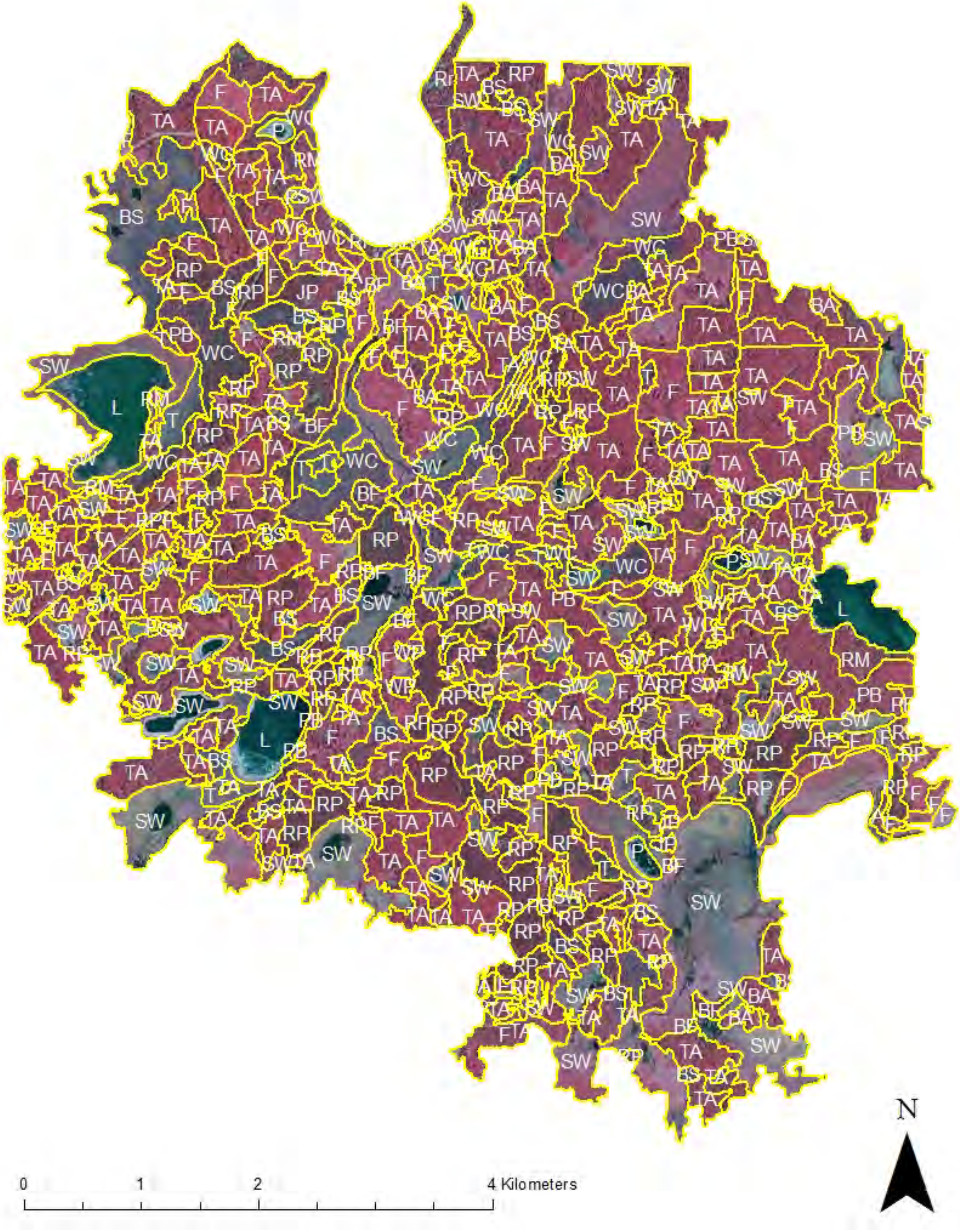


Figure 1. Helicopter net-gun capture locations of adult (≥1.5 yr), female white-tailed deer at the Inguadona Lake (46 km^2) and Elephant Lake (76 km^2) study sites, northcentral and northeastern Minnesota, 10−11 March 2018 and 5−8 February 2019. One deer was captured via Clover trap at Inguadona Lake in the first winter.


0
1
2
4 Kilometers
N

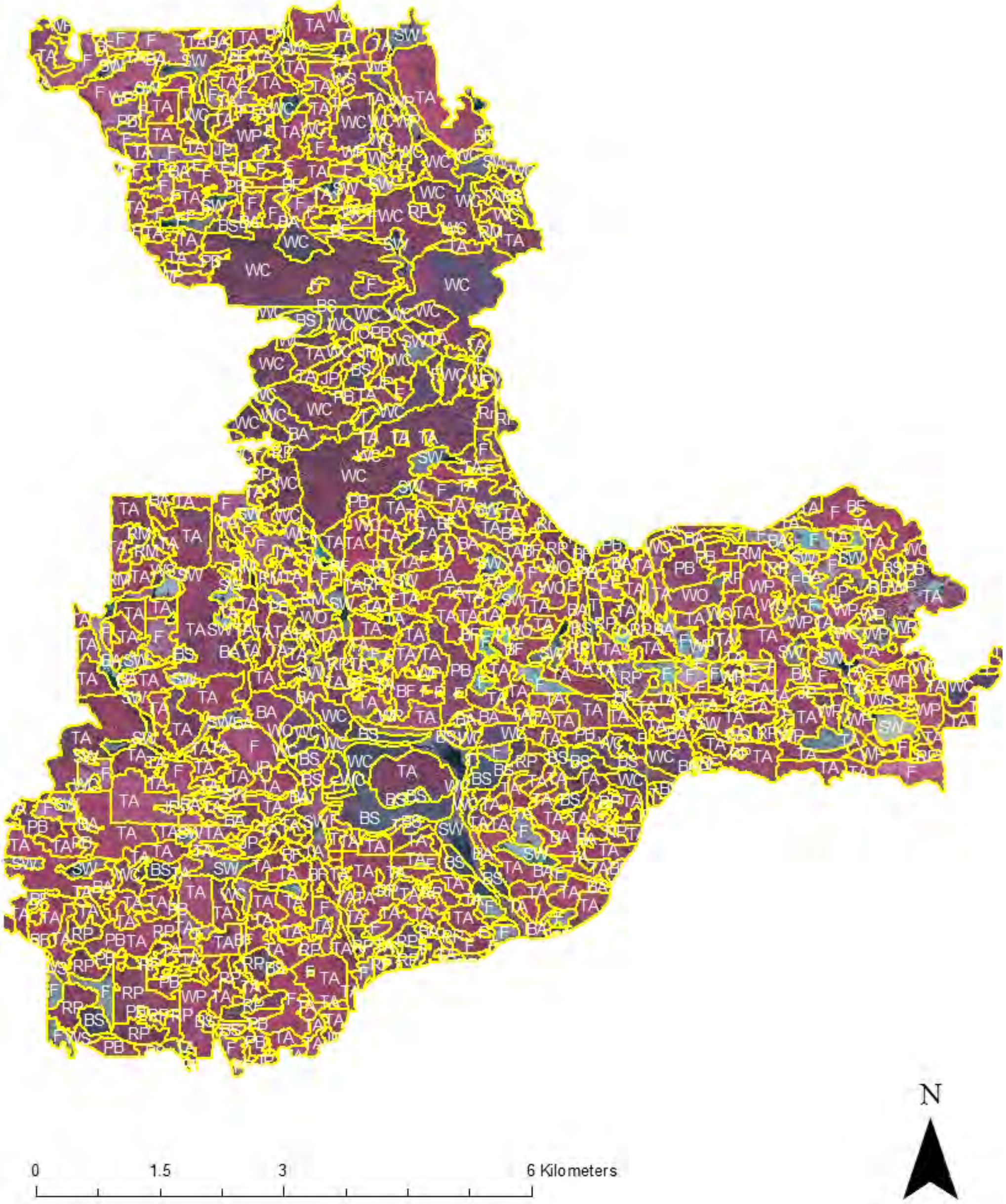
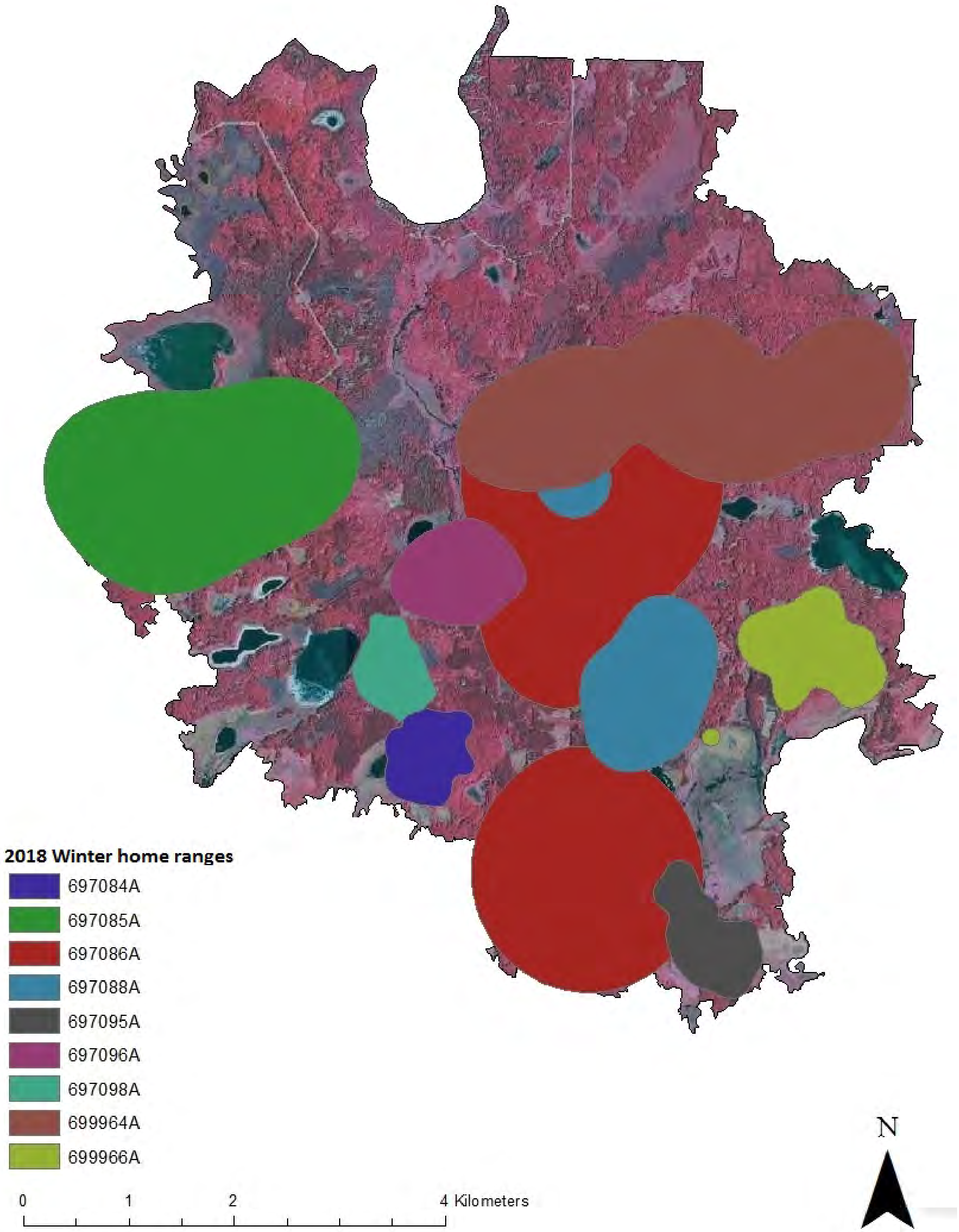


Figure 2. Classification and inventory of adult (≥1.5 yr), female white-tailed deer habitat at the stand level (only dominant tree species and forage are presented here) at the Inguadona Lake (top) and Elephant Lake (bottom) study sites, northcentral and northeastern Minnesota, winters 2017−2018 and 2018−2019, accomplished by air photointerpretation and Light Detection and Ranging (LiDAR). Stands and non-forest cover types were classified to a minimum size of 0.5 hectares. Cover type codes are presented in Table 1 in Smith et al. 2019.


2018 Winter home ranges
697084A
697085A
697086A
697088A
697095A
697096A
697098A
699964A
699966A
0
1
2
4 Kilometers
N

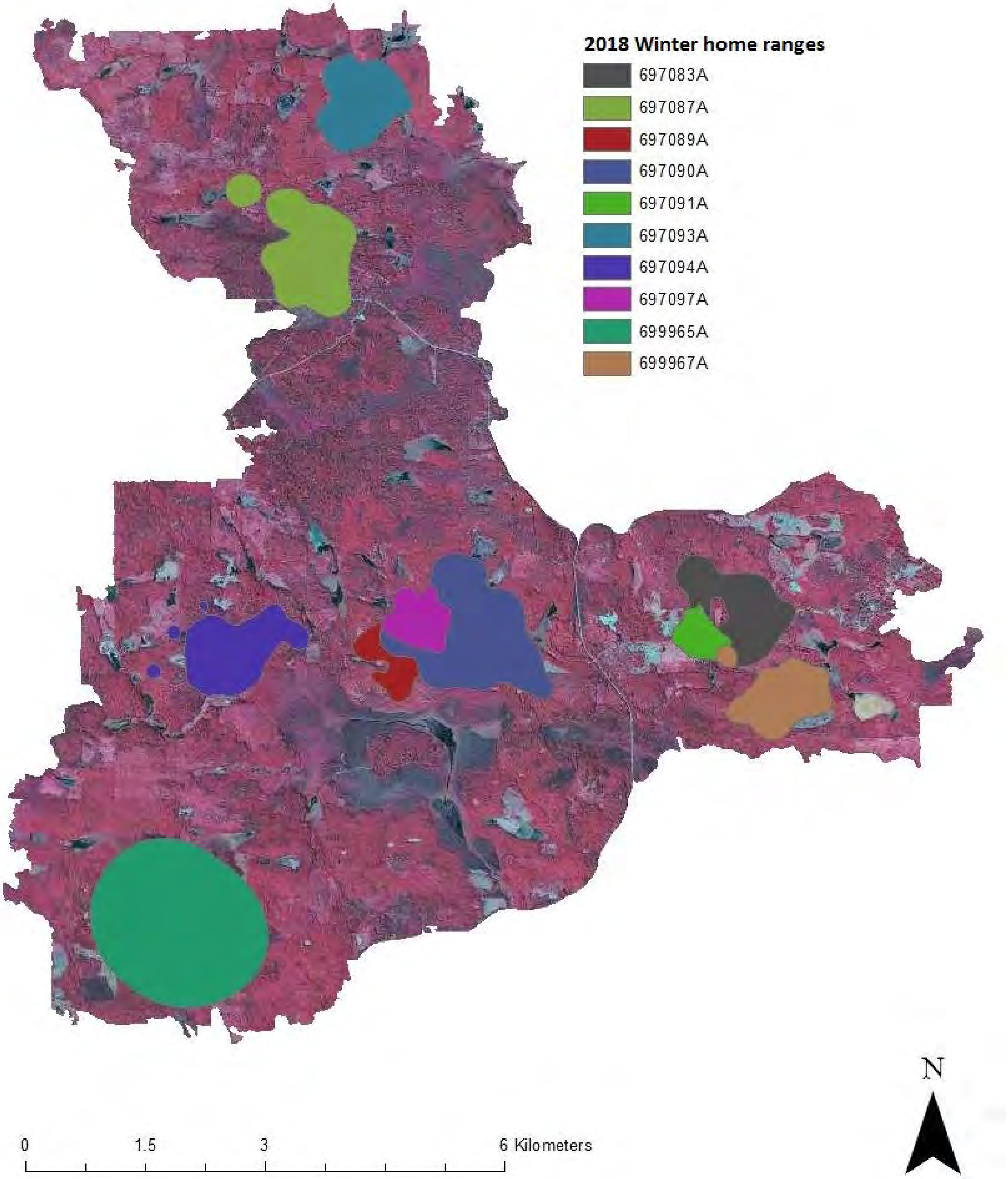


Figure 3. Winter home ranges (95% Kernel Density Estimate) of adult (≥1.5 yr), female white-tailed deer at the Inguadona Lake (top, *n* = 9) and Elephant Lake (bottom, *n* = 10) study sites, northcentral and northeastern Minnesota, 12 March−1 May 2018.

WINTER SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF WHITE-TAILED DEER IN NORTHERN MINNESOTA: AN UPDATE

Glenn D. DelGiudice, Bradley D. Smith,[1] and William J. Severud[1]

SUMMARY OF FINDINGS

Ongoing studies that examine the influences of environmental, intrinsic, and demographic factors on survival and cause-specific mortality rates of white-tailed deer (*Odocoileus virginianus*) have been critical to enhancing our understanding of population performance and to improving management. A recent evaluation report from the Office of the Legislative Auditor recommended that the "…DNR should conduct field research to collect and utilize more information about Minnesota's deer… and inform the department's vital rate estimates of deer births and deaths, and better reflect deer population dynamics" to improve our understanding of demographics and habitat requirements. Using cutting-edge global positioning system (GPS) collars, and remote sensing and geographic information system (GIS) technologies, we recently launched a study that will inform a level of understanding of habitat requirements and drivers of population performance required by managers to prescribe forest manipulations that best support population goals. Herein, our objectives are to compare winter survival rates and cause-specific mortality (and influential factors) of adult (≥1.5 yr) female deer residing on study sites in northcentral (Inguadona Lake) and northeastern (Elephant Lake) Minnesota. We predicted that survival, percent winter mortality, and the impact of wolf (*Canis lupus*) predation would be influenced by winter severity in a way that is consistent with our understanding of this relationship garnered from a previous long-term (1991−2005) study in northcentral Minnesota. The *natural* mortality rate during the first winter (2017−2018) was high; 6 of 19 (31.6%) GPS-collared adult female deer (3 at each site) were all killed by wolves during 10 April to 31 May 2018. Overall survival had decreased to 0.68 (95% confidence interval [CI] 0.50−0.93) by then. But this was a *pilot year*, so the survival estimate was limited by small sample sizes (10 collared deer per site) and represented only the late–winter season (12 March to 28 May 2018) due to delayed capture operations. However, during the second winter (2018−2019), with more than twice the sample size (n = 51), the *natural* mortality rate was also high (36.7%); 17 of 49 deer were preyed upon wolves and 1 by bobcat (*Felis rufus*) between 1 November 2018 and 20 April 2019 (cutoff for analysis for this annual report). Eight mortalities occurred at Inguadona Lake and 9 at Elephant Lake. The overall survival rate was 0.70 (95% CI = 0.57−0.86). The wolf predation rates during the 2 winters (31.6% and 34.7%) notably exceeded what we had expected based on the documented relationship of the previous long-term study. Typically, adult female deer enter winter in better physical condition than fawns and adult males, and thus have the highest probability of surviving winter. Our findings at least suggest that during both winters overall mortality rates at the population level, across sex and age classes, were likely higher than indicated by our adult female data. Ongoing federal protection of wolves in

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Minnesota limits Minnesota Department of Natural Resources (MNDNR) management options and has at least contributed to the estimated wolf population almost doubling from winter 1988−1989 (1,521 wolves) to the present (~2,900 wolves). Caution may be warranted in interpreting our preliminary findings, but they highlight the need for multi-year continuation of this study to better understand whether deer-habitat-wolf predation relationships have been changing since completion of the MNDNR's previous long-term study, a potentially significant consideration relative to implementation of the state's recently developed deer management plan.

INTRODUCTION

Studies that have examined the influences of extrinsic (e.g., habitat, predation, and human activities), intrinsic (e.g., age, sex, condition), and demographic (e.g., density) factors on survival and cause-specific mortality rates have enhanced our understanding of the dynamics of white-tailed deer (*Odocoileus virginianus*) and other ungulate populations in northern Minnesota and elsewhere (Nelson and Mech 1986*a*,*b*; Fuller 1990; Bartmann et al. 1992; DelGiudice 1998; Gaillard et al. 2000; DelGiudice 2002, 2006). A long-term (1991−2005) study of female deer, the reproductive component of populations, reported that the relative risk of mortality was strongly related to the severity of winter conditions in northcentral Minnesota (DelGiudice et al. 2002, 2006). Indeed, the risk of death increased as winters progressed, and by the end of winter was at least 10 times greater during the most severe winter (1995−1996) compared to the mildest winter (1990−1991). That study also documented that the relative risk of death of female deer by natural causes was consistently greater than by all other causes of mortality (e.g., hunting), and that wolf (*Canis lupus*) predation, directly related to snow depth, was the primary cause of mortality. Furthermore, the risk of mortality by wolves increases sharply for adults after 6 years of age.

The Office of the Legislative Auditor (OLA) recently issued an evaluation report of the Minnesota Department of Natural Resources' (MNDNR) management of the state's deer population (OLA 2016). This document focused on improving population estimates but emphasized that improved habitat management should be a key component of a new statewide deer management plan to establish and meet population goals. The OLA report recommends that the "…DNR should conduct field research to collect and utilize more information about Minnesota's deer… and inform the department's vital rate estimates of deer births and deaths, and better reflect deer population dynamics" to improve our understanding of demographics and habitat requirements. Partially in response to the OLA report, we recently launched a study using cutting-edge global positioning system (GPS)-collar, remote sensing, and geographic information system (GIS) technologies that will provide a level of understanding of habitat requirements and drivers of population performance (survival and reproduction) required by managers to prescribe forest manipulations that best support population goals (DelGiudice et al. 2017, 2019).

Fieldwork for this study was initiated during winter 2017−2018. In addition to an overall objective of establishing the technical feasibility of making fine-scale measurements of habitat use by deer at the forest stand level (see Smith et al. 2019), we monitored winter survival and cause-specific mortality as a means of assessing habitat quality (DelGiudice et al. 1989*a*,*b*) and to update input for state modeling of northern deer populations. We predicted that the influence of winter severity on crude winter mortality and the wolf predation rate of adult female deer would be consistent with findings from our previous long-term study (DelGiudice et al. 2002, 2006).

OBJECTIVES

1. To compare winter survival rates of adult female deer residing on the Inguadona Lake and Elephant Lake sites, located in northcentral and northeastern Minnesota, respectively, where there are differences in winter severity, habitat composition, and deer density; and

2. To determine specific causes of mortality and contributing factors.

STUDY AREA

As discussed above and in a companion research summary in this issue (Smith et al. 2019), the study included 2 deer winter range sites, Inguadona Lake (46 km^2) in northcentral and Elephant Lake (76 km^2) in northeastern Minnesota. These sites allow natural comparisons of potential influences of differences in winter severity, habitat composition, and deer density on habitat use and requirements and associated vital estimates. D'Angelo and Giudice (2015) reported pre-fawning deer densities of 7−9 and 3−5 deer/km^2 in the vicinity of the Inguadona Lake and Elephant Lake sites, respectively. The MNDNR calculates a winter severity index (WSI) throughout the state by accumulating 1 point for each day with an ambient temperature ≤−17.7° C and an additional point for each day when snow depth is ≥38 cm during 1 November−31 May. Generally, winters with maximum WSI values (by 31 May) <100, 100−180, and >180 are assessed as mild, moderately severe, and severe, respectively, relative to impacts on deer survival; however, multiple factors may influence this interpretation annually and geographically (DelGiudice et al. 2006). Maximum WSI at Inguadona Lake at the end of winter was 60 and 113 during 2017−2018 and 2018−2019, respectively. The maximum WSI values at Elephant Lake were 130 and 121. Additional details addressing site boundaries, location, topography, forest composition, long-term weather, and wolf and black bear (*Ursus americanus*) densities are provided in Smith et al. (2019).

METHODS

During 10−11 March 2018, 19 adult (≥1.5 yr old), female white-tailed deer were captured by net-gunning from helicopter (Hells Canyon Helicopters, Clarkston, Washington), 9 and 10 on the Inguadona Lake and Elephant Lake sites, respectively. A tenth deer was captured on 25 February by Clover trap at Inguadona (Clover 1956). Except for this deer (which was immobilized with xylazine and ketamine, and reversed with yohimbine), all deer were physically restrained for handling (Smith et al. 2019). Similarly, 20 adult female deer were captured on each site by net-gunning from helicopter (Quicksilver Air, Inc., Fairbanks, Alaska) during 5−8 February 2019 and handled following the same protocol. All deer were fitted with a Globalstar Recon GPS collar (Model IGW-4660-4; Telonics, Inc., Mesa, Arizona). We programmed all collars to collect 1 location-fix every 2 hours during December−June and 1 location-fix every 4 hours during July−November. The collar's mortality sensor relies on a 3-axis accelerometer. The unit samples this accelerometer every second. "Active" is recorded when the reading of any of the 3 axes changes by more than 0.3 g (gravitational force) since the last active second. Less than 5 accumulated seconds of activity during the previous 8-hours causes the unit to detect mortality, which triggers the collar to send a mortality notification through the Globalstar satellites, followed by sending an email to our team. This launches our field investigation. This also increases the very high frequency (VHF) pulse rate to notify researchers in the field. Our field investigations included a thorough search for site and carcass evidence to determine the specific cause of mortality. When available, we collected a mandible (to extract a fourth incisor) and femur (or other long bone as necessary) to age the deer to the year and assess body condition (Gilbert 1966, Mech and DelGiudice 1985). We conservatively assigned ultimate cause of death as "capture-related" when the mortality occurred within 7 days of capture, regardless of the proximate cause (e.g., wolf-kill; DelGiudice et al. 2002, 2006) or handling method (i.e., physically or chemically immobilized).

We calculated Kaplan-Meier survival estimates using the R package *KMsurv* (R Core Team 2017). We examined relationships between the WSI and percent winter mortality by simple linear regression analyses in Excel (Version 14.0.7153.5000, Microsoft Corporation 2010).

RESULTS AND DISCUSSION

Overall survival of our GPS-collared adult female deer decreased markedly to 0.68 (95% confidence interval [CI] 0.50–0.93) and 0.70 (95% CI 0.57−0.86) during winters 2017−2018 and 2018−2019, respectively, and was consistently similar at Inguadona Lake and Elephant Lake (Figure 1). The overall crude natural mortality rate (reported for consistency with the estimate from the previous study, but not to replace use of the aforementioned survival rates) was 31.6% (6 of 19 deer) and 36.7% (18 of 49 deer) during winters 2017−2018 and 2018−2019, respectively (Figure 2). These rates exclude 1 capture-related mortality during each season and an unrecovered hunter-harvest deer during the second year. Wolf predation rates were 31.6% (6 of 19) and 34.7% (17 of 49) during the 2 winters (Figure 2). Wolf predation accounted for all of the natural mortality during winter 2017−2018 and all but 1 of the mortalities during 2018−2019, which was a deer killed by a bobcat (*Felis rufus*). Many of the wolf-killed deer were in poor condition as indicated by a mean marrow fat content of 68.9% (± 9.1 [SE], range = 3−91%, *n* = 12; Watkins et al. 1991).

Given the low to moderate maximum WSI values during winters 2017−2018 and 2018−2019 at Inguadona (60 and 113) and Elephant Lake (130 and 121), reflecting mild to moderately severe conditions, these female mortality rates, overall and due to wolf predation specifically, were extraordinarily high (Figure 2). Winter wolf predation on northern deer is directly related to snow depth. Deepening snow increasingly impedes deer mobility and escape (deer have a heavier weight-load-on-track than wolves) and steadily compromises their energy balance and endurance (Moen 1976, Nelson and Mech 1986*b*, DelGiudice 1998, DelGiudice et al. 2002). Consequently, most winter mortalities, both in our previous long-term and present studies, occurred during March−May, when snow cover is deepest and body condition is poorest.

The preponderance of evidence in this study suggests that poor condition was a noteworthy contributing factor to the high mortality rates by wolf predation. Given that winter conditions were not particularly severe at either site in either year, as indicated by maximum WSI values, this prompts consideration of the role of winter habitat deficiencies compromising the ability of deer to adequately fulfill their biological requirements and avoid wolf predation as the season progressed. Indeed, it is additionally noteworthy that winter survival and wolf predation rates were similar on both sites during both winters. Increasing concerns about the limited quantity and quality of habitat (e.g., dense conifer cover) on deer winter range in northern Minnesota, in large part, prompted the need for the present study (T. Rusch, L. Petersen, and P. Backman, MNDNR, Section of Wildlife, personal communication), as did OLA's evaluation report strongly recommending continued deer research that enhances the MNDNR's understanding of vital estimates and associated population dynamics for improved management (OLA 2016). Typically, adult female deer enter winter in the best physical condition (i.e., most replete fat reserves), and thus have the lowest hazard risk or highest probability of surviving winter compared to fawns and adult males (Mautz 1984, Robbins 1993, DelGiudice et al. 2002). This at least suggests that overall winter mortality rates during these 2 years could have been even higher than indicated by our preliminary adult female data. Ongoing federal protection of wolves in Minnesota limits the MNDNR's management options, and consequently, has at least contributed to the estimated population almost doubling from winter 1988−1989 (1,521 wolves), just as the previous deer study was initiated (winter 1990−1991), to the present (~2,900 wolves, Erb and DonCarlos 2009, Erb et al. 2017). Caution may be warranted in interpreting our preliminary findings, but the accumulating evidence critically highlights the need for multi-year continuation of this study to better understand whether these complex deer-habitat-wolf predation relationships have been changing since completion of the previous long-term study (DelGiudice et al. 2002, 2006, 2007, 2013*a*,*b*).

ACKNOWLEDGMENTS

We appreciate the dedicated efforts of B. R. Matykiewicz, B. L. Wagner, M. Pike, and C. Marvet as field biology technicians whose duties included capturing and handling deer, collecting snow-urine samples for nutritional assessments, and gathering vegetation data for habitat classification and inventories. We thank P. Backman for her support with study site selection, deer mortality investigations, and overall logistics, and for her helpful knowledge of the Elephant Lake study site. We also appreciate the effort and technical skills of C. A. Humpal in the laboratory, analyzing snow-urine samples and deer bone marrow samples. Finally, we acknowledge the United States Forest Service LaCroix and Deer River Ranger Districts for providing housing during the winter field seasons. J. Erb loaned us field equipment as needed. This project is supported by the Minnesota Department of Natural Resources Section of Wildlife and the Wildlife Restoration (Pittman-Robertson) Program. The Minnesota Deer Hunters Association provided supplemental funding to support valuable post-doctoral research assistance.

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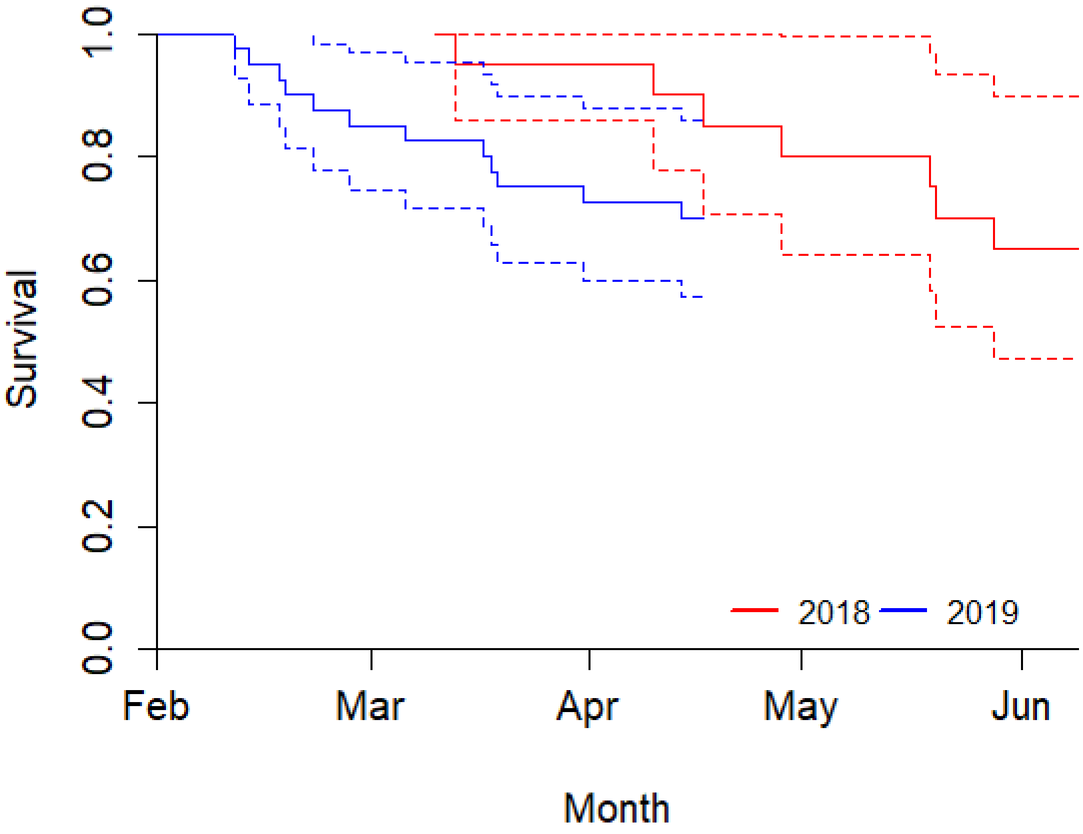

Figure 1. Kaplan-Meier survival curves of adult (≥1.5 yr) female white-tailed deer from date of capture, 10−11 March 2018 (*n* = 20) and 5−8 February 2019 (*n* = 40), to 31 May 2018 and 20 April 2019 (cutoff date for analysis included in this report), respectively, at the Inguadona Lake and Elephant Lake study sites (pooled), northcentral and northeastern Minnesota. Dashed lines represent 95% confidence intervals.

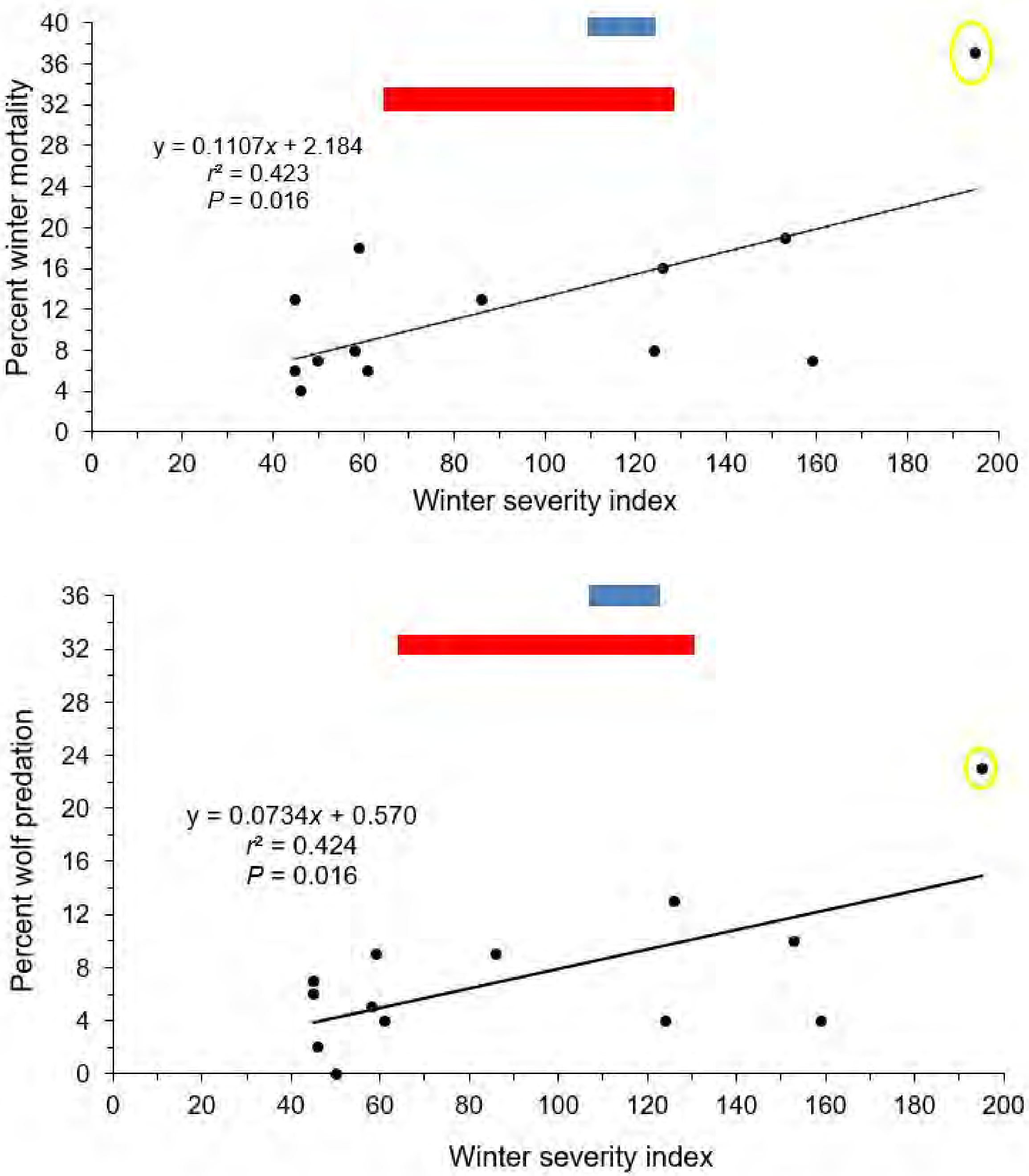

Figure 2. Comparison of crude winter mortality (top) and wolf predation rates (bottom) of adult (≥1.5 years) female white-tailed deer at the Inguadona and Elephant Lake sites (pooled) in northern Minnesota during winters 2017−2018 (red) and 2018−2019 (blue) to the long-term relationship of these rates for adult female deer to maximum winter severity index (WSI) in northcentral Minnesota during winters 1990−1991 to 2002−2003 (DelGiudice et al. 2006). The red bars represent deer (*n* = 19) during winter 2017−2018 and the blue bars represent deer (*n* = 49) during winter 2018−2019 and span WSI values for the 2 study sites each year. One capture-related mortality was excluded during each winter, and 1 unrecovered hunter-harvested deer was excluded from winter 2018-2019. The circled data point represents historically severe winter 1995−1996.

EVALUATING AMERICAN MARTEN HABITAT QUALITY USING AIRBORNE LIGHT DETECTION AND RANGING (LIDAR) DATA[1]

Michael Joyce[2], Dissertation Abstracts

CHAPTER 1: INDIVIDUAL DETECTION OF COARSE WOODY DEBRIS USING AIRBORNE LIDAR

SUMMARY OF FINDINGS

Coarse woody debris (CWD) is an essential component of forest ecosystems that provides habitat for diverse species, functions in water and nutrient cycling, and can be a potential surface fuel in wildfires. CWD detection and mapping would enhance forestry and wildlife research and management but passive remote sensing technologies cannot provide information on features beneath forest canopy, while field-based CWD inventories are not practical for mapping CWD over large areas. Airborne light detecting and ranging (LiDAR) is a remote sensing technology that provides detailed information on three-dimensional vegetation structure that could overcome limitations of field-based inventories. Our objectives were to evaluate whether airborne LiDAR could be used to detect individual pieces of CWD. We measured 1,968 pieces of CWD at 189 field plots from 2015 to 2016. We acquired high-density (~24 first returns/m^2) LiDAR data in 2014 and filtered out canopy and sub-canopy returns using a height threshold based on field measurements of CWD and used height-filtered data to determine which field-measured pieces of CWD were visible in the resulting point cloud. CWD pieces detected constituted 50% of plot CWD volume, and there was a strong, positive correlation between total plot CWD volume and volume of detected pieces ($r = 0.96$). Overall, we detected 23% of the individual pieces of CWD we measured. Large pieces of CWD were most likely to be detected, with the majority of pieces ≥30 cm diameter or ≥13.9 m long detected. Canopy density, shrub density, and forest type did not influence detection probability. CWD detection rates increased from 1 pulses/m^2 to 16 pulses/m^2, and CWD detection rate was constant from 16 pulses/m^2 to 24 pulses/m^2. Our results demonstrate that airborne LiDAR can be used to detect CWD. LiDAR-based detection and mapping of CWD will be most useful for applications that focus on larger and longer pieces of CWD or applications focused on total CWD volume.

CHAPTER 2: MEASURING FOREST CHARACTERISTICS USING LIDAR: HOW WELL DO LIDAR-DERIVED REGRESSION MODELS PERFORM WHEN APPLIED TO NEW DATA?

SUMMARY OF FINDINGS

Light detection and ranging (LiDAR) is an active remote sensing technology that has been used increasingly to measure topographic and vegetative structure for forestry and wildlife applications. Measurement of vegetation characteristics that cannot be measured directly from LiDAR data is typically accomplished through LiDAR forest inventory modeling, in which a statistical model is developed that relates LiDAR-derived explanatory variables to field-measured response

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[1] This project was sponsored by John Erb (MN DNR) and Ron Moen (UMD) with extensive support from Barry Sampson (MN DNR).

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variables. The successful use of LiDAR forest inventory models for forestry and wildlife applications relies on development of statistical models that provide accurate and precise estimates of the response variables of interest, particularly when the model is imputed across the landscape. Because the goal of LiDAR forest inventory modeling is generally to identify the best statistical model for prediction from many potential candidate models, investigators have often used exhaustive model-fitting techniques to select final models. Furthermore, not all investigators have adequately addressed potential issues associated with overfitting and collinearity while developing LiDAR forest inventory models. Our objectives were to evaluate how well LiDAR forest inventory models created with multiple regression techniques performed when used to make predictions on new data. We created regression models for 5 response variables: basal area, average tree diameter, maximum tree diameter, quadratic mean tree diameter, and tree density. We used cross validation and bootstrapping techniques to evaluate model performance on new data. Our results demonstrated that models generally performed well on new data, that including collinear variables did not substantially reduce model performance relative to models without collinear variables, and that model performance on new data varied among response variables. Taken together, our results suggest that LiDAR forest inventory models are likely to perform well when imputed across the landscape of interest, but also highlight the importance of including explicit testing of models using new data or internal validation techniques during model development phases.

CHAPTER 3: SPATIAL AND ANNUAL VARIATION IN HARVEST MORTALITY RISK FOR AMERICAN MARTENS

SUMMARY OF FINDINGS

Understanding animal-habitat relationships is a common focus of ecological research. Most studies of animal habitat selection focus on describing characteristics of sites used by animals relative to availability. Although these studies have improved our understanding of animal-habitat relationships, descriptive habitat analyses based on animal presence generally do not consider survival or reproductive output of the population and are unable to distinguish between the relative qualities of habitats used by different individuals within a population. Differences in mortality risk or reproductive success among areas of otherwise similar habitat can result in functional differences in habitat quality. Harvest is a major source of mortality for many wildlife species. Our objectives were to investigate spatial and annual variation in harvest mortality risk for American martens for application to a fitness-based understanding of habitat quality. We used data from radio-collared martens and harvest statistics to test whether harvest risk was influenced by marten age, sex, accessibility to trappers, and harvest levels. Harvest risk was higher for males than females and negatively correlated with average distance from roads. There was a weak positive effect of harvest intensity on harvest risk, but age class did not affect harvest risk. Areas with suitable habitat near roads may function as attractive sink-like habitat due to elevated mortality risk from harvest, while suitable habitat farther from roads may function as source-like habitat and be of higher overall quality. We suggest that spatial variation in harvest mortality risk is an important factor that contributes to population structure, source-sink dynamics, and gene flow.

CHAPTER 4: THE ROLE OF HABITAT STRUCTURE IN PREDATION OF AMERICAN MARTENS BY BOBCATS AND OTHER INTRAGUILD PREDATORS

SUMMARY OF FINDINGS

Intraguild predation occurs in many carnivore communities and can have profound effects on trophic interactions, community structure, and population regulation. Habitat can play an important role in modulating the frequency and outcome of encounters between intraguild predators and intraguild prey. Fine-scale habitat structure can reduce susceptibility of intraguild prey by providing concealment, escape cover, and refugia, or can increase predation risk by impeding detection of potential predators. American martens are small mustelid carnivores that are susceptible to predation by several predator species. Although predation risk is often used to explain habitat selection patterns of martens, there are few direct tests of the role of habitat structure on interactions of martens with their predators. Our objectives were to examine the role that habitat structure plays in mediating interactions between martens and predators. Because bobcats are frequent predators of martens, we focused our analysis primarily on marten-bobcat interactions. We used light detection and ranging (LiDAR) data to measure canopy and understory characteristics and compared characteristics of sites where martens were killed by predators to non-mortality telemetry locations. Sites where martens were killed by bobcats were closer to non-forested habitat and were near more non-forested habitat than non-mortality locations. The structural characteristics and types of non-forested habitats associated with mortality sites varied among carnivore species. Our results provide direct evidence that martens experience elevated mortality risk when in or near non-forested areas without tree canopy, including shrublands, wetlands, and young/regenerating forest.

WINTER NUTRITIONAL RESTRICTION AND DECLINE OF MOOSE IN NORTHEASTERN MINNESOTA, WINTERS 2013–2019

Glenn D. DelGiudice, William J. Severud,[1] Tyler R. Obermoller, and Bradley D. Smith[1]

SUMMARY OF FINDINGS

The moose (*Alces alces*) population in northeastern Minnesota has declined an estimated 53% from 2006 to 2019. As was the case in northwestern Minnesota's moose decline during mid-1980−2007, a number of complex ecological relationships between undernutrition, pathogens, predation, and environmental factors (e.g., habitat, temperature) are likely exerting pressure on moose and contributing to this recent decline. Nutrition is centrally related to our understanding of all other aspects of wildlife ecology, including population performance. Winter nutritional restriction of moose and other northern ungulates may be physiologically assessed by serial collection and chemical analysis of fresh urine in snow (snow-urine); urea nitrogen:creatinine (UN:C) ratios have shown the greatest potential as a metric of winter nutritional status with values <3.0, 3.0–3.4, and ≥3.5 mg:mg being indicative of moderate (normal), moderately severe, and severe nutritional restriction, respectively. During 4 January–22 March 2013–2019, we collected annual totals of 123, 307, 165, 189, 160, 332, and 190 moose snow-urine samples, respectively, and mean seasonal UN:C ratios were 3.7, 2.9, 2.9, 3.5, 3.7, 2.6, and 2.3 mg:mg for the 7 winters, respectively. The mean population UN:C ratios for winters 2013, 2016, and 2017 were above the threshold indicative of severe nutritional restriction (i.e., a starvation diet) and accelerated body protein catabolism. During 2014, 2015, 2018, and 2019 the corresponding values reflected moderate nutritional restriction. Most indicative of the unique severity of nutritional restriction in 2013, nearly one-third of all samples collected yielded UN:C ratios $\geq$3.5 mg:mg.

Perhaps the ultimate value to management of nutritional assessments of free-ranging animals is realized when the findings can be related to the performance and dynamics of the population and other ecological factors challenging that performance. Through 2017, our population-level nutritional assessments were closely tracking separate population estimates ($r^2 = 0.75$) of moose in northeastern Minnesota. However, this relationship weakened with the inclusion of the 2018 and 2019 population estimates and snow-urine data. This likely was attributable in part to the notable uncertainty associated with the annual population estimates and its continued statistical stability, but apparent decline. Biologically, the mean population-level UN:C ratio (2.6 and 2.3) and relatively low incidence of snow-urine samples with UN:C ratios indicative of severe nutritional restriction (14.8% and 6.8%) were consistent with the population's continued stability. Although nutritional restriction varied among the 7 winters, data suggested a level of deprivation not supportive of population growth. Climate change, reflected by the heat stress index for moose, and variation in winter conditions, as indexed by the Winter Severity Index (WSI), were not related to nutritional restriction of moose. For the first 5 winters (the only years for which survival estimates are available), we documented that the level of severe nutritional restriction was inversely related ($r = -0.86$) to variation of *natural* winter survival of global positioning system (GPS)-collared adult moose. While these relationships do

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not substantiate cause-and-effect, presently it provides the best preliminary empirical evidence that inadequate winter nutrition at the population level is intricately related to the declining trajectory of moose numbers in northeastern Minnesota.

INTRODUCTION

Declines in regional populations of moose (*Alces alces*) along the southern periphery of their global range have been common in recent decades (Timmerman and Rodgers 2017). In northeastern Minnesota the estimated 2019 population (4,180 moose) is 53% less than in 2006 (8,840 moose, DelGiudice 2019), exhibiting a trajectory similar to that documented previously for moose in northwestern Minnesota, where the population decreased from about 4,000 in the mid-1980s to <100 moose by 2007 (Murray et al. 2006). Furthermore, mean annual mortality rates of collared adult moose associated with the declines were similarly high (21%) in the northwest and northeast (Murray et al. 2006; Lenarz et al. 2009; R. A. Moen, unpublished data). In northwestern Minnesota, malnutrition and pathogens were identified as important factors influencing the population's decreasing trajectory (Murray et al. 2006). In northeastern Minnesota a recent (2013–2017) aggressive study of global positioning system (GPS)-collared adult moose reported a mean annual mortality rate of 14.7%, with health-related factors (e.g., parasites, disease) accounting for about two-thirds of the deaths, wolf (*Canis lupus*) predation for one-third, and complex interactions between the 2 categories being well-documented (Carstensen et al. 2018). In the earlier studies, climate change (i.e., warming temperatures) was implicated in both population declines (Murray et al. 2006; Lenarz et al. 2009, 2010).

Temperature-survival relationships are complex, and indicate that climate change can directly and indirectly impact ungulate populations (Bastille-Rousseau et al. 2016, Davis et al. 2016, Street et al. 2016). Moose are particularly well-adapted to cold climates, but temperatures that exceed "heat stress" thresholds of 14^{o} to 24^{o} C during summer and $–5^{o}$ C during winter may increase metabolic rates, induce energy deficits, and hasten deterioration of body condition (Renecker and Hudson 1986, 1990; Broders et al. 2012; McCann et al. 2013). These thresholds may be influenced by exposure to solar radiation and wind (Renecker and Hudson 1990, McCann et al. 2013). Nutritional and health status (e.g., disease, parasites), behavioral responses (e.g., altering movement, foraging, and bedding patterns), and quality of available habitat have the potential to affect the animal's ability to mitigate negative impacts from heat stress (Van Beest et al. 2012, McCann et al. 2016, Street et al. 2016).

Energy balance is central to animal fitness, which is critical to survival and reproduction, the 2 drivers of population performance (Robbins 1993). The natural "nutritional bottleneck" of winter typically imposes the greatest challenge to the supply side of energy budgets of moose and other northern ungulates (Mautz 1978, Schwartz and Renecker 2007). Gestation at this time increases energetic and nutritional demands, particularly during late-winter and early-spring (Robbins 1993). Although moose are generally well-adapted to this seasonal nutritional deprivation, elevated ambient temperatures exceeding heat stress thresholds, coupled with the influence of other compromising extrinsic factors (e.g., pathogens, poor quality forage and low availability of thermal cover, densities of conspecifics or other nutritionally competing species) can exacerbate energy deficits and associated consequences relative to adult and juvenile survival, subsequent reproductive success, and population dynamics (Robbins 1993; DelGiudice al. 1997, 2001).

Winter nutritional restriction of moose and other northern ungulates can be physiologically assessed at the population level by serial collection and chemical analysis of fresh urine voided in snow (snow-urine; DelGiudice et al. 1988, 1997, 2001; Moen and DelGiudice 1997; Ditchkoff and Servello 2002). Urea nitrogen (interpreted as a ratio to creatinine, UN:C), the end-product of protein metabolism, is one of many chemistries investigated for its value as a physiological metric of the severity of nutritional restriction (DelGiudice et al. 1991*a*,*b*, 1994). In healthy moose, urinary UN:C values decrease (N conservation) in response to diminishing intake of crude protein and digestible energy, but as dietary restriction and negative energy balance become more severe and fat reserves are depleted, ratios increase to notably elevated values in response to accelerated net catabolism of endogenous (body) protein. Snow-urine UN:C ratios exhibited differential effects of a winter tick (*Dermacentor albipictus*) epizootic and habitat differences on the severity of nutritional restriction of moose on Isle Royale, Michigan, and were strongly related to dynamics of the population, including a pronounced decline and recovery to historically high numbers (DelGiudice et al. 1997).

OBJECTIVES

1. To determine how nutritional restriction varies annually and as winter progresses; and

2. To examine potential relationships between the severity of nutritional restriction and the winter heat stress index (HSI) for moose, seasonal survival rates of GPS-collared adult moose, and annual population estimates.

We hypothesized that increasing winter ambient temperatures, exceeding the HSI threshold, are contributing to the severity of nutritional restriction and energy deficit of moose. We also predicted that the severity of nutritional restriction would be inversely related to the performance of the population in northeastern Minnesota, primarily through its effect on adult survival and possibly calf production. Findings will set the stage for additional work assessing nutritional relationships of moose to variations in habitat and other factors.

STUDY AREA

We assessed winter nutritional restriction of moose within a 6,068-km^2 study area located between 47°06'N and 47°58'N latitude and 90°04'W and 92°17'W longitude in northeastern Minnesota (Figure 1). Including bogs, swamps, lakes, and streams; lowland stands of northern white cedar (*Thuja occidentalis*), black spruce (*Picea mariana*), and tamarack (*Larix laricina*); and upland balsam fir (*Abies balsamea*), jack pine (*Pinus banksiana*), white pine (*P. strobus*), and red pine (*P. resinosa*), this region has been classified as Northern Superior Upland (Minnesota Department of Natural Resources [MNDNR] 2015). Trembling aspen (*Populus tremuloides*), white birch (*Betula papyrifera*), and conifers are frequently intermixed.

Wolves (*Canis lupus*) and American black bears (*Ursus americanus*) are predators of moose (Fritts and Mech 1981, Severud et al. 2015) with recent densities estimated at 4.0 wolves and 23 bears/100 km^2 (Garshelis and Noyce 2015, Erb et al. 2017). White-tailed deer (*Odocoileus virginianus*) are managed at pre-fawning densities of <4 deer/km^2, and are the primary prey of wolves in most of northern Minnesota (Nelson and Mech 1986, DelGiudice et al. 2002). The MNDNR assesses winter severity (1 November–31 May) by a Winter Severity Index (WSI), calculated by accumulating 1 point for each day with a temperature $\leq -17.7^{\circ}$ C (0° F, temperature-day) and 1 point for each day with snow depth ≥38 cm (15 inches, snow-day), for a potential total of 2 points per day. Maximum WSI values varied markedly across moose range, with values of 35–160, 184–245, 54–152, 31–142, 50−159, 50−179, and 100−159 for winters

2012–2013 to 2018–2019, respectively (Minnesota State Climatology Office 2018). Mean daily minimum and maximum temperatures varied markedly during November–April from 2012–2013 to 2018–2019 at Ely, Minnesota (Midwestern Regional Climate Center 2019; Figure 2). The heat stress index (HSI_{Min} and HSI_{Max}, see Figure 3) for moose during the "cold season" (November–March) was calculated by daily accumulation of degrees Celsius exceeding -5^{o} C for the maximum and minimum ambient temperatures, respectively (Renecker and Hudson 1986).

METHODS

We collected fresh snow-urine specimens of moose during 3 January−22 March 2013− 2019. Our field team drove (by truck or snowmobile) a route of approximately 201 km to distribute the sampling throughout the study area (Figure 1). Field technicians were not restricted to this route, rather they could deviate, particularly on foot, as dictated by the presence of fresh moose sign (e.g., tracks, urine specimens, pellets). Each field team used handheld GPS units loaded with several land coverages (R. G. Wright, Minnesota Information Technology @ MNDNR, Section of Wildlife) and a Superior National Forest map (US Forest Service) to navigate in the field.

Generally, sampling was conducted within 7 days of a fresh snowfall, most often within 2–4 days, so that we could associate urine chemistry data and nutritional assessments with specific narrow temporal intervals. Upon observing fresh moose sign, technicians tracked the individual(s) on foot as necessary until they found a fresh snow-urine specimen. The objective for the collections was to sample primarily adult (≥1.5 yr) moose (indicated by track and bed size). This was not particularly challenging, because by this time of year calves comprised only 13–17% of the population (DelGiudice 2019). We focused primarily on the adult age class to facilitate optimum comparability of physiological assessment data.

Specimens were collected and handled as described by DelGiudice et al. (1991*a*, 1997). A GPS waypoint was recorded for each snow-urine specimen collected. Date of the most recent snowfall and comments describing the presence of moose or other sign in the area also were recorded.

Snow-urine specimens were analyzed for UN and C (mg/dL for both) by a Roche Cobas Mira auto-analyzer (Roche Diagnostics Systems, Inc., Montclair, New Jersey) in the Forest Wildlife Populations and Research Group's laboratory. We used 0.1 and 3.0 mg/dL as reliable threshold concentrations for accurately measuring C and UN, respectively, with our auto-analyzer; samples with values below these thresholds were excluded (C. A. Humpal, MNDNR, personal communication). Data were compared as UN:C ratios to correct for differences in hydration, body size, and dilution by snow (DelGiudice et al. 1988, DelGiudice 1995).

Winter (January–March) was divided into 6, 2-week sampling intervals (~1–15 January, 16–31 January, 1–14 February, 15–28 February, 1–15 March, and 16–31 March). Sample sizes for the snow-urine collections varied by interval due to variability of weather (i.e., snow conditions), equipment availability, logistical challenges, and ease of finding samples. Most of the UN:C data are reported by the entire winter or by sampling interval as means (± standard error). Additionally, based on past work, urinary UN:C values were assigned to 1 of 3 levels of nutritional restriction: moderate or "normal," <3.0 mg:mg; moderately severe, 3.0–3.4 mg:mg; and severe, ≥3.5 mg:mg (DelGiudice et al. 1997, 2001, 2010). We report the percentage of samples with UN:C values falling within each of these categories. We examined relationships between proportions of snow-urine specimens with UN:C values indicative of severe nutritional restriction (≥3.5 mg:mg) and populations estimates, seasonal survival, and HSI by simple linear regression analyses in Excel (Version 14.0.7153.5000, Microsoft Corporation 2010).

RESULTS AND DISCUSSION

During January–March 2013–2019, 1,479 urine specimens from moose were collected and analyzed to assess nutritional status at the population level. Specifically, annual totals of 123, 307, 165, 189, 160, 332, and 190 moose snow-urine samples, respectively, were collected during 5–6, 2-week sampling intervals using our designated routes and were adequately concentrated for chemical analysis. The greater number of samples collected during 2014 was largely due to the early and prolonged deep snow cover, whereas during 2018, the greatest number of samples was attributable to more intense sampling during the middle of the winter sampling period.

Overall, mean UN:C ratios were 3.7, 2.9, 2.9, 3.5, 3.7, 2.6, and 2.3 mg:mg for winters 2013 to 2019, respectively (Figure 4). The mean population UN:C ratio for entire winters 2013, 2016, and 2017 were above the threshold indicative of severe nutritional restriction or a starvation diet (≥3,5 mg:mg) and accelerated body protein catabolism. But the elevated mean UN:C of 2016 and 2017 were influenced largely by a small number of collected samples that exhibited very high UN:C ratios indicative of a moribund condition (≥22.0 mg:mg), whereas during 2013, nearly one-third of all samples collected yielded UN:C ratios indicative of severe nutritional restriction (≥3.5 mg:mg, Figure 5). According to Figure 5 and the summed proportions of samples with UN:C ratios indicative of moderately severe and severe restriction, it appears that winters 2013 to 2015 were the most nutritionally challenging to moose, whereas during winters 2016 to 2019, UN:C ratios more consistently indicated moderate restriction to be most common.

Mean urinary UN:C ratios by 2-week interval of winter 2013 indicated that nutritional restriction was normal or moderate during late-January, but became severe throughout February and early-March, and was still assessed as moderately severe in late-March (Figure 6). As severe nutritional restriction of moose progresses with winter, those animals may be under-sampled as some eventually die directly from undernutrition or because they have become predisposed to another proximate cause of mortality (e.g., wolf predation, Carstensen et al. 2018), and those still alive urinate less, which is a physiological mechanism to conserve water and electrolytes. However, the percentage of samples with urinary UN:C ratios indicative of severe nutritional restriction peaked (73.3%) in early-February and remained relatively high through late-March (36%) during 2013 (Figure 7). Such elevated values have been associated with long-term fasting in controlled nutrition studies of captive white-tailed deer and starvation of free-ranging elk (*Cervus elaphus*), bison (*Bison bison*), and moose (DelGiudice et al. 1991*a*, 1994, 1997, 2001). The percentage of snow-urine specimens in 2013 with UN:C ratios indicative of moderately severe to severe nutritional restriction throughout the winter was 45.5% (Figure 5).

During 2014, mean urinary UN:C ratios in all 2-week intervals, except early February, remained just below the moderately severe category (Figure 6), and the percentage of samples with ratios indicative of severe nutritional restriction gradually decreased as this winter progressed (Figure 7), either due to an easing of conditions restricting access to forage or because severely stressed individuals were being under-sampled, which may be most plausible as previously explained. Adverse effects of the late, but prolonged conditions of winter 2013, including warm temperatures, may have contributed to the high spring-summer calf loss and absence of the need for dams to lactate (Severud et al. 2015). This also may have allowed surviving animals to rebound nutritionally more quickly and to fare better during winter 2014. This would not be unlike the documented effects on the nutritional status and survival of northern Minnesota deer during the consecutive severe winters of 1996 and 1997 (DelGiudice et al. 2006; G. D. DelGiudice, unpublished data). Overall in winter 2014, UN:C values of 64% of the collected snow-urine samples from moose classified nutritional restriction as moderate (normal), whereas 36% reflected moderately severe to severe restriction, which was less than in 2013 (Figure 5). Similar to winter 2014, severe nutritional restriction of moose was not as prevalent in 2015 as in 2013, but it was up slightly compared to 2014 (Figure 5). However, a higher percentage of moose appeared to be experiencing moderate or normal restriction and a smaller percentage moderately severe restriction than in 2013 and 2014 (Figure 5). Rapidly diminishing snow cover prevented collection of snow-urine samples or assessments during the last 2 weeks of March 2015, certainly a positive factor relative to moose nutrition at that time. Interestingly, the greatest percentage of samples (87%) reflecting moose experiencing moderate nutritional restriction occurred during winter 2018–2019, which corresponded with an apparent but not statistically significant increase (38%) in the estimated moose population. In an attempt to better understand within-winter temporal patterns of nutritional restriction across years, we will be conducting more detailed analyses of UN:C data relative to the temporal and spatial distributions of sampling, progressive winter conditions, and sample size.

According to maximum WSI values, winter 2014 was the most severe of the 7 in northeastern Minnesota's moose range, followed in order of severity by 2018, 2019, 2013, 2017, 2015, and 2016. Although the WSI numbers have value for annual comparisons of winter conditions, this WSI formula has far greater relevance to the size and energetics of white-tailed deer than for the much larger moose, which are not hindered as much by deep snow (DelGiudice et al. 2002, 2006; Schwartz and Renecker 2007). Furthermore, while the accumulation of snow-days and temperature-days has proven significant relative to the survival of white-tailed deer (DelGiudice et al. 2002), actual snow depth, its temporal occurrence and duration, may be of equal or greater importance for moose and deer (Telfer and Kelsall 1984, DelGiudice 1998, DelGiudice et al. 2002, Schwartz and Renecker 2007). During 2013, conditions became severe during mid- to late-winter; consequently, a high number of snow-days did not accumulate, but the season was prolonged. Severe nutritional restriction of moose in 2013 was most similar to that which occurred in moose during several winters (1988–1990) on Isle Royale, also associated with severe winter tick infestations and a steep population decline (DelGiudice et al. 1997). Abundant evidence from the field in the MNDNR's ongoing studies similarly indicated that the winter tick infestation of moose in northeastern Minnesota was notably more severe during winter 2013 than in any of those that followed through 2019 (Carstensen et al. 2014; M. Carstensen, MNDNR, personal communication).

Perhaps the ultimate value of nutritional assessments of free-ranging animals to management comes when the findings are related to the performance and dynamics of the population and other ecological factors challenging that performance (DelGiudice et al. 1997, Cook et al. 2004). During the first 5 years, our nutritional assessments closely tracked population estimates of moose from the annual aerial survey ($r^2 = 0.75$, DelGiudice et al. 2018). With the addition of the 2018 survey results and nutritional assessment data the relationship weakened markedly ($r^2 = 0.27$), but with inclusion of the 2019 data, that relationship has strengthened somewhat ($r^2 = 0.38$, Figure 8). This is likely due to several factors. First, there is a great deal of uncertainty (wide 90% confidence intervals) associated with the annual estimates of moose numbers (DelGiudice 2019). Second, there are spatial and temporal incongruences between data collection for the population estimates versus for the nutritional assessments. Relatively-speaking, the ~9-day aerial survey provides a population estimate that is a winter "snapshot," whereas sample collections for the nutritional assessments span early to late winter (90 days). Finally, we do not yet understand the timeframe associated with potential biological effects on these moose of variation in nutritional restriction within a season or the specific mechanisms involved. It is noteworthy that our population estimates indicate that moose numbers have been relatively stable since 2012, with the exception of 2013. During this winter the population appeared to decrease abruptly; however, general survey conditions were poor, and we could not quantify their potential influence as an artifact on the point estimate. Of the 7 winters, 2013 was the only one in which a severe winter tick infestation occurred and had uniquely strong nutritional consequences for moose at the population level, reflected by urinary UN:C ratios (Figure 5). As described earlier, this has been similarly documented on Isle Royale (DelGiudice et al. 1997). The incidence of samples with UN:C indicative of moderately severe to severe restriction was greatest during winters 2013−2015, whereas during 2016−2019, nutritional restriction has remained remarkably moderate and stable. Six points is the minimum number required for valid statistical assessments of these relationships (F. Martin, Department of Applied Statistics, University of Minnesota, personal communication). Presently, what appears most clear across years is that elevated UN:C values suggest a level of nutritional deprivation not supportive of positive population performance or growth. That said, the lowest percentage of annual samples indicative of moderately severe and severe nutritional restriction occurred during winter 2018−2019 and were associated with an increase (38%) in the estimated number of moose. Continued monitoring of population performance and dynamics and winter nutritional status, and primary factors influencing them, will increase our sample size and continue to improve our understanding of the mechanisms involved.

During 2013 to 2015, warming winter temperatures were strongly associated with variation in the nutritional status of moose. As the January and winter HSI_{Max} values increased, the incidence of severe nutritional restriction of moose increased ($r^2 \geq 0.93$, DelGiudice and Severud 2017). We believe that may have led to many of these animals becoming more vulnerable to predation and various health-related causes of mortality (DelGiudice et al. 1997, Carstensen et al. 2015). However, unexpectedly in 2016 and 2017, the years with the highest winter HSI values calculated with daily maximum (958 and 833) or minimum (220 and 194) ambient temperatures were associated with the smallest percentage of samples with UN:C ratios reflecting severe nutritional restriction and greatest percentage indicative of moderate restriction (Figure 5). Overall, the relationship between winter HSI_{Max} or HSI_{Min} and severe nutritional restriction collapsed. Absence of apparent relationships continued through winter 2019. However, the incidence of severe nutritional restriction at the population level remained inversely related to variation of *natural* winter survival ($r = -0.86$, $P = 0.061$) but not significantly related to winter-summer survival ($r = -0.65$, $P = 0.231$) of GPS-collared adult moose (Figure 9). Survival data collection was temporally more consistent (than population survey data) with data collection for the nutritional assessments, and both data sets have a high level of certainty. Unfortunately, completion of the 5-year study of adult moose did not permit a winter survival estimate for the sixth year. However, importantly, the 5-year relationship of winter nutritional restriction to winter survival supports a reasonably strong biological explanation of the winter nutritional influence on the population trajectory, and it suggests that the study cohort of GPS-collared moose was indeed representative of the free-ranging population in northeastern Minnesota. Clearly, there is still much to understand about these relationships.

In addition to the multi-year occurrence of severe nutritional restriction of moose, preliminary analyses reveal a vast spatial distribution throughout moose range of collected snow-urine specimens with UN:C ratios indicative of severe nutritional deprivation (Figure 10). The wide temporal and spatial distributions of severe nutritional restriction suggest that habitat deficiencies at the landscape scale may constitute a primary contributing factor. We continue to apply significant efforts into investigating the habitat-nutrition relationships, but habitat deficiencies related to forage availability and quality, vegetative species composition, or less-than-optimum arrangements of forage openings and forest stands affording seasonal thermal cover remain unclear. Data from future winter nutritional assessments are required to provide additional support for our conclusions or to refute them. But the current data set, in combination with data from other ongoing habitat and nutritional studies, should provide a basis for formulating management recommendations that may be implemented and evaluated in the near future.

ACKNOWLEDGMENTS

We thank M. Grunklee, D. Williamson, N. Lowing, S. Hurd, M. Pike, N. Martorelli, B. Matykiewicz, J. Ostroski, L. Kruse, M. Bowman, D. Dewey, R. Willaert, R. Ryan, R. Peterson, C. Olson, and K. Foshay for their dedication and strong efforts necessary to sampling these free-ranging moose, and R. Wright for contributing his GIS skills. We also are grateful to T. Rusch, D. Plattner, M. Meskill, L. Cassioppi, M. Magnuson, and N. Thom for their assistance and cooperation in setting up office space and securing key equipment. We appreciate and acknowledge the laboratory support and skills of C. Humpal. This study has been supported largely by the Minnesota Department of Natural Resources Section of Wildlife, Minnesota Environmental and Natural Resources Trust Fund (ENRTF), and Wildlife Restoration (Pittman-Robertson) Program. The Minnesota Deer Hunters Association has provided supplemental funding for stipends for full-time seasonal field biology volunteers.

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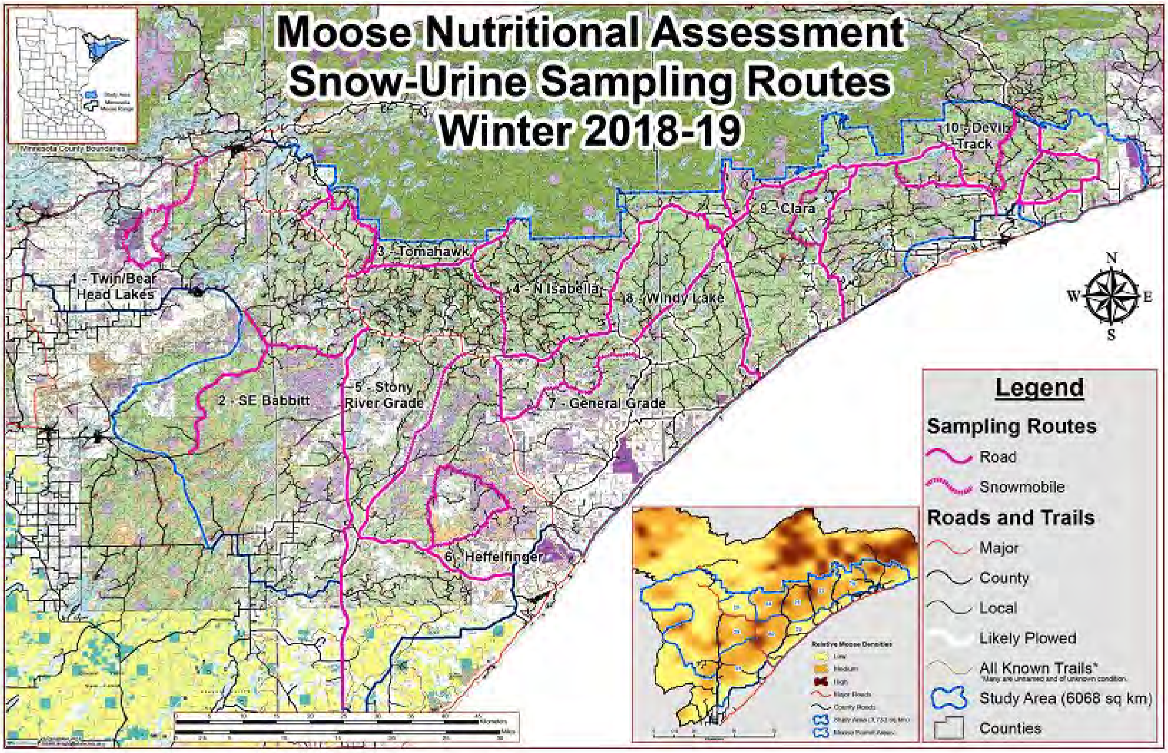
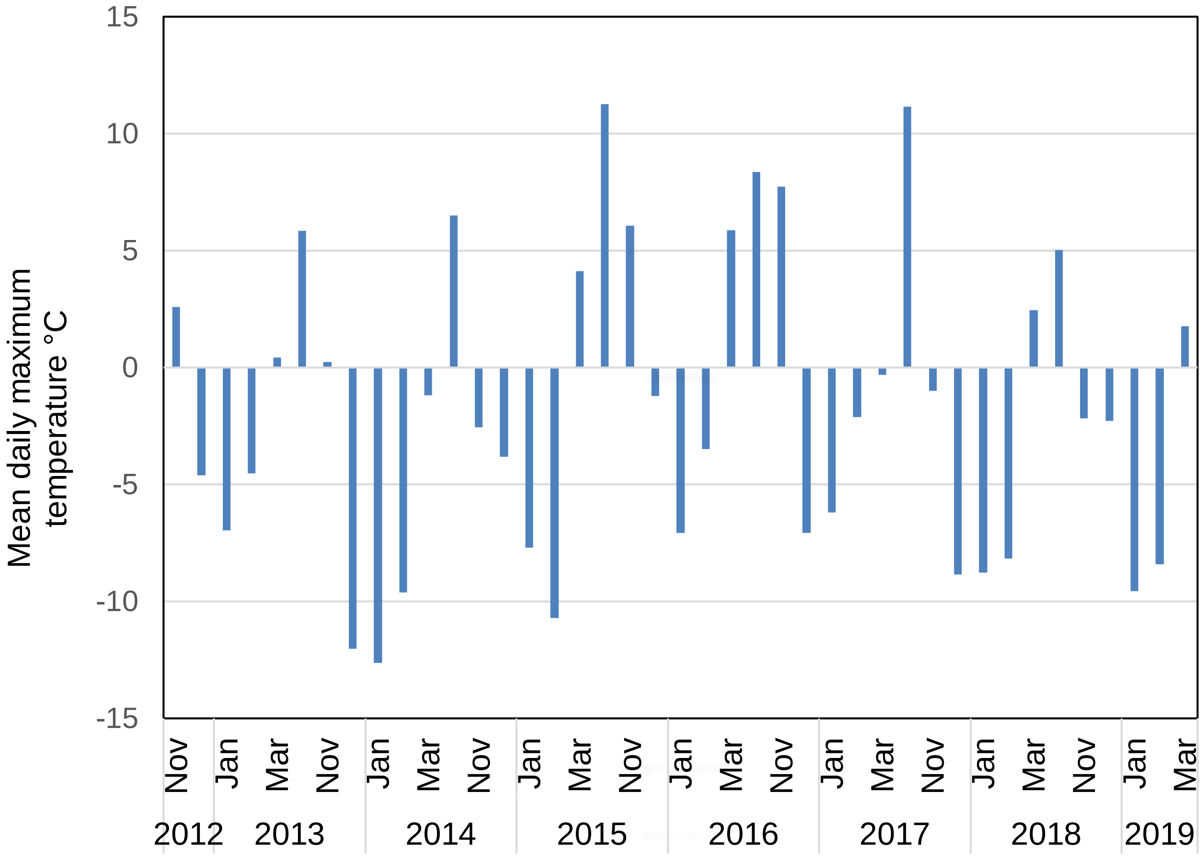


Figure 1. Map depicting the moose study area in northeastern Minnesota and the routes (i.e., roads and snowmobile trails in purple) used to distribute the sampling of fresh moose urine in snow (snow-urine) for nutritional assessments throughout the area, January–March 2013–2019.


Mean daily maximum temperature °C
15
10
5
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-5
-10
-15
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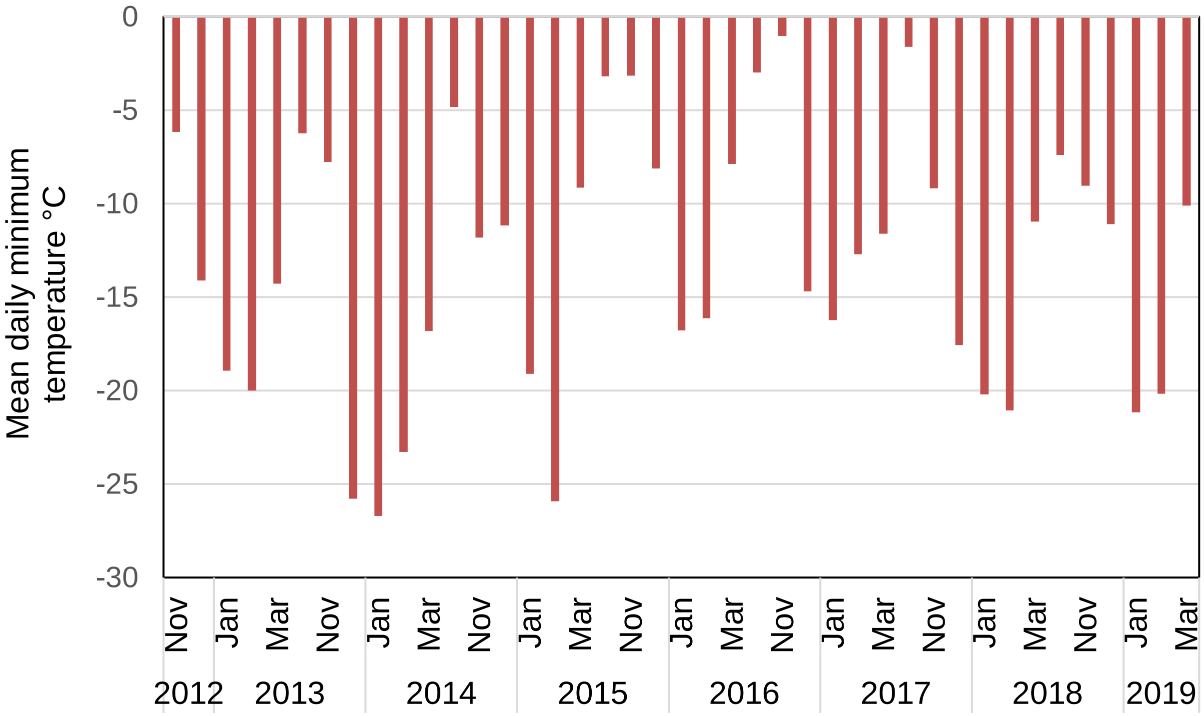
Mean daily minimum temperature °C
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Figure 2. Mean daily maximum (top) and minimum (bottom) ambient temperatures, Ely, Minnesota, November–April 2012–2019 (Midwestern Regional Climate Center 2019).

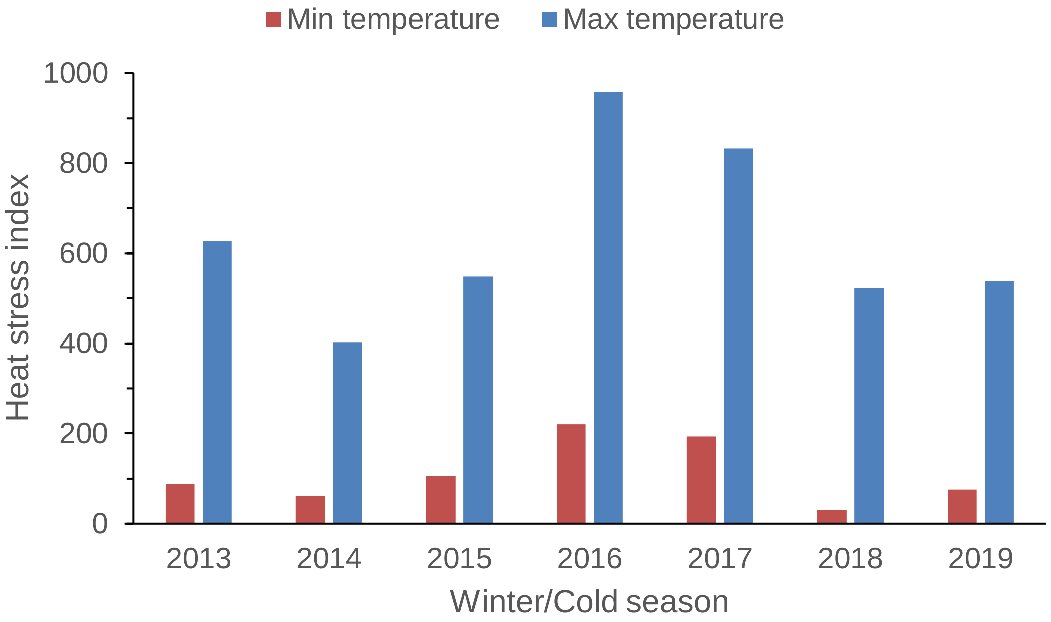


Figure 3. Winter heat stress index (HSI) for moose, calculated by accumulating daily ° C exceeding a −5° C threshold for the minimum and maximum ambient temperatures (1 November−31 March, Renecker and Hudson 1986, 1990) in northeastern Minnesota, 1 November−31 March 2012−2019.

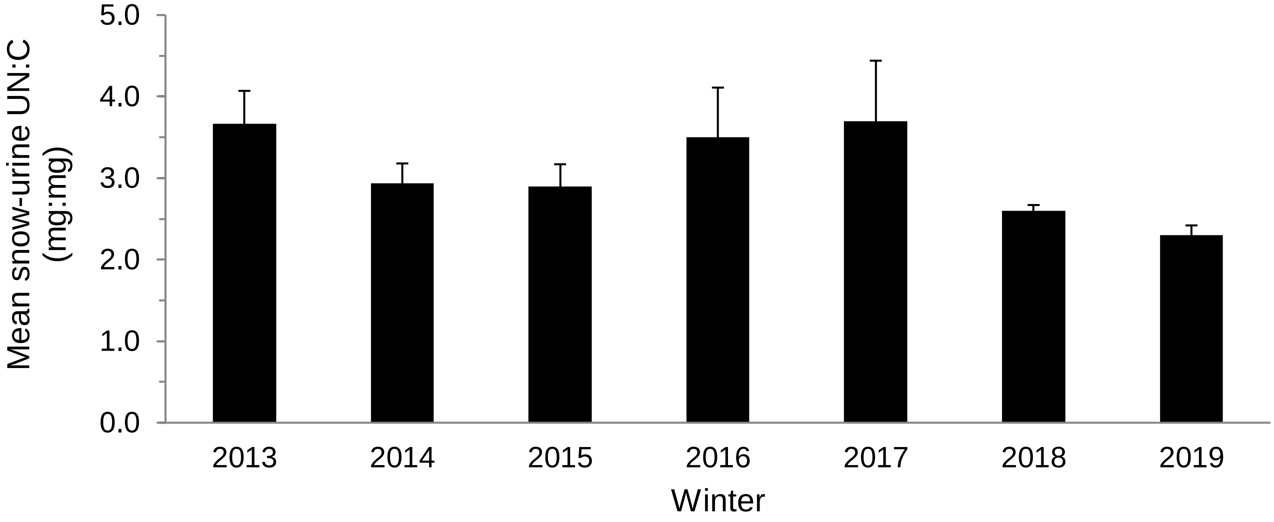


Figure 4. Overall mean (+SE) urea nitrogen:creatinine (UN:C) ratios of samples of fresh urine voided in snow (snow-urine) by moose and serially collected for assessments of nutritional restriction throughout northeastern Minnesota, January–March 2013–2019.

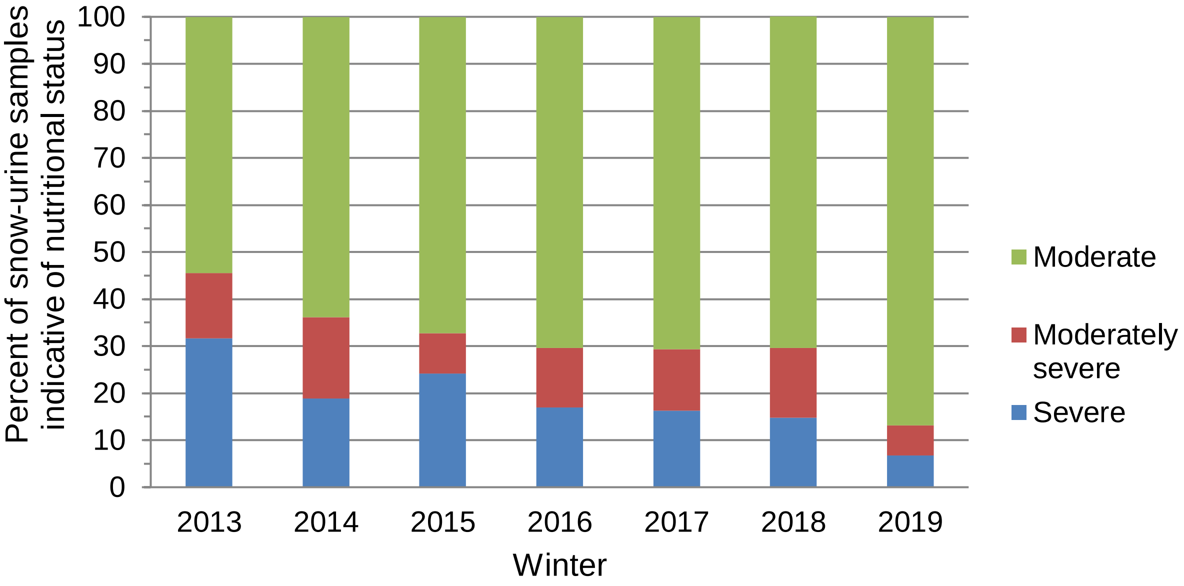


Figure 5. Overall percent of serially collected moose urine samples voided in snow (snow-urine) with urea nitrogen:creatinine (UN:C) ratios indicative of moderate/normal (UN:C <3.0 mg:mg), moderately severe (UN:C = 3.0–3.4 mg:mg), and severe nutritional restriction (UN:C ≥3.5 mg:mg) throughout northeastern Minnesota, January–March 2013–2019.

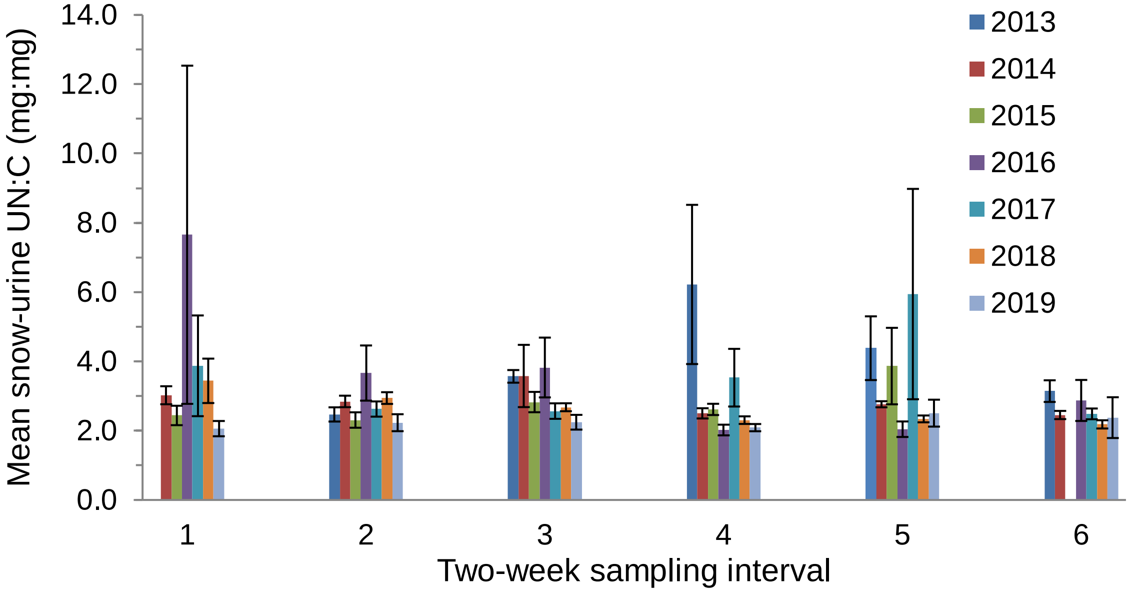


Figure 6. Mean (±SE) urea nitrogen:creatinine (UN:C) ratios of samples of fresh urine voided in snow (snow-urine) by moose and collected during 2-week sampling intervals for assessments of nutritional restriction throughout northeastern Minnesota, January–March 2013–2019.

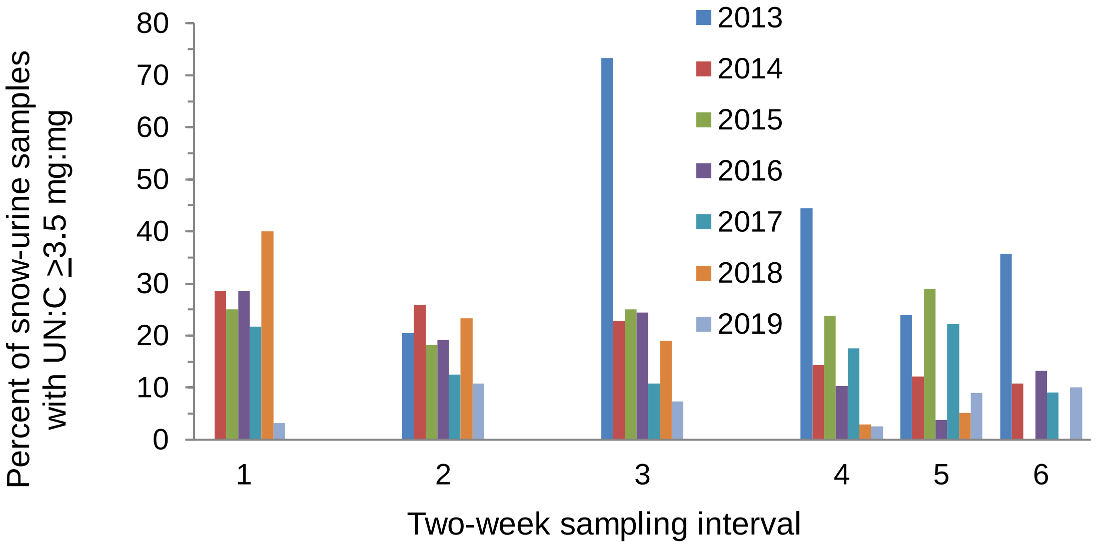


Figure 7. Percent of fresh urine samples voided in snow (snow-urine) by moose and collected during 2-week intervals with urea nitrogen:creatinine (UN:C) ratios indicative of severe nutritional restriction (UN:C ≥3.5 mg:mg) throughout northeastern Minnesota, January–March 2013–2019.

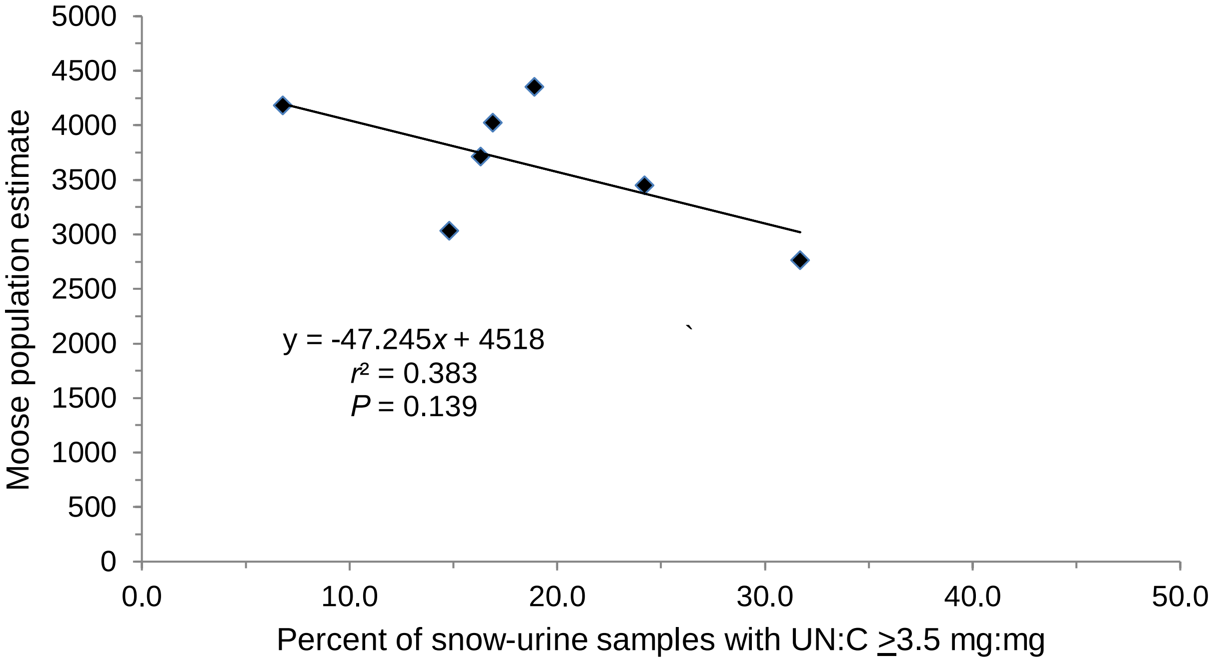
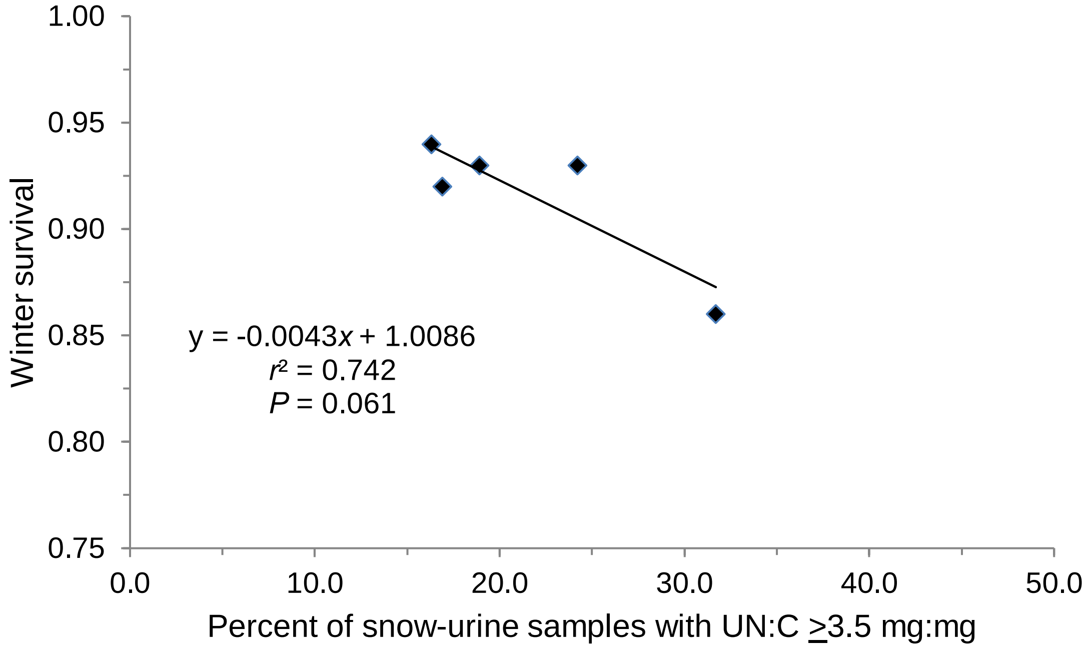


Figure 8. Relationship of the incidence of severe winter nutritional restriction of moose, indicated by the percentage of collected samples of urine in snow (snow-urine) with urea nitrogen:creatinine (UN:C) ratios ≥3.5 mg:mg, to annual population estimates of moose in northeastern Minnesota (estimates from DelGiudice 2018), January–March 2013–2019.



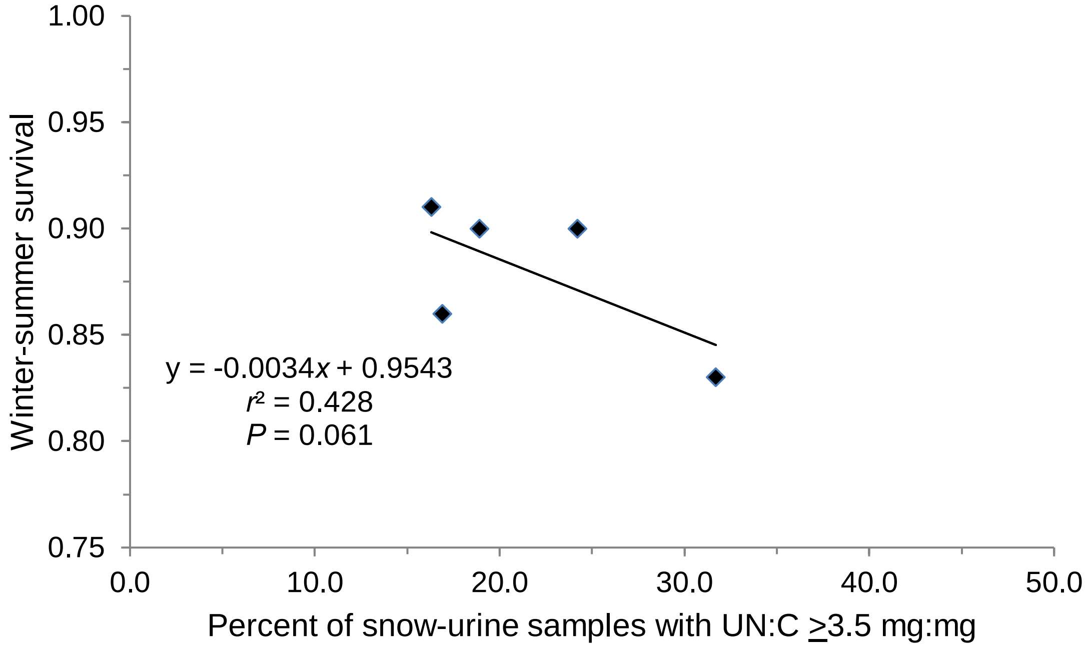


Figure 9. Relationships of the incidence of severe winter nutritional restriction of moose at the population level, indicated by the percentage of collected samples of urine in snow (snow-urine) with urea nitrogen:creatinine (UN:C) ratios ≥3.5 mg:mg, to winter (top, 1 November–31 May 2013−2017) and winter-to-summer (bottom, 1 November–31 August 2013−2017) survival of GPS-collared adult moose in northeastern Minnesota.

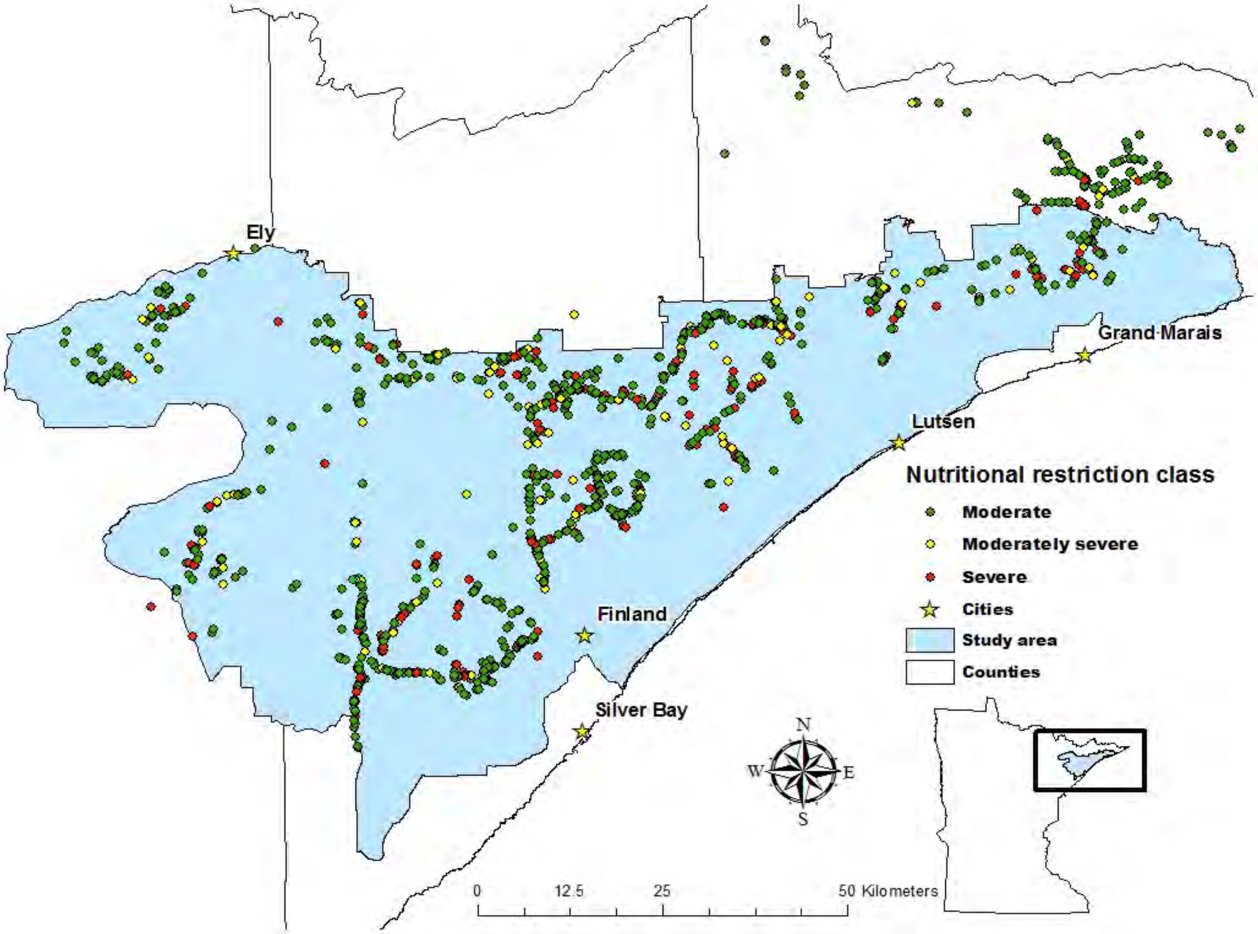


Figure 10. Spatial distribution of 1,479 samples of fresh urine in snow (snow-urine) from moose, serially collected for chemical analysis to assess the severity of winter nutritional restriction. Urinary urea nitrogen:creatinine (UN:C) ratios of <3.0, 3.0–3.4, and ≥3.5 mg:mg are indicative of moderate/normal (white circles), moderately severe (gray circles), and severe (black circles) nutritional restriction in northeastern Minnesota, January–March 2013−2019.

SHARP-TAILED GROUSE RESPONSE TO FALL PRESCRIBED FIRE AND MOWING

Charlotte Roy, Lindsey Shartell, and John Giudice

SUMMARY OF FINDINGS

We examined sharp-tailed grouse (i.e., sharptail, *Tympanuchus phasianellus*) responses to prescribed fire and mechanical treatment (i.e., mowing) in the fall (mid-August through November) during 2015-2018 in northwest Minnesota. We surveyed sharp-tailed grouse use at sites and measured vegetation before and after management at 16 mowing treatments and 12 prescribed burns, ranging in size from 5 to 664 ac (2–269 ha) and totaling 2,766 ac (1,118 ha). We also surveyed 21 control sites ranging in size from 6 to 460 ac (3–186 ha) and totaling 1,638 ac (663 ha) using a similar survey schedule. We surveyed sharp-tailed grouse use 0–28 (mean 9.1) days before (PRE), 1 week after (1WK), 1 month after (1MO), 1 year after (1YR), and 3 years after (3YR) management by conducting fecal pellet transects and documenting sharptails observed at the site. We detected sharp-tailed grouse pellets at 6 of the 28 treatment sites and 5 of the 21 control sites prior to treatment. Following treatment, sharp-tailed grouse pellets were detected in ≥1 fall survey (1WK or 1MO) at 13 treatment sites and 6 control sites. Sharptails were observed at only 1 treatment site and at 1 control site in PRE surveys, but in later fall surveys (1WK or 1MO) sharptails were observed at 4 treatment and 2 control sites. Sharp-tailed grouse use of treated sites nearly doubled 1 year after mowing and was highest in burned sites ≥1 year after management, but neither of these changes were significant in interim analysis due to high variability in site use. Our results thus far indicate that sharptail pellets provide a more useful indicator of site use than observations of grouse, and that sharptail use of treated sites is greatest ≥1 year after management. Vegetation results indicate that shrub height is lower 1 year following mowing treatments and that forb cover increases 1 year following burn treatments, however all other metrics measured did not differ 1 year after treatment. Additional field work is planned to complete 1YR and 3YR post-treatment surveys and improve statistical estimation of differences in occupancy, detection, and vegetation metrics.

INTRODUCTION

Sharp-tailed grouse (*Tympanuchus phasianellus*) rely on early successional habitats of open grass and brushland. Historically, these habitats were created and maintained through periodic wildfire. More recently, fire suppression has played a role in reducing habitat for sharp-tailed grouse (Berg 1997). Prescribed fire has become an important management tool for maintaining open grass and brushland habitats, but it can be difficult to implement effectively or safely under many conditions (e.g., too wet, windy, humid, dry) and can require considerable staff and resources to execute. Thus, wildlife managers supplement prescribed burning with mechanical habitat management tools (e.g., shearing, mowing) to maintain early successional habitats. Although mechanical treatments set succession back, they may not produce the same wildlife response as fire does. Wildlife managers have expressed concern that sharp-tailed grouse are not responding to management in the way they would expect if habitat were limiting.

Fall may be a particularly important season for management because juvenile sharptails disperse to surrounding habitat in the fall. Currently, most prescribed burns on state and other lands in the sharp-tailed grouse range occur in the spring (Roy and Shartell, unpubl. data from DNR Wildlife Managers). Region 1 (R1) regularly conducts fall burning, however Regions 2 and 3 (R2/3) have not been burning in the fall because of concerns about peat fires during drier conditions and challenges mobilizing a large number of fire-qualified staff on short notice during the fall (R1 has a Roving Crew to assist with prescribed fire treatments and R2 does not). This study aims to measure the response of sharptails to prescribed burning and mechanical treatments in the fall, as compared to untreated controls.

Historically, fires occurred throughout the year and maintained early successional habitats, such as open grass and brushland, on the landscape. Grassland fires were started by lightning during the growing season, and Native Americans set fires during both the spring and fall dormant seasons in both grasslands and forests to aid hunting (see review in Knapp et al. 2009). Stand replacing fires occurred at 0-10 year intervals in grass and shrub vegetation types, and in forest and woodland types, understory fires occurred at 0-10 year intervals, with more severe, stand-replacement fires occurring at less frequent intervals in Minnesota (Brown and Smith 2000).

Native Americans referred to the sharp-tailed grouse as the “fire grouse” or “fire bird” because of their association with habitats frequently burned, and kept open, by fire. Sharptails have been shown to respond to prescribed fire treatments. Kirsch and Kruse (1973) found that the numbers of broods hatched per 100 acres was higher in 2 burned areas compared to an unburned control 1 year after spring prescribed fires. Sexton and Gillespie (1979) reported that sharptails switched leks just 2 days after a spring burn, abandoning the former dancing ground in favor of the recently burned site 480 m to the north. Sharptails have also been observed returning to leks to dance the day after a burn (J. Provost, pers. comm.).

Burn season may have an effect on the response of sharptails to prescribed fire treatments. Burns conducted in the fall might attract dispersing juveniles searching for habitat. Numerous bird species are known to be attracted to fire, smoke, and recently burned areas (Smith 2000); smoke, flames, and dark burned ground could provide strong visual cues about habitat creation and its direction from a large distance. Young sharptails disperse during September and October (Gratson 1988), typically <6 km from brood rearing areas near nest sites. Sites burned in the fall are not followed by regrowth of vegetation during winter (Kruse and Higgins 1990) and could serve as lek sites the following spring. Sharp-tailed grouse also resume dancing at leks in the fall; Hamerstrom and Hamerstrom (1951) suggested that these fall dances, which include young males, might establish leks for the following spring.

Similar long-distance cues to habitat creation and maintenance are not provided by mechanical treatments. Thus, we might expect wildlife responses to management lacking these cues to be delayed or muted. In Florida shrub-grassland, burned plots were colonized by birds sooner than the mechanically treated plots, in which shrubs were chopped (Fitzgerald and Tanner 1992); birds were observed in burned plots the next day but not for months in chopped plots. Species richness and abundance remained lower in winter chop plots than in burned and control plots throughout this study. Fitzgerald and Tanner (1992) suggested that this was because burned plots provided more complex structure than mechanically treated plots.

Sharp-tailed grouse densities and responses to management treatments have been measured with numerous methods, but pellet counts are the simplest to execute. Pellet counts along transects have been shown to be indicative of the relative abundance of sage grouse (*Centrocercus urophasianus*, Hanser et al. 2011), density of red grouse (*Lagopus lagopus* *scoticus*, Evans et al. 2007), and habitat use of red grouse (Savory 1978). Pellet counts along transects in plots have been used to compare sage-grouse responses to mechanical and chemical treatments (Dahlgren et al. 2006). Schroeder and Vander Haegen (2014) used pellet counts along circular transects to examine the effects of wind farms on sage-grouse.

OBJECTIVES

- To compare sharp-tailed grouse use prior to and following fall management within burn, mow, and control treatments.
- To relate vegetation metrics to differences in sharp-tailed grouse use of burn, mow, and control treatments.

Hypotheses

- Sharp-tailed grouse use will increase following burning or mowing, with burned sites showing a greater increase in sharptail use than mowed sites, and both treatments having greater sharptail use than controls.
- Vegetation composition and structure will influence the use of treatment and control sites by sharp-tailed grouse, with increased use in early successional habitats.

METHODS

Study Areas

Our study was focused in the northwest sharp-tailed grouse region of Minnesota. Treated study sites were mainly on state lands, however 3 sites owned and managed by The Nature Conservancy (TNC) and 3 private land sites were included. In 2015, we conducted pre-treatment surveys at 16 sites that were planned to be treated and 15 control sites. Of these, 10 sites (6 mows and 4 prescribed burns) were treated (Table 1). In 2016 we conducted pre-treatment surveys at 9 sites that were planned for management and 6 control sites. Of these, 4 sites (1 mow and 3 prescribed burns) were treated (2016 was an unusually wet year which restricted management opportunities). In 2017, we conducted pre-treatment surveys at 13 sites that were planned for management and 8 control sites. Seven sites were treated (2 mows and 5 prescribed burns). In 2018, we conducted pre-treatment surveys at 16 sites planned for management and 10 control sites, with 7 mows completed.

Data Collection & Experimental Design

Treatment sites varied in size, date of management, vegetative composition, surrounding landscape, and local sharp-tailed grouse density. We attempted to match treatments in each DNR work area or sub-work area (some work areas are very large) with a control site of similar size and successional stage (e.g., crude habitat classification, visual assessment of percent cover shrubs and herbaceous vegetation, and average shrub height) *a priori* as determined by inspection of aerial imagery, conversations with managers, and site visits. Control sites were identified ≤6 km from treatment sites when possible (based on dispersal distances of young males in the fall; Gratson 1988). Control sites helped account for changes related to seasonal progression (i.e., changes in habitat use, social behavior, and vegetation) not related to management. Dahlgren et al. (2006) implemented a similar design to account for temporal differences in the application of management treatments for sage grouse. However we decided that a paired analysis was inappropriate due to the difficulty to closely match treatment and control sites. Thus, beginning in 2017 we selected 1 control for nearby sites treated on the same day. This also provided for a more balanced sample size among the 3 treatments (control, mow, and burn).

We surveyed treatment and control sites as close as possible in time, both before and after treatment (Smith 2002, also see Morrison et al. 2001:118-130). We walked systematically spaced parallel transects with a starting point placed on the site boundary and the transect traversing the treatment capturing both edge and interior portions. The sampling rate was standardized to 10 m of transect/ac (25 m/ha), with transects at least 150 m apart, based on placement of pellet transects in other studies (Evans et al. 2007, but half as dense as Dahlgren et al. 2006, Hanser et al. 2011). We counted sharp-tailed grouse pellet piles ≤0.5 m from the transect, removing all pellets encountered (Evans et al. 2007, Schroeder and Vander Haegen 2014). At each pellet pile we recorded pellet freshness and vegetation category (i.e., grass, shrub, forb, grass-shrub mix, grass-forb mix, etc.). We also recorded all sharp-tailed grouse observed (heard, flushed, tracks seen) at the site while walking transects.

We sampled transects 4 times at each site—once before treatment, targeting measurements within 2 weeks of treatment (PRE), and 3 times after treatment; 1 week after treatment (1WK), 1 month after treatment (1MO), and 1 year after treatment (1YR). Treatment and control sites were sampled within 21 days of each other. In 2018, we also conducted 3 year post-treatment surveys (3YR) at 4 sites treated in 2015, where additional management had not occurred since 2015.

To adjust naïve occupancy rates for detection differences among treatment groups, vegetation categories, and other sources, we conducted pellet detection assessments. We accomplished this by surveying transects with pellets placed in known locations (but unknown to observers) and estimated detection probabilities for each vegetation and management category. Dahlgren et al. (2006) reported detectability of pellets along transects to be very high and similar in different types of vegetative cover. However, their study was conducted on sage grouse in sage brush, and sharp-tailed grouse habitats in Minnesota differ considerably in vegetative composition and structure.

We sampled vegetation within treatments using point-intercept sampling (Levy and Madden 1933, Dahlgren et al. 2006) to determine percent cover and average height of broad vegetation classes (i.e., tree, shrub, forb, and graminoid) before and after treatment. We sampled vegetation along 20-m transects placed perpendicular to the pellet transect, with the number of transects based on the size of the site. We marked the start of each vegetation transect using ground staples with numbered aluminum tags and flagging, and we used Global Positioning System (GPS) coordinates to allow re-measurement following treatment. During 2015-2016, we recorded maximum height for each vegetation class every 0.5 m for a total of 40 points per transect. After exploratory analysis of data and considering logistical tradeoffs, we reduced the amount of vegetation data collected in 2017, recording height and class every 1.0 m for a total of 20 points per transect. We used a pole with graduated measurements every dm to determine the type of vegetation intercepted (touching the pole) and the highest point at which each vegetation class touched the pole. We also recorded whether the vegetation was dead/dormant, combining those categories because it was unclear due to natural plant senescence whether vegetation was dormant or dead in late-fall surveys. Following treatment, we classified cut vegetation as dead/dormant, recorded height, and noted that the vegetation was cut. If no vegetation was present, the substrate type was recorded. For the purpose of this study, moss and lichen were considered a substrate type rather than vegetation.

Vegetation metrics were calculated for each study site. Proportion of cover in each class and mean maximum height were compared among treatment types and between sites with and without sharptail use. In our preliminary analysis, we included both live and dead vegetation, using the maximum height of either type at each point. Significant differences among survey periods were tested for using Tukey's Honest Significant Difference, and significant differences between sites occupied and unoccupied by sharptails was tested for using T-tests. For both a significance level of $P < 0.05$ was used.

RESULTS AND DISCUSSION

Sharp-tailed grouse pellets were detected on transects at 6 (21%) of the 28 treatment sites and 5 (24%) of the 21 control sites prior to treatment (Table 2). Following treatment, sharp-tailed grouse pellets were detected in $\geq$1 fall survey (1WK or 1MO) at 13 treatment sites (46%) and 5 control sites (24%). Sharptail observations on transects prior to treatment exhibited similar patterns, with detections at only 1 treatment site (0.04%) and 1 control site (0.05%) in initial surveys. In later fall surveys (1WK, 1MO), however, sharptails were observed at 4 treatment sites (14%) and 2 control sites (10%, Table 3). In 1YR surveys (completed for 2015, 2016, and 2017 sites to date), we detected pellets on transects at 6 (21%) of 28 treatment sites and 3 (14%) of 21 control sites, and sharptails were observed on transects at 2 treatment sites (7%) and 1 control site (5%).

Our pellet survey results thus far suggest that our methods are capturing sharptail use of treatment and control sites. Naïve occupancy rates (i.e., site use) from data collected thus far suggest increases in sharptail use of sites following management (Figure 1). Although occupancy and detection are confounded in naïve estimates for the 1WK and 1MO surveys (due to treatment effects on screening cover), surveys conducted 1 year (1YR) and 3 years (3YR) following treatments should have similar detection rates to pre-treatment measurements due to regrowth of vegetation the next growing season, especially in burn sites. Thus, the PRE vs. 1YR and 3YR comparisons should be reasonably straightforward and informative, whereas results from other time comparisons are more tenuous to interpret from naïve occupancy rates due to large differences in detection (Figure 2). The naïve probability of sharp-tailed grouse site use at treated sites nearly doubled 1 year after mowing and was highest in burned sites, but neither of these changes was significant in preliminary analysis due to high variability in site use (Figure 3). We also found the area of the disturbance to be a predictor of the probability of site use in interim analysis (Figure 4).

General field observations of vegetation prior to treatment indicated that mowing might be applied to sites at a later successional stage than prescribed fire. Prior to treatment, mow sites had greater mean proportions of forb and shrub cover and taller shrubs than burn sites, however these differences were not significant (Table 4). The lack of significance could be due to the low sample size and high variability among sites.

Control sites had significantly lower graminoid height (P < 0.01) in 1MO surveys than in PRE surveys, which was likely the result of vegetation senescence (Table 5). One year later, we did not detect differences in vegetation cover or height at control sites compared to pre-treatment measurements (Table 5). At sites that were mowed, graminoid, forb, and shrub cover (P = 0.01, P < 0.01, P < 0.01) and height (P < 0.01, P < 0.01, P < 0.01) were all significantly lower in 1MO surveys, but in 1YR measurements only shrub height remained lower than PRE survey measurements (P < 0.01). At sites that were burned, graminoid cover, forb cover, and graminoid height (P < 0.01, P < 0.01, P < 0.01) were significantly lower in 1MO surveys, but in 1YR surveys the only significant difference was that forb cover was significantly greater (P < 0.01, Table 5). Sites occupied by sharp-tailed grouse had significantly shorter forbs and shrubs (when present) than unoccupied sites (P < 0.01, P = 0.02, Table 6).

This report includes the fall surveys for the fourth year of data collection (PRE, 1WK, 1MO) but not the 1YR surveys that will be conducted in fall 2019 or the 3YR surveys that will be collected through 2021. We anticipate data collection continuing for 3 more years to complete surveys of sites treated during 2015-2018. However, additional sites will not be added to the study. Results presented in this report are preliminary and subject to revision.

Managers throughout sharptail range in Minnesota have expressed a need for this type of information to more effectively manage for sharptails. Given the current sharptail population concerns in the east-central region, information on the effectiveness of various management options would be helpful for decision-making with finite resources for management. Managers in the northwest region are also interested in this information to ensure that their management actions are as effective as possible.

ACKNOWLEDGMENTS

We are thankful for the cooperation and support of Area Wildlife Managers, Assistant Managers, and staff at Aitkin, Baudette, Cloquet, Karlstad, Red Lake WMA, Roseau River WMA, Thief River Falls, Thief Lake WMA, and Tower. J. Eerkes at The Nature Conservancy in Karlstad was also very helpful in identifying sites and applying management. Special thanks to M. Schleif, V. Johnson, J. Colbaugh, A. Mosloff, A. Erickson, J. Brunner, A. Del Valle, L. Hause, N. Benson, and P. Coy for assistance with field data collection and data management. C. Scharenbroich loaded imagery into GPS units for field use. M. Larson provided comments that improved this report. J. Giudice provided statistical guidance and produced figures for this report. This study has been funded in part through the Wildlife Restoration (Pittman-Robertson) Program.

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Table 1. Management activities completed for sharp-tailed grouse habitat in northwest Minnesota during fall in 2015–2018 and associated control sites, in order of treatment date.

Site	Work area	Treatment	Treatment date	Treatment ac (ha)	Control ac (ha)
Roseau River	Roseau River	Mow	28 Aug–16 Sep 15	31 (12.5)	28 (11.3)
Skull Lake	Karlstad	Burn	1 Sep 2015	90 (36.4)	70 (28.3)
Halma	Karlstad	Mow	16–23 Sep 2015	41 (16.6)	39 (15.8)
Red Lake Mow	Red Lake	Mow	22 Sep 2015	12 (4.9)	22 (8.9)
Spooner	Baudette	Mow	28 Sep 2015	22 (8.9)	26 (10.5)
Caribou	Karlstad	Burn	28 Sep 2015	664 (268.7)	No control
TL 2015 Burn	Thief Lake	Burn	28 Sep 2015	58 (23.5)	31 (12.5)
Red Lake Burn	Red Lake	Burn	19 Oct 2015	152 (61.5)	176 (71.2)
Prosper	Baudette	Mow	19–30 Oct 2015	63 (25.5)	201 (81.3)
TL Mow	Thief Lake	Mow	30 Oct 2015	20 (8.1)	19 (7.7)
TL 2016 burn	Thief Lake	Burn	1 Sep 2016	31 (12.5)	37 (15.0)
Noracre[a]	Roseau	Burn	14 Sep 2016	71 (28.7)	22 (8.9)
Roseau brush	Roseau	Mow	27 Sep–7 Oct 16	23 (9.3)	29 (11.7)
Espelie	Thief River Falls	Burn	3 Oct 2016	443 (179.3)	460 (186.2)
Halma 2017	Karlstad	Mow	28 Aug–8 Sep 2017	62 (25)	61 (25)
Gates	Red Lake	Burn	8 Sep 2017	388 (157)	No control
K burn	Roseau	Burn	13 Sep 2017	90 (36)	93 (38)
F burn	Roseau	Burn	13 Sep 2017	99 (40)	Same as K
Prosper 2017	Baudette	Mow	27 Sep–26 Oct 2017	70 (28)	41 (17)
O burn	Roseau	Burn	9 Oct 2017	17 (7)	100 (40)
I burn	Roseau	Burn	9 Oct 2017	48 (19)	Same as O
Mow 3	Thief Lake	Mow	21 Sep – 10 Oct 2018	73 (29.5)	No control
Graceton mow	Baudette	Mow	1 – 11 Oct 2018	75 (30.4)	89 (36.0)
HQ brush mow	Roseau	Mow	4 – 5 Oct 2018	5 (2.0)	6 (2.5)
Mow 1	Thief Lake	Mow	24 Sep – 16 Oct 2018	53 (21.4)	45 (18.2)
TNC site 10	Karlstad	Mow	19 – 22 Oct 2018	11 (4.5)	Same as site 9
RWMA brush mow	Roseau	Mow	17 – 25 Oct 2018	9 (3.6)	Same as HQ control
TNC site 9	Karlstad	Mow	23 – 27 Oct 2018	45 (18.2)	43 (17.4)

[a] The Noracre burn site was treated again (burned in spring 2017 and sprayed with herbicide in spring and summer 2017) before the 1-year post-treatment (1YR) survey, so it is not clear whether observed use by sharp-tailed grouse in the 1YR survey was due to the initial fall burn or another treatment that was not part of the study.

Table 2. Sharp-tailed grouse pellet detections at treatment and control sites in northwest Minnesota during 2015-2018. Surveys were conducted before (PRE), 1 week (1WK), 1 month (1MO), and 1 year (1YR), and 3 years (3YR) after treatment. The number of pellet detections on transect are indicated numerically, and pellets detected off-transect are indicated with an OT, indicative of site use not captured in sampling. An asterisk indicates that snow impeded detection of pellets, and T indicates that tracks were detected in snow. Surveys with confirmed sharp-tailed grouse use through any source of sign are highlighted in gray. NS indicates that the 1YR or 3YR survey has not yet been completed for sites yet. TRT indicates that a 3YR survey was not conducted because of additional management conducted after the original treatment.

Fecal pellets	Treatment					Control				
Site	PRE	1WK	1MO	1YR	3YR	PRE	1WK	1MO	1YR	3YR
Red Lake mow	0	0	0	2	0	0	0	0	1	0
Thief Lake mow	0	0	0*	0	TRT	0	0	0*	0	TRT
Spooner mow	0	0	3	0	3	0	0	0	0	1
Roseau 2015 mow	2	1 OT	1	0	TRT	0	0	0	0	0
Halma mow	0	0	0	1 OT	1	1	1	2	0	0
TL 2015 burn	1 OT	0	1	1 OT	NS	0	0	0	0	TRT
Skull Lake burn	0	1	0	1	NS	0	0	0	0	NS
Red Lake burn	0	0	0	0	TRT	0	0	0*	0	NS
Prosper mow	0	1	0*	2	TRT	1	11	2T*	5 4 OT	TRT
Caribou burn	1	2	1 OT	0	32 16 OT	-	-	-	-	-
TL 2016 burn	0	1	4 7 OT	7 1 OT	NS	0	0	0	0	NS
Noracre burn[a]	0	9 3 OT	0	3T*	TRT	0	0	0	0	NS
Espelie burn	1	6	18 31 OT	1 3 OT	NS	1 1 OT	1 3 OT	4 5 OT	3* 2 OT	NS
Roseau 2016 mow	1 OT	0	0	0	TRT	0	0	0	0	NS
Halma 2017 mow	0	1 OT	1	0	NS	0	0	0	0	NS
Gates burn	0	3	0	0	TRT	0	-	-	0	NS
K burn	1 1 OT	0	7 11 OT	0	NS	-	-	-	-	-
F burn	4 1 OT	1	5 5 OT	14	NS	0	0	0	0	NS
Prosper 2017 mow	0	3T*	0	0	NS	0	0	0	0	NS
O burn	0	0	0	0	NS	-	-	-	-	-
I burn	0	0	0	1OT	NS	3 OT	1 OT	0	6 30+OT	NS
TL Mow 3	0	0	0*	NS	NS	-	-	-	-	-
Graceton mow	0	0	4T* 1OT*	NS	NS	4	2	3	NS	NS
HQ brush mow	0	0*	0	NS	NS	0	0*	0	NS	NS
TL Mow 1	0	0	0*	NS	NS	0	0	0	NS	NS
TNC site 10	0	0	2T*	NS	NS	3	1 1OT	0	NS	NS
RRWMA brush mow	1	0	1OT*	NS	NS	-	-	-	-	-
TNC site 9	0	0	3T* 1OT*	NS	NS	-	-	-	-	-

[a] The Noracre burn site was treated again (burned in spring 2017 and sprayed with herbicide in spring and summer 2017) before the 1YR survey, so it is not clear whether observed use by sharp-tailed grouse in the 1YR survey was due to the initial fall burn or another treatment.

Table 3. The number of sharp-tailed grouse observed at treatment and control sites in northwest Minnesota during 2015-2018. Surveys were conducted before (PRE), 1 week (1WK), 1 month (1MO), 1 year (1YR), and 3 years (3YR) after treatment. Sharp-tailed grouse observed while off-transect are indicated with OT, indicative of site use not captured in sampling. Surveys with confirmed sharp-tailed grouse use through observations of any birds at the site are highlighted in gray. NS indicates that the 1YR or 3YR survey has not been completed.

Grouse observations	Treatment					Control				
Site	PRE	1WK	1MO	1YR	3YR	PRE	1WK	1MO	1YR	3YR
Red Lake mow	0	0	0	0	0	0	0	0	0	0
Thief Lake mow	0	0	0	0	TRT	0	0	0	0	TRT
Spooner mow	0	0	11	3 OT	0	0	0	0	0	0
Roseau 2015 mow	2 OT	5 OT	2 OT	0	TRT	0	0	0	0	1
Halma mow	0	0	1	0	0	0	2	0	0	0
TL 2015 burn	4	0	0	0	NS	0	0	0	0	TRT
Skull Lake burn	0	0	0	0	NS	0	0	0	0	NS
Red Lake burn	0	0	0	0	TRT	0	0	0	0	NS
Prosper mow	0	0	0	1	TRT	0	0	0	12-20	TRT
Caribou burn	0	5	13	2 2 OT	27	-	-	-	-	-
TL 2016 burn	0	0	0	0	NS	0	0	0	0	NS
Noracre burn[a]	0	0	0	0	NS	0	0	0	0	NS
Espelie burn	0	1	2 OT	0	NS	5 OT	1	7 OT	0	NS
Roseau 2016 mow	6 OT	0	0	0	NS	0	0	0	0	NS
Halma 2017 mow	0	0	0	0	NS	0	0	0	0	NS
Gates burn	0	0	0	0	NS	-	-	-	-	NS
K burn	0	0	0	0	NS	-	-	-	-	NS
F burn	1 OT	0	0	0	NS	0	0	0	0	NS
Prosper 2017 mow	0	0	0	0	NS	0	0	0	0	NS
O burn	0	0	0	0	NS	-	-	-	-	NS
I burn	0	0	0	0	NS	0	0	0	0	NS
Mow 3	0	0	0	NS	NS	0	0	0	NS	NS
Graceton mow	0	0	0	NS	NS	1	0	0	NS	NS
North HQ	0	0	0	NS	NS	0	0	0	NS	NS
Mow 1	0	0	0	NS	NS	0	0	0	NS	NS
TNC site 10	0	0	0	NS	NS	0	1 OT	0	NS	NS
South HQ	0	0	0	NS	NS	0	-	-	-	-
TNC site 9	0	0	0	NS	NS	0	-	-	-	-

[a] The Noracre burn site was treated again (burned in spring 2017 and sprayed with herbicide in spring and summer 2017) before the 1YR survey, so it is not clear whether observed use by sharp-tailed grouse in the 1YR survey was due to the initial fall burn or another treatment.

Table 4. Mean pre-treatment vegetation cover and height for 4 vegetation classes at control (*n* = 22), mow (*n* = 16), and burn (*n* = 12) sites sampled for sharp-tailed grouse use in northwestern Minnesota during 2015-2018. No significant differences ($P < 0.05$) were observed.

	Control	Mow	Burn
Cover (proportion)			
Graminoid	0.94	0.94	0.98
Forb	0.31	0.41	0.22
Shrub	0.36	0.40	0.26
Tree	0.05	0.05	0.05
Height (m)			
Graminoid	0.51	0.51	0.54
Forb	0.32	0.34	0.28
Shrub	1.19	1.30	0.74
Tree	2.64	2.03	1.76

Table 5. Change in mean vegetation cover and height from pre-treatment to 1 month after (1MO, control *n* = 21, mow *n* = 16, and burn *n* = 12) and 1 year after (1YR, control *n* = 18, mow *n* = 9, and burn *n* = 12) at sites sampled for sharp-tailed grouse use in northwestern Minnesota during 2015-2018. Comparisons to 1YR surveys exclude sites that were treated in 2018. Significant differences (P < 0.05) between measurements pre- and post-treatment are indicated with an asterisk.

	Control	Control	Mow	Mow	Burn	Burn
	1MO	1YR	1MO	1YR	1MO	1YR
Cover (proportion)						
Graminoid	-0.02	0.01	-0.22*	-0.02	-0.43*	-0.04
Forb	-0.12	0.02	-0.30*	0.02	-0.18*	0.19*
Shrub	-0.06	0.04	-0.28*	-0.04	-0.10	-0.00
Tree	-0.02	-0.01	-0.03	-0.04	-0.02	0.02
Height (m)						
Graminoid	-0.14*	-0.03	-0.39*	-0.07	-0.25*	-0.05
Forb	-0.06	-0.01	-0.22*	-0.09	0.07	-0.05
Shrub	0.04	0.17	-1.09*	-0.84*	-0.02	-0.04
Tree	-0.58	0.10	-1.08	-1.19	0.20	-0.22

Table 6. Mean vegetation cover and height at sites occupied and unoccupied by sharp-tailed grouse during pre-treatment, 1 month, and 1 year surveys in northwestern Minnesota during 2015-2018 (occupied *n* = 48, unoccupied *n* = 90). Metrics at occupied sites that significantly differed ($P < 0.05$) from unoccupied sites are indicated with an asterisk.

	Occupied	Unoccupied
Cover (proportion)		
Graminoid	0.85	0.89
Forb	0.23	0.29
Shrub	0.25	0.32
Tree	0.04	0.04
Height (m)		
Graminoid	0.40	0.43
Forb	0.25*	0.30
Shrub	0.79*	1.05
Tree	2.47	1.95

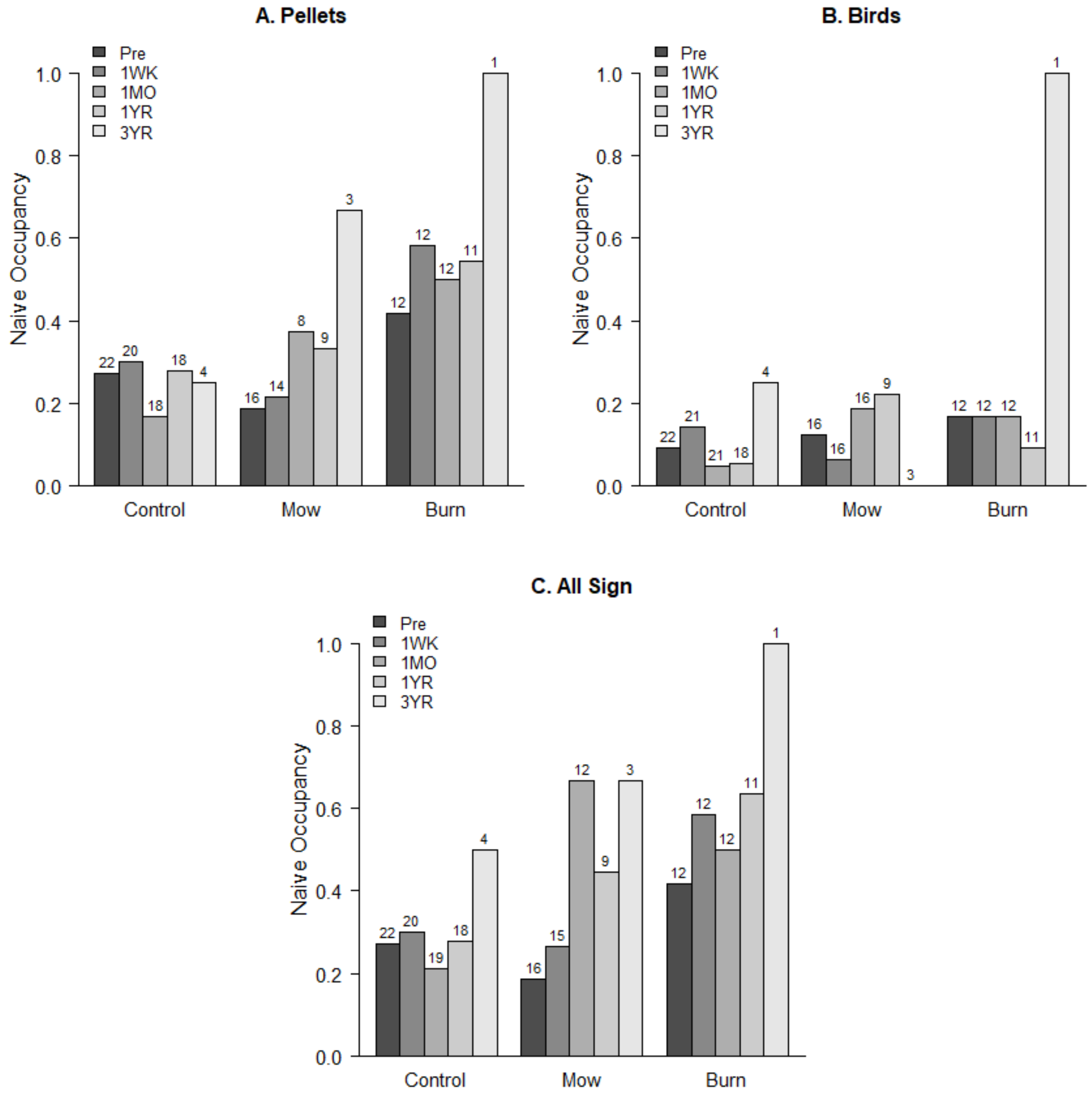

Figure 1. Naïve occupancy for sharptail pellets (A), sharptail observations (B), and all sign (includes off-transect detections, (C) during surveys conducted before (PRE), 1 week after (1WK), 1 month after (1MO), 1 year after (1YR), and 3 years after (3YR) treatment at sites managed during 2015–2018 in northwest Minnesota to assess the effects of prescribed burning and mowing compared to control sites.

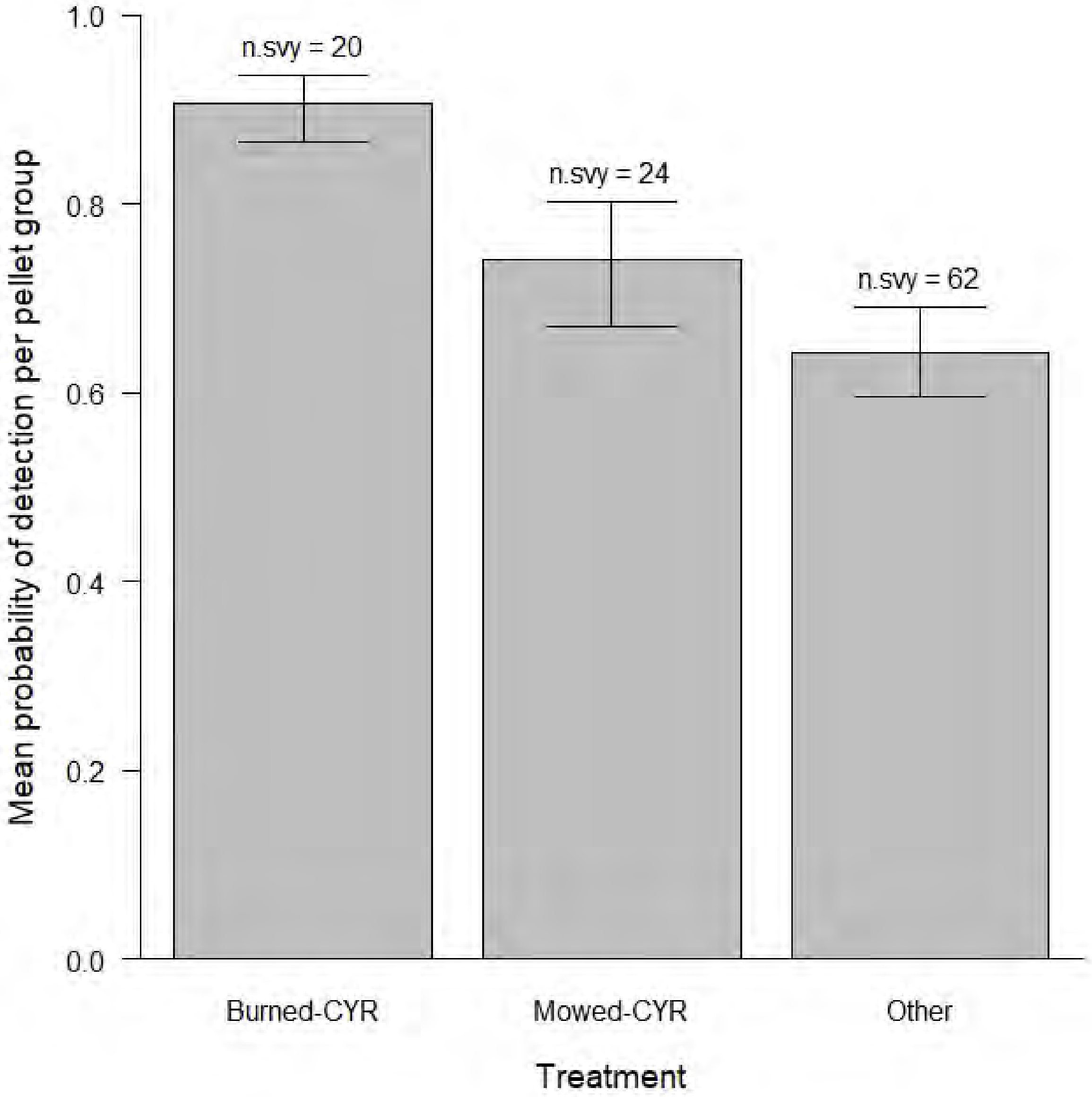

Figure 2. The probability of detecting sharp-tailed grouse fecal pellets given that they are present at sites burned in the current year (Burned-CYR), mowed in the current year (Mowed-CYR), and sites not recently treated prior to survey (OTHER) in northwest Minnesota during 2015-2018.

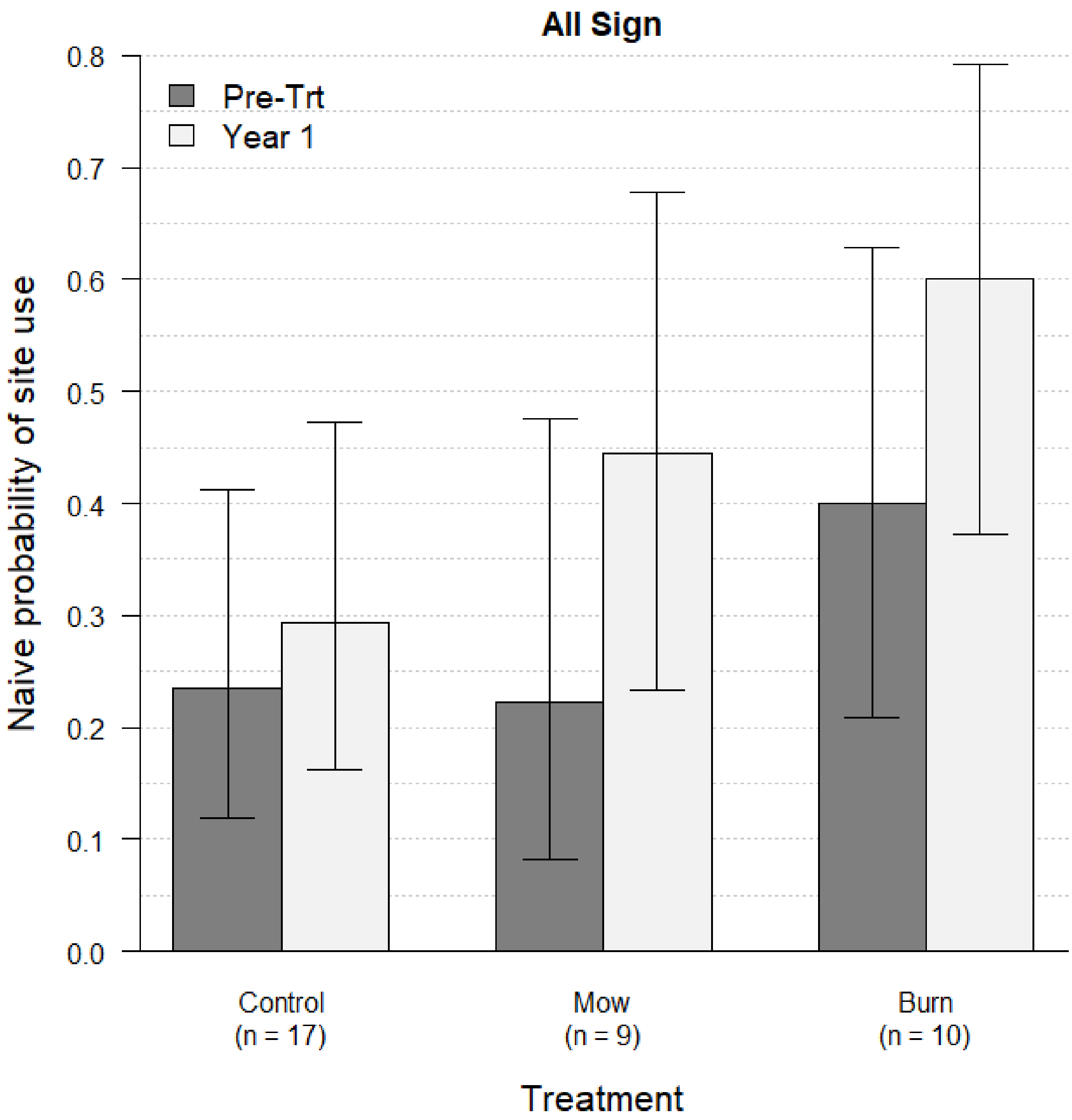

Figure 3. Mean naïve occupancy index of sharp-tailed grouse before and 1 year after management (with 85% confidence intervals) at sites in northwestern Minnesota during 2015–2017 based on a logistic regression model. One year post-treatment surveys have not yet been completed at sites treated in 2018.

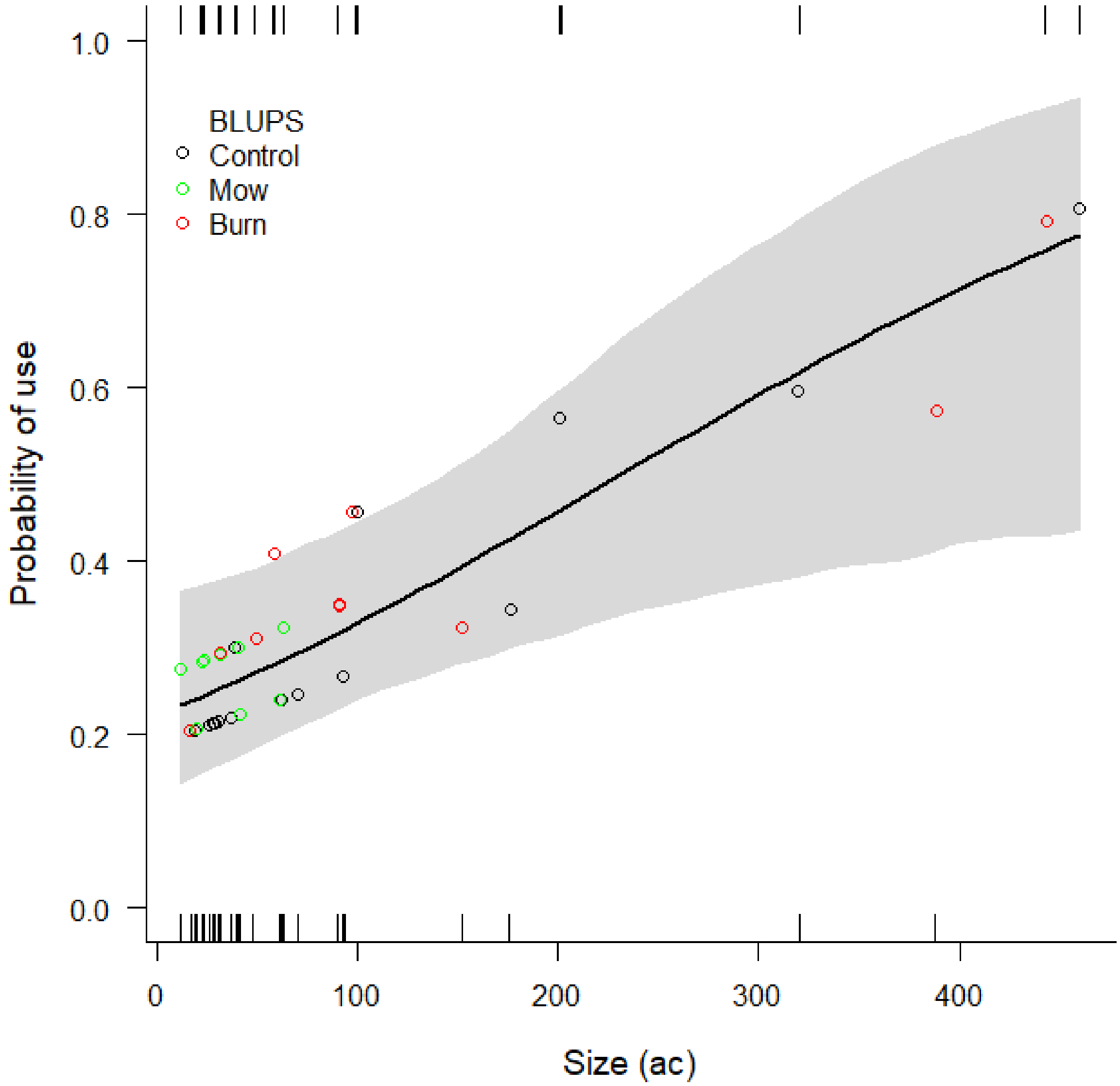

Figure 4. The relationship between the area disturbed (in acres) and the probability of sharp-tailed grouse use of control, mowed, and burned sites in northwest Minnesota during 2015-2018. Sample sizes of used (1) and unused (0) are indicated by tick marks at the top and bottom of the figure, respectively.

USE OF MANAGED FOREST WILDLIFE OPENINGS BY AMERICAN WOODCOCK

Lindsey Shartell

SUMMARY OF FINDINGS

American woodcock (*Scolopax minor*) were surveyed at permanently managed forest wildlife openings in northern Minnesota. Singing ground surveys conducted from mid-April through May during 2016, 2017, and 2018 indicated that 72% of openings were used by singing male woodcock during at least one year of study. Roosting ground surveys conducted from June through August during 2016 and 2018 indicated that 70% of openings were used by woodcock in at least one year of study. For both singing and roosting openings, woodcock were more likely to use larger openings with smaller perimeter-to-area ratios. In addition to woodcock surveys, vegetation data along transects were collected within openings to assess the relationship of vegetation metrics to woodcock use and management of habitat in openings. Openings used during singing ground surveys had a higher proportion of grass and lower proportions of herbaceous plants. Openings used by roosting woodcock had higher proportions of grass, lower proportions of shrubs, and shorter herbaceous vegetation. Years since management was not an important driver of singing ground use but was significant for roosting ground use. Openings managed more recently were more likely to have roosting woodcock and also had lower proportions of shrubs and shorter shrubs and trees. However, across all openings, shrubs and trees were uncommon, and most openings had been managed within the past 4 years. The proportion of woodcock habitat adjacent to openings was an important factor in determining singing ground use. The surrounding habitat may be a more important consideration than the vegetation within openings. Wildlife managers interested in creating and maintaining singing and roosting habitat for woodcock in forest-dominated areas should create openings at least 1 ac in size with a large core area, locate openings in areas with abundant woodcock habitat, and use management (mowing) as needed to decrease vegetation height and prevent the establishment of shrubs and trees.

BACKGROUND

The American woodcock (*Scolopax minor*) is a popular migratory game bird and a Species of Greatest Conservation Need in Minnesota [Minnesota Department of Natural Resources (MNDNR) 2015]. In 2015, Minnesota had an estimated 13,500 active woodcock hunters harvesting 25,600 woodcock, ranking Minnesota third highest in the country for both woodcock hunter and harvest numbers (Seamans and Rau 2016). Annual woodcock surveys have indicated a long-term (1968-2016) decline in singing male numbers across the full breeding range (Seamans and Rau 2016). These declines have been attributed to the loss of open and early successional forest and shrub habitat due to succession, lack of disturbance, and development (Dessecker and McAuley 2001).

Woodcock require a variety of habitat components including dense young forests or shrublands and open singing and roosting grounds (Wildlife Management Institute 2009). Woodcock move frequently between these habitat types, often being found in forests during the day and open sites at night (Sheldon 1967). In the spring, male woodcock use openings as breeding sites, called singing grounds, where they perform their courtship ritual. Females nest and raise broods in the forest surrounding these openings (Sheldon 1967). Both nest and brood locations have been found to be associated with short distances to openings or forest edges (Gregg and Hale 1977, Daly 2014). In the summer, woodcock make evening crepuscular flights to open habitats to roost. Open roosting grounds provide the benefit of reduced predation risk (Masse et al. 2013). Historically, disturbance by fire, wind, Native American activities, flooding, and beavers created openings and early successional habitat for woodcock (DeGraaf and Yamasaki 2003). Many of these disturbances that created and maintained open areas are now prevented. Pastures, fields, agricultural sites, and recent clearcuts (Hale and Gregg 1978, Long and Locher 2013) can all serve as open habitat for woodcock, but in areas dominated by forest cover, managed forest wildlife openings can be used to provide this habitat component.

The secretive nature and cryptic coloration of the woodcock makes it difficult to estimate population size and management effects. There have been past studies assessing the use of openings by woodcock, but most have been focused on the wintering grounds in the southeastern United States (for example Glasgow1958, Stribling and Doerr 1985, Berdeen and Krementz 1998). Fewer studies have explored woodcock use of summer roosting grounds in the northern part of their range (though see Sheldon 1961, Sepik and Derleth 1993, Masse et al. 2013), and even fewer have incorporated habitat characteristics and management into studies of use.

The Upper Great Lakes Woodcock and Young Forest Initiative published best management practices for woodcock in 2009. Their recommendations call for establishing eight singing grounds at least 0.5 acres in size and one roosting field at least 5 acres in size per 100 acres of land (Wildlife Management Institute 2009). Open sites should cover not more than 20 percent of the area, and the remaining land should consist of abundant feeding, nesting, and brood-rearing habitat (Wildlife Management Institute 2009). The MNDNR maintains permanent forest wildlife openings to provide singing and roosting grounds for woodcock, as well as habitat for a variety of other game and non-game species such as deer and bear. These openings require regular mechanical treatment to prevent the establishment of shrubs and trees. Management is most commonly fall mowing; however, mowing in other seasons, prescribed burning, and herbicide have also been used. Wildlife managers would like to improve their management of forest openings to maximize benefit, but do not know the optimal frequency of treatment. In addition, not all managed openings are used by woodcock. Understanding the factors that influence use, such as opening size and configuration, vegetation composition and structure, and surrounding landscape characteristics would improve the creation of forest openings and focus management on those openings expected to provide the greatest benefit. This information will allow for the development of better management practices for land managers and landowners interested in providing wildlife openings for woodcock and other wildlife.

OBJECTIVES

1. Assess woodcock use of managed forest wildlife openings with differing management history (years since mowing or burning).
2. Relate opening size and configuration, vegetation composition and height, and surrounding landscape characteristics to woodcock use and management history.
3. Develop recommendations to improve the current management of forest wildlife openings.

METHODS

Singing ground surveys for American woodcock were conducted from mid-April through May 2016 in forest openings within the Grand Rapids, Cloquet, and Red Lake work areas as part of a pilot project and from mid-April through May 2017 and 2018 in the Grand Rapids work area. Surveys followed Singing Ground Survey (SGS) protocol where possible (Seamans and Rau 2016). Surveys generally took place 15 to 60 minutes after sunset, when temperature was above 35 F, and there was no heavy precipitation or strong wind. Openings in close proximity were grouped to allow surveying multiple openings per evening. At each opening observers recorded their GPS location (UTM coordinates), time of sunset, cloud cover, temperature, wind speed, precipitation, and any noise disturbance present at the time of the survey. Observers listened for and recorded the number of different woodcock heard peenting or observed displaying (heard and/or seen) within and over the opening during a listening period of at least 5 minutes. Observers also recorded other observations of woodcock (not within the opening) along with time and approximate location (direction and distance) of the woodcock.

Roosting ground surveys were conducted at openings June through August 2016 and 2018 using crepuscular flight surveys and spotlighting within the opening (Glasgow 1958, Berdeen and Krementz 1998). Roosting surveys were not conducted in 2017 due to funding constraints. For crepuscular surveys, the observer was positioned on the edge of the opening and recorded the number of woodcock observed flying into the opening, over the opening, or heard in the opening (when not seen). Surveys were conducted from 20 minutes before sunset to 40 minutes after sunset (a one hour period). Observers recorded their GPS location (UTM coordinates), time of sunset, cloud cover, temperature, wind speed, precipitation, and any noise disturbance present at the time of the survey. After the survey window, observers systematically walked openings using spotlights and recorded the number of woodcock flushed or spotted.

Vegetation characteristics were sampled as close as possible in time to the survey date and repeated for roosting surveys. A point intersect method (Levy and Madden 1933) was used to determine proportion and height of 5 classes of vegetation (grass, herbaceous, woody, shrub, and tree). Two transects were sampled per opening, one placed across the widest part of the opening from edge to edge, and the second placed perpendicular to the first. Observers recorded vegetation class (or other substrate type if no vegetation present) and maximum height for each class every 1.0 m along the transect. Observers also described the habitat across the entire opening (e.g., number of trees, distribution of trees, percent shrub cover) and the surrounding habitat by type (e.g., upland forest, lowland forest, upland shrub), tree or shrub species, and coarse age class (young, middle, old). Presence of tansy, or other aggressive invasive species were recorded from a visual assessment across the opening as percent cover in 10% increments. Presence of a mowed or packed trail within the opening was noted, as these may provide persistent areas of short vegetation regardless of years since management. To obtain an accurate estimate of opening size and shape, the edge of the opening was walked using a GPS unit to digitize the boundary of the opening. Forest inventory data was used to determine the proportion of the opening adjacent to woodcock habitat. For this study, woodcock habitat included the cover types young deciduous forest ≤18 years old, lowland and upland brush, and lowland and upland grass.

To assess the frequency of use of openings in this study by other wildlife the presence of scat encountered within 0.5 m on either side of the vegetation transects was noted. Location along the transect and species (when identifiable) was recorded for each encounter.

Statistical Analysis

Singing and roosting ground use were analyzed separately. Because 89% of singing ground surveys and 76% of roosting ground surveys had woodcock counts ≤1, opening use was assessed as a binary response variable. Predictor variables included opening size (ac), opening perimeter (m), proportion of vegetation in 5 classes (grass, herbaceous, woody, shrub, and tree), average height (m) of vegetation in 5 classes, presence of a road or trail, years since management (mowing or burning), and proportion of the opening adjacent to woodcock habitat. In addition, because opening size (ac) and perimeter (m) were strongly correlated and both highly skewed toward low values, the perimeter-to-area ratio was used to represent both opening size and configuration. Student's t-tests with a significance level of $P < 0.05$ were used to explore the importance of predictor variables on woodcock use of openings. Linear regression was used to relate vegetation variables to years since management. All statistical analyses were conducted using the statistical package R (R Core Team 2018).

Following univariate analysis, models were run using the function "glmer" in the R package "lme4" (Bates et al. 2015, R Core Team 2018) to fit mixed-effect logistic regression models to the analysis dataset. To help with model convergence, size of opening, perimeter, and perimeter-to-area ratio were centered (mean subtracted) and scaled (divided by the standard deviation). Years since management was modeled as a discrete numeric variable. Because few sites had multiple surveys, we did not attempt to account for false zeros due to non-detection. Therefore, "probability of site use" is defined conservatively in this analysis. The R function "drop1" was used with AIC to conduct a stepdown variable-selection exercise. A NULL model that included an intercept term and a random effect for site was also computed. Second-order Akaike information criterion (AICc) was used to identify models with reasonable data support (i.e., ΔAICc < 2), and the function "r.squaredGLMM" in the R package "MuMIn" was used to compute a marginal coefficient of determination (pseudo R^2) for the fixed effects in the models.

For modeling singing ground use, a subset of 61 openings surveyed in both 2017 and 2018 and under good conditions (no or light precipitation, wind < 12 mph, and temperature > 35 F) was used. For singing grounds, a principal components analysis, with a correlation matrix, was used to reduce the dimensionality of vegetation-composition metrics associated with each opening producing pc1 and pc2 scores as potential model covariates. The simplest significant vegetation metric, proportion grass (pg), was also considered as an alternative to using principal component scores. Singing ground models included the following covariates as fixed effects: years since management (yrmg), perimeter-to-area ratio (par), proportion of the opening adjacent to woodcock habitat (hab), vegetation composition metrics from a principal components analysis (pc1 and pc2), and proportion grass (pg). All models also included a random effect for site, to accommodate repeated measurements over time.

For modeling roosting ground use, a subset of 49 openings surveyed in both 2016 and 2018 and under good conditions was used. For roosting grounds, a principal components analysis was not used and the 4 vegetation metrics most related to woodcock use (pg, ps, hh, and hs) were considered. Roosting ground models included the following covariates as fixed effects: years since management (yrmg), size of the opening (ac), proportion of the opening adjacent to woodcock habitat (hab), proportion grass (pg), proportion shrub (ps), average height of herbaceous vegetation (hh), and average height of shrub vegetation (hs). All models also included a random effect for site.

RESULTS

Singing Ground Use

In 2016, singing ground surveys were conducted at 85 forest openings, with singing males observed at 51 openings (60%). The majority of openings with woodcock (43 of 51) had only 1 male present, 6 openings had 2 males, and 2 openings had 3 males. In 2017, singing ground surveys were conducted at 64 openings, with males observed at 41 openings (64%). At 33 openings 1 male was observed and 8 openings had 2 males present. In 2018, singing ground surveys were conducted at the same 64 openings as the previous year, with males being observed at 34 openings (53%). At 28 openings 1 male was observed, and 2 males were observed at 6 openings. Across the duration of the study, 71 of the 99 unique openings (72%) were used by singing male woodcock during at least one year of study. A limited number of repeat visits were conducted within the same year to aid in assessing detection ability. A total of 24 openings were surveyed on 2 occasions during the sample period. Presence or absence of woodcock was the same for 20 openings (83%, 9 presence and 11 absence). At the remaining openings, 2 indicated presence during the first survey and absence in the second, and 2 indicated absence in the first survey and presence in the second.

Openings surveyed ranged in size from 0.48 to 16.33 ac, with a mean size of 2.79 ± 0.29 ac. Opening size and configuration were significantly related to woodcock use. Openings used by woodcock were larger in area ($t = -4.39$, $P < 0.01$) and had greater perimeter length ($t = -4.01$, $P < 0.01$). However, perimeter-to-area ratio was significantly smaller for openings used by woodcock (i.e., shorter perimeter and larger area, $t = 4.53$, $P < 0.01$), suggesting that area is the more important driver. Despite this, woodcock were observed using openings as small as 0.63 ac.

Vegetation composition in the opening showed some relationship to woodcock use for singing (Table 1). Openings used by woodcock for singing had a significantly higher proportion of grass ($P < 0.01$) and a lower proportion of herbaceous vegetation ($P = 0.02$). Most openings were dominated by grass with few shrubs and trees. Height of the vegetation in each class was not found to be significantly related to woodcock use (Table 1). The proportion of the opening adjacent to woodcock habitat (openland, brushland, and young deciduous forest) was significantly related to woodcock use ($t = -4.62$, $P < 0.01$), with woodcock using openings with a greater proportion of adjacent habitat.

Woodcock were anecdotally observed using un-vegetated or packed down roads and trails in openings. Presence of a road or trail, however, was not significantly related to woodcock use ($t = 1.54$, $P = 0.13$). Number of years since management (mowing or burning) was also not significantly related to woodcock use ($t = 1.13$, $P = 0.26$), however nearly all surveys took place at sites that had been managed within the past 4 years, and the majority (62%) of surveys were conducted at sites managed within 2 years. Vegetation composition and height within openings was related to years since management (Table 2). The proportion of grass significantly decreased with years since management ($P < 0.01$), whereas the proportion of shrub significantly increased ($P < 0.01$). Woody height and shrub height also significantly increased with years since management ($P = 0.02$, $P < 0.01$).

The best supported model predicting probability of use of openings as singing grounds included perimeter-to-area ratio (par), proportion of the opening adjacent to woodcock habitat (hab), proportion of grass in the opening (pg), and a random intercept for site (Table 3). Smaller perimeter-to-area ratio, higher proportion of adjacent habitat, and higher proportion of grass increased the probability of woodcock use. Other supported models ($\Delta AICc < 2$) were the model including perimeter-to-area, habitat, and site and the model including perimeter-to-area, habitat, vegetation metrics (pc1), and site.

Roosting Ground Use

Roosting ground surveys were conducted at 63 openings in 2016 and at 64 openings in 2018. In 2016, woodcock were observed at 42 openings (67%) during crepuscular surveys, and roosting woodcock were spotlighted and/or flushed at 14 openings (22%). In 2018, woodcock were observed at 27 openings (42%), and roosting woodcock were spotlighted and/or flushed at 15 openings (23%). Both survey methods provided useful information on woodcock use, thus openings with woodcock observations during either survey were used for the analysis. In 2016, 44 openings (70%) were used by woodcock and in 2018, 28 openings (44%) were used.

Across the duration of the study, 52 of the 74 unique openings (70%) were used by woodcock during at least one year of study. Roosting surveys were not repeated due to time limitations, but other research has found that the frequency of roosting field use by individual woodcock varies by month and by age and sex, peaking in June and July (Sepik and Derleth 1993). However, there was no significant relationship found between date and the number of woodcock observed at roosting openings in this study in which surveys were conducted from June through August.

Openings used by woodcock were larger in area ($t = -3.13$, $P < 0.01$) and had greater perimeter length ($t = -2.66$, $P < 0.01$). The perimeter-to-area ratio was significantly smaller for used openings ($t = 2.28$, $P = 0.01$). Despite these findings, woodcock were observed using openings as small as 0.59 ac.

Openings used by roosting woodcock showed some relationship to vegetation metrics (Table 4). Used openings had a significantly higher proportion of grass ($P < 0.01$) and a lower proportion of shrub ($P = 0.04$). Herbaceous vegetation height was also significantly lower in openings used by woodcock ($P = 0.01$). The proportion of the opening adjacent to woodcock habitat was not significantly related to woodcock use ($t = -1.86$, $P = 0.07$). Openings used by woodcock had been managed more recently (i.e., fewer years since management, $t = 2.58$, $P = 0.01$). Years since management was also positively related to proportion of shrub ($P < 0.01$) and shrub and tree height ($P < 0.01$, $P = 0.03$), but was not related to other vegetation classes (Table 5).

The best supported model predicting probability of use of openings as roosting grounds included size of the opening (ac), shrub height (hs), and a random intercept for site (Table 6). Larger area and shorter shrub height increased the likelihood of woodcock use. Other supported models ($\Delta AICc < 2$) were the model including opening size, shrub height, herbaceous height, and site and the model including opening size, shrub height, herbaceous height, proportion grass and site.

Use of Openings by Other Wildlife

Across 3 years of study, scat from bear, coyote, deer, fox, goose, moose, opossum, porcupine, rabbit, raccoon, ruffed grouse, and wolf, as well as a woodcock nest and mallard nest, were observed along vegetation transects in forest wildlife openings. Scat was observed in at least one year of study at 74 of 99 unique openings (75%). Deer scat was the most commonly observed, with 60 openings (61%) having deer scat in at least 1 year.

DISCUSSION

The use of forest wildlife openings as singing grounds followed expected outcomes in that woodcock were more likely to use larger openings and openings with greater amounts of surrounding woodcock habitat. However, results suggested that frequency of mowing openings (within a 5 year time period) is not important in determining use as singing grounds. Management on a longer time frame is likely still important. The majority of openings in this study had been treated within the last 5 years and were relatively free of shrubs and trees.

Vegetation metrics generally were not strongly related to woodcock use. This could be partly attributed to the low variation in vegetation metrics across openings. However, other studies have suggested that quantifying the structure and composition of the singing ground opening may be of little value compared to the surrounding habitat (Sepik et al. 1993). Male woodcock select openings with surrounding nesting habitat as they have a higher probability of attracting females (Dwyer et al. 1988). Thus, higher consideration should be placed on the surrounding habitat and its management when selecting openings for woodcock.

The use of openings as roosting grounds was more likely for larger openings, which is consistent with other studies and best management practices (Wildlife Management Institute 2009). However, best management practices recommend roosting grounds of at least 5 acres, whereas in this study woodcock were found using much smaller openings. The availability of openings on the landscape may play a role in the use of smaller openings and should be explored. In addition, females are known to use smaller forest openings for roosting to remain closer to feeding areas, and males may remain in smaller forest openings through the summer to maintain or establish singing grounds (Sepik and Derleth 1993). Nevertheless, providing small forest openings when large roosting grounds are not available will offer roosting habitat for woodcock.

Roosting grounds had higher proportions of grass and lower proportions of shrub. Open roosting grounds are thought to provide a habitat where woodcock can become aware of and escape from predators. Management frequency at openings was also related to shrub abundance and the height of shrubs and trees. Thus woodcock likely preferred to use sites managed more recently, as they provided more open habitat. Unlike singing grounds, openings used for roosting were not significantly related to adjacent woodcock habitat. Woodcock are known to make longer flights to roosting areas, so this was not unexpected.

It was noted anecdotally in 2016 that sites heavily invaded by common tansy (*Tanacetum vulgare*), an exotic invasive plant, tended to have no woodcock present. In 2017 presence of tansy and other invasive species was recorded; however, few sites had invasive plants dominating the opening, restricting statistical analysis. Future research could explore the effects of invasive species on woodcock use of openings.

Researchers have also studied the use of aspen clearcuts in Wisconsin and young pine plantations in Arkansas by woodcock in spring and summer, finding that woodcock utilize these areas (Hale and Gregg 1978, Long and Locher 2013). Forest harvest in Minnesota is common, and these areas are likely an important component of woodcock habitat. Recent forest harvests were adjacent to and in close proximity to some of the wildlife openings in this study. It was noted that woodcock were seen using the harvest sites for singing and roosting. Additional research comparing the use and characteristics of temporary openings such as clearcut harvests to permanent openings for both singing and roosting grounds would improve our understanding and provide context for management in Minnesota.

Wildlife managers interested in providing singing and roosting habitat for woodcock should continue to create and maintain wildlife openings in forest-dominated areas. Opening size should continue to follow best management practices from the Upper Great Lakes Woodcock and Young Forest Initiative (at least 0.5 ac in size), however larger openings of at least 1 ac in size with a large core area are preferred. Perhaps of more importance, openings should be located in areas with abundant woodcock habitat (young deciduous forest and brushland). Annual management (mowing) of forest wildlife openings is not necessary, however mowing should be used as needed to decrease vegetation height and prevent the establishment of shrubs and trees. A management regime consisting of mowing every 4 to 5 years should be sufficient at sites where shrub and tree establishment is slow. Forest wildlife openings are not only frequently used by woodcock, many other species of wildlife were shown to use these openings, and they also offer opportunities for wildlife viewing and hunting. Incorporating forest wildlife openings in to forest habitat management can provide multiple benefits.

ACKNOWLEDGMENTS

This project was made possible thanks to the dedicated fieldwork of Andrew Erickson and Nicole Benson and the cooperation of wildlife staff, especially Jeff Hines, Bruce Anderson, Charlie Tucker, and Ted Dick. Guidance on statistical design and analysis was provided by

John Giudice. This study was funded in part through the Wildlife Restoration (Pittman-Robertson) Program.

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Table 1. Mean values for proportion and height (m) of 5 vegetation classes at used and un-used forest wildlife openings sampled for singing woodcock from 2016-2018 in Minnesota and results of student's t-tests for differences. Asterisk indicates a significant difference between used and un-used openings.

Metric	Mean used	Mean un-used	t	p-value
Proportion grass (pg)	0.82	0.72	-3.89	< 0.01*
Proportion herbaceous (ph)	0.61	0.69	2.34	0.02*
Proportion woody (pw)	0.14	0.16	1.63	0.10
Proportion shrub (ps)	0.12	0.12	0.14	0.89
Proportion tree (pt)	0.02	0.03	1.37	0.17
Height grass (hg)	0.12	0.12	0.28	0.78
Height herbaceous (hh)	0.10	0.11	1.01	0.31
Height woody (hw)	0.26	0.25	-0.10	0.92
Height shrub (hs)	0.72	0.68	-0.67	0.50
Height tree (ht)	1.91	1.46	-1.87	0.06

Table 2. Linear regression results for proportion and height (m) of 5 vegetation classes varying with years since management at forest wildlife openings surveyed for singing woodcock use from 2016-2018 in Minnesota. Asterisks indicate a significant trend.

Metric	Estimate	p-value
Proportion grass (pg)	-0.03	< 0.01*
Proportion herbaceous (ph)	0.002	0.90
Proportion woody (pw)	0.004	0.56
Proportion shrub (ps)	0.02	< 0.01*
Proportion tree (pt)	0.002	0.28
Height grass (hg)	0.003	0.49
Height herbaceous (hh)	0.006	0.14
Height woody (hw)	0.02	0.02*
Height shrub (hs)	0.10	< 0.01*
Height tree (ht)	0.11	0.28

Table 3. Mixed-effect logistic regression modeling results for woodcock use of forest wildlife openings as singing grounds from 2016-2018 in Minnesota. Model variables include fixed effects for years since management (yrmg), perimeter-to-area ratio (par), proportion of the opening adjacent to woodcock habitat (hab), vegetation composition metrics from a principal components analysis (pc1 and pc2), proportion grass (pg), and a random effect for site.

Model	K[a]	AICc[b]	ΔAICc[c]	ModelLik	AICcWt[d]	LL[e]	Pseduo R^2	Model structure
M5	5	139.7	0.00	1.000	0.361	-64.60	0.39	par + hab + pg + (1 | site)
M4	4	140.3	0.59	0.746	0.269	-65.98	0.35	par + hab + (1 | site)
M3	5	140.9	1.18	0.556	0.200	-65.19	0.38	par + hab + pc1 + (1 | site)
M2	6	141.9	2.16	0.340	0.123	-64.57	0.39	par + hab + pc1 + pc2 + (1 | site)
M1	7	143.8	4.05	0.132	0.048	-64.39	0.40	yrmg + par + hab + pc1 + pc2 + (1 | site)
M0	2	160.0	20.30	0.000	0.000	-77.96	---	1 + (1 | site)

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

[b] AICc = second-order Akaike information criterion.

[c] ΔAICc = difference in AICc relative to the best performing model.

[d] AICcWt = Akaike weight representing relative model support.

[e] LL = log likelihood value.

Table 4. Mean proportion and height (m) metrics for vegetation at used and un-used wildlife openings sampled for roosting woodcock during 2016 and 2018 in Minnesota and results of student's t-tests for differences. Asterisk indicates a significant difference between used and un-used openings.

Metric	Mean used	Mean un-used	t	p-value
Proportion grass (pg)	0.78	0.70	-2.70	< 0.01*
Proportion herbaceous (ph)	0.84	0.85	0.38	0.70
Proportion woody (pw)	0.21	0.15	-1.64	0.10
Proportion shrub (ps)	0.23	0.30	2.09	0.04*
Proportion tree (pt)	0.04	0.03	-1.10	0.28
Height grass (hg)	0.67	0.64	-0.66	0.51
Height herbaceous (hh)	0.67	0.79	2.57	0.01*
Height woody (hw)	0.38	0.40	0.55	0.58
Height shrub (hs)	0.80	0.92	1.94	0.06
Height tree (ht)	0.83	0.78	-0.46	0.65

Table 5. Linear regression results for proportion and height (m) of 5 vegetation classes varying with years since management at forest wildlife openings surveyed for roosting woodcock use during 2016 and 2018 in Minnesota. Asterisks indicate a significant trend.

Metric	Estimate	p-value
Proportion grass (pg)	-0.01	0.17
Proportion herbaceous (ph)	-0.002	0.81
Proportion woody (pw)	-0.01	0.34
Proportion shrub (ps)	0.04	< 0.01*
Proportion tree (pt)	0.002	0.55
Height grass (hg)	0.004	0.77
Height herbaceous (hh)	0.03	0.10
Height woody (hw)	0.009	0.55
Height shrub (hs)	0.10	< 0.01*
Height tree (ht)	0.06	0.03*

Table 6. Mixed-effect logistic regression modeling results for American woodcock use of forest wildlife openings as roosting grounds during 2016 and 2018 in Minnesota. Model variables include fixed effects for years since management (yrmg), size of the opening (ac), proportion of the opening adjacent to woodcock habitat (hab), proportion grass (pg), proportion shrub (ps), average height of herbaceous vegetation (hh), average height of shrub vegetation (hs), and a random effect for site.

Model	K[a]	AICc[b]	ΔAICc[c]	ModelLik	AICcWt[d]	LL[e]	Pseduo R^2	Model structure
M5	4	124.8	0.00	1.00	0.31	-58.20	0.29	ac + hs + (1 | site)
M4	5	125.1	0.27	0.88	0.27	-57.23	0.33	ac + hs + hh + (1 | site)
M3	6	125.7	0.83	0.66	0.20	-56.37	0.34	ac + hs + hh + pg + (1 | site)
M2	7	127.3	2.45	0.29	0.09	-56.02	0.35	ac + hs + hh + pg + yrmg + (1 | site)
M1	8	129.5	4.62	0.10	0.03	-55.92	0.35	ac + hs + hh + pg + ps + yrmg + (1 | site)
M0	2	137.8	12.96	0.00	0.00	-66.84	---	1 + (1 | site)

[a] K = number of parameters in the model.
[b] AICc = second-order Akaike information criterion.
[c] ΔAICc = difference in AICc relative to the best performing model.
[d] AICcWt = Akaike weight representing relative model support.
[e] LL = log likelihood value.

GAINING A DEEPER UNDERSTANDING OF CAPTURE-INDUCED ABANDONMENT OF MOOSE NEONATES[1]

Glenn D. DelGiudice, William J. Severud[2], Tyler R. Obermoller, Véronique St-Louis

ABSTRACT

Capture-induced abandonment of ungulate neonates has been poorly understood until recently, likely often underestimated, and anecdotally reported to occur at variable rates. This complex maternal behavior adversely affects the accuracy, efficiency, cost-effectiveness, and consequently the overall value of behavioral and survival studies. To follow-up on a previous study where we reported an 18.4% rate of abandonment of moose (*Alces alces*) neonates following helicopter-assisted capture in Minnesota, USA, we tracked the movement behavior of 12 and 13 moose neonates fitted with global positioning system (GPS) collars during 8–15 May 2014 (phase 1) and 21 May–19 June (phase 2), respectively. These efforts were part of an overall study of reproductive success and cause-specific mortality in Minnesota's remaining viable but declining moose population. During phase 1, 7 (3 M, 4 F) of 12 (6 M, 6 F) neonates were abandoned by 5 of 9 dams. Our capture-induced abandonment contingency plan and monitoring of hourly location fixes of the GPS-collared newborns and their dams allowed us to recover 6 of the 7 abandoned neonates alive and in good condition. During phase 2, we reduced our capture team from 3 to 4 to 2 persons and limited handling to fitting the GPS collar and sexing the neonate (mean = 0.7 min). Capture-induced abandonment decreased to 1 of 10 dams abandoning a set of twins. Mean distance of dams to capture site (calving site) 1 hour pre- and 1 hour post-capture did not indicate a predisposition to abandonment. However, differences in distances of dam to capture site, dam to neonate(s), and neonate to capture site over 48–96 hours post-capture suggested a clear pattern of capture-induced abandonment. None of the birth, capture, neonate, or dam characteristics examined indicated a predisposition to capture-induced abandonment at the study cohort level. However, minimizing capture-induced abandonment through rapid handling of neonates will greatly increase the overall value of field studies that rely on the capture of animals. © 2017 The Wildlife Society.

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ASSESSING EXPANDABLE GLOBAL POSITIONING SYSTEM COLLARS FOR MOOSE NEONATES[1]

Tyler R. Obermoller, Glenn D. DelGiudice, William J. Severud[2]

ABSTRACT

Deploying Global Positioning System (GPS) collars on ungulate neonates would offer notable advantages to examining their life history and influence on population performance. During 2013 and 2014, we deployed expandable GPS collars on 74 moose (*Alces alces*) neonates in Minnesota, USA, to estimate survival and cause-specific mortality during their first year. Collars slipped from 10.5% and 62.5% of calves at 15.8 (± 4.5 [SE]) and 27.9 (± 8.1) days postcapture in 2013 and 2014, respectively, from premature deterioration of the breakaway mechanism or excessive band expansion. We conducted various controlled tests on the bands to quantify potential design flaws. We placed 8 bands (with GPS package) around a polyvinyl chloride (PVC) pipe outdoors (exposed to weather) with clear plastic tubing (sleeve) to prevent neck abrasions, 7 collars outdoors with no sleeve, and 7 collars indoors with no sleeve. We dropped each pipe 10 cm 50 times in the morning and in the afternoon daily for 4 weeks to simulate animal movement and test elastic memory. Circumference of bands from the 3 treatment groups increased 14.6 (± 2.5), 8.5 (± 2.9), and 3.9 (± 2.4) cm, respectively, with 41.9% attributed to the sleeve, 26.9% to simulated animal movement, and 31.2% to weather exposure. Circumference of control group bands (indoors, not bounced) did not change. After design modifications were made to the collar, the band length increased only 1.5 ± 0.6 cm during a 4-week trial. Subsequently, we placed 6 of these collars on confined and sheltered Holstein dairy calves; 5 retained their collar during an 8-week test. After increasing the strength of the expandable loops via sewing, we placed 4 collars on pastured Angus beef calves. Three of 4 slipped their collars at 42.4 (± 8.9) days. Our results indicate additional modifications of the band are needed before GPS-collaring of moose neonates is resumed. © 2018 The Wildlife Society.

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MATERNAL BEHAVIOR INDICATES SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF MOOSE CALVES[1]

Tyler R. Obermoller, Glenn D. DelGiudice, William J. Severud[2]

ABSTRACT

Continuing research on cause-specific mortality and annual survival of moose (*Alces alces*) calves in northeastern Minnesota, USA, is important to understanding the long-term trajectory of the population. In 2013 and 2014, we observed global positioning system (GPS)-collared, female moose exhibit a specific behavior (i.e., mortality movement) associated with the death of their GPS-collared neonate. The females made a rapid, long-distance movement (flee), followed by a return to the calf mortality site. We used characteristics of this movement in 2013–2014 (n = 46) to develop models for assessing calf survival, and then evaluated these models using female movement rates (*n* = 49) in 2015−2016. Using this behavior as an indicator of calf mortality in 2016, we conducted field investigations, leading to evidence of 15 mortalities at a mean age of 30.6 ± 15.5 (SE) days (range = 3–243 days). We launched 21 investigations in response to a mortality movement and they resulted in confirmation of 11 of the 15 calf mortalities. Specific causes of mortality included 9 wolf (*Canis lupus*)-kills, 3 black bear (*Ursus americanus*)-kills, 1 unknown predator-kill, and 2 deaths following vehicle collisions. The mean distance females fled after a mortality was 1,873 ± 412 m (range = 126–5,805 m, *n* = 14). Females that made return visits returned a mean 2.8 ± 0.5 times (range = 1–5, *n* = 8) to within a mean 106 ± 22 m (range = 34–230 m, *n* = 8) of the mortality site. Calf survival to 30 days of age was 67 ± 8% (95% CI = 53–84%, *n* = 36) but declined to 53 ± 8% (95% CI = 39–72%, *n* = 36) by 3 months of age. We developed 2 population-level movement models to improve the efficacy of using the mortality movement to identify and locate calf mortalities in real time via field investigations. The first approach, a temporal-based model, used a 3-day average movement velocity threshold (118 m/hr) for all females to indicate calf mortality and accurately predicted survival status in 51% (*n* = 105) of the cases. The second approach, an age-specific model using different thresholds (28–135 m/hr) for females relative to calf age, was 80% (n = 231) accurate. Using movement behavior of females to assess calf mortality yielded important insights into mechanisms influencing the population decline that will inform future management decisions. © 2019 The Wildlife Society

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[1] Abstract from published paper: Obermoller, T. R., G. D. DelGiudice, and W. J. Severud. 2019. Maternal behavior indicates survival and cause-specific mortality of moose calves. Journal of Wildlife Management 83:790–800.

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ASSOCIATION OF MOOSE PARTURITION AND POST-PARTURITION HABITAT WITH CALF SURVIVAL[1]

William J. Severud[2], Glenn D. DelGiudice, Tyler R. Obermoller

ABSTRACT

Habitat use during calving and the energetically demanding post-parturition period can be an important determinant of neonatal survival. The moose (*Alces alces*) population in northeastern Minnesota, USA declined 65% from 2006 to 2018. During 2013–2015, annual survival of calves was estimated as low as 28%. We remotely monitored global positioning system (GPS)-collared adult female moose and their neonates during the calving and post-parturition seasons to examine calving movements, birth-sites, habitat use, survival, and cause-specific mortality of neonates. Identifying the association of specific landscape characteristics with neonate survival should yield insight into mechanisms contributing to the declining moose population and serve as a basis for an ecologically sound management response. We compared habitat characteristics of pre-calving, calving, peak-lactation, and mortality sites at a fine and broad scale. We also compared calving sites of females that successfully reared a calf to winter to those that did not. In general, females tended to move to areas of more conifer cover to calve. During peak-lactation, females and their calves used steeper areas with abundant forage and high concealment but less conifer cover. Mortalities occurred at sites that were more level than other site types. Females that successfully reared a calf to 1 February typically calved in areas with more deciduous forest and less forested wetland cover than females whose calves died before 9 months of age. Habitat improvement projects for moose should consider forage requirements and placement on the landscape in relation to cover and slope. © 2018 The Wildlife Society.

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[1] Abstract from published paper: Severud, W. J., G. D. DelGiudice, and T. R. Obermoller. 2019. Association of moose parturition and post-parturition habitat with calf survival. Journal of Wildlife Management 83:175–183.

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SURVIVAL AND CAUSE-SPECIFIC MORTALITY OF MOOSE CALVES IN NORTHEASTERN MINNESOTA[1]

William J. Severud[2], Tyler R. Obermoller, Glenn D. DelGiudice, John R. Fieberg[2]

ABSTRACT

Ungulate reproductive success (calf production and survival) influences population performance. The moose (*Alces alces*) population in northeastern Minnesota, USA, has declined 65% from 2006 to 2018 but has begun to stabilize. Because causes of this decline were largely unknown, we investigated production, survival, and cause-specific mortality of calves of the global positioning system (GPS)-collared females in this population. In 2013 and 2014, we GPS-collared 74 neonates and monitored them for survival. In 2015 and 2016, we monitored 50 and 35 calving females for signs of neonatal mortality using changes in adult female velocities and assessed seasonal calf survival by aerial surveys. In 2013 and 2014 (pooled), survival to 9 months was 0.34 (95% CI = 0.23–0.52) for collared calves, and in 2015 and 2016 (pooled) survival was 0.35 (95% CI = 0.26–0.48) for uncollared calves. Mortality in all 4 years was high during the first 50 days of life. In 2013 and 2014 (pooled), calving sites were relatively safe for collared neonates; predator-kills occurred a median 17.0 days after departure and a median 1,142 m from calving sites. Predation was the leading cause of death of collared calves (84% of mortalities), with wolves (*Canis lupus*) accounting for 77% of these. Other forms of mortality for collared and uncollared calves included drowning, infection, vehicle collision, and natural abandonment. We documented higher wolf predation than other recent studies with similar predator communities. Identifying specific causes of calf mortality and understanding their relations to various landscape characteristics and other extrinsic factors should yield insight into mechanisms contributing to the declining moose population in northeastern Minnesota and serve as a basis for ecologically sound management responses.

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[1] Abstract from paper published in early view: Severud, W. J., T. R. Obermoller, G. D. DelGiudice, and J. R. Fieberg. 2019. Survival and cause-specific mortality of moose calves in northeastern Minnesota. Journal of Wildlife Management 83:in early view.

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COMPARING SURVEY AND MULTIPLE RECRUITMENT-MORTALITY MODELS TO ASSESS GROWTH RATES AND POPULATION PROJECTIONS[1]

William J. Severud[2], Glenn D. DelGiudice, Joseph K. Bump[2]

ABSTRACT

Estimation of population trends and demographic parameters is important for fundamental ecology and species management, yet these data are often difficult to obtain. The northeastern Minnesota moose (*Alces alces* Linnaeus, 1758) population declined 58% during 2006–2017, yet aerial surveys indicated stability during 2012–2017. In response to the decline, the Minnesota Department of Natural Resources (MNDNR) initiated studies of adult and calf survival. We estimated population growth rate (λ) using adult survival and calf recruitment data from demographic studies and the Recruitment-Mortality (R-M) Equation, and compared these estimates to those calculated using data from aerial surveys. We then projected population dynamics 50 years using each resulting λ, and used a stochastic model to project population dynamics 30 years using data from the MNDNR's studies. Calculations of λ derived from 2012−2017 survey data and the R-M Equation indicated growth (1.02 ± 0.16 [SE] and 1.01 ± 0.04, respectively). However, the stochastic model indicated a decline in the population over the next 30 years (λ = 0.91 ± 0.004). The R-M Equation has utility, but supporting information from demographic collaring studies helps to address management questions. Furthermore, estimates of λ calculated using collaring data were less uncertain and more reflective of current conditions. Long-term monitoring using collars would better inform population performance predictions and demographic responses to environmental variability.

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[1] Abstract from paper in revision: Severud, W. J., G. D. DelGiudice, and J. K. Bump. In revision. Comparing survey and multiple recruitment-mortality models to assess growth rates and population projections.

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SURVEILLANCE AND MANAGEMENT OF CHRONIC WASTING DISEASE IN MINNESOTA

Kelsie LaSharr, Erik Hildebrand, Michelle Carstensen, Margaret Dexter, Patrick Hagen, Chris Jennelle, and Lou Cornicelli

SUMMARY OF FINDINGS

In fall 2018, mandatory surveillance for chronic wasting disease (CWD) in hunter-harvested white-tailed deer (*Odocoileus virginianus*) continued across 3 surveillance areas in the state. In the north-central and central surveillance areas, sampling occurred over the opening weekend of firearm season for a second consecutive year, in response to positive cervid farms discovered in Crow Wing and Meeker counties. We collected 888 and 462 samples from hunter-harvested deer in the north-central and central surveillance areas, respectively; no CWD was detected. In southeast Minnesota, 3,122 samples were collected during opening weekends of the firearms seasons in deer permit areas (DPAs) outside the CWD Management Zone (DPA 603); 3 new CWD-positive cases were detected. This marked the first time CWD was detected outside of the CWD Management Zone, which was established in 2016. In DPA 603, we tested 1,250 hunter-harvested deer and detected 9 positive cases. Additionally, two deer that were found dead by hunters also had CWD. Disease prevalence in DPA 603 had doubled from the previous fall, from 0.46% to 0.84%. In response to both the increased in CWD prevalence and spread into new areas, the Minnesota Department of Natural Resources (MNDNR) implemented additional management actions post-season to curb the spread of disease, including special late hunts, landowner shooting permits (LSP), and agency culling. Late hunts accounted for another 1,004 deer harvested and 4 new cases of CWD; 3 in DPA 603 and 1 in DPA 346. Shooting permits were mailed to 3,559 landowners in Fillmore county; however, only 245 permits were utilized to harvest a total of 409 deer. Shooting permits were also mailed to 235 landowners in Winona and Houston counties, resulting in only 33 additional deer taken. Agency culling removed 493 deer in DPA 603 (12 were CWD-positive) and 47 in DPA 346 (2 were CWD-positive). Thus, post-season efforts in the southeast resulted in a total of 1,986 additional samples with 19 new positives. In February 2019, an adult doe that was found dead less than a half mile from a CWD-infected cervid farm in Crow Wing county was confirmed with the disease, marking the first occurrence of CWD in a wild deer in northern Minnesota. Through a combination of landowner shooting permits, agency culling, and opportunistic sampling, 115 deer were tested from February-April in the area immediately surrounding the infected farm and no CWD was detected. To date, 52 wild deer have been confirmed CWD positive in Minnesota since surveillance efforts began in 2002.

INTRODUCTION

Chronic Wasting Disease (CWD) belongs to a family of infectious diseases known as transmissible spongiform encephalopathies, or prion diseases. Members of the cervid family are susceptible and infection always results in death, with no vaccine or treatment available (Williams 2005). While a link between CWD and neurological illnesses in humans has not been detected (MaWhinney et al. 2006, Sandberg et al. 2010), the Center for Disease Control recommends testing deer or elk intended for human consumption and abstaining from eating known CWD positive meat (cdc.gov/prions/cwd/prevention.html).

Since 2002, Minnesota Department of Natural Resources (MNDNR) has tested over 71,000 wild white-tailed deer (*Odocoileus virginianus*), elk (*Cervus canadensis*) and moose (*Alces alces*) across the state. From 2002-2004, Minnesota completed a statewide surveillance effort, sampled 28,000 deer, and detected no CWD cases. In 2005, MNDNR adopted risk-based surveillance for 1) any suspect deer displaying neurological symptoms consistent with CWD (opportunistic deer), 2) new infections found near neighboring states' borders, and 3) surveillance of hunter-harvested deer as a response to CWD-positive cervid farms. Through risk-based surveillance, 43,000 wild deer were sampled from 2005 to present. To continue mitigating disease spread, in 2016 MNDNR expanded the carcass importation ban to disallow the movement of whole cervid carcasses into Minnesota from anywhere outside its borders.

Using risk-based surveillance, the first case of CWD in a wild deer occurred in 2010 while conducting hunter harvested surveillance near an infected farmed elk facility in Pine Island. From 2011–2013, MNDNR implemented the 2011 CWD Response Plan (MNDNR 2011) to manage the discovery of the disease. After sampling 4,000 deer between 2011 to 2013, no new cases of the disease were found and the disease management zone boundary was dissolved (Hildebrand et al. 2013). For the next 6 years, no further CWD was found in Minnesota's wild herd. In 2016, MNDNR conducted risked-based surveillance using hunter-harvested deer in the southeast as a response to CWD found across state borders in Iowa and Wisconsin; 3 positive wild deer were found near Preston, MN. Over the winter, MNDNR created a disease management zone, called Deer Permit Area (DPA) 603, established carcass movement restrictions, implemented a recreational feeding ban, and liberalized hunting opportunities. In addition, a special late hunt, landowner shooting permits, and targeted agency culling were conducted and yielded 8 additional CWD-positive wild deer. In fall 2017, 6 more CWD-positive deer were detected within DPA 603 during fall harvest (Hildebrand et al. 2018).

The Minnesota Board of Animal Health (BAH) currently manages 371 captive cervid farms that contain approximately 10,000 animals, primarily white-tailed deer and elk (L. Glaser, BAH Assistant Director, personal communication, 27 June 2018). In Minnesota, farmed cervids are classified as livestock and are subject to certain regulations including mandatory CWD testing of all deceased adult animals (https://www.bah.state.mn.us/deer-elk/#chronic-wasting-disease). Since 2002, CWD has been detected on 8 captive cervid farms. The most recent detections occurred in Crow Wing county in 2016, and Meeker and Winona counties in 2017 (the Meeker county farm traced-out from the Crow Wing county farm). As part of the CWD Response Plan, MNDNR began conducting precautionary surveillance around the Crow Wing and Meeker county farms in 2017, in addition to implementing a ban on recreational feeding in all adjacent counties. Precautionary surveillance of wild deer began in fall 2018 for the area surrounding the CWD-infected Winona county farm.

The cervid farms in Meeker and Winona counties were depopulated after discovering the disease, but the Crow Wing county farm elected to remain in business. Additional positive cases were found on that farm in 2018, and the United States Department of Agriculture (USDA) indemnified and depopulated the remaining animals in April 2019; 9 additional deer tested positive for CWD. In February 2019 an emaciated wild deer was discovered dead less than a half-mile from the CWD-infected farm and confirmed positive for CWD. A necropsy revealed this deer died from CWD and MNDNR immediately enacted its CWD Response Plan to try to contain the disease.

METHODS

Fall Hunter-Harvested Surveillance

MNDNR staff and students from 5 universities conducted mandatory sampling of hunter-harvested deer in fall 2018 to monitor changes in CWD prevalence and spread. Staff and students collected medial retropharyngeal lymph nodes for CWD testing and additional samples (e.g., muscle sample, front incisor) were collected in DPA 603, where there was a greater chance for detecting an individual with the disease. Hunter contact information, harvest location, and age/sex of the deer were also recorded. Lymph node samples were sent to Colorado State University (CSU) for testing using enzyme-linked immunosorbent assay (ELISA), and all suspect cases were confirmed with immunohistochemistry (IHC). Test turnaround times were typically 3-4 business days for samples taken within the disease management zone.

We also worked with 27 taxidermists across the 3 surveillance areas to collect samples from animals that had a higher probability of incubating CWD: older, mature bucks. MNDNR also developed tracking forms to allow hunters who harvested trophy bucks to have their deer tested at a later time. In addition, the dual deer feeding and attractant use ban grew to 6 counties in the southeast and remained at 11 counties for north-central and central (attractants excluded) surveillance areas.

Southeast

In southeast Minnesota, the CWD surveillance zone consisted of 10 DPAs, including 341, 342, 343, 344, 345, 346, 347, 348, 349, and 255 (Figure 1). Sampling occurred on opening weekends of both A and B firearms seasons (3-4 and 17-18 November 2018) at 23 sampling stations. Due to increased concerns from hunters and landowners around the CWD-positive Winona county cervid farm, self-service sampling stations were placed in DPA 346 during the remaining firearms season (outside of mandatory testing weekends) to allow hunters to submit voluntary samples.

In DPA 603, liberalized hunting opportunities in the fall were available through elimination of antler point restrictions and providing unlimited disease management tags for antlerless deer. Mandatory testing was required for adult deer throughout all hunting seasons. Five self-service sampling stations were available during archery and muzzleloader seasons and 3 sampling stations were continually staffed during the two 9-day firearm seasons. Carcass movement restrictions remained in place for adult deer, while fawns could leave the zone after age was confirmed by staff during firearm season. MNDNR and Bluffland Whitetails Association provided a dumpster, tripod, and shed at the Preston Forestry Office to provide hunters a place to quarter their deer and comply with carcass movement restrictions. A self-service refrigerated semi-trailer was provided by MNDNR for hunters to store carcasses while awaiting test results during the firearm season.

Central and North Central

For fall 2018, MNDNR reduced the size of the surveillance zones from fall 2017 to approximately a 15-mile radius surrounding each CWD-positive cervid farm in Crow Wing and Meeker counties. Surveillance occurred over opening weekend of firearm season (3-4 November 2018). For the north-central surveillance area, the 2 DPAs included 242 and 247 and utilized 5 sampling stations (Figure 2). For the central surveillance area, the surveillance zone had 4 sampling stations and included portions of DPAs 277 and 283 east of State Highway 4, DPA 219 south of State Highway 55, and DPA 285 north of State Highway 7 (Figure 2).

Winter CWD Surveillance and Management Efforts

Due to increased incidence of CWD during fall hunter-harvest surveillance efforts, MNDNR implemented its 2011 CWD Response Plan and created more opportunities to harvest deer post-season, including special late-season hunts, landowner shooting permits (LSP), and agency culling. To address these opportunities with the community, public meetings were held in Chatfield on 18 December 2019, Winona on 15 January 2019, and Brainerd on 4 March 2019.

Special hunts were designed to increase harvest through liberalized regulations, and included options such as unlimited disease management tags ($2.50) for either-sex deer, elimination of antler point restrictions, permission to cross-tag bucks, no bag limits, and the ability to use any unfilled deer license from earlier in the 2018 deer seasons. The first special hunt occurred over two weekends (21-23 & 28-30 December 2018) in an extended hunt boundary surrounding DPA 603 and consisted of DPAs 603, 347, 348, and portions of 343 and 345 south of Interstate 90. A second special hunt occurred in DPA 346 over two weekends (25-27 January & 1-3 February 2019).

Landowner shooting permits allowed a landowner to apply for a permit and designate shooters to take an unlimited number of deer off their property. The first LSP phase occurred within the extended hunt boundary which included DPAs 603, 347, 348, and parts of 343 and 345 (1-3 January 2019). The second LSP phase occurred within two miles of the first positive deer found in DPA 346, hereafter called Looney Valley (12-20 Jan 2019). The third LSP phase took place within two miles of the positive deer found during the special hunt which was adjacent to the CWD-infected cervid farm in DPA 346, hereafter called Cedar Valley (23 February to 10 March 2019). Following the discovery of the CWD-positive wild deer in Crow Wing, a final LSP phase permitted landowners within two miles of the positive deer and adjacent CWD-infected cervid farm to take additional deer from their properties (2-24 March 2019).

MNDNR contracted with USDA-Wildlife Services (WS) to conduct agency culling directed at areas with known CWD-positive deer near Preston, Winona, and Brainerd, MN (22 January to 29 March 2019). Priority areas were determined as sections with a high number of positives, positive female deer (considered to be disease anchors), or areas with high deer densities in close proximity to known positives. In the southeast, agency culling efforts were expected to last for at least 2 months with several hundred deer culled. Carcass movement restrictions meant few options were available for meat processing within DPA 603. MNDNR staff collected samples and processed agency culled deer via gutless-quartering. Deer were skinned and chest cavities, with the viscera intact, were disposed of into a gasket-sealed, 20-yard dumpster and brought to a lined landfill. Deer quarters were stored in food-grade, wax lined boxes labeled with sample identification, and held in the refrigerated semi-trailer until test results were returned. MNDNR collaborated with a local sportsman's group, Bluffland Whitetails Association, to pioneer Share the Harvest Program – a venison donation program that distributed meat to the public. Additional tissue samples (submandibular lymph nodes, parotid lymph nodes, tonsils, feces, blood, and muscle) were taken in partnership with the University of Minnesota to aid in the development of a rapid diagnostic test procedure to identify CWD prions. Due to the late-winter discovery of the CWD-positive wild deer in Crow Wing County, USDA-WS efforts in that area were only anticipated to last for a couple weeks. Eviscerated carcasses were brought to a local meat processor in Emily, MN for processing and venison was distributed to special interest groups and the public via the Share the Harvest Program in collaboration with Turn-in-Poachers.

RESULTS

From 1 July 2018 – 30 June 2019, 8,040 deer were tested and 34 new positive cases were discovered (Table 1). During fall 2018, MNDNR tested a total of 5,722 total deer, including 378 samples from participating taxidermists. In the southeast, 3,122 deer were tested outside of DPA 603 and 3 new CWD detections occurred for the first time outside the CWD Management Zone; hunter compliance was 86% for the first weekend and 88% for the second weekend, a marked improvement from the previous year. In DPA 603, 1,250 deer were tested and 9 positive cases were discovered (Figure 1). In the north-central surveillance area, 888 deer were tested with no CWD detections; hunter compliance rate was 92% (Figure 2). In the central surveillance area, 462 deer were tested with no CWD detections; hunter compliance rate was incalculable, as the surveillance area included portions of 4 separate DPAs (Figure 2). Throughout the year and especially in areas of concern, routine monitoring of opportunistic deer displaying CWD symptoms was conducted. In DPA 603, 25 opportunistic deer were tested in the fall and 2 deer found dead by landowners tested positive for CWD. The apparent prevalence of CWD within DPA 603 using hunter-harvested and special late hunt samples is 0.84%, up from 0.46% in 2017.

Combined across the southeast, MNDNR sampled an additional 1,986 deer through special hunts, landowner shooting permits, and targeted agency culling. These efforts led to the removal of another 18 CWD-positive deer off the landscape (Figure 3). During the first special hunts, 644 deer were tested and 3 new positives were discovered. For the special hunts in DPA 346, 360 deer were tested and one new positive was discovered 2.5 miles from the Winona county CWD-positive cervid farm. During the LSP phase in the extended boundary around DPA 603, 3,559 permits were mailed or written; of those, 245 permits were utilized and 409 deer were sampled with no new disease detections. Note, during this LSP timeframe, a deer was found dead within DPA 603 and tested positive. During the second phase of LSPs in Looney Valley, 80 permits were issued and 18 deer were sampled with no new disease detections. For the third phase of LSPs in Cedar Valley, 155 permits were issued and 15 deer were sampled with no new disease detections. In the Preston area, 49 landowners granted USDA-WS access to their property (about 9,600 acres of private land and 7,000 acres of public land). In Winona county, 10 landowners allowed USDA-WS to access their property to remove deer (about 1,500 acres of private property). Targeted agency culling around Preston resulted in 493 culled deer with 12 new CWD-positives while Winona county resulted in 47 culled deer and 2 new positive cases.

In the north central area, MNDNR sampled 80 additional deer through LSP and targeted agency culling phases; no additional CWD positives were detected (Figure 4). During LSP, MNDNR issued permits for 139 landowners and 14 deer were sampled with no new positives. In Crow Wing county, 15 landowners allowed USDA-WS access to their property, (about 900 acres of private land); 66 deer were culled with no new positives. During winter 2019, area staff tested 41 opportunistic deer in the Crow Wing county area, which included the CWD-positive found dead deer that initiated the winter surveillance efforts.

The first year for the Share the Harvest Program was a great success. During the 2018-2019 season, 705 people signed up to receive venison, 553 deer were distributed from all efforts, and about 280 individuals received deer.

Between 1 July 2018 and 30 June 2019, MNDNR collected 252 opportunistic samples statewide across 53 different DPAs. Of those, 92 were found dead, 77 were vehicle-killed, 46 were reported sick, and the remainder fell into miscellaneous categories. Four found dead deer were positive: two in DPA 603 during fall 2018, and one each in DPAs 603 and 247 in January 2019.

DISCUSSION

For 2018, the statewide carcass importation ban remained in place to prevent movement of potentially infected carcass remains into Minnesota and further spread of the disease. Recreational deer feeding and attractant bans were expanded in some areas of the state based on risk factors. Increased communication efforts included new quartering and caping videos for social media posting, an increased outreach effort at the Minnesota State Fair, creation of 2 informative pamphlets about CWD, an informational website overhaul, and a new widget on the website that allowed every hunter who submitted a sample to check their results online. In addition, mandatory testing requirements and sampling station locations were printed in the 2018 Minnesota Hunting & Trapping Regulations booklet and posted online.

Total expenditures for 2018 and 2019 CWD surveillance efforts came to $1,524,906. During fall 2018, CWD surveillance occurred at 35 sampling stations in 3 areas of the state. In total, $482,179 was spent planning, collecting, and testing 5,722 samples during the fall (Aug. 1-Nov. 30, 2018), roughly $84.27/sample. However, all winter management efforts, including special late hunts, landowner shooting permits, and USDA-WS agency culling, resulted in 2,318 samples at a cost of $1,003,135, or $432.76/sample. Between fall and winter surveillance, wildlife staff worked 259 shifts and students filled 180 shifts.

An aerial survey was conducted from 13-15 February 2019 in DPA 603 to estimate deer density. Fixed-wing aerial surveys were also conducted in Winona and Crow Wing counties following the discovery of positives in those areas in winter 2019.

Future Surveillance Plans

MNDNR drafted an updated CWD Response Plan (MNDNR 2019), as the original plan had not laid out decision metrics to determine if CWD became endemic in an area of the state. The new plan outlined management actions for 3 stages of infection: 1) initial detection, 2) persistent infection, and 3) endemic disease. The plan lays out goals and options for handling the disease at each stage of infection as well as transition points between stages. MNDNR utilized public engagement with legislative, tribal, agency, and stakeholder group leaders as well as public meetings to explain the plan and collect input.

Following the discovery of CWD outside of DPA 603, new disease management zone boundaries were drawn in the southeast and north central. Using information collected from the Southeast Deer Movement study (Jennelle et al., 2018), new disease management boundaries were drawn 15-miles around each positive wild deer. In the southeast, CWD positive deer across the border in Vernon county (WI) and apparent spread within Minnesota meant portions of 8 different DPAs were affected by the 15-mile boundaries. DPAs affected by that buffer were converted to a 600-series delimiter. DPA 603 was dissolved and the following DPAs were adopted: 643, 645, 646, 647, 648, 649, and 655. Within the disease management zone, mandatory testing is required for adult deer and carcass movement restrictions affects all deer, including fawns for the first time. Additionally, a novel CWD Control Zone was established to surround the disease management zone; DPAs 255, 343, and 344 will have carcass movement restrictions on all deer, including fawns, as well as mandatory testing for all adult deer harvested over opening weekends of firearms A and B seasons. This control zone allows carcasses to move into management zone, but whole carcasses cannot leave the management zone without “not detected” test results (Figure 5).

In the north central area, MNDNR created a new disease management zone, DPA 604, to enforce mandatory testing of all adult deer as well as carcass movement restrictions for all deer, including fawns (Figure 5). DPA 604 was established using a 15-mile buffer around the positive cervid farm and found dead wild deer with boundary lines imposed by enforceable roads.

In fall 2019, self-service sampling stations will be available in the disease control and management zones during archery and muzzleloader season and the stations will be staffed during the firearms seasons. Mandatory surveillance will occur in the central surveillance area over opening weekend for at least one more year and that area will be dissolved if no CWD positive deer are found. The recreational deer feeding and attract ban will be expanded to include 24 counties.

During the 2019 Minnesota legislative session, $1.87 million was awarded from the state's general tax fund to help monitor and manage CWD. In addition, a dumpster program was initiated by state legislators to mandate a partnership between MNDNR, Department of Health, Pollution Control Agency, waste haulers/disposal sites, taxidermists, and meat processors to create guidelines for proper carcass disposal. Within both disease management and control zones, dumpsters and quartering stations will be available to hunters to process their deer and abide by carcass movement restrictions.

ACKNOWLEDGMENTS

We would like to thank all MNDNR Wildlife and Enforcement staff who assisted in the surveillance this past year. We especially thank the Ready Reserve wildlife staff who assisted with the planning process, area staff who helped manage the winter efforts particularly Robby Rathburn and Ray Munson. A special thanks goes to Todd Froberg who went above and beyond to gain buy-in from special interest groups and individuals in the southeast MN community. Bluffland Whitetails Association and Turn-in-Poachers helped make a successful first year of the Share the Harvest Program. We thank all of the sampling station locations who allowed staff to provide a testing service for hunters. Special thanks goes to the Preston Forestry office for allowing the bulk of our southeast efforts to occur within their heated facility. We thank our students who provided enthusiasm and help. MNIT staff provided all of our GIS, mapping, and website needs: Bruce Anderson, Steve Benson, Rob David, Julie Hines, Chris Scharenbroich, and Bob Wright. Our communication efforts were assisted by Denny Behr, Deb Beyer, Kristi Coughlon, Nyssa Gesh, Harland Hiemstra, Dan Ruiter, David Schueller, Shamiah Specht, Pete Takash, and Cheri Zepplin. The DNR Info Center helped with their continued triage of questions and concerns from the public. Finally, we thank USDA-WS for their long hours and success from their targeted culling efforts – especially John Hart and Tim White for taking leadership roles during these phases.

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Table 1. MN wild white-tailed deer samples submitted for testing and resulting positive CWD cases detected by operational phase between 1 July 2018 and 30 June 2019.

Dates	Operational phase	Number of landowners who received a Landowner Shooting Permit	Number of deer sampled	Resulting positives
Nov. 3-4, Nov. 17-18	Fall - Southeast		3,122	3
Nov. 3-4	Fall - North central		888	0
Nov. 3-4	Fall - Central		462	0
Sept. 15 – Dec. 31	Fall – DPA 603		1,250	9
Dec. 21-23, Dec. 28-30	Special late hunts – extended boundary DPA 603		644	3
Jan. 1 -13	Landowner Shooting Phase - extended boundary DPA 603	3,559	409	0
Jan. 12-20	Houston county LSP – Looney Valley	80	18	0
Jan. 25-27, Feb. 1-3	346 Special late hunt		360	1
Feb. 23 - Mar. 10	Winona county LSP - Cedar Valley	155	15	0
Jan. 22 - Mar. 29	USDA-WS Preston		493	12
Mar. 7-29	USDA-WS Winona		47	2
Mar. 2 -24	Crow Wing LSP	139	14	0
Mar. 17-29	USDA-WS Crow Wing		66	0
July 1-June 30	Opportunistic statewide		252	4
Totals			8,040	34

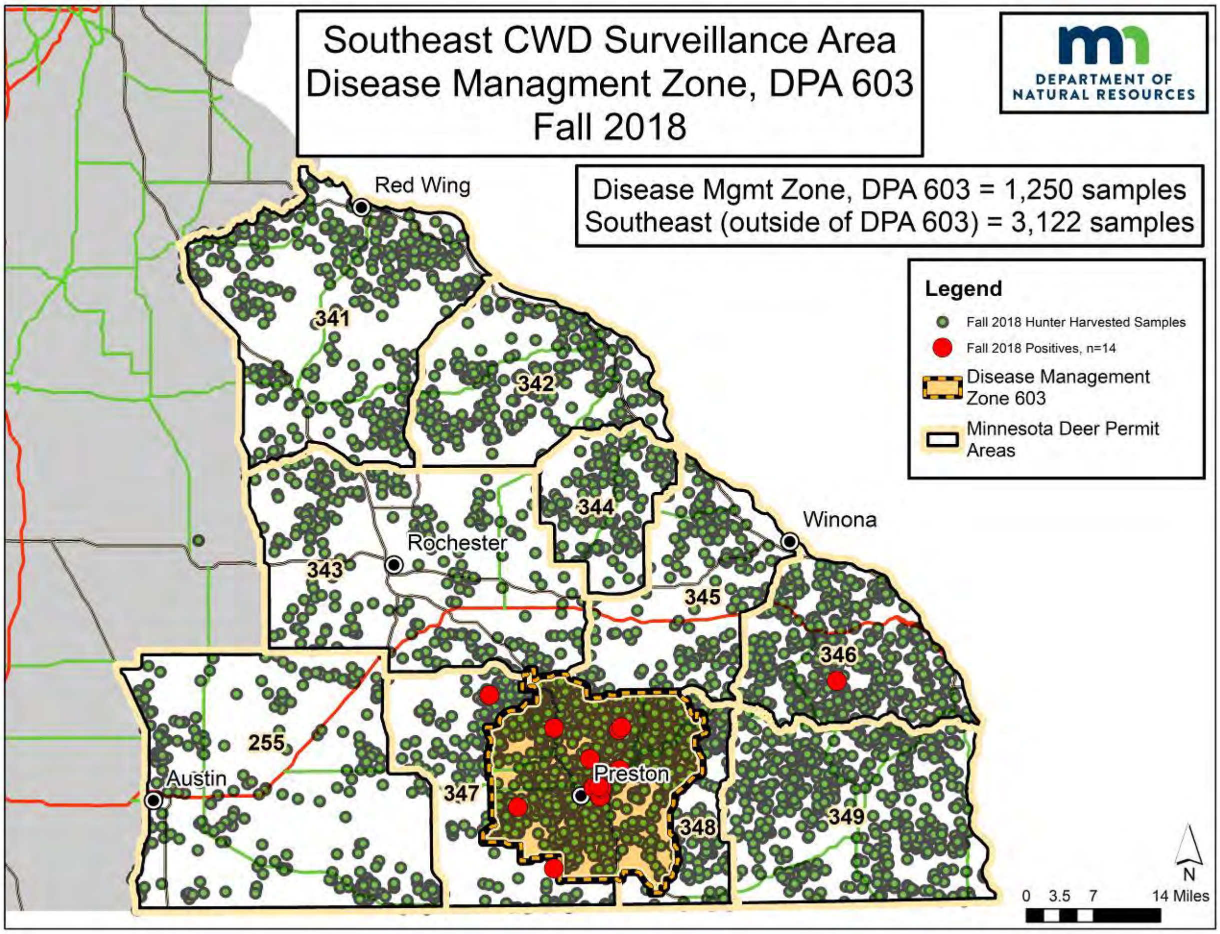

Figure 1. Fall 2018 sampling results of hunter harvested white-tailed deer for the southeast and DPA 603. DPAs for the southeast surveillance area included 341, 342, 343, 344, 345, 346, 347, 348, 349, and 255. Outside of DPA 603, 3,122 samples were collected and 3 new positives were found; 2 in DPA 347 and 1 in DPA 346. Within DPA 603, 1,250 samples were collected and 9 positive hunter harvested deer and 2 found dead deer were found.

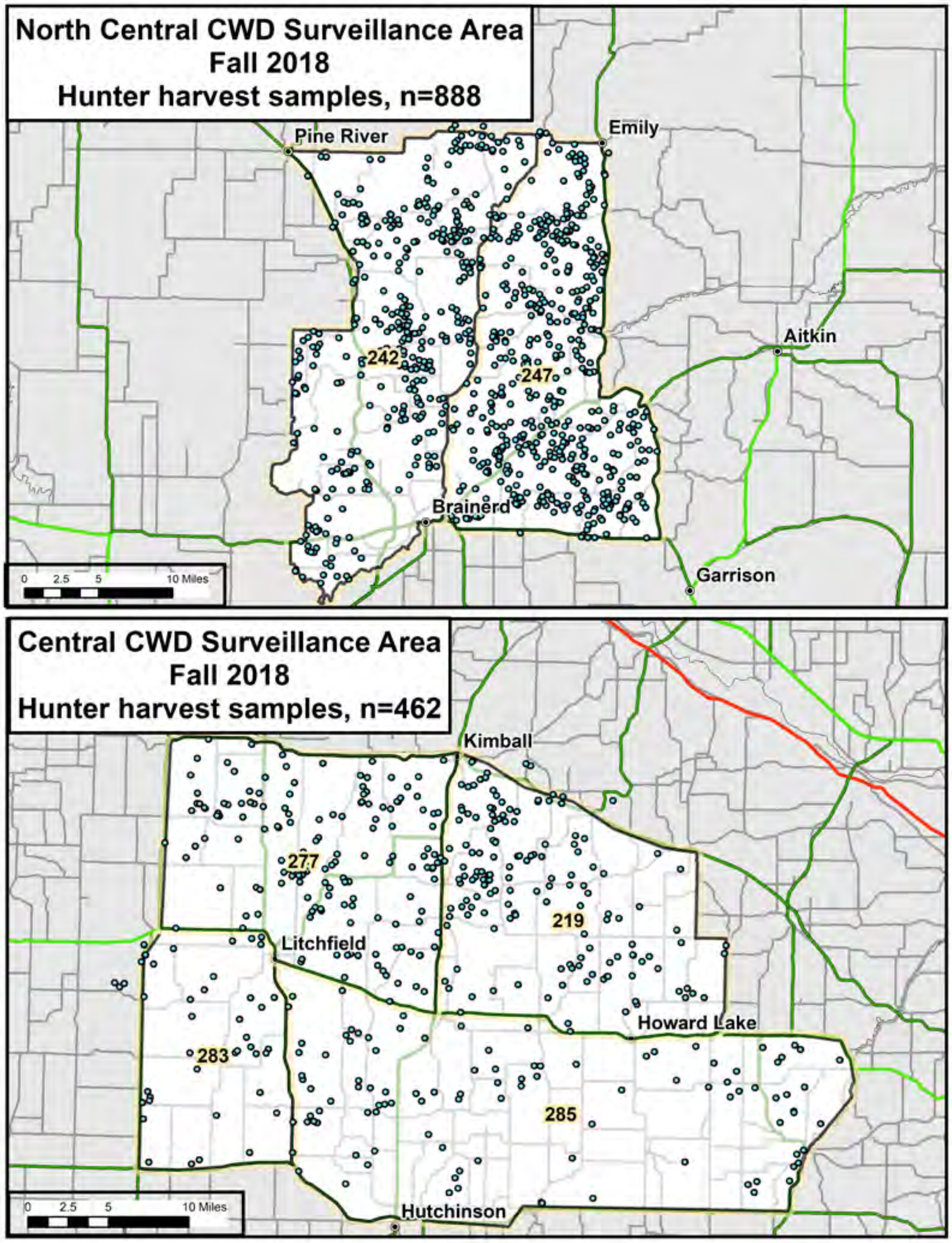

Figure 2. Fall sampling results of hunter harvested white-tailed deer from north central and central surveillance on 3-4 November 2018. North central surveillance included DPAs 242 and 247; 888 samples were taken and no new positives were detected. Central surveillance area included portions of DPAs 277 and 283 east of State Highway 4, DPA 219 south of State Highway 55, and DPA 285 north of State Highway 7; 462 samples were collected and no new positives were found.

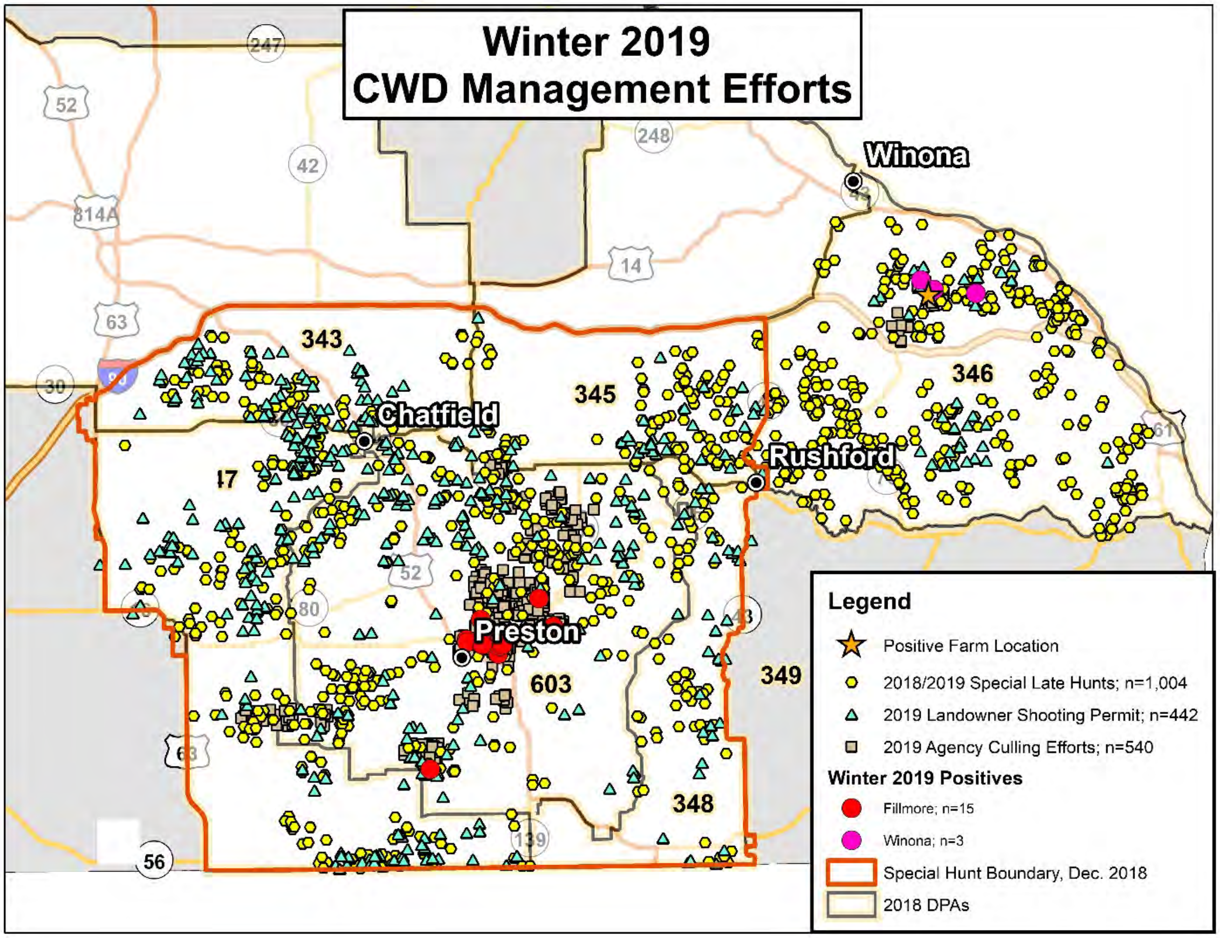

Figure 3. During winter 2019, MNDNR sampled an additional 1,986 white-tailed deer through special hunts, landowner shooting permits, and targeted agency culling. These efforts led to the removal of another 18 CWD-positive deer off the landscape. During the special hunts, 1,004 deer were tested and 4 new positives were discovered. During the three LSP phases in this area, 442 deer were harvested and 1 found dead deer tested positive. Agency culling removed 493 deer in DPA 603 with 12 positives and 47 deer in DPA 346 with 2 positives.

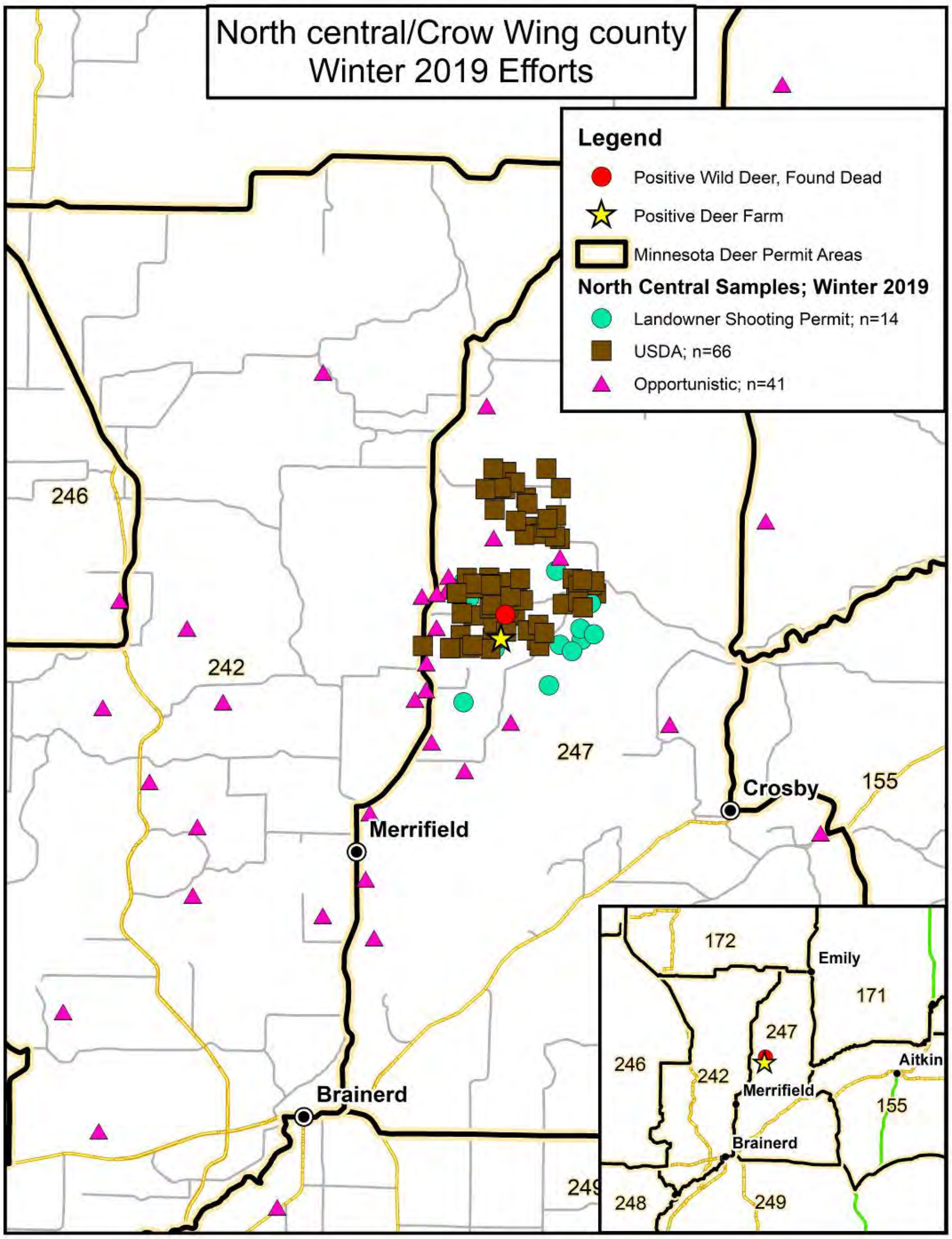

Figure 4. Following the discovery of the positive deer found dead in Crow Wing county, landowner shooting permits were issued to 138 landowners and 14 deer were tested. Immediately following, USDA-WS culled 66 deer from private and public properties surrounding the farm and positive deer. Throughout all of winter 2019, area wildlife staff tested 41 opportunistic deer in the surrounding area. No additional positives were discovered from these efforts.

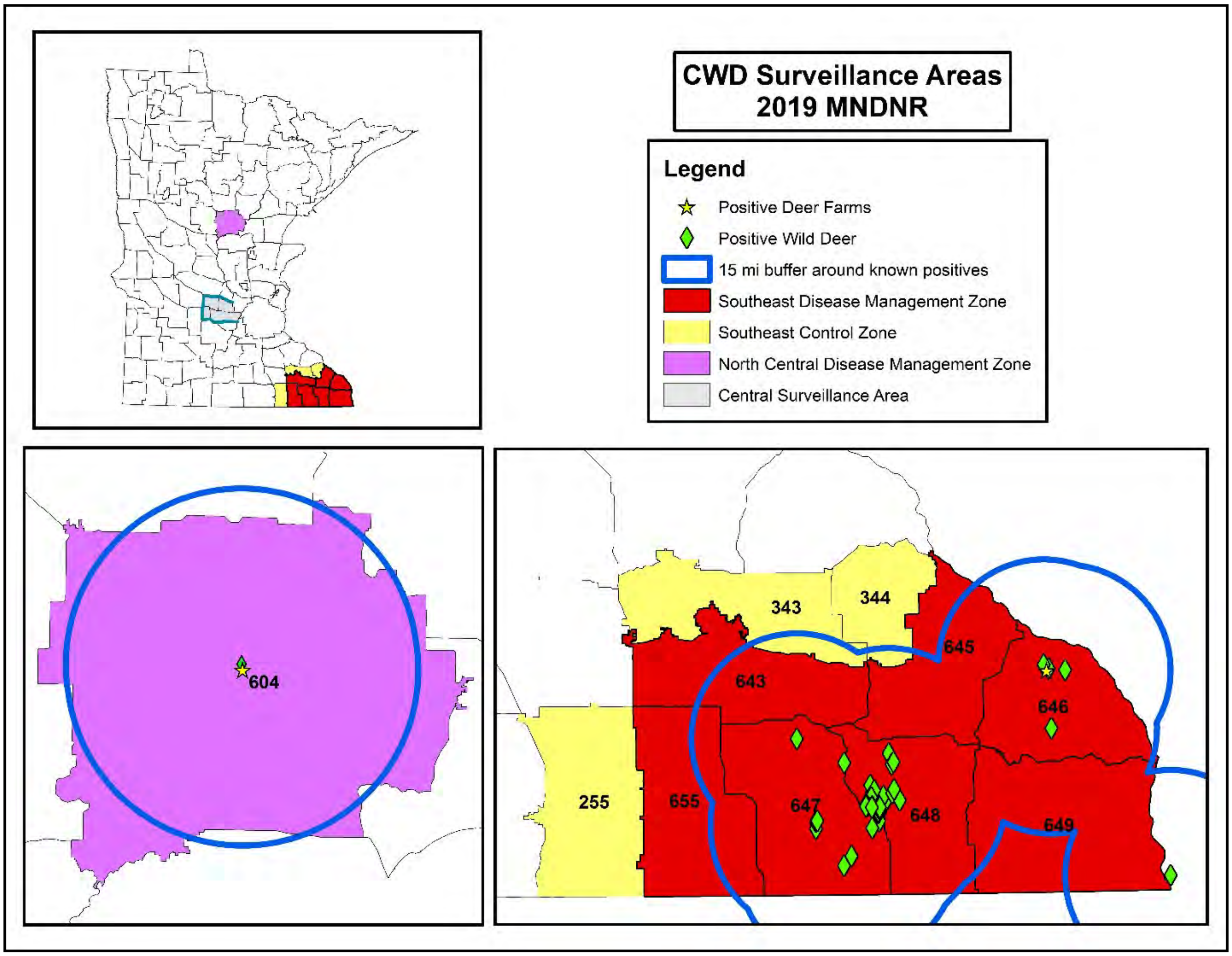

Figure 5. For fall 2019, new disease zone boundaries were drawn 15-miles around each positive wild deer. The southeast management zone consists of DPAs 643, 645, 646, 647, 648, 649, and 655. Additionally, a CWD Control Zone was established for the DPAs surrounding the new 600-series; DPAs 255, 343, and 344. This control zone allows carcasses to move into the 600-series, but whole carcasses cannot leave the 600-series without “not detected” test results. In the north central area MNDNR created a new disease management zone, DPA 604.

INVESTIGATION OF MOVEMENT DYNAMICS OF WILD DEER IN SOUTHEASTERN MINNESOTA TO UNDERSTAND POTENTIAL SPREAD OF CHRONIC WASTING DISEASE

Chris Jennelle, Kelsie LaSharr, Michelle Carstensen, Lou Cornicelli, Margaret Dexter, Todd Froberg, Patrick Hagen, Erik Hildebrand, Tyler Obermoller, and Ryan Tebo

SUMMARY OF FINDINGS

Now in its second year, the Minnesota Department of Natural Resources (MNDNR) is continuing a study to investigate the movement dynamics of wild white-tailed deer (*Odocoileus virginianus*) in southeastern Minnesota. The detection of chronic wasting disease (CWD) in fall 2016 in Fillmore County motivated this project to 1) understand potential pathways of CWD spread on the landscape by movement of wild deer, and 2) increase our likelihood of managing the outbreak in this and other areas of Minnesota. In February 2019, we captured and fitted GPS collars to 64 white-tailed deer (39 female fawns and 25 male fawns) in our study area centered around the CWD Management Zone. A total of 173 deer have been collared since the study began in March 2018, but as of 3 August 2019, only 66 animals remain available for tracking. There have been 45 known mortalities due to hunter-harvest (n=14), poor health (n=6), vehicle collision (n=5), agency culling (n=4), unknown cause (n=4), and capture-related issues (n=12). A significant number of collars from the 2018 release cohort failed (n=80) due to either hardware malfunction or collar expansion failure; however, only one collar from the 2019 release cohort has failed. We considered movements during the fall, excursions, or temporary movements outside of an established adult home range. During fall 2018, females (n=6/11) had a slightly higher rate of excursions than males (n=6/26) at 55% versus 23%, respectively. The median excursion distance traveled by females and males was 4 km and 7 km, respectively. We estimated average natal (1 March 2019 -14 April 2019) home range size for fawns as 1.84 km^2 and 2.65 km^2 for female and male deer, respectively. Preliminary assessment of spring dispersal (15 April 2019 - 15 July 2019) suggests that dispersal probability of females (44%, n=34) was nearly equal to males (45%, n=22). Median dispersal distance travelled was also nearly equal between sexes at 10 km (n=15) and 11 km (n=10) for females and males, respectively. These dispersal rates and distances traveled are comparable with those estimated from 2018. These valuable data are informative for understanding potential CWD spread in wild deer in southeastern Minnesota and enable MNDNR to adjust surveillance and management activities more effectively to counter CWD spread in Minnesota.

INTRODUCTION

Chronic wasting disease (CWD) is a fatal infectious disease first characterized in the late 1960s in Colorado that affects elk *(Cervus canadensis)*, mule deer *(O. hemionus)*, white-tailed deer, reindeer (*Rangifer tarandus*), and moose *(Alces alces)*. It has been detected in wild and captive cervids in 26 states and 3 Canadian provinces in North America, as well as Finland, Norway, Sweden, and South Korea. Recent work has demonstrated that CWD can cause population declines in white-tailed deer and mule deer in the western US, particularly when high prevalence levels occur in a population (Edmunds et al .2016, DeVivo et al. 2017). In the upper Midwestern US, an ongoing study of CWD in white-tailed deer in Wisconsin has shown that CWD-infected deer die at 3x the rate of uninfected deer (Wisconsin DNR 2018). In the same

CWD system, research has shown that deer regularly die from CWD in the wild, although they typically go undetected by people (Samuel and Storm 2016).

It has been nearly 3 years since CWD was discovered during the 2016 regular hunting season in Fillmore County, Minnesota, and through 3 August 2019, the MNDNR has detected 52 CWD-positive wild deer in Minnesota (including 1 from Olmsted County in 2010). Just in the last year alone, of 8,040 deer tested across the state, 34 new positive cases were detected in Fillmore, Winona, Houston, and Crow Wing Counties. With the exception of the recent Crow Wing and Winona County cases, which are likely sourced to CWD-positive captive cervid farms, the distribution of new cases suggests spatial spread of disease radiating from the core area in DPA 603 (Figure 1). While it is not clear how CWD was introduced into Fillmore County, potential routes of introduction include movement of infectious deer from neighboring states (e.g., Wisconsin, Iowa, or Illinois), contact between wild deer and prior CWD-positive captive cervid facilities, or contamination of the environment with infectious cervid carcass material facilitated by out-of-state hunters disposing of butchering remains on their Minnesota property. This study aims to better understand natural deer movement ecology and in particular detect seasonal corridors of movement that may inform our management of CWD spatial spread in southeastern Minnesota.

As infected and non-infected deer interact and move across the landscape, they transmit infectious prions through direct contact with other deer or indirectly through environmental deposition (Almberg et al. 2011). Limited information exists about deer contact rates and their relationship to transmission rates. The presumed main driver of spatial spread among wild deer are natural movements, which vary as a function of season, sex, age, habitat, underlying deer density, and other variables (Nixon et al. 2007, Long et al. 2008, Long et. al. 2010, Lutz et al. 2015, Peterson et al. 2017). Besides this current effort, there is only one published source of information (Simon 1986) that informs the extent to which deer may move across the landscape and interact with each other in southeastern Minnesota. However, the Wisconsin Department of Natural Resources has an ongoing study in its third year designed to understand the impacts of CWD on survivorship and movements of deer in Wisconsin, which will be very helpful to compare our study results with.

Deer behavior and movements vary by sex, season, and landscape features, along with deer population demographics and social structure. Three types of movement likely facilitate disease spread across the landscape including dispersal events, recurrent seasonal movements, and temporary excursion events. The most substantial long-distance movements involve dispersal from natal to adult ranges, most likely to occur in 1-year-old deer during spring, although males may also disperse in fall. We define dispersal in this context as an asymmetric movement from the natal home range to a distinct and non-overlapping adult range. While dispersal usually occurs once, there are cases of 2-stage dispersals when a deer makes a second asymmetric movement to a second adult home range, but this is rare. Recurrent seasonal movements can include migratory behavior and movements between summer and winter ranges. Excursions (or synonymously forays) are temporary transient movements out of an established home range that typically occur in fall. Because deer densities and movement behavior can be altered by management actions, a better understanding of both deer density and movement activities related to density will enhance our ability to effectively manage disease risk in the Minnesota deer population. The importance of this research is underscored by the unabated spread of CWD both between and within states, and the need to find management solutions to suppress the spread of disease arising from natural deer movements.

METHODS

Study Area

The study area, approximately 7,250 km^2, is centered on DPA 603, also referred to as the CWD management zone, in Fillmore County, Minnesota (Figure 2). The study area limits are flexible and have been established as approximately a 20-mile buffer outside of and including DPA 603. We included extensive areas around DPA 603 to capture and release GPS-collared deer, so that our collared sample is representative of the deer population inside and surrounding the CWD management zone. Given the increase in number of cases detected within DPA 603 in 2017-18, and our expanded harvest efforts in response (both hunter-harvest and agency culling), we chose to limit capture of animals in the second year (2019) to areas outside of DPA 603. This choice increased the likelihood that collared animals would not be pre-maturely harvested during late season hunts and agency culling efforts before significant information regarding their movements could be obtained.

The study area is composed of a matrix of agricultural lands interspersed with deciduous forest upon a landscape of rolling hills and in some cases very steep ridges and valleys. There is considerable heterogeneity in landscape topography and land use, particularly as one moves from east to west. The eastern part of the study area is composed of forested blufflands and steep ridges and moving west and south, the landscape transitions to be flat and dominated by agriculture. More than 90% of the landscape is held in private ownership, and there is significant heterogeneity in deer density due to both habitat heterogeneity and localized refugia (i.e., parcels with viable deer habitat where hunter harvest is not permitted).

Since most of the region is in private ownership, our pre-capture efforts were heavily focused on securing permission to access private property in the study area. We secured permissions to use 115,259 acres, consisting of private (72,398 ac) and public (42,861 ac) properties, for search and capture of white-tailed deer in southeastern Minnesota (Figure 3). This amount reflects an additional 10,000 acres secured for access compared to the first year of the study. We could not have achieved our sampling goals without the enormous outpouring of support from private landowners in the study area (about 224). Public properties included state-owned wildlife management areas, forests, and natural areas. For future deer captures, we hope to add to our permission list and increase available properties for capture efforts. Increasing the size of contiguous blocks of property access increases the probability that we can capture deer there. We focused on securing permission to access properties that are forested (where deer may be flushed) with adjacent open fields (where deer may be captured and a helicopter may safely land).

Sampling Design and Data Collection

Given the breadth of the study area, we divided it into 10 quadrants (Figure 3) from which we established a baseline target goal of capturing 3-4 fawns (≈ 7-9 months old) of each sex per quadrant for 2019. Our goal was to capture and collar 64 deer; 32 male fawns and 32 female fawns.

We contracted with Quicksilver Air Inc. (Peyton, CO) to capture deer by net-gunning from a Robinson R44 Raven 2 helicopter. A highly experienced capture crew of 3 personnel from the company performed all deer handling procedures including deer capture, collar placement, ear tag placement, collection of auxiliary measurements (body temperature, age class, sex, and body condition), and an ear punch for genetic analysis. Helicopter pursuit time of animals did not exceed 3.5 minutes, and the crew was instructed to abort chase of an animal if pursuit time exceeded 5 minutes. Average handling time per animal was approximately the same at 3 minutes.

We programmed GPS collars for males (Iridium TL330 with expandable collar, Lotek Wireless Inc, Newmarket, Canada) and females (Iridium 420, Lotek Wireless Inc, Newmarket, Canada) to collect location coordinates every day at an increased rate during spring dispersal and fall rut periods. The rate of GPS location fixes was programmed to occur once every 85 minutes (approximately once per hour) between 15 April through 15 July and 1 September through 15 December. During all other time periods, collars were scheduled to collect positional data every 3 hours and 45 minutes or approximately 6 locations per day. We chose these periods in part based on seasonal movements recorded from yearling males in Wisconsin. To ensure that location data were collected across the entire 24-hour day distribution, we included a 15 minute offset from an hour (e.g., 1 hr 15 min, 3 hr 45 min) so programmed GPS fixes occurred on a staggered schedule that changed every day.

The collars included timed-release drop-off mechanisms, which after 130 weeks (2.5 years) will cause the collars to detach and can be retrieved and potentially re-furbished. In addition, for male collars only, we included a line-of-sight mechanism that permits the collars to be detached remotely in line of sight to the animal (within 200m). We added this feature on male collars because male necks expand and contract with season, and during the rut when their necks are largest in diameter, there is a risk that collars could be too tight if the expansion mechanism fails. If hunters are in the field and may come across collared deer with suspected tight collar issues and then report them to us, we can make efforts to locate and remotely release these collars.

Due to hardware failures from our first release cohort of 115 GPS collars in March 2018, the manufacturer (Lotek) warrantied 73 failed collars (63%) and provided us with replacements at no cost. As of 3 August 2019 seven additional GPS collars from the 2018 release cohort have gone off the air for unknown reasons (although these were not under warranty). The manufacturer made modifications to on-board software, corrected quality-control issues with production of the collars, and modified the expansion design of male collars (by our direction and input) to improve performance of the equipment.

Data Analysis

We define dispersal as having occurred if an individual displayed a permanent, asymmetric movement from a natal range to a distinct adult range (Kenward et al. 2001, 2002), such that pre-dispersal locations do not overlap post-dispersal locations (Long et al. 2005, Lutz et al. 2015). All recorded spatial locations were vetted before incorporated into any analysis because the accuracy of a location is influenced by the number of satellites available in the sky that communicate with a collar and how a deer is juxtaposed in the landscape (i.e., influence of physical barriers). The vetting process involved omitting any spatial location from further consideration if less than 3 satellites were used to derive a location. These 2-dimensional location coordinates resulted in highly biased altitude above sea level estimates (around 0) and high Dilution of Precision values (> 4) indicative of inaccurate locations (generally on the order of > 500m based on controlled tests). We estimated natal and adult home ranges using minimum convex polygons (MCP) (Mohr 1947). We assumed that we captured fawns on their natal range during the initial capture period in February 2019. We calculated dispersal distance as the straight-line distance between adult home range and natal home range centroids (Kenward et al. 2002). We performed all spatial data analysis and characterization using R software (R Core Team 2017), R package adehabitatHR (Calenge 2006), and ArcMap 10.6 (Environmental Systems Research Institute, Redlands, CA, USA).

We classified a movement as an excursion (or foray event) if it was a temporary movement clearly outside the boundary of a home range, with subsequent return to the respective home range.

We estimated the distance of an excursion as the straight-line distance between the farthest excursion location outside of the home range and the centroid of the home range. We examined seasonal differences in dispersal and excursion distances.

Collars were programmed to transmit a mortality text and email message if inactive for 12 hours. Mortality events were investigated within 48 hours of mortality notification whenever possible. Sometimes a triggered mortality event was the result of a slipped or broken collar, in which case responding staff simply retrieved the collar from the field. In cases of true mortalities, responding staff routinely collected medial retropharyngeal lymph nodes for CWD testing, a muscle sample for potential genetic testing, and a front incisor tooth for age confirmation. Upon inspection of carcasses, staff were instructed to collect additional samples if any tissues or organs appeared abnormal, and these were submitted for additional diagnostic testing at the University of Minnesota Veterinary Diagnostic Laboratory. If an animal died within the first 2 to 3 weeks following capture, every effort was made to retrieve the entire carcass and submit it to the University of Minnesota Veterinary Diagnostic Laboratory to determine cause of death. In these cases, we were particularly interested in determining whether an animal died due to capture myopathy, which results from extensive muscle damage due to extreme exertion, struggle, or stress of capture. Capture myopathy is an unfortunate reality when handling wildlife, and we make every effort to avoid excessive animal handling during capture. Outside of an approximate 3-week window following capture when capture myopathy is most likely to occur, staff performed field investigations to determine likely cause of death. Using all evidence available from a carcass (e.g., broken bones, bite marks, body condition) and the area surrounding a death site (e.g., evidence of struggle), staff assigned probabilities of cause of death including hunter harvest, agency culling, vehicle collision, starvation, health-related, capture-related, predation, or uncertain.

Collared deer were not protected from legal harvest during hunting seasons, and we encouraged hunters to select animals for harvest based on their personal preference regardless of whether the hunter noticed a collar on the deer. Hunters who harvested a collared deer were asked to contact MNDNR and return the collar.

RESULTS AND DISCUSSION

From 18-21 February 2019, we captured and outfitted 64 deer with Iridium GPS collars: 39 female fawns and 25 male fawns (Figure 4). During the capture period, 3 male fawns and 1 female fawn were able to kick off their collars just after initial collar fitting, and we were able to retrieve these collars to redeploy them on other animals. One female fawn accidentally broke its neck upon capture, and we were able to donate the meat from this animal to the Share the Harvest donation program (for details on the program, see https://www.dnr.state.mn.us/cwd/share-harvest.html). One male was able to kick its collar off within about a month of capture. Seven deer (6 females, 1 male) have died since capture (Table 1). Two animals are suspected to have died due to capture myopathy based on examination at the UMN Veterinary Diagnostic Lab, 1 female was killed by vehicle collision on US Hwy 63 just south of Stewartville, MN, and it is unclear what caused the death of the remaining four individuals. There has only been 1 collar failure to date (2%) from the second release cohort of 64 GPS collars, which so far suggests that the modifications made to improve collar performance have been successful. As of 3 August 2019, we have 66 deer actively being monitored including 38 females and 28 males.

We have amassed over 450,000 records of deer location data from 23 March 2018 through 1

September 2019. By September 2018, most of the fawns from the March 2018 release cohort were expected to have established an adult home range. We used data from September through December 2018 to examine fall movements, particularly excursions or temporary movements outside the home range. From the deer available in the study at that time (n=37), we found that 55% of females (n=6/11) and 23% (n=6/26) of males underwent excursions or temporary movements outside of their established adult home range (Table 2). The median distance traveled for females and males was about 4.3 km and 7.3 km, respectively, (Table 2) in the fall. So, although females had a higher likelihood of making excursions from their home range, they tended to travel a shorter distance compared with males.

Prior to the spring dispersal period between April and July 2019, the average winter home range size of deer from the 2019 release cohort were similar at 1.84 km^2 for female fawns and 2.65 km^2 for male fawns (Table 3). These winter home range estimates align with our expectations of deer home range at this time of year, and were similar to 2018 estimates. During the spring dispersal period of 2019 (approximately 15 April through 15 July), female deer had a higher than expected apparent dispersal probability (44%, n=15/34), although it was comparable with males (46%, n=10/22). These proportions were not appreciably different than estimates from 2018. The median dispersal distance travelled was 10.1 km (n=15) and 11.2 km (n=10) for females and males, respectively (Table 3). These estimates align almost exactly with estimates from 2018 when females and males traveled a median distance of 12 km and 12.5 km, respectively. Given our small sample sizes, we choose the median (as opposed to mean) as a measure of central tendency because of the non-normal distribution of distances that deer traveled. Such non-normality causes extreme outliers (which we have) to skew distance distributions, artificially inflating the mean.

From the 2019 release cohort, we found that 2 males and 1 female have apparently dispersed to Iowa, although it is not clear yet if they have established an adult range in that state. We saw similar movements from animals in the 2018 release cohort that appeared to be seasonal movements between winter and summer ranges. The majority of mortalities arose from harvest – either by hunters (n=14) or agency personnel (n=4) (Table 1). Capture related issues (n=12), poor health (n=6), vehicle collision (n=5), and unknown causes (n=4) made up the remaining causes of death (Table 1). The total number of deer mortalities we were able to document are likely an underestimate because of the failure of 73% of our 2018 release cohort collars, which precluded us from determining their fates.

While male dispersal is typically regarded as the primary force driving potential disease spread (CWD) on the landscape (Grear et al 2006, Oyer et al. 2007), evidence suggests that females orphaned at a young age (Etter et al. 1995) or high underlying deer density (Lutz et al. 2015) can drive females to disperse. Given the relatively high rate and extent of female dispersal and high pre-fawn deer densities in the farmland-forest transition zone of our study area at around 22 deer/mi^2 (Norton and Giudice 2017), we hypothesize that this phenomenon may be occurring in southeastern Minnesota. This highly productive landscape favors high deer survival and fecundity, given extensive food resources, winter cover, and relatively mild winters. Additional years of collaring female and male fawns representative of southeastern Minnesota will further inform our understanding of dispersal and movement activities as it relates to potential spread of CWD prions on the Minnesota landscape.

We have provided outreach materials both for landowners that have provided us with permission to use their properties for deer capture and for the general public. We continue to inform participating landowners twice per year with deer movement updates and maps of the collared deer in the study, and provide a summary of study findings and expectations for future work. Similarly, we continue to update a dedicated website to this research project at https://www.dnr.state.mn.us/cwd/deer-movement-study.html. This site provides information about the purposes of the study, periodic updated findings, and information about how readers can assist and contribute to our efforts. We encourage the public to provide us with trail camera photos of collared deer they may encounter, and with their permission, we make these pictures available on our website. There have also been almost two dozen popular press articles covering this study in various media outlets. Overall, we strive to continually improve how we communicate science to the public, and provide transparency in all of the work that we conduct.

Future Capture and GPS-Collaring Efforts

Between January and February 2020, we plan to capture and GPS-collar between 80 and 90 white-tailed deer fawns in the study area to maintain a sample size of about 100 deer for location monitoring at any given time. We will aim to collect equal sample sizes between sex, but this depends in large part on chance as there is no way to verify sex until the capture crew captures and processes a deer in the field. Like previous years, we hope to capture approximately 4-5 deer of each sex in each of 10 quadrants around the study area.

ACKNOWLEDGMENTS

We extend warm thanks to all of the participating landowners in southeastern Minnesota that gave us permission to access and conduct capture operations on their properties. We thank all the MNDNR Wildlife and Enforcement staff, who assisted in contacting landowners and fielding questions about the study to the public including Don Ramsden, Mike Tenney, and Mitch Boyum. Special thanks to Julie Hines and Bob Wright for their great work assisting us with our GIS mapping needs; Chris Scharenbroich and Pete Takash for their efforts with helping us update the webpage dedicated to this project on the DNR website; Rushford Municipal Airport (Airport Manager Mike Thurn) and Fillmore County Airport (Airport Manager Isaac Deters) in Preston, Minnesota for use of their facilities. Without the support of these and many more people behind the scenes, this project would not be possible. We also thank the Legislative-Citizen Commission on Minnesota Resources (LCCMR), the Environment and Natural Resource Trust Fund (ENRTF) - Emerging Issues account (M.L. 2015, Chp. 76, Sec. 2, Subd. 10) that we received grant funding from provided the needed funds to get this project off the ground. Additional funding was provided in part by the Wildlife Restoration (Pittman-Robertson) Program.

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Table 1. As of 3 August 2019 there have been 45 known mortalities from both release cohorts of deer in 2018 (n=109) and 2019 (n=64) in southeastern Minnesota. Given the hardware and electronic failure of 80 collared deer from the 2018 cohort and 1 collared deer from the 2019 cohort, we can only detect if one of these animals died by reports provided by hunters or the public that might have harvested them or come across their carcasses in the field. Thus, the observed mortalities provided below are likely an underestimate of total collared deer deaths.

Cohort-Sex	Capture-related	Hunter-harvest	Agency-culled	Poor condition/ health	Vehicle collision	Unknown
2018-Female	3	3	0	1	1	1
2018-Male	6	11	4	2	3	2
2019-Female	2	0	0	3	1	1
2019-Male	1	0	0	0	0	0
TOTAL	12	14	4	6	5	4

Table 2. Mean proportion (and 95% confidence interval) of available white-tailed deer in southeastern Minnesota collared in March 2018 undergoing excursions during fall (Sept through December 2018) or temporary movements from their adult home range, and the median distance (Distance – km) traveled during excursions. The distance estimates do not account for non-linear pathways traveled, forward and backwards movements along pathways, and only describe straight-line distances.

Cohort	n-total	% Excursions (95% C.I.)	n-Excursion	Distance (min, max)
Females	11	54.5 (24.6, 81.9)	6	4.3 (3.8, 7.0)
Males	26	23.1 (9.8, 44.1)	6	7.3 (2.3, 35.0)
TOTAL	37		12	

Table 3. Mean estimate (and 95% confidence interval) of winter home range (HR - km^2), apparent spring dispersal probability (Pr. Dispersal), and median apparent spring dispersal distance (Distance – km) of white-tailed in southeastern Minnesota collared in March 2019. The distance estimates do not account for non-linear pathways traveled, forward and backwards movements along pathways, and only describe straight-line distances.

Cohort	n-HR	HR (95% C.I.)	n-Dispersal	Pr. Dispersal (95% C.I.)	Distance (min, max)
Female fawns	34	1.84 (1.46, 2.19)	34	0.44 (0.28, 0.62)	10.1 (4.8, 47.1)
Male fawns	22	2.65 (1.69, 3.45)	22	0.46 (0.25, 0.67)	11.2 (4.0, 86.9)
TOTAL	56		56		

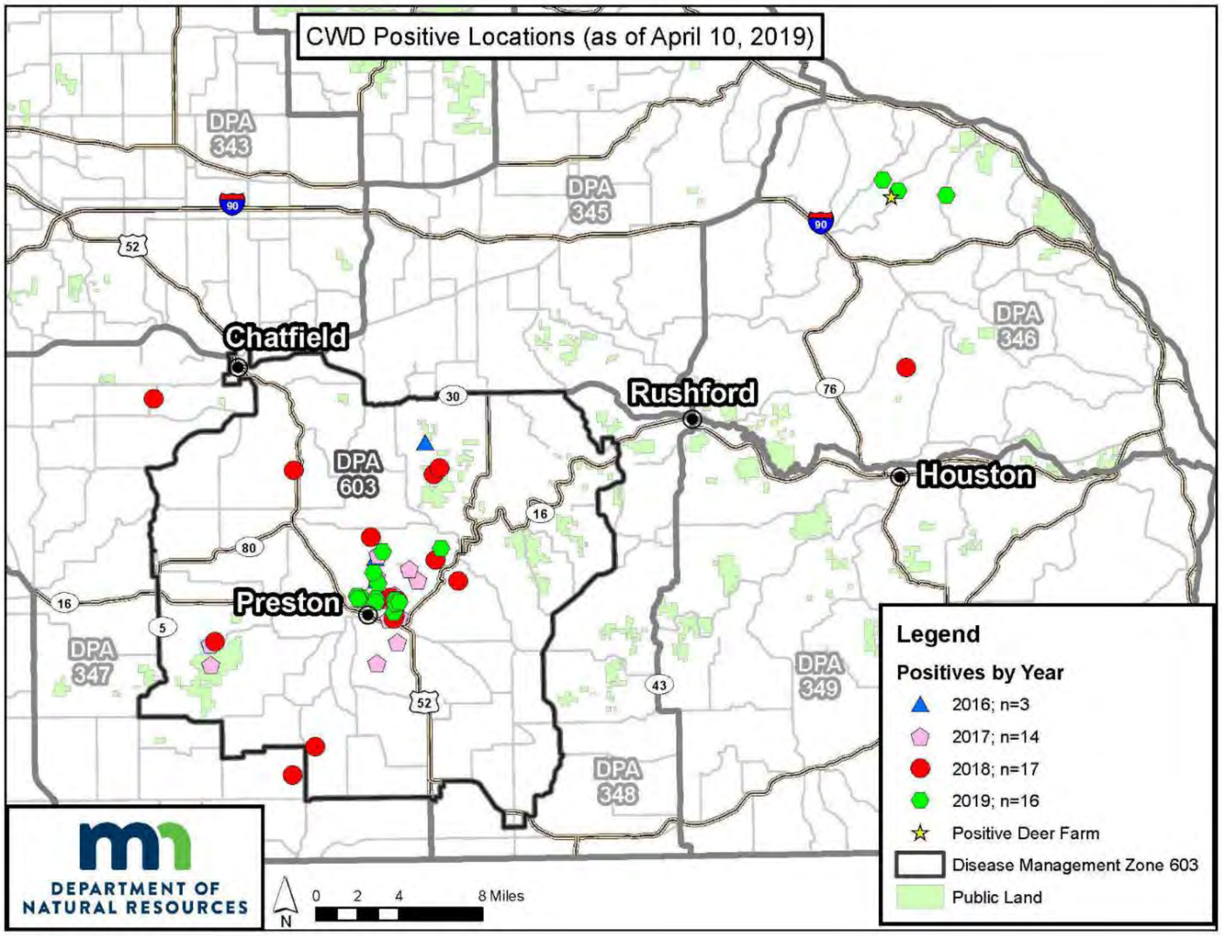

Figure 1. Spatial distribution of wild white-tailed deer confirmed with CWD infection in southeastern Minnesota as of 03 August 2019. There have been 50 wild white-tailed deer confirmed positive with CWD in southeastern Minnesota since fall 2016. The grey-labelled areas represent deer permit areas (DPA), which have recently been re-designated in the 600 series representing disease management zones. DPA 603 is outlined in black and is to be phased out completely (used here as a visual reference).

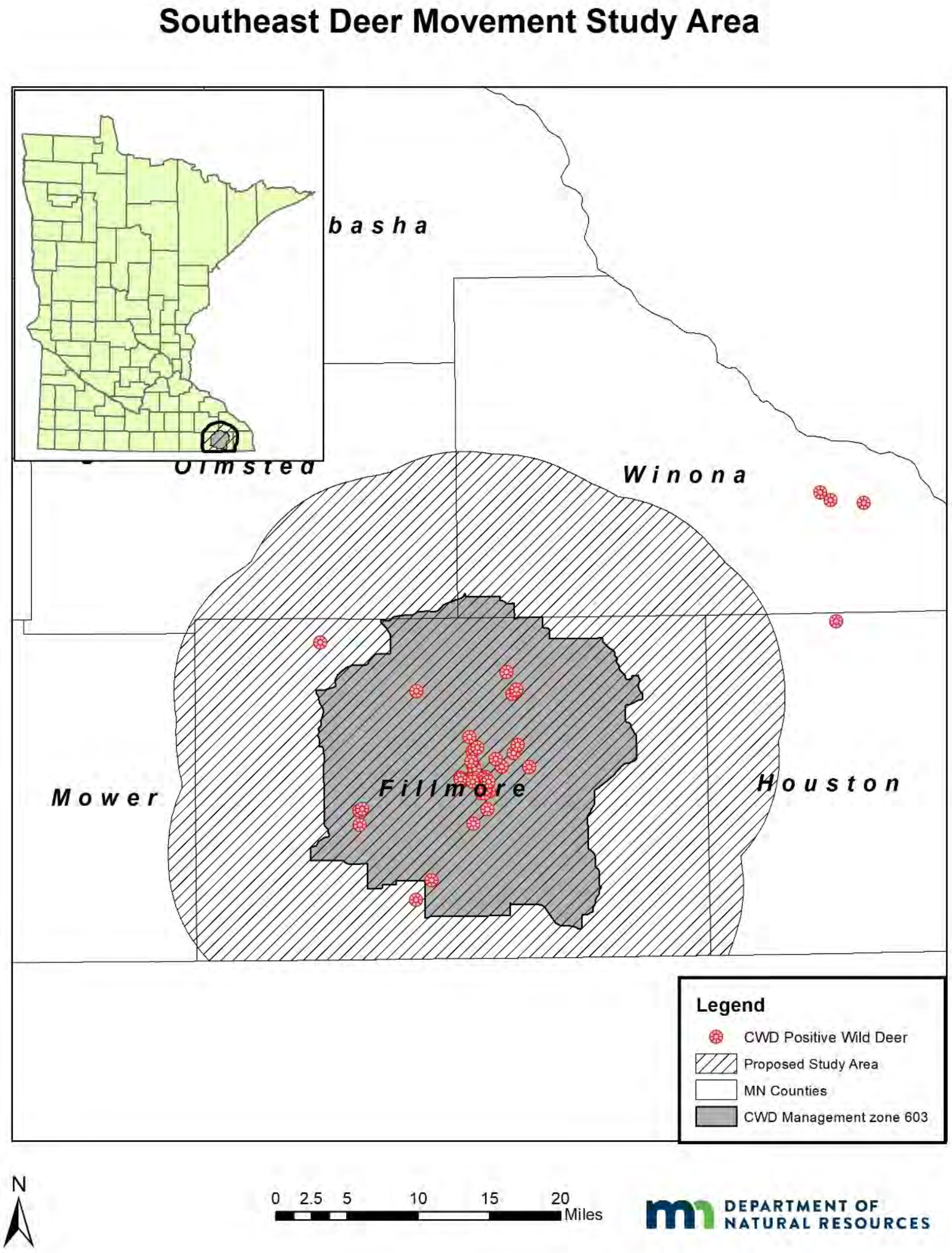

Figure 2. Approximate study area boundaries in and around the chronic wasting disease management zone (Deer Permit Area 603). Also shown are locations of CWD positive wild deer (n=50) from 2016 through September 2019 in southeastern Minnesota. This area is largely private land, so the final disposition of sampling locations for GPS collaring deer will depend on permissions we receive from cooperating landowners, weather patterns, and local scale landscape characteristics that facilitate helicopter capture of wild white-tailed deer.

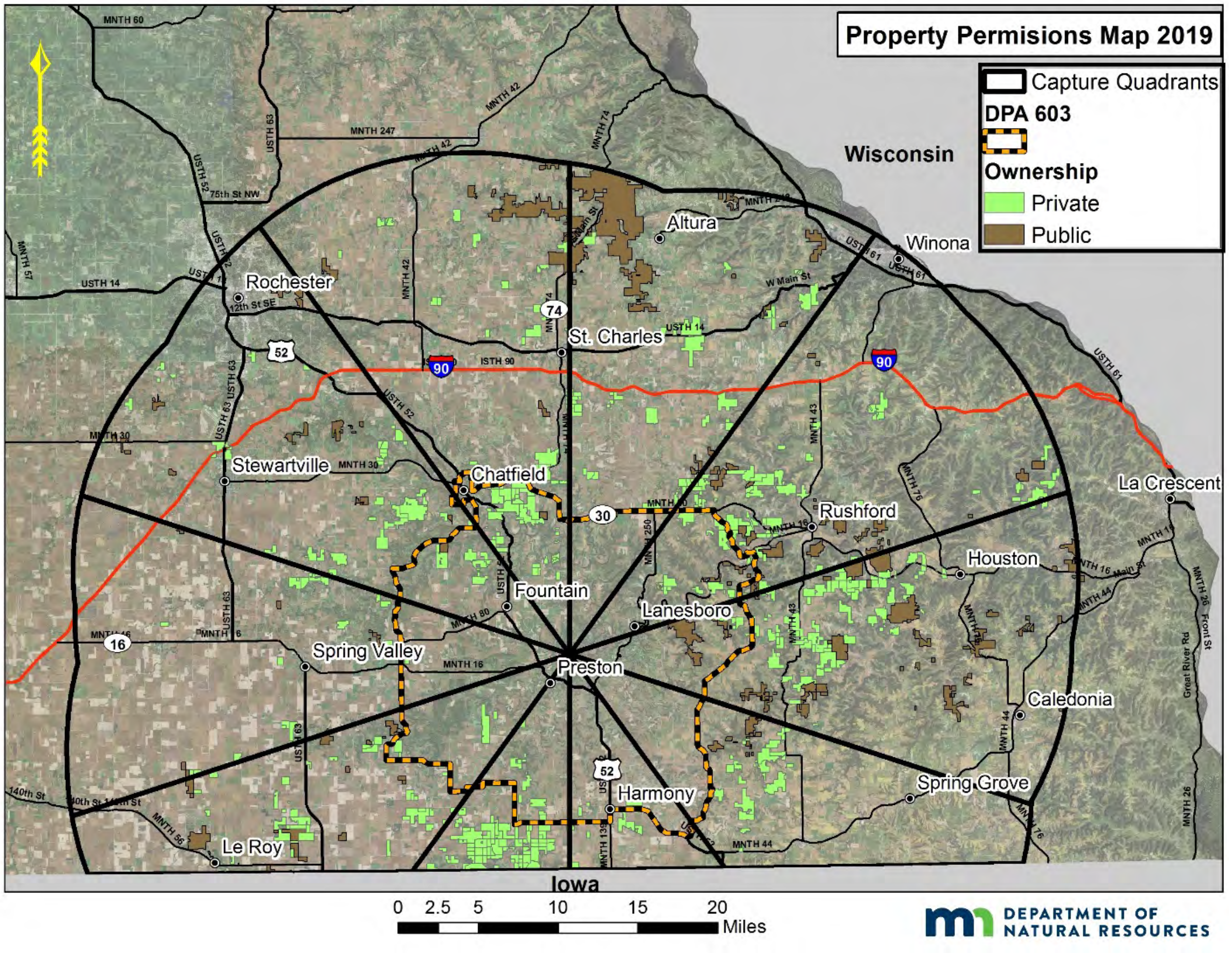

Figure 3. Spatial distribution of study area capture quadrants used as a basis for establishing the February 2019 deer capture goals in southeastern Minnesota. The target optimal capture distribution was established as 3-4 male and 3-4 female white-tailed deer fawns captured per quadrant. We secured permissions to access 115,259 acres of property, consisting of private (72,398 ac) and public (42,861ac) lands – over 180 mi^2.

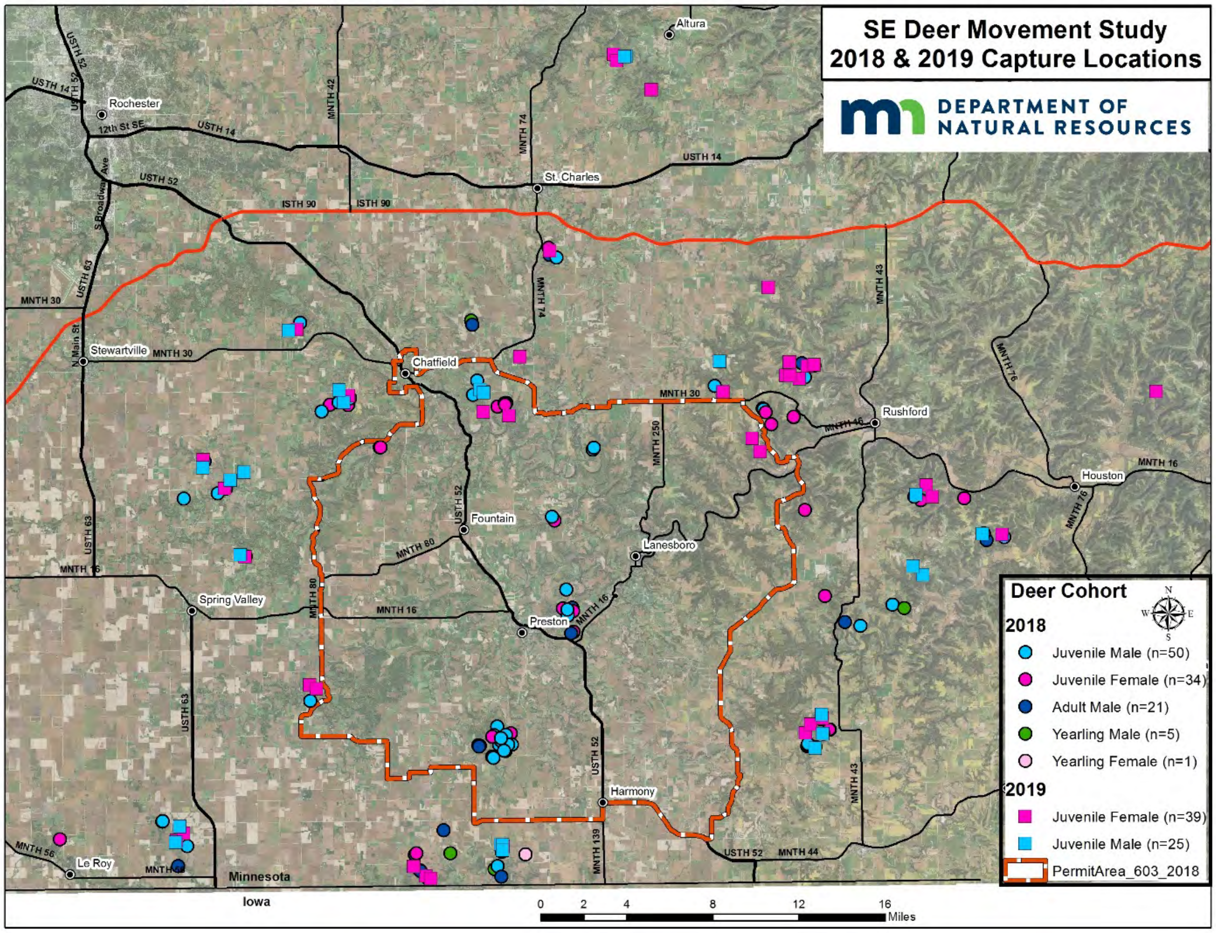

Figure 4. Spatial distribution of all deer captured and GPS-collared in southeastern Minnesota during March 2018 (n=109) and February 2019 (n=64). Points represent the locations where white-tailed deer were captured, collared with GPS units, and released in the study area centered on CWD management zone 603 in Fillmore County, Minnesota.

USING MOVEMENT ECOLOGY TO INVESTIGATE MENINGEAL WORM RISK IN MOOSE[1]

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ABSTRACT

Anthropogenic habitat change and moderating climatic conditions have enabled the northward geographic range expansion of white-tailed deer (*Odocoileus virginianus*) and the parasitic nematode (meningeal worm) it carries, *Parelaphostrongylus tenuis*. This expansion can have consequences in dead-end host species for other ungulates because meningeal worm reduces health, causes morbidity or direct mortality, and has been attributed to population declines. In northeastern Minnesota, which marks the southern extent of the bioclimatic range for moose (*Alces alces*), the population has declined more than 50% in the last decade, with studies detecting *P. tenuis* in 25 to 45% of necropsied animals. We took a novel, top-down approach for assessing the factors that are most associated with meningeal worm infection by linking moose movement ecology with known *P. tenuis* infection status from necropsy. Moose were outfitted with GPS-collars to assess their space use and cause-specific mortality. Upon death, a necropsy was performed to determine cause of death and document meningeal worm infection. We then created statistical models to assess the relationship between meningeal worm infection and exposure to hypothesized factors of infection risk based on the space-use of each moose by season. Predictors included landcover types, deer space use and density, environmental conditions, and demographics of individual moose (age and sex). Moose had a greater risk of infection when their home ranges contained higher proportions of wetter environments and their fall home ranges included more upland shrub/conifer. In contrast, the strongest relationships showed that higher proportions of mixed and conifer forest within spring home ranges resulted in lower risk of infection. Relationships between exposure and infection were strongest in the spring models, potentially due to moose foraging on ground vegetation during spring. By incorporating the movement of moose into disease ecology, we were able to test hypothesized components of infection risk with actual spatial and temporal exposure of individual necropsied moose. The probability of infection for moose in northeastern Minnesota was not influenced by deer density, although deer densities did not vary greatly within the study area (2 – 4 deer/km^2), highlighting the importance of both moose space use and environmental conditions in understanding infection risk. We suggest management strategies that use a combination of deer and land management prescriptions designed to limit contact rates in susceptible populations.

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[1] Journal of Mammology: *in press.*

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MICROBIAL ASSOCIATIONS AND SPATIAL PROXIMITY PREDICT NORTH AMERICAN MOOSE (ALCES ALCES) GASTROINTESTINAL COMMUNITY COMPOSITION[1]

Nicholas Fountain-Jones[2], Nicholas Clark[3], Amy Kinsley[2], Michelle Carstensen[4], James Forester[5], Tim Johnson[6], Elizabeth Miller[6], Seth Moore[7], Tiffany Wolf[2], and Meggan Craft[2]

ABSTRACT

Microbial communities are increasingly recognized as crucial for animal health. However, our understanding of how microbial communities are structured across wildlife populations is poor. Mechanisms such as interspecific associations are important in structuring free-living communities, but we still lack an understanding of how important interspecific associations are in structuring gut microbial communities in comparison to other factors such as host characteristics or spatial proximity of hosts. Here we ask how gut microbial communities are structured in a population of North American moose (*Alces alces*). We identify key microbial interspecific associations within the moose gut and quantify how important they are relative to key host characteristics, such as body condition, for predicting microbial community composition. We sampled gut microbial communities from 55 moose in a population experiencing decline due to a myriad of factors, including pathogens and malnutrition. We examined microbial community dynamics in this population utilizing novel graphical network models that can explicitly incorporate spatial information. We found that interspecific associations were the most important mechanism structuring gut microbial communities in moose and detected both positive and negative associations. Models only accounting for associations between microbes had higher predictive value compared to models including moose sex, evidence of previous pathogen exposure, or body condition. Adding spatial information on moose location further strengthened our model and allowed us to predict microbe occurrences with ~90% accuracy. Collectively, our results suggest that microbial interspecific associations coupled with host spatial proximity are vital in shaping gut microbial communities in a large herbivore. In this case, previous pathogen exposure and moose body condition were not as important in predicting gut microbial community composition. The approach applied here can be used to quantify interspecific associations and gain a more nuanced understanding of the spatial and host factors shaping microbial communities in non-model hosts.

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MYOCARDIAL CYSTICERCOSIS (PRESUMPTIVE *CYSTICERCUS TARANDI/TAENIA OVIS KRABBEI*) IN A MOOSE (*ALCES ALCES*)[1]

Arno Wünschmann[2], Anibal G. Armién[2], and Michelle Carstensen[3]

History: On January 24, 2013, this free-ranging moose cow was captured, equipped with a collar by field biologists of the Minnesota Department of Natural Resources (MDNR) and released. The moose was part of a MDNR study to examine the sudden decline of Minnesota's moose population. The collar had GPS capabilities and a device that triggered a mortality signal when the animal did not move for 6 hours. The animal was aged at 11 years at capture based on analysis of the annual cementum layer of an extracted incisor tooth. A mortality signal was emitted at 12.25AM on November 25, 2015. The carcass of the moose was extracted from the field and a necropsy was performed at the Minnesota Veterinary Diagnostic Laboratory (MVDL) within 36 hours of the first notification.

Gross Findings: The animal weighed 369kg. It only had very scant internal (e.g. perirenal) fat stores and no measurable subcutaneous adipose tissue stores. The abdominal cavity contained 5 liters of clear watery colorless fluid with a small amount of delicate beige stringy elastic material (interpreted as fibrin strands). The heart weighed 2.7kg. The myocardium of the left ventricular free wall, septum and right ventricular free wall had numerous (approximately 50 in total), scattered 1 to 2cm long and approximately 1.5cm in diameter cysts (Fig. 1). These cysts contained watery clear fluid and a white, spherical, approximately 0.5cm in diameter structure. In addition, rare collapsed cysts, approximately 1cm in diameter, with greenish pasty material were present within the myocardium. Similar cysts were present in high number within the skeletal muscles particularly in the movers of the head, the masticatory muscles and the esophagus. Approximately 10% of the liver parenchyma were replaced by up to 5cm in diameter cysts containing brownish pasty material that were bordered by fibrous caspule. This lesion is consistent with fluke (*Fascioides magna*)-induced hepatitis. Few flukes (presumptive *Paramphistomum cervi*) were attached to the ruminal mucosa. The animal was pregnant with an approximately 8cm long (crown to rump length) fetus.

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CEREBRAL HYDATID CYST (*ECHINOCOCCUS GRANULOSUS*) IN A MOOSE (*ALCES ALCES*)[1]

Arno Wünschmann[2], Anibal G. Armién[2], and Michelle Carstensen[3]

History: On January 30, 2013, a free-ranging, 11-year old, female moose (*Alces alces*) was captured, equipped with a satellite-linked Global Positioning System (GPS) collar by the Minnesota Department of Natural Resources (MDNR) and released. The moose was part of a study to examine the causes of mortality in Minnesota's declining moose population. A mortality signal was emitted from the GPS collar on December 07, 2013. The carcass was found intact, without signs of predation or scavenging, and was extracted from the field and underwent necropsy to the Minnesota Veterinary Diagnostic Laboratory (MVDL) 2 days after the mortality signal was first received.

Gross Findings: The animal weighed 439kg and had moderate internal fat stores although measurable subcutaneous fat stores were absent in the rump region and near the base of the tail. An approximately 7 cm by 5 cm by 3 cm unilocular cyst replaced large portions of the frontal and parietal lobe of the right cerebral hemisphere (Fig. 1). The cyst was slightly raised over the meningeal surface and extended through the entire cortex abutting against and distorting the right lateral ventricle. The inner surface of the right parietal bone had a slight depression that conformed to the raised aspect of the cyst. The cyst was bordered by an approximately 2 mm thick opaque wall. The cyst contained clear watery fluid with sandy material. The right caudate nucleus was softened and slightly discolored (Fig. 2). The brain parenchyma neighboring the cyst was compressed and the midline of the cerebrum was deviated to the left.

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EXPANDING GIS ANALYSES TO MONITOR AND ASSESS NORTH AMERICAN MOOSE DISTRIBUTION AND DENSITY[1]

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ABSTRACT

Development of long-term geographic information system (GIS) databases of species densities and distributions, combined with biological, ecological, and management-related metrics, can help guide research and management strategies. Here we summarize 3 decades of North American moose (*Alces alces)* population and harvest densities collected at the management unit scale for the years 1980, 1990, 2000, and 2010. A summary analysis of these data indicates that moose have both expanded and contracted along their southern range boundary in recent decades - including the Prairie Provinces and states, and much of the northeastern United States. A narrow band of relatively stable and high-density moose populations extends from central Alaska across the Prairie Provinces, and east to the Maritime Provinces and upper New England states. Distributions in 2010 indicate that moose now occupy an area >9,492,000 km^2 in North America. We also identified that a core range of boreal habitat, only 30% of the occupied range across the continent, supports 89% of the estimated 1 million moose in North America. Time-series analyses can offer a simple and cost-effective approach to monitor the status of moose populations in North America, and might be particularly insightful given the current and predicted future influences of climate change on moose. Other analyses might address population dynamics, habitat, environmental constraints, and harvest management, among other issues. We encourage jurisdictions to cooperate strategically in implementing and coordinating GIS analyses to monitor, assess, and manage the North American moose population.

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[1] Alces. 2018. 54: 45-54.

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CHRONIC WASTING DISEASE DETECTION AND MORTALITY SOURCES IN A SEMI-PROTECTED DEER POPULATION[1]

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ABSTRACT

Surveillance for wildlife diseases is essential for assessing population dynamics of ungulates, especially in free-ranging populations where infected animals are difficult to sample. Chronic wasting disease (CWD) is an emerging infectious disease of concern because of the potential for substantial negative effects on populations of cervids. Variability in the likelihood that CWD is detected could invalidate traditional estimators for prevalence. In some instances, deer located after death cannot be tested for infectious diseases, including CWD, because of lack of availability or condition of appropriate tissues. We used various methods to detect infectious diseases that could cause mortality for deer *Odocoileus* spp. residing in Wind Cave National Park, South Dakota, USA, and we report survival estimates for animals in this population. We included 34 monthly encounters of deer resightings and 67 mortalities. We tested live deer by tonsillar biopsy for CWD and estimated pooled prevalence (mean ± SE) at 5.6 ± 3.0% over the three-year study. Live deer potentially had exposure to several infectious diseases, including bluetongue, epizootic hemorrhagic disease, bovine viral diarrhea, West Nile virus, and malignant catarrhal fever, but no apparent morbidity or mortality from those diseases. We tested survival and influence of covariates, including age and sex, using known-fate analysis in Program MARK. Those data best supported a model with time-invariant encounter probability and an annual survival of 72.8%. Even without direct pressure from hunting within the park, average life expectancy in this population was 3.2 years. Only 68% of mortalities contained sufficient material for CWD sampling (because of predation and scavenger activity) and >42% of these were CWD-positive. These findings underscore the possible biases in postmortem surveillance estimates of disease prevalence because of potential for subclinical infected animals to be removed by predators and not tested.

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[1] Wildlife Biology. 2018(1). https://doi.org/10.2981/wlb.00437.

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LIMITED DETECTION OF ANTIBODIES TO CLADE 2.3.4.4 A/GOOSE/GUANGDONG/1/1996 LINEAGE HIGHLY PATHOGENIC H5 AVIAN INFLUENZA VIRUS IN NORTH AMERICAN WATERFOWL[1]

David E. Stallknecht[2], Clara Kienzle-Dean[2,3], Nick Davis-Fields[2], Christopher S. Jennelle[4], Andrew S. Bowman[5], Jacqueline M. Nolting[5], Walter M. Boyce[6], James M. Crum[7], Jefferson J. S. Santos[8], Justin D. Brown[9], Diann J. Prosser[10], Susan E. W. De La Cruz[11], Joshua T. Ackerman[12], Michael L. Casazza[12], Scott Krauss[13], Daniel R. Perez[2], Andrew M. Ramey[14], Rebecca L. Poulson[2]

ABSTRACT

During 2014, highly pathogenic (HP) influenza A viruses (IAV) of the A/Goose/Guangdong/1/1996 lineage (GsGD-HP-H5) were detected in domestic poultry and wild birds in Canada and the United States. These clade 2.3.4.4 GsGD-HP-H5 viruses included reassortants possessing North American lineage gene segments; were detected in wild birds in the Pacific, Central, and Mississippi flyways; and caused the largest HP IAV outbreak in poultry in United States history. To determine if an antibody response indicative of previous infection with clade 2.3.4.4 GsGD-HP-H5 IAV could be detected in North American wild waterfowl sampled before, during and after the 2014 - 2015 outbreak, sera from 2793 geese and 3725 ducks were tested by bELISA and hemagglutination inhibition tests using both clade 2.3.4.4 GsGD-HP-H5 and North American lineage low pathogenic (LP) H5 IAV antigens. We detected an antibody response meeting a comparative titer-based criteria (two dilution difference in titer) for previous infection with clade 2.3.4.4 GsGD-HP-H5 IAV in only five birds, one blue-winged teal (*Spatula discors*) sampled during the outbreak and three mallards (*Anas platyrhynchos*) and one Canada goose (*Branta canadensis*) sampled during the post-outbreak period. These serological results are consistent with the spatiotemporal extent of the outbreak in wild birds in North America during 2014 and 2015 and limited exposure of waterfowl to GsGD-HP-H5 IAV, particularly in the central and eastern United States.

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Publications Lists

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Bischof, Stefan. M., Brian R. Herwig Stephen D. Sebestyen **Mark A. Hanson** Kyle D. Zimmer James B. Cotner Timothy J. Kroeger 2019. Further Development of a Specific Conductivity Approach to Measure Groundwater Discharge Area within Lakes, First published:01 February 2019 https://doi.org/10.1111/1752-1688.12730

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