Susitna–Watana Hydroelectric Project
(FERC No. 14241)

Terrestrial Furbearer Abundance and Habitat Use
Study Plan Section 10.10

Study Completion Report

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF ACRONYMS, ABBREVIATIONS, AND DEFINITIONS

Abbreviation	Definition
AEA	Alaska Energy Authority
AIC	Akaike Information Criterion
AICc	Akaike Information Criterion (corrected for small sample size)
CIRWG	Cook Inlet Regional Working Group
DNA	deoxyribonucleic acid
DNPP	Denali National Park and Preserve
DSLS	days since last snowfall
FERC	Federal Energy Regulatory Commission
GIS	Geographic Information System
GPS	Global Positioning System
ILP	Integrated Licensing Process
ISR	Initial Study Report
PME	Protection, Mitigation, and Enhancement
Project	Susitna–Watana Hydroelectric Project
RSP	Revised Study Plan
SECR	spatially explicit capture–recapture
SPD	Study Plan Determination
UAF	University of Alaska Fairbanks

1. INTRODUCTION

The Terrestrial Furbearer Abundance and Habitat Use Study, Section 10.10 of the Revised Study Plan (RSP) approved by the Federal Energy Regulatory Commission (FERC or Commission) for the Susitna–Watana Hydroelectric Project, FERC Project No. 14241, focuses on providing current information on the abundance and habitat use of four species of terrestrial furbearers: coyote (*Canis latrans*), red fox (*Vulpes vulpes*), lynx (*Lynx canadensis*), and marten (*Martes americana*).

A summary of the development of this study, together with the Alaska Energy Authority's (AEA) implementation of it through the 2013 study season, appears in Part A, Section 1 of Initial Study Report (ISR) 10.10 filed with FERC in June 2014 (UAF 2014a). As required under FERC's regulations for the Integrated Licensing Process (ILP), the ISR describes AEA's "overall progress in implementing the study plan and schedule and the data collected, including an explanation of any variance from the study plan and schedule" (18 CFR 5.15(c)(1)).

Since filing the ISR in June 2014, AEA has continued to implement the FERC-approved plan for the Terrestrial Furbearer Abundance and Habitat Use Study (Terrestrial Furbearer Study). For example:

- The study team completed field work (winter and summer).
- The study team completed aerial track surveys.
- The study team completed laboratory analyses of DNA from hair and scat samples.
- The study team developed population estimates of coyotes and red foxes through fecal genotyping and genetic capture–recapture modeling.
- The study team assessed density of snowshoe hares and voles using pellet counts and live captures, respectively.
- The study team compiled furbearer habitat data using aerial and ground-based surveys.
- The study team developed occupancy probabilities for all target furbearers using ground-based survey data.
- On October 21, 2014, AEA held an ISR meeting for the Terrestrial Furbearer Abundance and Habitat Use Study, along with meetings for each of the other wildlife studies.

In furtherance of the next round of ISR meetings and FERC's SPD expected in 2016, this report contains a comprehensive discussion of results of the Terrestrial Furbearer Study from the beginning of AEA's study program in 2012, through the end of calendar year 2014. It describes the methods and results of the Terrestrial Furbearer Study and explains how the study objectives set forth in the Commission-approved Study Plan have been met. Accordingly, with this report, AEA has now completed all field work, data collection, data analysis, and reporting for this study.

2. STUDY OBJECTIVES

The five objectives of this study are established in RSP Section 10.10.1:

1) Develop population estimates of coyotes and red foxes through fecal genotyping and genetic capture–recapture analyses, using scats collected along trails and rivers throughout the study area during winter months (January–March) in 2013 and 2014;

2) Develop a population estimate of marten through DNA-based capture–recapture analysis, using hair samples collected in the reservoir inundation zone with hair-snag tubes;

3) Develop a population estimate of lynx through DNA-based capture–recapture analysis, using hair samples collected throughout the study area with hair-snag plates;

4) Assess prey abundance in the study area by conducting snowshoe hare pellet counts and estimating vole density using a mark–recapture framework from live-trapping sessions;

5) Compile habitat-use data for the furbearer species being studied, using aerial track surveys.

3.   STUDY AREA

As established by RSP Section 10.10.3, the Terrestrial Furbearer Study Area (Figure 3-1) includes all terrestrial areas that are safely accessible by snowmachine within a 10-km (6.2-mile) buffer zone surrounding the areas that may be directly altered or disturbed by the proposed Project construction and operations, including facility sites, laydown/storage areas, the reservoir inundation zone, and access road and transmission-line corridor alternatives.

As described in the ISR Overview (Section 1.4) filed in June 2014 and subsequently the *Proposal to Eliminate the Chulitna Corridor from Further Study* filed with FERC September 17, 2014, AEA explained that it had decided to pursue the study of an additional alternative north/south-oriented corridor alignment for transmission and access from the proposed dam site to the Denali Highway, referred to as the "Denali East Corridor Option," and to eliminate the Chulitna Corridor from further study.

4.   METHODS AND VARIANCES

The methods implemented for each of the four major study components and variances are described below.

4.1.   Sample Collection

AEA implemented the methods as described in the Study Plan (RSP Section 10.10.4.1), with the exception of variances explained below (Section 4.1.1).

The study team established a network of seven survey transects totaling approximately 311 km (193 mi). Transects ranged in length from approximately 15 to 80 km (9.5–50 mi), and were established along proposed transmission corridors and natural corridors of animal movement in the study area, such as creeks, rivers, and the Denali Highway (Figure 4.1-1). Transects along the Denali Highway and along the Denali East and Denali West access corridor options were relatively long (70–80 km), while shorter transects extended up several tributary drainages (Watana, Tsusena, Jay, Deadman, and Butte creeks). Transect placement ensured roughly equal coverage of the accessible survey area, but notable gaps existed in areas of the reservoir zone and

the Chulitna and Gold Creek corridors which were either inaccessible on snowmachine or located on Cook Inlet Regional Working Group (CIRWG) lands. The study team aimed to survey at least one transect per day on a rotating basis, such that each transect was traveled approximately every week during January 7–April 15, 2013, and January 8–April 2, 2014, collecting all carnivore scats seen along the transects. Deviations from this survey schedule only occurred during periods of heavy and continuous snowfall when scats would have been buried and undetectable.

When a carnivore scat was encountered, a GPS location was recorded and the scat was collected using a ziplock bag. The ziplock bag was then placed in an autoclave bag or whirlpack labeled with an ID number. Double-bagging prevented cross-contamination of fecal DNA among samples during storage. The maximum age of the scat (in hours) was estimated based on snowfall and travel history. The carnivore species that made the scat was identified based on characteristic morphology (size and shape) and associated snow tracks. Observers rated their certainty in species identification (values ranged from 35–95 percent certainty). All species identifications were later verified using molecular analyses. Scats were stored frozen in the field until transport to the lab, where they were stored at –80°C until DNA was extracted.

The study team deployed hair-snag stations every 5 km (3.1 mi) along transects. Stations were placed near fresh lynx trails and nailed to trees at a height of 50 cm. Lynx hair-snags were constructed from a carpet pad imbedded with a wire tube brush that was soaked in a lure of beaver castor and catnip. An aluminum pie plate was hung above the hair snag and used as a visual attractant (Figure 4.1-2). This design was based on the National Lynx Detection Protocol (McKelvey et al. 1999). Hair-snag stations were checked approximately twice a month during January 29–April 12, 2013, and January 17–April 1, 2014. Hair samples were removed from the wire brush using tweezers and placed in a coin envelope for storage. Sealed coin envelopes were then placed in a larger bag that contained silica desiccant beads to remove moisture from the samples. After hairs were removed from the snag, a pocket lighter was used to burn the wire brush and clean off any remaining hairs or particles that could contaminant future samples, and additional lure was added to the carpet pad. Hair samples were stored in silica to preserve DNA, and the bags containing silica and hair samples were kept frozen at –20°C as an added measure to preserve DNA until extraction.

Five marten hair tubes were deployed in forested locations considered likely to be used by marten to test the effectiveness of the sampling method during the 2013 field season, and 43 marten hair tubes were deployed during February 4–March 26, 2014. Marten tubes were constructed using the design described by Pauli et al. (2008), which was successfully used to obtain marten hair samples on Admiralty Island in Southeast Alaska. A 35-cm-long piece of PVC tube (10.2 cm in diameter) was fitted with a polycarbonate door on one end and a piece of bait (chicken) hung at the opposite end. A stainless-steel tube brush was inserted into the middle of the tube trap to collect hair samples from the marten as it entered the front of the trap and moved to the back to access the bait (Figure 4.1-3). Hair samples were removed from the wire brush using tweezers and placed in a coin envelop for storage. Sealed coin envelops were then placed in a larger bag that contained silica desiccant beads, which removed moisture from the samples. After hairs were removed from the brush, a pocket lighter was used to burn the wire brush and clean off any remaining hairs or particles that could contaminant future samples and the chicken was replaced, if necessary. Hair samples were stored in silica to preserve DNA, and

the bags containing silica and hair samples were kept frozen at –20°C as an added measure to preserve DNA until extraction.

The study team estimated snowshoe hare abundance from counts of fecal pellets in 15 survey grids spaced throughout the survey area (Figure 4.1-4). Each grid was made up of 50 circular plots with a radius of 0.5 m (1.6 ft) spaced 15 m (49.2 ft) apart and arranged in a rectangular array (10 plots by 5 plots). All pellets were counted and cleared from the plots during each survey. Pellets were aged, based on appearance, to estimate whether they were more or less than a year old (Prugh and Krebs 2004). Pellet grids were placed in contiguous areas of hare habitat (spruce forest and riparian shrub) located throughout the survey area. Creek drainages and portions of the Project area, including the Denali East and West corridors and reservoir inundation zone, were specific areas of interest. Three grids established in August 2012 were resurveyed and 12 new grids were established for field sampling during July 15–24, 2013. All 15 grids from 2013 were resurveyed during July 10–31, 2014.

Pellet counts provide a reliable index of snowshoe hare density, but the specific form of the relationship between pellets and density can vary regionally (Krebs et al. 2001, Murray et al. 2002). Therefore, the study team used the relationship between pellet and hare density estimated by a study of snowshoe hares in a nearby area of the central Alaska Range in 1999. In that study, a density estimate of hares was obtained from a 5-night trapping session on a 9.4-hectare trapping grid, and pellet density was obtained by conducting pellet counts in 126 circular plots located on the trapping grid (Prugh 2005). The following conversion factor calculated by Prugh (2005) was used:

$$D_h = 0.03 * D_p$$

where D_h is the density of hares (number per hectare) and D_p is the density of pellets (number per m^2). Because this relationship was estimated during a single year, confidence intervals around the conversion factor could not be estimated. The conversion of pellets to hare density was therefore approximate, but variation in pellet density among plots and years should have reflected changes in hare abundance accurately.

The study team estimated abundance of voles using live-trapping on 15 grids (Figure 4.1-4). One meadow grid (Watana Creek) established in August 2012 was resurveyed and 14 new grids were established for field sampling during August 2–13, 2013. All 15 grids were resurveyed during July 11–28, 2014. Each grid was composed of 100 live-trap locations, spaced at 10-m (32.8-ft) intervals arranged in a square array (10 traps by 10 traps). The study team deployed Sherman live traps (H. B. Sherman Traps, Inc.; model LFA, 3×3.5×9 inches) for one night at each grid. Traps were covered with roofing paper for rain protection and a wad of upholstery cotton was provided inside each trap for bedding and insulation. Traps were baited with sunflower seeds at 20:00 local time and checked the following morning at 08:00. Each captured animal was identified to genus or species, sexed, weighed, and released.

1.1.1.   Variances

1.1.1.1.   2013 Season

Study 10.10 ISR, Part A, Section 4.1.1 (UAF 2014a) outlined the following variances in the 2013 study effort. The survey area was modified for the following reasons: (1) much of the study area was located far from the 2013 winter base of operations on the Denali Highway; (2) physical barriers in 2013 and 2014 prevented safe travel by snowmachine along the sections of the Susitna River downstream from the proposed Watana dam site; and (3) access to CIRWG lands was precluded during the winter survey seasons in both 2013 and 2014.

The lack of a suitable base of operations centrally located within the large study area in 2013 made it impossible to access the entire study area. Because the only feasible option for a base of winter operations in 2013 was a lodge at Mile 68 of the Denali Highway (Figure 3-1), sampling sites along and near that road were accessible but the proposed Watana dam site and the area west of Watana Creek were too far away for routine access. A temporary tent camp was established near Watana Creek in March 2013, allowing limited sampling of areas closer to the proposed reservoir inundation zone.

The study team modified the deployment and use of the lynx hair snags to increase sampling efficiency in the field and to create a survey layout that allowed better comparison of the lynx survey data with those from the canid scat collection effort. Rather than subdividing the entire study area into 50 blocks as proposed in the Study Plan, lynx stations were deployed along the major sampling transect routes that were established for scat collections. Stations were systematically deployed every 5 km (3.1 mi) along those routes to maintain a similar sampling density to that described in the Study Plan (Figure 4.1-5). This method of station layout and deployment allowed the field crew to check the hair stations while simultaneously looking for scats, thereby increasing the efficiency of data collection. Creating spatial overlap of the different types of sample collection locations provided additional descriptive data concerning abundance of canids and lynx as well as potential interspecific interactions in the shared sampling area. These variances had no impact on the study team's ability to meet study objectives because sampling routes tended to be located along drainages, therefore encompassing most of the available lynx habitat in the study area. Areas between sampling routes were generally higher elevation alpine habitats, which are considered less suitable for lynx (Ruggiero et al. 2000).

Collection of marten hair samples was not accomplished in 2013 as proposed in the Study Plan because of the difficulty of snowmachine access in the proposed reservoir inundation zone, which included a large proportion of the suitable marten habitat (spruce forest) present in the surveyed areas, and the lack of access to CIRWG lands. The inundation zone was identified in the Study Plan as the primary area to be surveyed for marten in both years of study.

Snowshoe hare pellet surveys were conducted primarily as described in the Study Plan, although the study team changed the way that the sample grid locations were allocated to better account for variability of habitats throughout the survey area. Instead of dividing the study area into equal-sized blocks as described in the Study Plan, grids were established in parts of the study area where the desired habitat elements (spruce forest or riparian shrubs) occurred in contiguous

patches. Habitat maps and aerial scouting were used to pinpoint the specific locations that fit these habitat requirements. The study team used stratified randomization (stratified by the two major habitat types, spruce forest and riparian shrubs) to distribute plots in hare habitat throughout the study area. The study team also established grids in accessible portions of the Project area, such as the dam and camp infrastructure area and the Denali West Corridor option. The study team increased the number of sampling locations from the 8–10 grids proposed in the Study Plan to a total of 15 grids, an increase in sampling effort that was considered necessary because of the large size of the study area and high level of variability in pellet density among grids in 2012.

The vole live-trapping survey in 2013 also included variances from the Study Plan. As proposed, trapping grids were established in spruce and meadow habitats. These grids were set up in pairs (one grid in spruce and one in meadow) throughout several major drainages and the Denali West corridor (see Section 5.1 below). One grid (Deadman Mountain Meadow) was set up in a meadow without a paired forest grid because of the lack of suitable spruce habitat in that location. Trapping nights were reduced from the one to five nights proposed in the Study Plan to a single night per grid. This reduction in effort was justified by the strong correlation ($r = 0.85$; L. Prugh, unpublished data) between the number of voles caught on the first night of trapping and the vole density estimated from 5-night mark–recapture trapping sessions in a similar study in DNPP (Prugh 2005). The original plan was to trap 2 grids for 5 nights to obtain mark-recapture density estimates, and then to estimate density on 6–8 additional grids that would be trapped for 1 night only. Density was to be estimated on these additional grids using data from the 2 mark-recapture grids by attempting to relate the number of voles caught on the first night to the density estimate from the 5-night survey (following Prugh 2005). Conducting 1-night sessions on all grids allowed abundance estimates to be generated in 15 areas rather than 8–10 as originally proposed.

Although vole live-trapping in 2013 was modified from the Study Plan (reducing trap nights to a single night per grid), as described above, this proposed method did not allow the validity of the first-night captures as indices of density to be assessed. Therefore, a long-term (1992–2002) vole-trapping data set in Denali National Park and Preserve (DNPP) was analyzed, which revealed a strong relationship between the number of voles captured on the first night of trapping and the density estimate from the full 5-night mark–recapture session ($n = 43$ grid-years, $R^2 = 0.852$; L. Prugh, unpublished analyses):

$$D_v = 0.5157*N_1 - 0.0684$$

where D_v is the density of voles (number per hectare) estimated from a 5-night mark–recapture session and N_1 is the number of voles caught on the first night of the 5-night session. The study team used identical trap arrays and trapping protocols as in the DNPP study area, which was adjacent to the study area and where the same species of voles were captured. The relationship estimated from the DNPP data set was more robust than the conversion factor that would have been created using the data from this study.

1.1.1.2.   2014 Season

The procedural variances from 2013 were continued in the second winter of study in 2014 and the modifications to the Study Plan, as outlined in the Study 10.10 ISR, Part C, Section 7.1.2 (UAF 2014b) and described below, were implemented. As described above under Study Area, AEA decided to pursue the study of an additional alternative north/south-oriented corridor alignment for transmission and access from the dam site to the Denali Highway, referred to as the "Denali East Corridor Option," and to eliminate the Chulitna Corridor from further study. The addition of the Denali East Corridor Option did not affect survey locations for this study. Winter field surveys were completed before this new corridor option was added and summer 2014 prey survey locations were not changed from the locations established in 2013. Portions of the Denali East Corridor have been sampled by some of the existing furbearer and prey survey locations, so information was available for this new corridor option.

The lack of access to CIRWG lands prevented the field crew from sampling in the western portion of the proposed reservoir inundation zone during both 2013 and 2014. Access to CIRWG lands was granted after both winter field seasons had been completed. The combination of access restrictions with physical barriers along the Susitna River (cliffs, steep slopes, and unstable ice conditions) made it impossible or unsafe to cross from the north side of the Susitna River to the south side in the reservoir inundation zone.

A change in base camp location in 2014 (Figure 3-1) improved the study team's ability to travel throughout the study area, but areas south of the river still remained inaccessible by snowmachine. Sampling was conducted in as much of the study area as possible, but no surveys were conducted in the Chulitna or Gold Creek corridors. To maximize sampling effort in areas accessible by snowmachine from the 2013 winter base of operations, the survey area was expanded to include areas northeast of the study area (Figure 3-1). The study team extended track transects farther south down Deadman Creek in winter 2014 to sample more of the study area near the proposed dam site. These variances allowed sampling to be conducted more efficiently in areas that may be most vulnerable to the impacts of project construction, such as the Deadman Creek drainage and Denali Corridor. Although logistical difficulties made it impossible to survey the entire study area proposed in the Study Plan (RSP Section 10.10.3), the study team used results from accessible areas to extrapolate analytical results across the inaccessible portions of the study area.

Because very few hair samples were obtained from lynx hair snags in 2013, the study team also backtracked fresh lynx tracks that were discovered while checking lynx hair snags in 2014 in an effort to increase the sample size of hair samples. Winter backtracking has been shown to be an effective way to locate hair samples that have been rubbed off on tree bark or left in bedding areas (McKelvey et al. 2006).

Snowmachine access remained unsafe and CIRWG lands remained off-limits during the winter field season in 2014, so marten hair traps were deployed on accessible lands north of the originally proposed survey area. Rather than deploying hair tubes using a grid-based system as described in the Study Plan, hair tubes were deployed at approximately 1-km (0.62 mi) intervals along major sampling transect routes that were established for scat collection, as described above for lynx (Figure 4.1-5). Because marten home ranges are small and a comprehensive survey of

the entire study area would be impractical, the marten survey was restricted to heavily forested areas near the inundation zone that were on accessible lands (no access to CIRWG lands was possible in winter 2014, as was the case in winter 2013). The study team surveyed an area of approximately 125 km² (48.3 mi²) north of the proposed Watana dam site and inundation zone in winter 2014. This marten survey area was divided into 25 5-km² (1.9-mi²) blocks, roughly corresponding in size to the home range of female martens reported in the study area during the APA Project studies in the 1980s (3–6 km² [1.2–2.3 mi²]; Buskirk 1983, Buskirk and McDonald 1989). Marten hair tubes were deployed in those areas closest to the proposed inundation zone in areas of dense spruce forest, similar to habitats found in the inundation zone. A total of 43 hair tubes were deployed in 2014, creating a trap density that was greater than the originally proposed study design, in an attempt to increase detection.

4.2.   Genetic Analyses

AEA implemented the methods as described in the Study Plan (RSP Section 10.10.4.2) with no variances.

1.1.2.   DNA Extraction

To extract DNA, scats were removed from the –80˚C freezer and placed on ice to defrost slowly. After the outer surface of each scat had thawed (~30 minutes), the outer surface of each scat was rubbed with the end of a wooden craft stick (Mumma et al. 2015). The end of the craft stick that contained the sample was snapped off into a 1.5-ml tube so that no part of the stick extended above the top of the tube. Hair samples were removed from coin envelopes using sterilized forceps and placed in tubes. DNA was extracted from scats and hairs using Qiagen DNA Investigator Kits (Qiagen Inc., Valencia, CA) with a negative control included in each batch to monitor for contamination.

1.1.3.   Species Identification

Each sample was identified to species using a modification of a previously developed mitochondrial DNA test (De Barba et al. 2014). Primer pair DL1F and DL5R (Palomares et al. 2002) and a forward primer, Gulo1F (Dalen et al. 2004), were combined with the primers SIDL (Murphy et al. 2000), H3R (Dalen et al. 2004), and H16145 (Murphy et al. 2000), to amplify DNA fragments of species-specific lengths. Diagnostic fragment lengths for each species were as follows: red fox = ~346 base pairs (bps), coyote = ~363 bps, lynx = ~125 bps, marten = ~318 bps, wolf = ~368 bps, and wolverine = ~242 bps. The conditions for 15 μL reactions were 0.2 μM DL1F, 0.2 μM DL5R, 0.2 μM Gulo1F, 0.4 μM SIDL, 0.4 μM H3R, 0.2 μM H161453, 3 μL H20, 1.26 μL TE buffer, 7.5 μL 1x Qiagen Master Mix, 1.5 μL Q solution, and 1.5 μL of DNA extract. Reactions were later scaled down to 7 μL to reduce costs, because testing indicated no loss of quality in results from lower-volume reactions. Primer concentrations were maintained while adjusting the remaining solution volumes to 0.69 μL dH20, 0 μL TE buffer, 3.5 μL 1x Qiagen Master Mix, 0.7 μL 0.5x Qiagen Q solution, and 2 μL of DNA extract. The PCR profile for both the 15 and 7 μL reactions consisted of an initial denaturation step of 95˚C for 15 minutes followed by 30 cycles of 95˚C for 15 seconds, 46˚C for 90 seconds, 72˚C for 60 seconds with a final elongation step of 72˚C for 15 minutes. Fragment sizes were determined using an Applied

Biosystems 3730xl Genetic Analyzer (Applied Biosystems Inc., Foster City, CA) and associated GENEMAPPER 3.7 software.

1.1.4. Individual Identification

All scats verified as red fox or coyote were genotyped twice using a canid PCR multiplex (C1) consisting of five microsatellite primer pairs (FH2328, FH2054, FH2010, FH2088, and FH2001; Breen et al. 2001; Guyon et al. 2003; Moore et al. 2010). High quality coyote samples (≥3 matching loci) were genotyped up to two additional times for C1 and up to four times for a second canid multiplex (C2) depending upon how many loci amplified consistently. C2 included five primer pairs (FH2137, FH2140, FH2159, FH2096, and CXX2235; Breen et al. 2001; Guyon et al. 2003) as well as control and two sex-determining loci (DBX and DBY; Seddon 2005). High quality red fox samples (≥3 matching loci) were genotyped up to two additional times for C1 and up to four times for a fox multiplex (V2). V2 was designed after finding that red fox samples failed to amplify at certain loci contained in C2. V2 included four primer pairs (INU055, FH2140, REN105L03, and CXX2235; Breen et al. 2001; Guyon et al. 2003; Moore et al. 2010) and two sex determining loci (CF-hprt and VV-sry; Berry et al. 2007). Scats verified as lynx were genotyped four times using a lynx PCR multiplex (L1) consisting of five microsatellite primer pairs (LC106, LC109, LC110, LC111, and LC120; Carmichael et al. 2000). Scats verified as wolverine or marten were genotyped four times using a mustelid PCR multiplex (M1) consisting of five microsatellite primer pairs (MA2, MA8, MA19, GG7, and GG14; Davis and Strobeck 1998).

Similar to the species verification test, primer concentrations were maintained for multiplexes while reducing the PCR volume from 20 to 7 μL to limit costs. All 7 μL reactions consisted of 0.65 μL dH2O, 3.5 μL 1x Qiagen Master Mix, 0.7 μL 0.5x Qiagen Q solution, and 2 μL of DNA extract, along with the following primer concentrations for each multiplex: C1 = 0.22 μM FH2328, 0.18 μM FH2054, 0.2 μM FH2010, 0.22 μM FH2088, and 0.22 μM FH2001; C2 = 0.2 μM FH2137, 0.2 μM FH2140, 0.2 μM FH2159, 0.22 μM FH2096, 0.18 μM CXX2235, 0.2 μM DBX, and 0.2 μM DBY; V2 = 0.2 μM INU055, 0.2 μM FH2140, 0.2 μM REN105L03, 0.2 μM CXX2235, 0.2 μM CF-hprt, and 0.07 μM VV-sry; L1 and M1 = 0.16 μM of each primer.

The PCR profile for all individual identification multiplexes began with a denaturation step of 95°C for 15 minutes followed by a touchdown of 10 cycles at 94°C for 30 seconds, 68°C for 30 seconds (annealing), and 72°C for 45 seconds with a 1°C decrease in the annealing temperature at each cycle followed by 25 cycles at 94°C for 30 seconds, 58°C for 30 seconds, and 72°C for 45 seconds and a final elongation step of 60°C for 15 minutes. A negative control was included for each batch of PCR reactions. Allele sizes were determined using an Applied Biosystems 3730xl Genetic Analyzer (Foster City, CA, USA) and GENEMAPPER 3.7 software.

Consensus genotypes were generated for each locus by comparing replicate PCRs for each sample. Consensus required ≥2 matching replicate PCR runs for heterozygous loci (i.e., loci with different alleles) and ≥3 matching PCR runs for homozygous loci (i.e., loci with two copies of the same allele). Stricter criteria for consensus were used for homozygous loci because the probability of an allelic dropout (which can erroneously lead to a homozygous loci) is greater than the probability of a false allele (Buchan et al. 2005). To construct reliable multi-locus genotypes for each sample, consensus genotypes at ≥5 loci were required for each lynx and

mustelid, ≥6 loci were required for each fox, and ≥7 loci were required for each coyote sample. These criteria were developed after first determining the minimum number of loci necessary to assure a low probability (≤0.01) of misidentifying two first-order relatives as the same individual (Waits et al. 2001) using the software GENALEX6 (Peakall and Smouse 2006). GENALEX6 was then used to identify individuals by matching the completed samples from this study, based on the consensus multi-locus genotypes. Samples that had matching genotypes or a mismatch at only one locus were recorded as being from the same individual. Grouping samples with single mismatches avoided falsely inflating the number of individuals, because it was more likely that single locus mismatches would occur from allelic dropouts or false alleles than it was for the samples to represent two individuals (Peakall and Smouse 2006).

1.1.5.   Density Estimation

The study team used spatially explicit capture–recapture models implemented with the *R* software package SECR to estimate coyote and red fox density (Borchers and Efford 2008; Efford 2011). SECR uses the spatial records of each genotyped scat (i.e., "capture") to produce density estimates based on theoretical home range locations within the study area. SECR allows a variety of techniques to be implemented, including those used in non-invasive sampling. The study team used the "count" detector option which is designed for studies that capture animals using traps that do not restrict animal's movements, and which allows multiple individuals to be captured at one location during the same occasion, making it ideal for scat collection studies. This method assumes that captures (detections) occur at a specified location within a sampling grid cell. Therefore, a grid of 1×1-km cells was overlaid on the study area and the center of each cell was specified as the "trap location". The relatively small cell size ensured good spatial coverage of the study area and allowed mapping of individual home ranges at a scale that produced more precise density estimates than the larger 2×2-km cells used for occupancy surveys. *ArcGIS* software (version 10.1, ESRI, Redlands, CA) was used to identify the cell associated with each genotyped scat. In addition, each cell was categorized based on the level of survey effort (termed "usage" in SECR). Cells that intersected primary scat travel routes were classified as high usage, cells located within areas that were accessible during backtracking surveys or opportunistic collections were classified as low usage, and cells that were never searched or inaccessible were classified as unused.

SECR models estimate three response parameters: density (D), the probability of detection for a detector at the center of the home range (g0), and a scaling parameter (σ). Together, g0 and σ define the model for detection probability as a function of location. Models may be constructed to estimate parameters based on automatically generated predictor variables such as survey occasion (*t*; the discrete sampling event) and survey session. A survey session in SECR is defined as a set of occasions over which a population is considered closed to immigration, emigration, births, and deaths. Survey year (YEAR) was used to create two sessions. Surveys were divided into two occasions each session (T), the first spanning from January through February, and the second consisting of March and early April. Occasions were set up to divide the survey season in half, both in terms of timing and survey effort. The study team chose not to separate occasions by month (January, February, March) because January often produced fewer scat samples due to lack of daylight available for searching. The above two predictors (session and occasion) were used to create six models for each species to estimate g0 and σ while holding D constant. AICc was then used to identify the most parsimonious detection model for each

species. Models for D, the true parameter of interest, were then constructed while maintaining a fixed set of detection parameters.

When modeling D, user-defined covariates and habitat masks may be used to estimate density across a group or landscape gradient. The study team included a habitat mask using the same broad habitat categories used in occupancy models: forest, shrub, and open tundra. The habitat mask was used to create models that estimated density as a function of habitat type (HABITAT). The study team also modeled density differences between years. All models were ranked using AICc.

1.1.6.   Estimation of Survival, Recruitment, and Population Growth

Population models were constructed using program MARK (Version 8.0) to estimate population growth, apparent survival, and recruitment rates of coyote and red fox populations in the study area between 2013 and 2014. The "Pradel models including robust designs" option was used, with the fecal genotyping capture history constructed for SECR models used as the input data. These models estimated five parameters: phi (apparent survival, or the probability of surviving and remaining in the study area between 2013 and 2014), f (recruitment during the interval between 2013 and 2014 sessions), lambda (annual population growth rate from 2013 to 2014), *p* (recapture probability within years), and N (population size) in 2013 and 2014.

4.3.   Habitat Use

The study team implemented the methods as described in the Study Plan (RSP Section 10.10.4.3), with the exception of variances explained below (Section 4.3.1).

Habitat use was evaluated using two methods: (1) aerial snow track surveys and (2) ground-based snow track surveys. Helicopter surveys of carnivore tracks in the snow were conducted in 2013 on February 26, March 27, and April 19, and in 2014 on February 17 and March 25. The survey design was based on the helicopter-based track surveys that were conducted in the Project area in 1980 (Gipson et al. 1984), using the same 14 transect lines (Figure 4.3-1) to facilitate comparison of current and historical data. An experienced observer (L. Prugh) flew along the transect lines at low altitude (100–200 ft) and slow speed (20–40 mph) in a Robinson R44 helicopter. The two helicopter pilots used on different surveys (T. Cambier and R. Swisher) were experienced at furbearer track identification and also served as observers during the surveys. A global positioning system (GPS) receiver was used to record the locations of all furbearer tracks encountered. Associated data included the species that made the tracks and field descriptions of the habitat in which the tracks were found, using the same habitat categories as in the historic surveys (Gipson et al. 1984).

1.1.7.   Variances

As described in the Study 10.10 ISR, Part A, Section 4.3.1 (UAF 2014a), additional data on habitat use and species occupancy (beyond the aerial surveys described in the Study Plan) were collected during the ground-based track surveys in winter 2013. These variances continued in 2014.

Ground-based track surveys were used to examine habitat associations. Using *ArcGIS* software, a grid of 2×2-km cells was overlain on the 2013 survey area. Cells were classified as being majority shrub, forest, or open tundra/alpine habitats, based on the percentage of each vegetation type within that cell as shown on existing vegetation mapping layers produced by Ducks Unlimited in association with Bureau of Land Management and Fish and Wildlife Service (Boggs et al. 2012).

In 2013, 110 survey cells were selected from the grid using proportional sampling to select cells randomly within each habitat stratum based on the availability of each habitat type across the survey area. Of the 110 selected cells, the survey team was able to access and survey a total of 60 cells in 2013. The study team evaluated the efficacy of three survey techniques: linear transects ($n = 22$ cells), square transects ($n = 15$ cells), and remote cameras ($n = 23$ cells). Square transects were 1 km on each side and were surveyed in a single visit, whereas linear transects were 1.87 km long on average (SE = 0.063) and were surveyed repeatedly throughout the winter (range = 2–10 repeats, mean = 3.88). The linear transect method used temporal replication to estimate detection probabilities, whereas the square transects used spatial replication, which eliminated the need for return visits to the cell (MacKenzie et al. 2006). At each camera station, a motion-triggered camera (Reconyx® PC800 HyperFire Professional) was placed along a likely travel route within the cell to maximize chances of detection. Cameras were deployed for periods of 2–3 weeks and baited with a scent (commercially available skunk lure) and a bird (grouse or ptarmigan) wing as attractants. Cells were randomly assigned a survey method, which was subsequently modified based on logistical constraints, when necessary. Cells that were difficult to access repeatedly (e.g., located far from the base camp) were surveyed by square transects, whereas cells that were possible to access repeatedly were surveyed by either linear transects or cameras.

Surveys were conducted after a minimum of 24 h after the last track-obliterating snowfall to allow adequate time for tracks to accumulate, and no more than seven days after a snowfall to prevent tracks from becoming too melted out, windblown, or otherwise disrupted. All furbearer tracks encountered along linear and square ground transects were recorded, along with species identity and a GPS waypoint. Vegetation and snow characteristics were recorded every 250 m during a ground track survey, and at every point that a track was encountered. Snow characteristics consisted of depth and compaction. Depth was measured from the ground to the surface of the snow with a probe to the nearest 0.5 cm, and compaction was measured by dropping a 200 gram cylinder weight (diameter = 8.2 cm, height = 4.2 cm) from 50 cm above ground level and recording the sink depth. Vegetation (microhabitat) was recorded as the percent cover of trees and shrubs within a 10-m radius along with the dominant tree and shrub species. Tracks of hares, squirrels, voles, and ptarmigan/grouse were also counted and recorded at 250-m intervals. In this way, every cell had habitat, snow, and prey information associated with it.

In 2014, the study team re-surveyed the randomly generated cells from 2013 and also surveyed those cells that were crossed en route to the random cells. This trail network comprised the scat-collection transects. All cells in 2014 ($n = 90$) were surveyed using linear transects with temporal replication, because analysis of 2013 data indicated that results from square and linear transects had similar detection rates, and cameras provided too few photographs to be useable. This design allowed an increase in sample size and survey efficiency because the study team could more efficiently collect habitat use data from ground tracking while simultaneously collecting scat and

hair samples (Figure 4.3-2). The use of linear surveys for occupancy analyses has been well supported in literature despite spatial autocorrelation between neighboring cells, because spatial autocorrelation can be accounted for in models (e.g., Hines et al. 2010, Whittington et al. 2014). Although some bias in trail placement is inevitable due to topographic constraints, the trail network was more random than other similar surveys because it was created to access randomly selected cells. Information collected from the cameras in 2013 was not included in the final occupancy analyses because too few photographs were obtained to estimate detection and occupancy probabilities. The inclusion of ground-based track surveys improved overall knowledge of furbearer habitat use and distribution. Species occupancy is a metric of species abundance that can be used as a baseline metric to monitor population trends and habitat use (MacKenzie et al. 2006, Clare et al. 2015).

The study team had planned to conduct three aerial surveys of furbearer tracks in 2014, but unusually poor snow conditions (infrequent snowfall and high temperatures leading to melted-out tracks) precluded a third survey.

4.4.   Statistical Analyses and Data Interpretation

AEA implemented the methods as described in the Study Plan (RSP Section 10.10.4.4), with the exception of variances explained below (Section 4.4.1).

1.1.8.   Variances

The Study Plan did not propose to include occupancy modeling in the study design; rather, the study team included this additional analytical element during final project planning. Ground-based track data were used to assess furbearer habitat associations using occupancy models. Single-season occupancy models were used to estimate occurrence (use) probabilities of the target mesocarnivore species. The single-season occupancy model provides estimates of two response parameters: probability of site occupancy (ψ), and detection probability (p). Occupancy is the probability that the species occupied each survey cell during the survey period (e.g., ψ = 0.6 indicates that 60 percent of the survey cells are occupied, or within the home ranges, of individuals of a given species). Because the survey cells were smaller than the home range sizes of all target species except marten (Gipson et al. 1984), the assumption of closure was violated during this study (i.e., animals with a given survey cell inside their home range were often outside the cell). Reported occupancy probabilities are therefore best interpreted as probabilities of use (i.e., the probability that the survey cell is used by a given species), as is recommended in such cases (MacKenzie et al. 2002). Detection is the probability that an individual of the target species is detected (i.e., crosses a transect and is identified during a survey) given that it uses the 4-km^2 survey cell as part of its home range (MacKenzie et al. 2002). Although two seasons of data were included in the analysis, a single-season model framework was used and study year was included as a covariate in the candidate model set because the primary interest was in determining factors affecting use, not occupancy dynamics that would require estimation of colonization and extinction parameters. Similarly, species was included as a covariate, which allowed the study team to combine data from all species into one database and develop models to examine species-specific as well as guild-wide patterns. All analyses were performed in program *R* version 3.1.0 (*R* Development Core Team 2014) using packages unmarked (Fiske and Chandler 2011) and AICcmodavg (Mazerolle 2015).

Two survey-specific covariates and seven site-specific covariates were included in the models using a logit link function. Survey covariates were (1) number of days since the last snowfall (DSLS) and (2) total distance (km) surveyed within a sample cell (DIST). Site-specific covariates were (1) study year (YEAR); (2) survey method (METHOD); (3) habitat type (HABITAT); (4) species (SPECIES); (5) average snow depth over all survey occasions (DEPTH); (6) average snow compaction over all survey occasions (COMPACTION); and (7) average total combined prey species abundance per km surveyed, adjusted for days since last snowfall (PREY). Average snow depth, snow compaction, and prey abundance within each habitat type were calculated, and a Pearson's correlation matrix was used to test for correlation between continuous covariates. These continuous covariates were standardized before inclusion in the occupancy models.

The study team used a three-step process to develop a finalized candidate model set that was both biologically relevant and analytically feasible. Models were ranked based on AICc and QAICc (Burnham and Anderson 2002). First, models were constructed to estimate p while holding ψ constant (Schuette et al. 2013), using all combinations of DSLS, DIST, METHOD, YEAR, and SPECIES. The top-ranking model contained all of the available covariates, therefore the study team used the most parameterized model in the next steps to account for all relevant predictors of p.

In the second step, all combinations of HABITAT, SPECIES, YEAR, DEPTH, COMPACTION, and PREY were used to build models that estimated ψ while p was modeled as a function of covariates from step one. This resulted in a full candidate set of 64 additive models. This full set of models was then used to calculate summed individual covariate weights to assess the relative importance of each predictive covariate on guild-wide occupancy. Covariates with weights > 50 percent were considered important (Burnham and Anderson 2002). In step three, the study team used a Δ AICc $\leq$ 2 cutoff to reduce the full candidate set down to the top four models. This model set was supplemented with five interaction models and a null model, for a final candidate model set of 10 models. Each of these five models contained an interaction term between SPECIES and one of the other five covariates used to model occupancy. These models were developed to directly assess the influence of snow, habitat type, prey, and study year on furbearer occupancy. The top-ranking model was used to produce estimates of furbearer species occupancy for the study area.

These variances benefit the study by increasing the amount of data available to describe furbearer populations and habitat use. These additions (ground-based snow track surveys and occupancy models) were carried out during both study seasons and will help to achieve study objective 5 by providing additional data to estimate habitat use and study objectives 1–3 by estimating a new population level parameter to describe current furbearer population status.

5.   RESULTS

Data developed in support of this study are available for download in the following file at:

http://gis.suhydro.org/SIR/10-Wildlife/10.10-Terrestrial_Furbearer/

See Table 5-1 for details.

5.1.   Sample Collection

Samples of scats or hairs were collected from all four of the targeted furbearer species and additional non-target species samples were also collected opportunistically (Figure 5.1-1). The study team collected 138 scats in 2013 and 305 scats in 2014 (Table 5.1-1). An increase in samples during the 2014 field season was attributed to a more centrally located field site which allowed the study team more time surveying and less time traveling to and from field camp locations. In addition, the study team was more efficient at traveling throughout the survey area during the second field season and was aided by relatively infrequent snowfall events which kept scats exposed for a longer period of time. Fewer samples of lynx hair were collected from hair snags than expected during both winters, but backtracking supplemented the total sample size in 2014 (Table 5.1-2). Hair samples were poorer in quality than described in a lynx study by McKelvey et al. (2006) in the Rockies but similar to the low success reported by Mumma et al. (2015) in Quebec. Marten hair tubes were deployed for trial purposes only in 2013, and sample collection was therefore limited to 2014 (Table 5.1-2). The functionality of the marten tube traps was successfully tested during the 2013 season, but the lack of a complete survey provided the study team with little information regarding the quality of hair samples that would be produced from this method. The 2014 samples that were collected were often small hair samples that were broken off rather than pulled from the root.

Surveys of prey species during the summer field seasons indicated that snowshoe hare and vole densities varied by study year and survey location. Snowshoe hare pellet surveys were conducted in the Jay, Watana, Butte, Deadman, Tsusena, Seattle, and Brushkana creek drainages. Several areas of high-density use were located (e.g., the Jay Creek shrub grid, the Deadman Creek forest grid, and the Oshetna Creek forest grid; Table 5.1-3), as well as areas with little or no hare sign. Across habitat types, a paired t-test revealed that estimated snowshoe hare densities were marginally greater in 2013 (mean = 0.31 hares/ha, variance = 0.19) than in 2014 (mean = 0.19 hares/ha, variance = 0.06) in the areas surveyed; (t(14) = 2.27; p = .04). No significant differences were found in snowshoe hare density between forested areas and shrub areas during the 2013 (t(6) = 1.80; p = 0.12) or 2014 (t(6) = 2.18; p = 0.07) seasons (Table 5.1-3).

Vole trapping during 2012 consisted of only three trapping grids (two forest and one meadow), only one of which (Watana Creek meadow) was resampled during 2013 and 2014. A total of 8 voles (seven red-backed voles and one meadow vole) were captured in 2012 for an average vole density of 11.1 voles/ha in forests, 3.2 voles/ha in meadows, and 12.6 voles/ha overall. Across habitat types and species, a paired t-test revealed that overall estimated vole densities were significantly greater in 2014 (mean = 21.14 voles/ha, variance = 173.61) than in 2013 (mean = 2.67 voles/ha, variance = 4.74) in the areas surveyed (t(12) = –4.97; p = 0.0003). No significant differences in vole density occurred between forested areas and shrub areas during the 2013 (t(6) = 1.0; p = .36) or 2014 (t(5) = 2.57; p = .73) seasons (Table 5.1-4). Survey areas included the Jay, Watana, Butte, Deadman, Tsusena, and Seattle creek drainages and Deadman Mountain. The vole species captured included red-backed vole (*Myodes rutilus*), meadow vole/tundra vole (*Microtus pennsylvanicus/Microtus oeconomus*, which are not readily distinguishable in the hand), and singing vole (*Microtus miurus*). Red-backed voles were the most commonly trapped species in both survey years, making up 79 percent of all captures in 2013 and 77 percent of all

captures in 2014. Red-backed voles were found in both forest and meadow trapping locations. Both meadow voles and singing voles were caught primarily in meadow locations, with the only exception being a single meadow/tundra vole captured in the Tsusena Creek Forest location in 2014. Meadow/tundra voles represented 14 percent of all captures in 2013 and 15 percent in 2014. Singing voles were least common and made up the remaining 7 percent and 8 percent of all captures in 2013 and 2014, respectively. Due to the consistency in the numbers of each species trapped over the two-year study, these percentages are likely accurate estimates of the relative proportion of each species in the survey area. Although trap-related mortality can be a common occurrence during small mammal surveys, precautions were taken to minimize this risk by including cotton material for nesting, supplying ample food, and checking traps as often as possible. No mortalities of captured voles occurred during the 2013 sampling. An unusual cold spell during the 2014 trapping season resulted in 8 (9 percent) mortalities of captured voles.

5.2. Habitat Use and Furbearer Occupancy

1.1.9. Aerial Surveys

A total of 1,360 sets of tracks from 12 furbearer species were recorded during the five helicopter surveys in 2013 and 2014, 865 of which were of the four target furbearer species (Table 5.2-1). Note that these track counts were indices and did not represent the number of individuals, because tracks from individuals were likely counted multiple times if animals crossed transects repeatedly. The species with the highest track counts (in descending order) were marten, weasels (*Mustela erminea* and *M. nivalis*), wolverine (*Gulo gulo*), lynx, and red fox (Table 5.2-1). Totals of 570 marten, 161 lynx, 113 red fox, and 21 coyote tracks were recorded over both years (Table 5.2-1). Marten track density was greatest at Deadman Creek (Transect 6; Figure 5.2-1), whereas lynx, fox, and coyote tracks were more abundant along transects from Watana Creek upstream to the Oshetna River (Transects 8–14; Figure 5.2-1).

Tracks of mustelids and lynx were more abundant in 2013 than in 2014, whereas tracks of coyotes and foxes were more abundant in 2014 than in 2013 (Table 5.2-2). No coyote tracks were seen on the aerial survey transects in 2013, but 21 sets of coyote tracks were recorded in 2014. On average, the number of tracks counted per survey was higher in 2013 than in 2014, in terms of both total track counts and tracks per DSLS (Table 5.2-2). In particular, weasel tracks were detected less frequently in 2014, likely due to poor snow conditions because their tracks were smaller and may not have made visible prints in areas that had melted and frozen. Likewise, several tracks were categorized as "unknown furbearer" due to poor snow conditions in 2014, whereas in 2013 the study team was able to identify all tracks (Table 5.2-2).

Marten and lynx tracks occurred primarily in forested habitat types: 88 percent of marten tracks and 82 percent of lynx tracks were detected in forests (Figure 5.2-2). Marten tracks were most common in black spruce forest (>60 percent cover) and black spruce woodland (10–60 percent cover), whereas lynx tracks were most common in white spruce forest and black spruce forest (>60 percent cover, Table 5.2-3). Coyote tracks were found on the frozen Susitna River and in tall shrub habitat (Table 5.2-3). Red fox tracks were found in a wide variety of habitat types, most commonly in spruce, alder, and alpine habitat types (Figure 5.2-2).

1.1.10. Ground Surveys

Furbearer track detections along snowmachine survey transects were used to create a presence/absence record of each target species across the survey areas to model probabilities of occupancy and detection. The top-ranked detection model included all covariates hypothesized to affect detection (model *p*(*dist+dsls+species+method+year*); see Section 4.4 above). Detection probabilities ranged from a low of 0.12 ± 0.04 for coyotes in 2014 to a high of 0.40 ± 0.08 for marten in 2013, and detection probabilities were lower for all species in 2014 than in 2013 (Figure 5.2-3). Occupancy probabilities (*model: ψ*(*species*year*) *p*(*dist+dsls+species+method +year*)) for the target species in the survey area during the winters of 2013 and 2014 ranged from 0.28 ± 0.07 (marten) to 0.84 ± 0.37 (lynx). Most species had similar occupancy probabilities between years (Table 5.2-4), with the exception of lynx. Lynx occupancy probability in 2014 (0.84 ± 0.37) was dramatically higher than in 2013 (0.35 ± 0.12). Cell-specific occupancy probabilities across both years were generated for all encountered furbearers (model: *ψ*(*species*) *p*(*dist+dsls+species+method+year*)) to depict the spatial variability in furbearer occupancy patterns throughout the survey area (Figure 5.2-4). The models used to produce these overall and year-specific occupancy probabilities did not take into account any additional predictive covariates that may have explained furbearer occupancy probabilities (e.g., snow, habitat, prey), and therefore these data should be used as a baseline to look at changes in overall species distribution and space use over time.

Considering all species together, summing individual covariate weights showed that habitat and snow conditions had the most influence on furbearer occupancy, whereas prey abundance (indexed by track counts of hares, squirrels, voles, and ptarmigan/grouse; see Methods) was a poor predictor (Table 5.2-5). Combining species into one analysis allowed the study team to assess influential covariates on the occupancy patterns of all furbearers in the survey area. Individual species occupancy estimates were evaluated as a function of snow conditions, habitat type, prey abundance, and study year in those models that included interaction terms with the SPECIES covariate (Table 5.2-6). The top-ranking model included such an interaction term, which suggests that occupancy probability varied by furbearer species and was most strongly affected by the level of snow compaction (Table 5.2-6). Predictions from this top model showed that coyotes had the highest winter occupancy and marten had the lowest winter occupancy when evaluating species at the mean snow compaction level (Figure 5.2-5). This model also showed that occupancy probabilities of coyotes and red foxes were negatively affected by fluffy snow ($\beta = -0.99 \pm 0.55$, and 0.51 ± 0.62, respectively), whereas lynx and marten had higher occupancy probabilities in areas of fluffy snow ($\beta = 1.73 \pm 0.70$, and 3.03 ± 0.87, respectively).

Based on the highest-ranking model containing the habitat covariate *ψ* (*habitat+compaction+ species*), the probability of occupancy (logit-transformed model coefficients $\pm$ SE) was highest in forest ($\beta = 5.65 \pm 1.955$) and lowest in open tundra landscapes ($\beta = 1.41 \pm 2.13$) across all species. Species-specific habitat use was described by the model *ψ* (*habitat * species*) and showed that marten and lynx use was greatest in forested areas, whereas canid use patterns were more evenly distributed across habitat types (red fox) or were concentrated in shrub habitats (coyotes) (Figure 5.2-6). The support for these habitat models was not strong relative to those models that included snow compaction (Table 5.2-6); however, the estimates from this model provide habitat use information for each target furbearer species.

5.3. Genetic and Statistical Analyses

The study team obtained reference tissue samples from specimens of known species identity archived at the University of Alaska Museum of the North to screen a set of microsatellite DNA markers. The study team then optimized those DNA markers and began DNA fingerprinting of scats. The study team used the reference tissues and scats to develop a species identification protocol, which was more difficult to develop than expected due to the large suite of carnivore species present in the area. This difficulty delayed large-scale processing of scats and hairs because species identity needed to be determined for each species prior to individual identification.

Although hair samples were successfully collected in the field, DNA extractions from lynx and marten hair samples had remarkably low success, so true density estimates for these species were impossible to obtain. Of the 84 hair samples collected from marten tubes, lynx hair snags, or backtracking, species identification was successful for only 9 samples: 6 wolverine, 2 lynx, and 1 red fox. Of those 9 samples, only 1 wolverine hair sample successfully produced an individual ID. The failure to extract genetic material from the hair samples was likely a result of poor sample quality. Several of the "samples" collected from lynx hair snags appeared to be fibers from the carpet pads used to construct the hair snags, and most of the hair samples collected from marten tubes likely were vole hairs, based on the high degree of vole activity observed at marten hair traps by the survey team. Many of the samples were of single hairs, often underfur, rather than high-quality guard hairs with roots attached.

In contrast, DNA extraction and amplification of canid scats and scats from other carnivore species was successful. Of the 448 total scats collected, molecular species identification was successful for 383 scats (85 percent success). Of these, 231 scats were positively identified as red fox (Table 5.1-1), and the study team obtained reliable multi-locus genotypes from 137 of those scats (59 percent), representing 56 individuals which were used for density estimation. Of the 56 identified foxes, the study team identified the sex of 52, 26 of which were male and 26 of which were female. Of the 73 scats positively identified as coyote (Table 5.1-1), reliable multi-locus genotypes were obtained from 50 scats (68 percent) representing 14 individuals, 7 males and 7 females.

Although lynx, marten, and wolverine do not mark trails with feces as often as canids do, the study team collected scats opportunistically from these species and attempted to identify individuals. All 8 lynx scats were successfully genotyped, representing 5 individuals. Likewise, all 3 marten scats were successfully genotyped, representing 3 individuals. Of the 35 collected wolverine scats, 17 were successfully genotyped, representing 9 individuals. The wolverine that was identified from a hair sample at a lynx hair station (Gulo05) was also identified from a scat sample.

A set of six SECR models was created for each canid species to determine population density by year and habitat type, and models were ranked based on AICc (Table 5.3-1). Model-averaged density estimates showed that canid densities did not vary greatly between years and that red fox densities were greater than coyote densities across all habitat types (Figure 5.3-1). Habitat type seemed to have a minimal influence on coyote density, whereas red foxes had significantly lower densities in forested habitats than in shrub or open tundra areas (Figure 5.3-1). Overall density

estimates (D ± SE animals per 1,000 km^2) for foxes and coyotes were produced by evaluating the top ranking model that held the D parameter constant (Table 5.3-1). Red fox density (15.9 ± 2.3 foxes per 1,000 km^2) was roughly four times greater than coyote density (3.8 ± 0.9 coyotes per 1,000 km^2).

Pradel open mark–recapture models were used to examine changes in fox and coyote populations among years. Estimates of population growth (lambda, λ) indicated the red fox population grew by 21 percent (λ = 1.21, SE = 0.28), from 49 foxes (95 percent CI = 37–77) in 2013 to 60 foxes (95 percent CI = 46–92) in 2014 in the study area (Table 5.3-2). In contrast, the coyote population was stable (λ = 1.04, SE = 0.37), estimated to be 11 coyotes (95 percent CI = 9–24) in 2013 and 12 coyotes (95 percent CI = 10–22) in 2014. The probability of recapturing a coyote (i.e., collecting scats from an individual during multiple survey occasions within a year) was higher (p = 0.43, SE = 0.11) than the probability of recapturing a fox (p = 0.24, SE = 0.05), and likewise the apparent survival rate (i.e., the probability of surviving and remaining in the study area between years) of coyotes was higher than foxes (phi = 0.61 and 0.38, respectively). In contrast, recruitment of coyotes between years was lower than recruitment of foxes (f = 0.43 and 0.83, respectively).

6.   DISCUSSION

6.1.   Coyote

Coyote density was extremely low (3.8 coyotes per 1,000 km^2) and changed very little between survey years. The study was conducted during the low phase of the snowshoe hare population cycle (Krebs et al. 2013). Hares, which are the primary prey of coyotes in northern ecosystems (O'Donoghue et al. 1998), were at low densities during both years. This factor may explain why coyote densities were low and stable during the study period. No coyotes were detected during aerial surveys along the Susitna River in 2013, although ground-based track surveys indicated that coyotes were relatively common in other nearby drainages. Ground-based track surveys indicated that coyotes were found primarily in areas of compact, shallow snow. Snow depth along the Susitna increases substantially downsteam of the Oshetna River, which may prevent coyotes from routinely using areas within the inundation zone. Snow conditions were unusually shallow and compact due to high temperatures in the winter of 2014, allowing coyotes to expand into this area, and the study team detected coyotes during aerial surveys in 2014. These results are generally consistent with findings from aerial and ground-based furbearer surveys conducted in this area by Gipson et al. (1982), who did not detect any signs of coyotes upstream of Devils Canyon but noted the presence of coyotes in surrounding areas. As in this study, the work conducted in this area in the 1980s occurred during the low phase of the snowshoe hare cycle, facilitating comparisons. Results from the present study indicate that the distribution and abundance of coyotes may have increased slightly since the 1980s, but coyotes remain relatively rare within the inundation zone. Two other studies have estimated coyote densities in Alaska: one study was conducted during the peak and decline phase of the hare cycle in the central Alaska Range during 1999–2002 (Prugh et al. 2005) and the other study was conducted concurrently with the present study (2013–2014) in DNPP (Pozzanghera 2015). Densities of 14–25 coyotes per 1,000 km^2 reported by Prugh et al. (2005) and 19.7 coyotes per 1,000 km^2

reported by Pozzanghera (2015) both were substantially higher than density estimates from the current study.

Due to the scarcity of coyote data from the aerial track surveys along the Susitna River and the relatively low sample size of genotyped coyote scats ($n = 50$), information about coyote habitat use is best obtained from the ground-based track surveys. Coyote occupancy probabilities were highest in shrub habitats, moderate in forested habitats, and low in the tundra (Fig 5.2-5).

6.2. Red Fox

Results supported the study team's expectation that red foxes would occur in higher densities across the study area than would coyotes. Red foxes are smaller-bodied, require less overall prey biomass, and can therefore maintain smaller home ranges than coyotes (Peters 1986, Sargeant et al. 1987, Harrison et al. 1989). Indeed, the spatially explicit density estimates for red foxes were four times higher than estimates for coyotes (15.9 foxes vs. 3.8 coyotes per 1,000 km^2). While consistent with predictions, these estimates were dramatically lower than densities reported from other parts of these species' distributions, which are as high as 910 foxes and 710 coyotes per 1,000 km^2 (Hein and Andelt 1995, Henke and Bryant 1999a, Heydon et al. 2000, Sarmento et al. 2009). Low densities may be partly related to low snowshoe hare numbers during this study, but large red fox home ranges reported in other northern boreal regions may be an indication that these mesocarnivores continually persist at low densities in boreal ecosystems (Jones and Theberge 1982).

Fox density in this study was remarkably similar to the estimate from this area in the 1980s, which was 12.2 foxes per 1,000 km^2 (Gipson et al. 1982). Fecal genotyping analyses indicated that fox densities increased from 2013 to 2014, and likewise the number of fox tracks counted per survey nearly doubled from 2013 to 2014 despite poor tracking conditions in 2014. Aerial track surveys along the Susitna River indicated that fox tracks were distributed among a variety of habitat types and were most common along Transects 7–13 between Watana Creek and the Oshetna River, which is similar to findings from aerial surveys conducted along the same transects in 1980 (Gipson et al. 1984). These findings indicate the current distribution and abundance of red foxes in this area closely resembles patterns of fox distribution and abundance in the same area three decades ago.

Gipson et al. (1982) reported that red foxes primarily used higher elevation tundra and shrub habitats. Likewise, red fox occupancy probabilities from ground-based track surveys and spatially-explicit density estimates from fecal genotyping indicated that red foxes were found primarily in shrub or open tundra areas. Aerial track surveys indicated slightly higher use of forested areas than other methods did, but this discrepancy is likely due to the predominance of low-elevation forested habitat available in areas surveyed by aerial transects compared to the areas surveyed on the ground. This discrepancy between aerial and ground-based habitat use patterns was also noted by Gipson et al. (1982). Track analyses may underrepresent furbearer use of open tundra areas due to windy conditions easily covering up tracks, but the correspondence of fox habitat-use patterns estimated from ground-based snow tracks and by fecal genotyping suggested this bias was likely minimal. Future studies using these noninvasive methods will be able to produce estimates that are directly comparable to those found during this study and assess any change in population size and habitat influences on these furbearers.

6.3.   Marten

Although hair samples from lynx and marten were collected during the winter field seasons, the study team was unable to extract DNA successfully from most of those samples; therefore, population density estimates could not be generated for these two species. DNA extraction and amplification from hair is most successful when the roots of hair follicles remain attached to the samples and multiple hairs are available with each sample (Foran et al. 1997). Large guard hairs that have been pulled from an animal, and not broken off, are the ideal sample. The samples collected during this study were generally of single hairs or patches of underfur, fluffy hair that is shed more easily and does not contain a hair follicle root. These samples contained insufficient amounts of DNA and therefore could not successfully be identified to the species or individual levels. Previous studies that have successfully demonstrated the use of lynx and marten hair-snag snares have used this method during summer or fall months (McKelvey et al. 1999, McDaniel et al. 2000, Pauli et al. 2008, Williams et al. 2009). These methods have not been well-tested during winter months, and it has been noted that difficulties of winter hair-snaring for lynx may arise from changes in snow depths leading to inconsistencies in the rub pad height on trees, and inability of lure to properly volatilize in extreme cold (Long et al. 2012). In addition, differences exist in the density and structure of furbearer hair during winter and summer months (Maurel et al. 1986, Korhonen 1988), and quality guard hairs may be more likely to be shed onto a rub pad during late summer than during the depths of winter when lynx and marten coats are at their densest and strongest.

Although density estimates could not be produced with the methods used, the addition of occupancy analysis to this study may provide a potential index of density (Clare et al. 2015), and may be referred to over time to assess changes in population size. Occupancy analysis from snow-tracking surveys during the winters of 2013 and 2014 produced useful data on furbearer abundance and habitat use that supplemented helicopter survey results and non-invasive genetic data. Marten detectability during the 2013 and 2014 survey seasons was low compared with studies from other parts of the marten range (Moriarty et al. 2011). Likewise, marten occupancy estimates in the survey area were low ($\psi = 0.33 \pm .09$; based on the top-ranking model) compared with other findings (Smith et al. 2007, Moriarty et al. 2011), suggesting that this species exists at low densities compared with other regions. Marten had lower occupancy probabilities than the other species during ground-based track surveys (Figure 5.2-3) and were mainly restricted to forested areas (Figure 5.2-5).

Despite their relatively low occupancy probabilities in the large area surveyed on the ground in 2013 and 2014, marten tracks were far more abundant than were tracks of the other three furbearer species during aerial surveys of the inundation zone (Figure 5.2-1). Results from the 1980s likewise indicated that marten used forested/woodland areas more frequently than such habitat types occurred across the study region (Buskirk 1983) and that marten have relatively small home ranges (Buskirk and McDonald 1989) that restrict them to a small portion of the survey area. The abundance of marten along the 14 aerial transects showed similar patterns as those reported along the same transects in 1980 (Buskirk 1983), in that marten tracks were most common in black spruce forests and along Transects 4–6, peaking near Deadman Creek, and with another increase in abundance along Transect 14, upstream of the Oshetna River. As in the 1980 survey, marten tracks were more abundant than were tracks of other furbearers, but

numbers cannot be compared directly because the 1980 survey was conducted only once and did not indicate the number of days since the last snowfall.

Although the number of marten tracks counted per survey decreased substantially from 2013 to 2014, a pattern also seen in weasel and wolverine track data, this pattern may have been due to poor tracking conditions rather than an actual decline in abundance. Alternatively, the compact snow conditions in 2014 may have restricted marten movements relative to 2013 (which was characterized by frequent snowfalls and cold temperatures), based on results from ground-based occupancy surveys. These surveys indicated that marten occurred most commonly in areas of deep, fluffy snow, whereas coyotes and foxes were more common in areas of shallower, more compact snow. Thus, marten movements may have been more restricted in 2014 to avoid encounters with canids. Climate change will likely increase snow compaction due to higher temperatures (Olsson 2009), which may negatively affect marten in the future and increase their vulnerability to other disturbances such as habitat loss and human activity. The difference in snow conditions between 2013 and 2014 may explain why marten had greater occupancy in 2013 than 2014, while the opposite pattern was seen for other target species. Indeed, marten were the only study species to show increased occupancy probabilities in areas of deep snow.

6.4. Lynx

Population estimates of lynx could not be generated for this species, as is explained above under Section 6.3. Lynx detectability was consistent with previous findings from other similar studies (Walpole et al. 2012), and lynx occupancy in the survey area ($\psi = 0.46 \pm 0.11$; based on top ranking model) was very similar to occupancy estimates from other northern studies (Walpole et al. 2012). Lynx abundance may have increased in the surveyed area since the 1980s. Gipson et al. (1982) did not record lynx tracks during their aerial track surveys and they indicated that lynx sign was uncommon. In contrast, lynx were widely distributed throughout the area surveyed in both years of this study, including within the inundation zone. During aerial surveys, lynx tracks were rarely detected on transects downstream of Deadman Creek (Transects 1–7) and were most abundant at the confluences of Watana Creek (Transect 8) and the Oshetna River (Transect 13). Track counts per survey decreased from 2013 to 2014, but it is unclear whether this pattern represented a decline in abundance or resulted from poor tracking conditions in the second year.

Both ground-based track surveys and aerial track surveys indicated that lynx were most commonly found in forests and rarely found in tundra, a finding that is similar to other studies (Ruggiero et al. 2000; Zielinski et al. 2005). Ground-based surveys also showed that lynx commonly used shrub habitats, whereas aerial surveys rarely recorded lynx in areas with shrubs. This discrepancy is likely due to the relatively low availability of shrubs in areas surveyed aerially compared to ground surveys. Ground-based surveys indicate how the occupancy probability of each species varies with respect to predictor variables such as habitat and therefore automatically take availability into account. Results from occupancy models are therefore a better reflection of habitat selection than are the results from the aerial surveys.

6.5. Interspecies Comparisons

Occupancy estimates for coyotes and red foxes were consistent between years and indicated that foxes had similar occupancy probabilities to coyotes, yet density analyses showed a clear

difference in population size between these canids. Many studies have shown spatial partitioning of sympatric coyotes and red foxes (Sargeant et al. 1987, Harrison et al. 1989, Gosselink et al. 2003). Taking into account canid density estimates, it is plausible that coyotes exist at a lower density yet require larger home ranges and end up using an equal amount of the landscape as the more numerous red foxes across the survey area. Using both density and occupancy provides a more complete picture of the current status of these populations than using either metric alone.

The study team assessed a number of predictive covariates on occupancy and evaluated those covariates at the guild-wide level (combining all study species into one analysis) and at the species-specific level. When looking at guild-wide responses, habitat type was a strongly influential predictor of species occupancy, along with snow compaction. When assessing species independently, habitat did not have the same predictive strength. This difference is likely an indication that the target species' occupancy probabilities were influenced by the different habitat types in similar ways. For example, all species tended to have higher occupancy probabilities in forest and shrub habitats than in open tundra habitats. These results are consistent with previous findings, especially those that describe lynx and mustelid habitat use (e.g., Ruggiero et al. 1994, Kilström 2004, Squires et al. 2010). The study team found highest numbers of winter prey tracks in forest habitats, which could explain higher levels of mesocarnivore use in forested areas. Habitat use by each species did not vary among years, based on results of both the ground-based and the aerial track surveys.

Snow conditions, especially snow compaction, provided strong predictive power of furbearer occupancy at both the guild-wide and species-specific level. This finding indicates that snow conditions may be driving the distribution of furbearers in the survey areas. Coyotes and red foxes occupied areas of more densely compacted snow, whereas marten and lynx occupied areas of less compacted, fluffy snow. Densely compacted snow can be created along snowmachine trails, areas that canids have been shown to frequent in previous studies (Kamler and Gipson 2000, Perrine 2005). Lynx and marten may avoid those areas to avoid competitive interaction with canids or simply because they are better adapted to deep and fluffy snow conditions (Raine 1983, Ruggiero et al. 2000, Zielinski et al. 2005). Because the study team was able to collect detailed information about snow depth and compaction during ground-based occupancy surveys, this method provided information about the response of these furbearer species to climate-related factors that would not have been possible to examine using aerial surveys and density estimates alone. Snow conditions strongly affected the distribution of furbearers, with each species preferring different conditions. The study team therefore expects climate change to strongly affect the distribution of each furbearer species in the future, but each species will likely be affected differently. Marten and lynx may be negatively affected, because they preferred fluffy snow, and warming temperatures should increase snow compaction (Chapin et al. 2014), thus favoring the canids. The study team encountered dramatically different snow conditions during the two primary winter survey seasons. During 2013, interior and south-central Alaska experienced frequent snowfalls with a late spring that resulted in an unusually long-lasting snowpack and greater snow coverage into the end of the survey season. In contrast, winter weather in 2014 was much warmer than average and snowfall events were infrequent (Alaska Climate Research Center 2015).

6.6.   Small Mammal Abundance

Hare and vole abundance were generally low during both 2013 and 2014, although vole numbers increased markedly between years. A strong increase in the vole populations could be extremely important for furbearer species in the study area. Indeed, the red fox population increased substantially between 2013 and 2014, likely in response to the increase in voles, which are a primary food source for foxes (Sivy 2015). Although this study has only produced three years of prey abundance data, the nearly eight-fold increase in vole abundance from 2013 to 2014 highlights the volatility of that prey resource. Similarly, results from ongoing studies in nearby Denali National Park have documented large annual fluctuations in vole abundance (Oakley et al. 1999, Rexstad and Kielland 2006). In contrast to the increase in vole abundance, results of this study indicated that hare abundance decreased from 2013 to 2014. Lower abundance of hares in 2014 could be a result of the shallow snow depth and lack of spring snow cover (as observed by the study team during the end of the 2014 survey season) for camouflage. Hares may be more vulnerable to predation during seasonal transition periods when their white winter coat stands out against dark, snowless landscapes (Mills et al. 2013).

7.   CONCLUSIONS

From 2012 to 2014, AEA completed an investigation of the abundance and habitat use of terrestrial furbearers, including population estimates and occupancy analyses for coyotes and red foxes, occupancy analyses for marten and lynx, and an assessment of prey abundance for snowshoe hares and voles. The field work, data collection, data analysis, and reporting for this Terrestrial Furbearer Abundance and Habitat Use Study successfully met three of the five study objectives in the FERC-approved Study Plan. Although the two objectives pertaining to population estimates of marten and lynx could not be fulfilled due to laboratory analytical problems, sufficient data on habitat use, occupancy, and abundance were obtained to be able to assess Project impacts and develop PME measures. The results of the Terrestrial Furbearer Abundance and Habitat Use Study are reported herein and earlier by AEA (UAF 2014a). With this report, AEA has now completed the Terrestrial Furbearer Abundance and Habitat Use Study.

Results of this study highlight the importance of forested habitats to mesocarnivore populations, especially marten. Of the four species examined, marten were the most restricted to forests, whereas the canids were least dependent on forest habitats. Coyote and red fox population estimates indicated that these populations were fairly stable and at relatively low densities, with substantially higher densities of red foxes compared with coyotes. Studies conducted in the 1980s reported that coyotes and lynx were rare. These species may therefore have increased in distribution or abundance during the past three decades. Although lynx and marten population densities were not determined using the genetic capture–recapture study design originally proposed, the spatially explicit ground-based occupancy surveys that were added to this study provided useful information on the habitat use, current distribution, and relative abundance of these species.

8. LITERATURE CITED

Alaska Climate Research Center. 2015. Season/Annual Summaries—Fairbanks. September 15, 2015. Available online: http://akclimate.org/.

Berry, O., S. D. Sarre, L. Farrington, and N. Aitken. 2007. Faecal DNA detection of invasive species: the case of feral foxes in Tasmania. *Wildlife Research* 34: 1–7.

Breen, M., S. Jouquand, C. Renier, C. S. Mellersh, C. Hitte, N. G. Holmes, A. Cheron, N. Suter, F. Vignaux, A. E. Bristow, C. Priat, E. McCann, C. Andre, S. Boundy, P. Gitsham, R. Thomas, W. L. Bridge, H. F. Spriggs, E. J. Ryder, A. Curson, J. Sampson, E. A. Ostrander, M. M. Binns, and F. Galibert. 2001. Chromosome-specific single-locus FISH probes allow anchorage of an 1800-marker integrated radiation–hybrid/linkage map of the domestic dog genome to all chromosomes. *Genome Research* 11: 1,784–1,795.

Boggs, K., T. V. Boucher, T. T. Kuo, D. Fehringer, and S. Guyer. 2012. Vegetation map and classification: northern, western and interior Alaska. Alaska Natural Heritage Program, University of Alaska Anchorage. 88 pp.

Borchers, D. L., and M. G. Efford. 2008. Spatially explicit maximum-likelihood methods for capture–recapture studies. *Biometrics* 6: 377–385.

Buchan, J. C., E. A. Archie, R. C. Van Horn, C. J. Moss, and S. C. Alberts. 2005. Locus effects and sources of error in noninvasive genotyping. *Molecular Ecology Notes* 5: 680–683.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: A Practical Information–Theoretic Approach, 2nd ed. Springer–Verlag, New York, NY.

Buskirk, S. W. 1983. The ecology of marten in southcentral Alaska. Ph.D. dissertation, University of Alaska Fairbanks.

Buskirk, S. W., and L. L. McDonald. 1989. Analysis of variability in home-range size of the American marten. *Journal of Wildlife Management* 53: 997–1,004.

Carmichael, L. E., W. Clark, and C. Strobeck. 2000. Development and characterization of microsatellite loci from lynx (*Lynx canadensis*) and their use in other felids. *Molecular Ecology* 9: 2,197–2,199.

Chapin, F. S., III, S. F. Trainor, P. Cochran, H. Huntington, C. Markon, M. McCammon, A. D. McGuire, and M. Serreze. 2014. Climate Change Impacts in the United States: The Third National Climate Assessment. United States Global Change Research Program, Washington, DC.

Clare, J. D. J., E. M. Anderson, and D. M. MacFarland. 2015. Predicting bobcat abundance at a landscape scale and evaluating occupancy as a density index in central Wisconsin. *Journal of Wildlife Management* 79: 469–480.

Dalen, L., A. Gotherstrom, and A. Angerbjorn. 2004. Identifying species from pieces of faeces. *Conservation Genetics* 5: 109–111.

Davis, C. S., and C. Strobeck. 1998. Isolation, variability, and cross-species amplification of polymorphic microsatellite loci in the family Mustelidae. *Molecular Ecology* 7: 1,776-1,778.

De Barba, M., J. R. Adams, C. S. Goldberg, C. R. Stansbury, D. Arias, R. Cisneros, and L. P. Waits. 2014. Molecular species identification for multiple carnivores. *Conservation Genetics* 6: 821–824.

Efford, M. G. 2011. Estimation of population density by spatially explicit capture–recapture analysis of data from area searches. *Ecology* 92: 2,202–2,207.

Fiske, I. & Chandler, R. 2011. Unmarked: An *R* package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software* 43: 1–23.

Foran, D. R., S. C. Minta, and K. S. Heinemeyer. 1997. DNA-based analysis of hair to identify species and individuals for population research and monitoring. *Wildlife Society Bulletin* 25: 840–847.

Gipson, P. S., S. W. Buskirk, and T. W. Hobgood. 1982. Susitna Hydroelectric Project environmental studies, Subtask 7.11: furbearers—Phase I report. Report by Alaska Cooperative Wildlife Research Unit, University of Alaska, Fairbanks, for Terrestrial Environmental Specialists, Inc. 81 pp.

Gipson, P. S., S. W. Buskirk, T. W. Hobgood, and J. D. Woolington. 1984. Susitna Hydroelectric Project furbearer studies: Phase I report update. Final report by Alaska Cooperative Wildlife Research Unit, University of Alaska, Fairbanks, for Alaska Power Authority, Anchorage. 100 pp.

Gosselink, T. E., T. R. V. Deelen, R. E. Warner, and M. G. Joselyn. 2003. Temporal habitat partitioning and spatial use of coyotes and red foxes in east-central Illinois. *Journal of Wildlife Management* 67: 90–103.

Guyon, R., T. D. Lorentzen, C. Hitte, L. Kim, E. Cadieu, H. G. Parker, P. Quignon, J. K. Lowe, C. Renier, B. Gelfenbeyn, F. Vignaux, H. B. DeFrance, S. Gloux, G. G. Mahairas, C. Andre, F. Galibert, and E. A. Ostander. 2003. A 1-MB resolution radiation hybrid map of the canine genome. *Proceedings of the National Academy of Sciences* 100: 5,296–5,301.Harrison, D. J., J. A. Bissonette, and J. A. Sherburne. 1989. Spatial relationships between coyotes and red foxes in eastern Maine. *Journal of Wildlife Management* 53: 181–185.

Hein, E. W., and W. F. Andelt. 1995. Estimating coyote density from mark–resight surveys. *Journal of Wildlife Management* 59: 164–169.

Henke, S. E. and F. C. Bryant. 1999. Effects of coyote removal on the faunal community in western Texas. *Journal of Wildlife Management* 63: 1,066–1,081.

Heydon, M. J., J. C. Reynolds, and M. J. Short. 2000. Variation in abundance of foxes (*Vulpes vulpes*) between three regions of rural Britain, in relation to landscape and other variables. *Journal of Zoology* 251: 253–264.

Hines, J. E., J. D. Nichols, J. A. Royle, D. I. MacKenzie, A. Gopalaswamy, N. S. Kumar, and K. Karanth. 2010. Tigers on trails: occupancy modeling for cluster sampling. *Ecological Applications* 20: 1,456–1,466.

Jones, D. M., and J. B. Theberge. 1982. Summer home range and habitat utilisation of the red fox (*Vulpes vulpes*) in a tundra habitat, northwest British Columbia. *Canadian Journal of Zoology* 60: 807–812.

Kamler, J. F., and P. S. Gipson. 2000. Space and habitat use by resident and transient coyotes. *Canadian Journal of Zoology* 78: 2,106–2,111.

Kilström, Å. 2004. The wolverine population in the boreal forest area. Ph.D. dissertation, Uppsala University, Sweden.

Korhonen, H. 1988. Seasonal comparison of body composition and hair coat structure between mink and polecat. *Comparative Biochemistry and Physiology Part A: Physiology* 91: 469–473.

Krebs, C. J., K. Kielland, J. Bryant, M. O'Donoghue, F. Doyle, C. McIntyre, D. DiFolco, N. Berg, S. Carriere, R. Boonstra, S. Boutin, A. J. Kenney, D. G. Reid, K. Bodony, J. Putera, H. K. Timm, and T. Burke. 2013. Synchrony in the snowshoe hare (*Lepus americanus*) cycle in northwestern North America, 1970–2012. *Canadian Journal of Zoology* 91: 562–572.

Krebs, C. J., R. Boonstra, V. Nams, M. O'Donoghue, K. E. Hodges, and S. Boutin. 2001. Estimating snowshoe hare population density from pellet plots: a further evaluation. *Canadian Journal of Zoology* 79: 1–4.

Long, R. A., P. MacKay, J. Ray, and W. Zielinski. 2012. Noninvasive Survey Methods for Carnivores. Island Press, Washington, DC.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, J. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. *Ecology* 83: 2,248–2,255.

MacKenzie, D. I., J. Nichols, J. Royle, K. Pollock, L. Bailey, and J. Hines. 2006. Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Elsevier Academic Press, Oxford, UK.

Maurel, D., C. Coutant, L. Boissin-Agasse, and J. Boissin. 1986. Seasonal moulting patterns in three fur-bearing mammals: the European badger (*Meles meles*), the red fox (*Vulpes vulpes*), and the mink (*Mustela vison*): a morphological and histological study. *Canadian Journal of Zoology* 64: 1,757–1,764.

Mazerolle, M. J. 2015. AICcmodavg: model selection and multimodel inference based on (Q)AIC(c). *R* package version 2.0-3. Available online: http://CRAN.R-project.org/package=AICcmodavg.

McDaniel, G. W., K. S. McKelvey, J. R. Squires, and L. F. Ruggiero. 2000. Efficacy of lures and hair snares to detect lynx. *Wildlife Society Bulletin:* 28: 119–123.

McKelvey, K. S., J. J. Claar, G. W. McDaniel, and G. Hanvey. 1999. National lynx detection protocol. U.S. Forest Service, Rocky Mountain Research Station, Missoula, MT.

McKelvey, K. S., J. Von Kienast, K. B. Aubry, G. M. Koehler, B. T. Maletzke, J. R. Squires, E. L. Lindquist, S. Loch, and M. K. Schwartz. 2006. DNA analysis of hair and scat collected along snow tracks to document the presence of Canada lynx. *Wildlife Society Bulletin* 34: 451–455.

Mills, L. S., M. Zimova, J. Oyler, S. Running, J. T. Abatzoglou, and P. M. Lukacs. 2013. Camouflage mismatch in seasonal coat color due to decreased snow duration. *Proceedings of the National Academy of Sciences of the United States of America* 110: 7,360–7,365.

Moore, M., S. K. Brown, B. N. Sacks. 2010. Thirty-one short red fox (*Vulpes vulpes*) microsatellite markers. *Molecular Ecology Resources* 10: 404–408.

Moriarty, K. M., W. J. Zielinski, and E. D. Forsman. 2011. Decline in American marten occupancy rates at Sagehen Experimental Forest, CA. *Journal of Wildlife Management* 75: 1,774–1,787.

Mumma, M. A., C. Zieminski, T. K. Fuller, S. P. Mahoney, and L. P. Waits. 2015. Evaluating noninvasive genetic sampling techniques to estimate large carnivore abundance. *Molecular Ecology Resources* 15: 1,133–1,144.

Murphy, M. A., L. P. Waits, and K. C. Kendall. 2000. Quantitative evaluation of fecal drying methods for brown bear DNA analysis. *Wildlife Society Bulletin* 28: 951–957.

Murray, D. L., J. D. Roth, E. Ellsworth, A. J. Wirsing, and T. D. Steury. 2002. Estimating low-density snowshoe hare populations using fecal pellet counts. *Canadian Journal of Zoology* 80: 771–781.

O'Donoghue, M. O., S. Boutin, C. J. Krebs, G. Zuleta, D. L. Murray, and E. J. Hofer. 1998. Functional responses of coyotes and lynx to the snowshoe hare cycle. *Ecology* 79: 1193–1208.

Oakley, K. L., E. M. Debevec, and E. A. Rexstad. 1999. Development of a long-term ecological monitoring program in Denali National Park and Preserve, Alaska (USA). Pages 307–314 *in* North American Science Symposium: Toward a Unified Framework for Inventorying and Monitoring Forest Ecosystem Resources. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO.

Olsson, R. 2009. Boreal Forest and Climate Change. Air Pollution and Climate Series 23. Air Pollution and Climate Secretariat and Taiga Rescue Network, Göteborg, Sweden.

Palomares, F., J. A. Godoy, A. Piriz, S. J. O'Brien, and W. E. Johnson. 2002. Faecal genetic analysis to determine the presence and distribution of elusive carnivores: design and feasibility for the Iberian lynx. *Molecular Ecology* 11: 2171–2182.

Pauli, J. N., M. B. Hamilton, E. B. Crain, and S. W. Buskirk. 2008. A single-sampling hair trap for mesocarnivores. *Journal of Wildlife Management* 72: 1650–1652.

Peakall, R. O. D., and P. E. Smouse. 2006. GENALEX 6: genetic analyses in Excel. Population genetic software for teaching and research. *Molecular Ecology Notes* 6: 288–295.

Perrine, J. D. 2005. Ecology of red fox (*Vulpes vulpes*) in the Lassen Peak region of California, USA. Ph.D. dissertation, University of California, Berkeley.

Peters, R. H. 1986. The Ecological Implications of Body Size. Cambridge University Press. New York, NY.

Pozzanghera, C. B. 2015. Non-invasive methods for obtaining occupancy probabilities and density estimates of interior Alaska's mesocarnivore populations. M.S. thesis, University of Alaska Fairbanks.

Prugh, L. R. 2005. Coyote prey selection and community stability during a decline in food supply. *Oikos* 110: 253-264.

Prugh, L. R., and C. J. Krebs. 2004. Snowshoe hare pellet-decay rates and aging in different habitats. *Wildlife Society Bulletin*, 32: 386–393.

Raine, R. M. 1983. Winter habitat use and responses to snow cover of fisher (*Martes pennanti*) and marten (*Martes americana*) in southeastern Manitoba. *Canadian Journal of Zoology* 61: 25–34.

Rexstad, E., and K. Kielland. 2006. Mammalian herbivore population dynamics in the Alaskan boreal forest. Pp.121–132 *in* D.L. Verbyla (ed.). Alaska's Changing Boreal Forest. Oxford University Press, New York, NY.

Ruggiero, L. F., J. R. Squires, S. W. Buskirk, K. B. Aubry, K. S. McKelvey, G. Koehler, and C. J. Krebs. 2000. Ecology and Conservation of Lynx in the United States. University Press of Colorado, Boulder, CO.

Ruggiero, L. F., K. B. Aubry, S. W. Buskirk, L. J. Lyon, and W. J. Zielinski. 1994. The scientific basis for conserving forest carnivores: American marten, fisher, lynx, and wolverine in the western United States. General Technical Report RM-254. United States Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. 184 pp.

Sargeant, A. B., S. H. Allen, and J. O. Hastings. 1987. Spatial relations between sympatric coyotes and red foxes in North Dakota. *Journal of Wildlife Management* 51: 285–293.

Sarmento, P., J. Cruz, C. Eira, and C. Fonseca. 2009. Evaluation of camera trapping for estimating red fox abundance. *Journal of Wildlife Management* 73: 1,207–1,212.

Schuette, P., A. P. Wagner, M. E. Wagner, and S. Creel. 2013. Occupancy patterns and niche partitioning within a diverse carnivore community exposed to anthropogenic pressures. *Biological Conservation* 158: 301–312.

Seddon, J. M. 2005. Canid-specific primers for molecular sexing using tissue or non-invasive samples. *Conservation Genetics* 6: 147–149.

Sivy, K. J. 2015. Direct and indirect effects of wolves on mesopredator communities in interior Alaska. M.S. thesis, University of Alaska Fairbanks.

Smith, J. B., J. A. Jenks, and R. W. Klaver. 2007. Evaluating detection probabilities for American marten in the Black Hills, South Dakota. *Journal of Wildlife Management* 71: 2,412–2,416.

Squires, J. R., N. J. Decesare, J. A. Kolbe, and L. F. Ruggiero. 2010. Seasonal resource selection of Canada lynx in managed forests of the Northern Rocky Mountains. *Journal of Wildlife Management* 74: 1,648–1,660.

UAF (University of Alaska Fairbanks). 2014a. Terrestrial Furbearer Abundance and Habitat Use Study Plan Section 10.10; Initial Study Report Part A: Sections 1–6, 8–10; Susitna–Watana Hydroelectric Project (FERC No. 14241). Report for Alaska Energy Authority, Anchorage, by University of Alaska Fairbanks, Institute of Arctic Biology. 17 pp.

UAF. 2014b. Terrestrial Furbearer Abundance and Habitat Use Study Plan Section 10.10; Initial Study Report Part C: Executive Summary and Section 7; Susitna–Watana Hydroelectric Project (FERC No. 14241). Report for Alaska Energy Authority, Anchorage, by University of Alaska Fairbanks, Institute of Arctic Biology. 4 pp.

Waits, L. P., G. Luikart, and P. Taberlet. 2001. Estimating the probability of identity among genotypes in natural populations: cautions and guidelines. *Molecular Ecology* 10: 249–256.

Walpole, A., J. Bowman, D. Murray, and P. Wilson. 2012. Functional connectivity of lynx at their southern range periphery in Ontario, Canada. *Landscape Ecology* 27: 761–773.

Whittington, J., K. Heuer, B. Hunt, M. Hebblewhite, and P. Lukacs. 2014. Estimating occupancy using spatially and temporally replicated snow surveys. *Animal Conservation* 18: 92–101.

Williams, B. W., D. R. Etter, D. W. Linden, K. F. Millenbah, S. R. Winterstein, and K. T. Scribner. 2009. Noninvasive hair sampling and genetic tagging of co-distributed fishers and American martens. *Journal of Wildlife Management* 73: 26–34.

Zielinski, W. J., R. L. Truex, F. V. Schlexer, L. A. Campbell, and C. Carroll. 2005. Historical and contemporary distributions of carnivores in forests of the Sierra Nevada, California, USA. *Journal of Biogeography* 32: 1,385–1,407.

9. TABLES

Table 5-1. Server Location and File/Folder Names for the Field Data for Terrestrial Furbearers Collected in 2013–2014.

Server Pathway or File/Folder Name	Description
http://gis.suhydro.org/SIR/10-Wildlife/10.10-Terrestrial_Furbearer/	Pathway to data files
TFUR_10_10_Data_2013_2014_UAF.gdb	Geodatabase file containing spatial layers of study areas, scat collection locations, survey cell locations, hair snag locations, historic aerial transects, historic aerial transect endpoints, occupancy estimates.
TFUR_10_10_Data_2013_2014_UAF.zip	A zip file of Excel tables of ground track survey data, hair sample data, occupancy data, scat collection data, snowshoe hare pellet data, vole capture data, aerial survey data.

Table 5.1-1. Furbearer Scat Samples Collected during the Terrestrial Furbearer Study, Winter 2013 and 2014.

Species[1]	Number of Scats Collected in an Unknown Year	Number of Scats Collected in 2013	Number of Scats Collected in 2014	Total Scats Collected	Number of Scats Successfully Genotyped (% of total)
Red Fox	2	75	154	231	137 (59%)
Coyote	1	28	44	73	50 (68%)
Marten	0	02	1	3	3 (100%)
Lynx	0	0	8	8	8 (100%)
Wolverine	0	11	24	35	17 (49%)
Wolf	0	3	30	33	n/a
Failed	2	19	44	65	n/a
Total Success	3	119	261	383	n/a
Grand Total	5	138	305	448	215

Notes:
1.   Samples were identified to the species level in the Prugh Lab at UAF using DNA extraction techniques outlined in the RSP (section 10.10.4.2).
2.   n/a indicates Not Applicable (scats from wolves were not genotyped to the individual level).

Table 5.1-2. Hair Samples Collected during the Terrestrial Furbearer Study, Winter 2013 and 2014.

Species[1]	Number of Samples Collected in 2013	Number of Samples Collected in 2014 at Hair Snag Stations	Number of Samples Collected in 2014 by Backtracking
Lynx	23	18	22
Marten	0	21	0
Total	23	39	22

Notes:

1   Samples were identified in the field based on hair coloration and size and the presence of furbearer tracks near the hair snag station. These counts represent field collection data only, as DNA extractions failed for most of the collected samples.

Table 0. Average Number of Hare Pellets per Survey Plot and Average Hare Densities* at 15 Survey Plots, Summer 2012–2014.

Survey Plots	Mean Number of Pellets per Plot in 2012	Estimated Hare Density in 2012 (hares/ha)	Mean Number of Pellets per Plot in 2013	Estimated Hare Density in 2013 (hares/ha)	Mean Number of Pellets per Plot in 2014	Estimated Hare Density in 2014 (hares/ha)
**Lower Watana Creek Forest	3.1	0.12	n/a	n/a	n/a	n/a
**Lower Jay Creek Mixed Forest	13.4	0.51	n/a	n/a	n/a	n/a
**Lower Watana Crk Mixed Forest	12.3	0.47	n/a	n/a	n/a	n/a
**Lower Tsusena Mixed Forest	0.0	0.0	n/a	n/a	n/a	n/a
**Lower Tsusena Forest	0.2	0.01	n/a	n/a	n/a	n/a
1) Watana Creek Shrub	1.1	0.04	2.0	0.08	1.32	0.05
2) Watana Creek Forest	n/a	n/a	3.3	0.12	3.36	0.13
3) Jay Creek Forest	0.4	0.01	2.2	0.09	0.74	0.03
4) Jay Creek Shrub	n/a	n/a	45.2	1.41	16.90	0.62
5) Tsusena Creek Shrub	n/a	n/a	8.7	0.30	3.96	0.15
6) Deadman Creek Forest	n/a	n/a	25.8	0.74	10.38	0.39
7) Upper Butte Creek Forest	n/a	n/a	0.5	0.02	0.10	0.00
8) Upper Butte Creek Shrub	n/a	n/a	1.3	0.05	0.64	0.02
9) Seattle Creek Shrub	n/a	n/a	3.8	0.09	4.08	0.16
10) Seattle Creek Forest	n/a	n/a	0.3	0.01	0.10	0.00
11) Butte Lake Forest	n/a	n/a	0.6	0.02	0.82	0.03
12) Butte Lake Shrub	n/a	n/a	16.5	0.39	6.46	0.25
13) Southern Butte Creek Forest	n/a	n/a	6.2	0.18	2.64	0.10
14) Southern Butte Creek Shrub	n/a	n/a	3.3	0.11	1.76	0.06
15) Oshetna Creek Forest	9.5	0.36	29.8	1.14	29.40	0.86
Average - Shrub	1.1	0.04	11.5	0.35	5.0	0.19
Average - Forest	5.5	0.21	8.6	0.29	5.9	0.19
Average - Overall	5.0	0.19	10.0	0.32	5.5	0.19

Notes:
n/a: Plots were not surveyed.
* Density conversion equation: $D_h = 0.03 * D_p$ (Prugh 2005).
** Plots were only surveyed in 2012 and were inaccessible during subsequent years due to land-access constraints.

Table 5.1-4. Number of Voles Captured and Estimated Vole Density* at 15 Survey Plots, Summer 2013 and 2014.

Survey Plots	Red-backed Voles Caught, 2013	Singing Voles Caught, 2013	Meadow/ Tundra Voles Caught, 2013	Total Number of Voles Caught, 2013	Vole Density 2013 voles/ha	Red-backed Voles Caught, 2014	Singing Voles Caught, 2014	Meadow/ Tundra Voles Caught, 2014	Total Number of Voles Caught, 2014	Vole Density 2014 voles/ha
** Watana Lower Forest	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
** Jay Lower Forest	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
1) Watana Creek Forest	1	0	0	1	3.2	3	0	0	3	9.5
2) Watana Creek Meadow	0	0	0	0	0	10	0	0	10	31.6
3) Jay Creek Forest	0	0	0	0	0	9	0	0	9	28.4
4) Jay Creek Meadow	0	1	1	2	6.3	2	0	12	14	44.2
5) Tsusena Creek Forest	1	0	0	1	3.2	8	0	1	9	28.4
6) Tsusena Creek Meadow	0	0	0	0	0	11	0	0	11	34.8
7) West Tsusena Creek Forest	0	0	0	0	0	n/a	n/a	n/a	n/a	n/a
8) West Tsusena Creek Meadow	3	0	0	3	9.5	n/a	n/a	n/a	n/a	n/a
9) Upper Butte Creek Forest	1	0	0	1	3.2	7	0	0	7	22.1
10) Upper Butte Creek Meadow	1	0	0	1	3.2	2	0	0	2	6.3
11) Upper Watana Creek Forest	1	0	0	1	3.2	4	0	0	4	12.6
12) Upper Watana Creek Meadow	0	0	1	1	3.2	0	7	0	7	22.1
13) Seattle Creek Forest	1	0	0	1	3.2	9	0	0	9	28.4
14) Seattle Creek Meadow	2	0	0	2	6.3	2	0	0	2	6.3
15) Deadman Mountain Meadow	0	0	0	0	0	0	0	0	0	0
Average - Meadow	0.8	0.1	0.3	1.1	3.6	3.9	1.0	1.7	6.6	20.8
Average - Forest	0.7	0.0	0.0	0.7	2.3	6.7	0.0	0.2	6.8	21.6
Average - Overall	0.7	0.1	0.1	0.9	2.9	5.2	0.5	1.0	6.7	21.1

Notes:

n/a   Plots were not surveyed.

*   Density conversion equation (D_v = voles per hectare; N_1 = number of voles caught on first trap night; see Methods): $D_v = 0.5157 * N_1 - 0.0684$.

**   Plots were only surveyed in 2012 and were inaccessible during subsequent years due to land access constraints.

Table 5.2-1. Furbearer Track Counts During Five Aerial Surveys, Winter 2013 and 2014.

Species	2013			2014		
	Feb 26	**Mar 27**	**Apr 19**	**Feb 17**	**Mar 25**	**Total**
Marten	93	105	193	70	109	570
Weasels	68	43	91	13	20	235
Wolverine	14	40	53	22	33	162
Lynx	22	53	39	19	28	161
Red Fox	13	28	11	15	46	113
Wolf	9	0	11	0	37	57
Coyote	0	0	0	10	11	21
Unknown Furbearer	0	0	0	3	16	19
River Otter	2	6	4	2	3	17
Bear	0	0	2	1	0	3
Beaver	1	0	0	0	0	1
Mink	0	1	0	0	0	1
Total Tracks	222	276	404	155	303	1,360
DSLS[1]	2	4	9	5	6	--
Tracks / DSLS	111.0	69.0	44.9	31.0	50.5	--

Notes:
1   DSLS = days since last snowfall.

Table 5.2-2. Average Furbearer Track Counts and Tracks Per DSLS[1] from Aerial Surveys Summarized by Year, Winter 2013 and 2014.

Species	Tracks per Survey		Tracks per DSLS[1]	
	2013	**2014**	**2013**	**2014**
Marten	130.3	89.5	31.4	16.1
Weasels	67.3	16.5	18.3	3.0
Wolverine	35.7	27.5	7.6	5.0
Lynx	38.0	23.5	9.5	4.2
Red Fox	17.3	30.5	4.9	5.3
Wolf	6.7	18.5	2.9	6.2
Coyote	0	10.5	0	1.9
Unknown Furbearer	0	9.5	0	1.6
River Otter	4	2.5	1.0	0.5
Bear	0.7	0.5	0.2	0.2
Mink	0.3	0	0.3	0
Average	27.3	20.8	6.9	4.0

Notes:
1   DSLS = days since last snowfall.

Table 5.2-3. Track Counts from Aerial Furbearer Surveys by Habitat Type, Winter 2013 and 2014.

Habitat Type	Coyote	Lynx	Marten	Red Fox
Forested				
Black Spruce Forest	0	28	94	17
Black Spruce Woodland	0	24	165	13
Deciduous Forest	0	0	1	0
Mixed Forest	0	18	65	4
Mixed Woodland	0	2	7	0
White Spruce Forest	0	34	84	13
White Spruce Woodland	0	26	84	3
Forest Total	0	132	500	50
Shrub				
Alder	0	9	11	12
Low Shrub	0	1	18	7
Tall Shrub	3	7	15	4
Shrub Total	3	17	44	23
Other				
Alpine	2	1	5	32
Marsh	0	4	9	2
Creek	0	2	3	2
River	16	2	0	2
Lake	0	3	4	0
Missing Data	0	0	5	2
Other Total	18	12	26	40
Grand Total	21	161	570	113

Table 5.2-4. Overall Furbearer Occupancy Probabilities (ψ) by Survey Year. Occupancy estimates generated from model *ψ(species*year) p(dist + dsls + species + method + year)*.

Species	Survey Year	Occupancy Probability ($\psi \pm$ SE)
Coyote	2013	0.54 ± 0.14
Coyote	2014	0.65 ± 0.28
Lynx	2013	0.35 ± 0.12
Lynx	2014	0.84 ± 0.37
Marten	2013	0.36 ± 0.08
Marten	2014	0.28 ± 0.08
Red fox	2013	0.46 ± 0.09
Red fox	2014	0.46 ± 0.11

Table 5.2-5. Individual Covariate Influence (Summed AICc Weight) on Furbearer Occupancy Probabilities (ψ), Winter 2013 and 2014.

Covariate Name	Description	Summed AICc Weight (%)
Species	Target species (coyote, lynx, marten, red fox)	92.7
Habitat	Majority habitat type within sample cell (forest, shrub, tundra)	90.4
Compaction	Survey cell average snow compaction. Standardized to mean of 0	90.7
Depth	Survey cell average snow depth. Standardized to mean of 0	62.4
Prey	Average prey species abundance per cell. Standardized to mean of 0	30.2
Year	Study year (2013 or 2014)	27.6

Table 5.2-6. Terrestrial Furbearer Occupancy Model Selection Table, Winter 2013 and 2014. The top-ranked detection model was used for all occupancy models, *p(dist+dsls+species+method+year)*.

Model Name[1]	Model Parameters	QAICc	Delta_QAICc	QAICcWt
ψ(species*compaction)	17	531.08	0.00	0.98
ψ(species*prey)	17	541.08	9.99	0.006
ψ(habitat+species+compaction)	16	541.59	10.51	0.005
ψ(habitat+species+compaction+snow)	17	542.02	10.93	0.004
ψ(species*habitat)	21	542.22	11.13	0.004
ψ(habitat+species+compaction+snow+prey)	18	544.11	13.02	0.001
ψ(habitat+species+compaction+snow+year)	18	544.19	13.11	0.001
ψ(species*snow)	17	546.12	15.04	0.000
ψ(species*year)	17	549.97	18.89	0.000

Notes:

1   dist = distance; DSLS = days since last snowfall; compaction = snow compaction; method = square or linear transects; prey = total number of prey tracks per km per DSLS.

Table 5.3-1. Spatially Explicit Capture–Recapture Model Selection Table for Coyote and Red Fox. Models estimate density (D) and detection (g0 and sigma) parameters at constant conditions (1), by habitat type (veg), by study year (year), or by survey occasion (t). The six models for each species are listed in order of their AICc ranking.

Species	Model	Model Parameters	AICc	Delta AICc	AICcwt
Coyote	D~1 g0~1 sigma~1	3	603.28	0	0.572
	D~1 g0~year sigma~1	4	606.06	2.77	0.143
	D~year g0~1 sigma~1	4	606.42	3.14	0.119
	D~1 g0~t sigma~1	4	606.56	3.28	0.111
	D~veg g0~1 sigma~1	5	608.91	5.63	0.034
	D~year g0~year sigma~1	5	609.92	6.63	0.021
Red Fox	D~1 g0~year sigma~1	4	1507.52	0	0.661
	D~year g0~ year sigma~1	5	1509.62	2.11	0.230
	D~1 g0~1 sigma~1	3	1513.03	5.52	0.042
	D~veg g0~1 sigma~1	5	1513.77	6.25	0.029
	D~1 g0~t sigma~1	4	1514.40	6.88	0.021
	D~ year g0~1 sigma~1	4	1514.91	7.40	0.016

Table 5.3-2. Estimates of Population Growth Rate (Lamda), Apparent Survival (Phi), Recruitment (f), Recapture Probability (p), and Abundance (N) for Red Foxes and Coyotes in the 2013 and 2014 Survey Areas. Estimates were produced from Pradel open mark–recapture models and include standard errors (SE) and lower (LCI) and upper (UCI) confidence intervals.

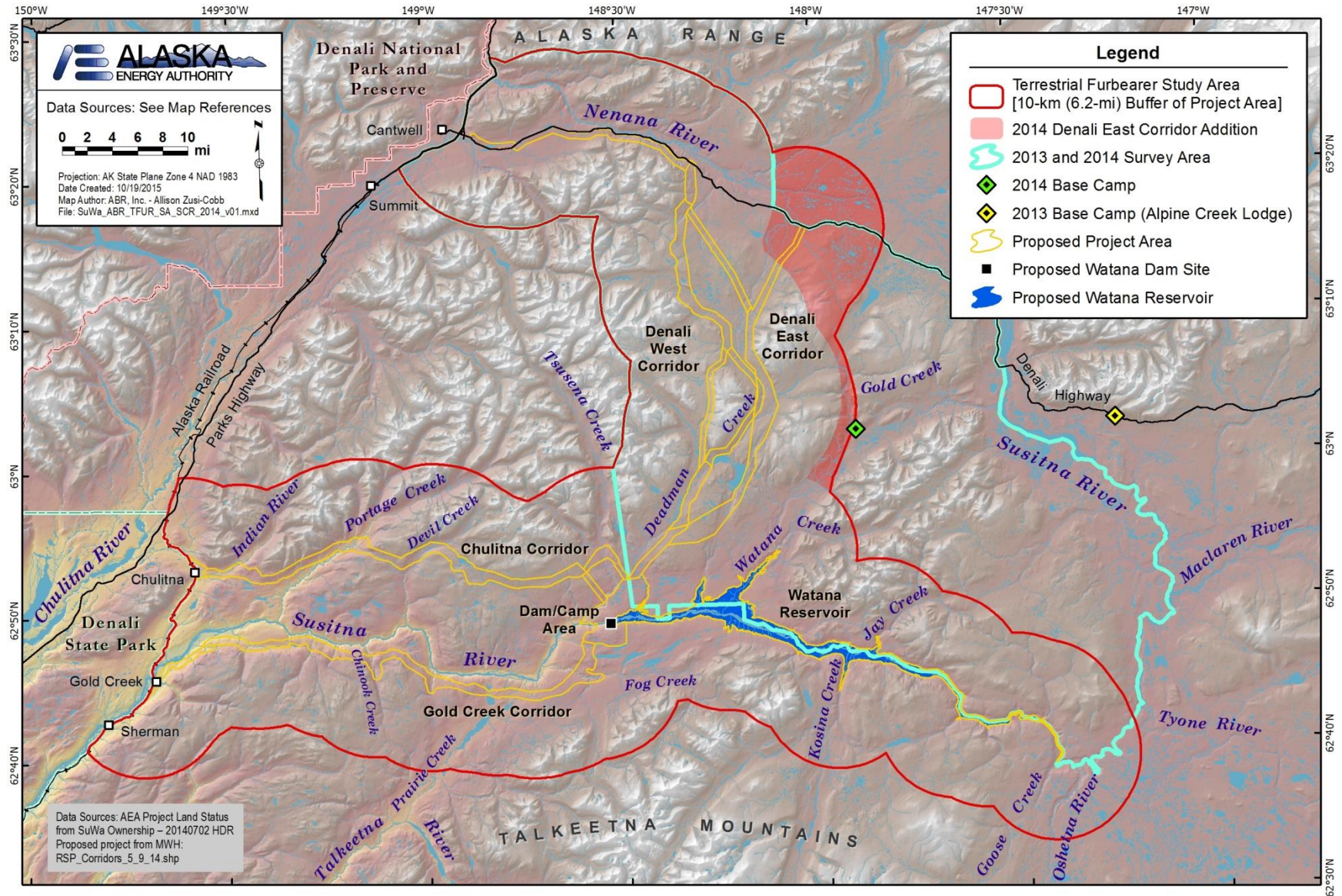| Species	Parameter	Mean	SE	LCI	UCI
Coyote	Lambda	1.04	0.37	0.53	2.05
	Phi	0.61	0.23	0.19	0.91
	f	0.43	0.37	0.04	0.93
	p	0.43	0.11	0.25	0.64
	N 2013	11	3	9	24
	N 2014	12	2	10	22
Red Fox	Lambda	1.21	0.28	0.78	1.89
	Phi	0.38	0.15	0.15	0.68
	f	0.83	0.28	0.09	1.00
	p	0.24	0.05	0.16	0.35
	N 2013	49	9	37	77
	N 2014	60	11	46	92

10. FIGURES

Figure 3-1. Terrestrial Furbearer Study Area and Survey Area for the Susitna–Watana Hydroelectric Project.
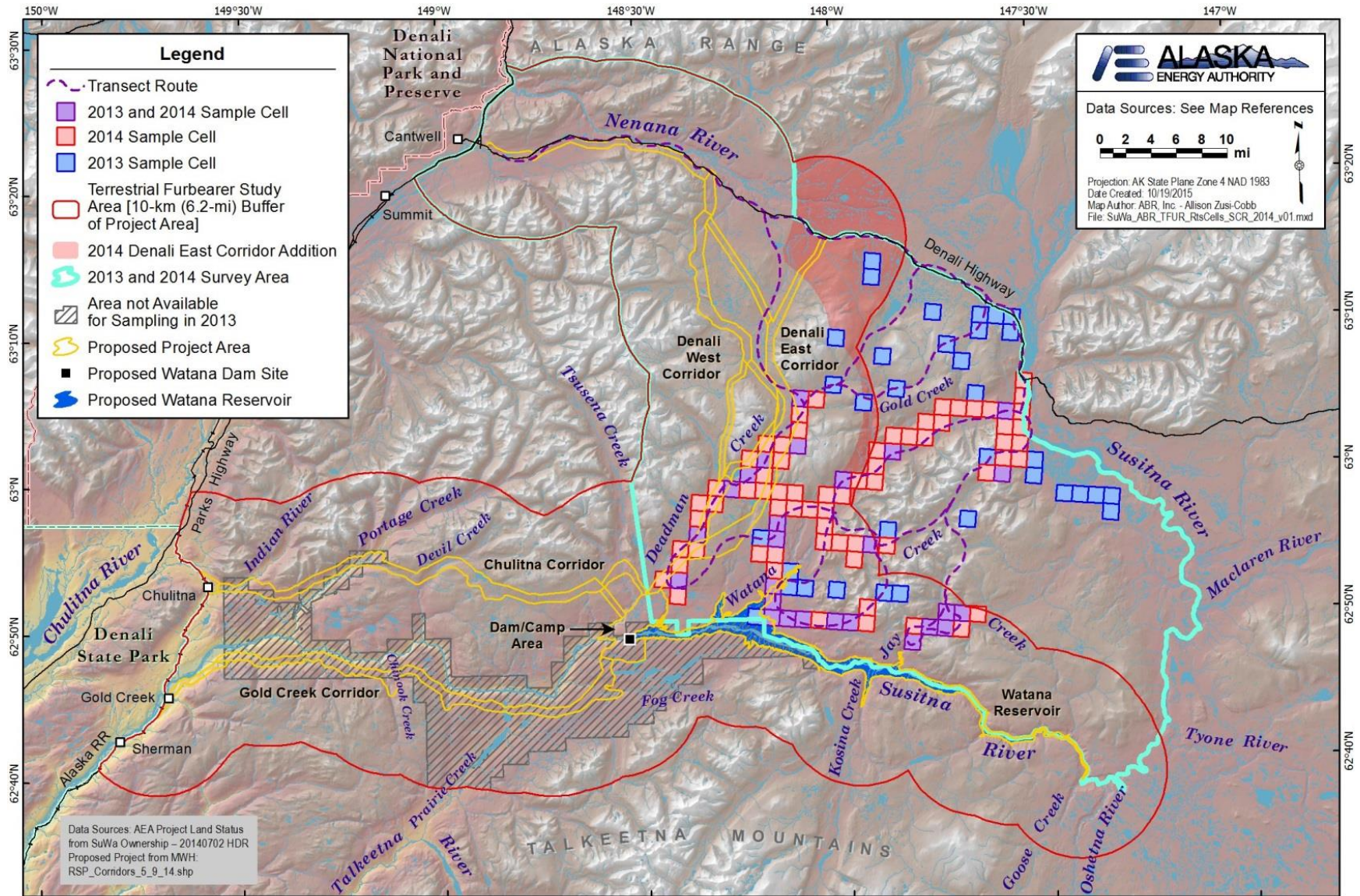
Figure 4.1-1. Location of Ground-based Transect and Occupancy Survey Cells Sampled in Winter 2013 and 2014.

Figure 4.1-2. Example of a Lynx Hair-snag Station in the Study Area during the 2013 Survey Season. Aluminum pie plates were used as a visual attractant and carpet pads imbedded with wire tube brushes were fixed to trees and scented with catnip and beaver castor oil.

Figure 4.1-3. Example of a Marten Hair Tube Deployed during the 2014 Survey Season. Tubes were constructed of PVC pipe embedded with a steel tube brush and were baited with chicken.

Figure 4.1-4. Plot and Grid Locations Sampled for Snowshoe Hare and Vole Abundance in Summer 2013 and 2014.
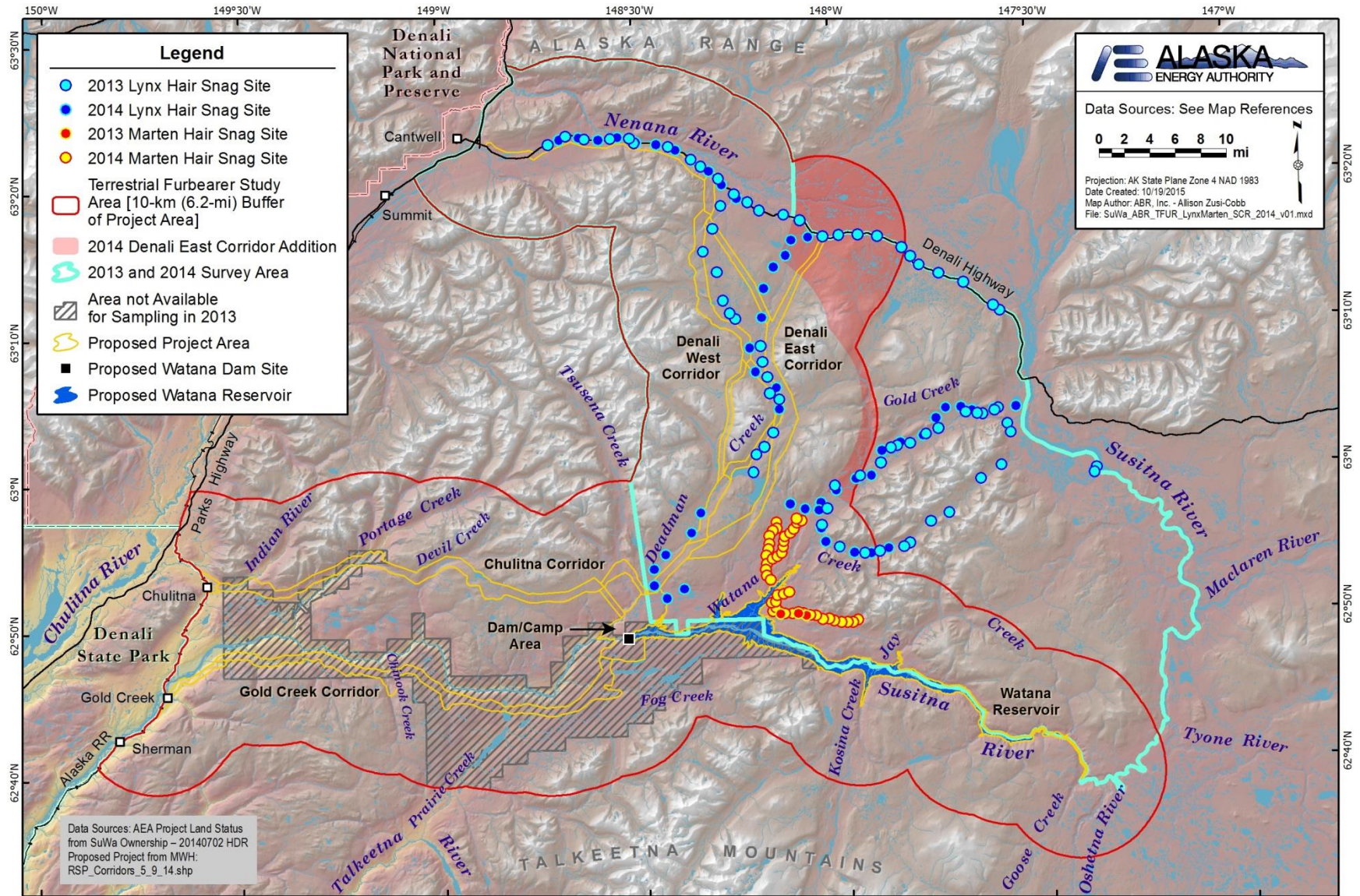
Figure 4.1-5. Locations of Lynx and Marten Hair-snag Sites in Winter 2013 and 2014.
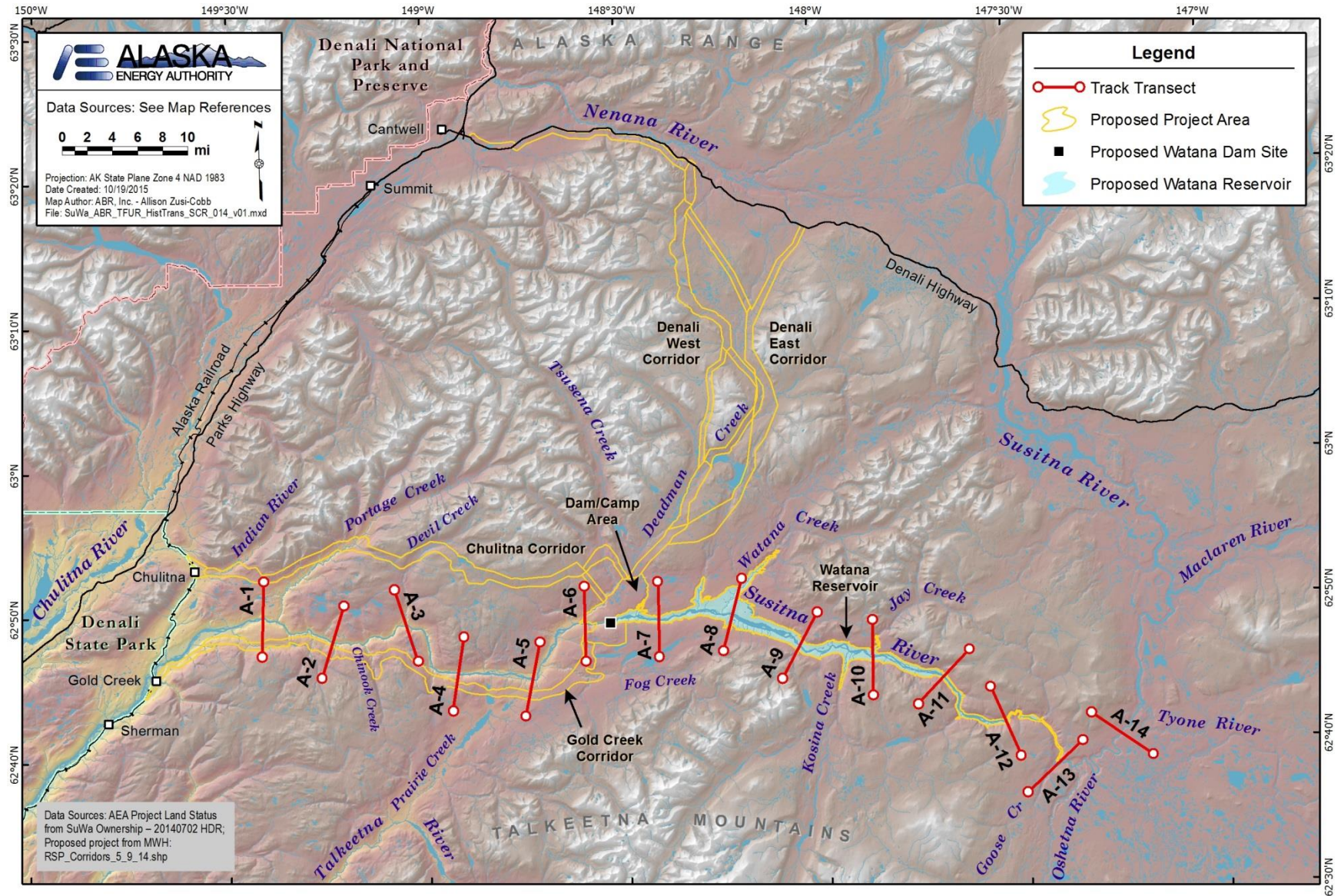
Figure 4.3-1. Aerial Transects for Track Surveys of Terrestrial Furbearers in Winter 2013 and 2014.
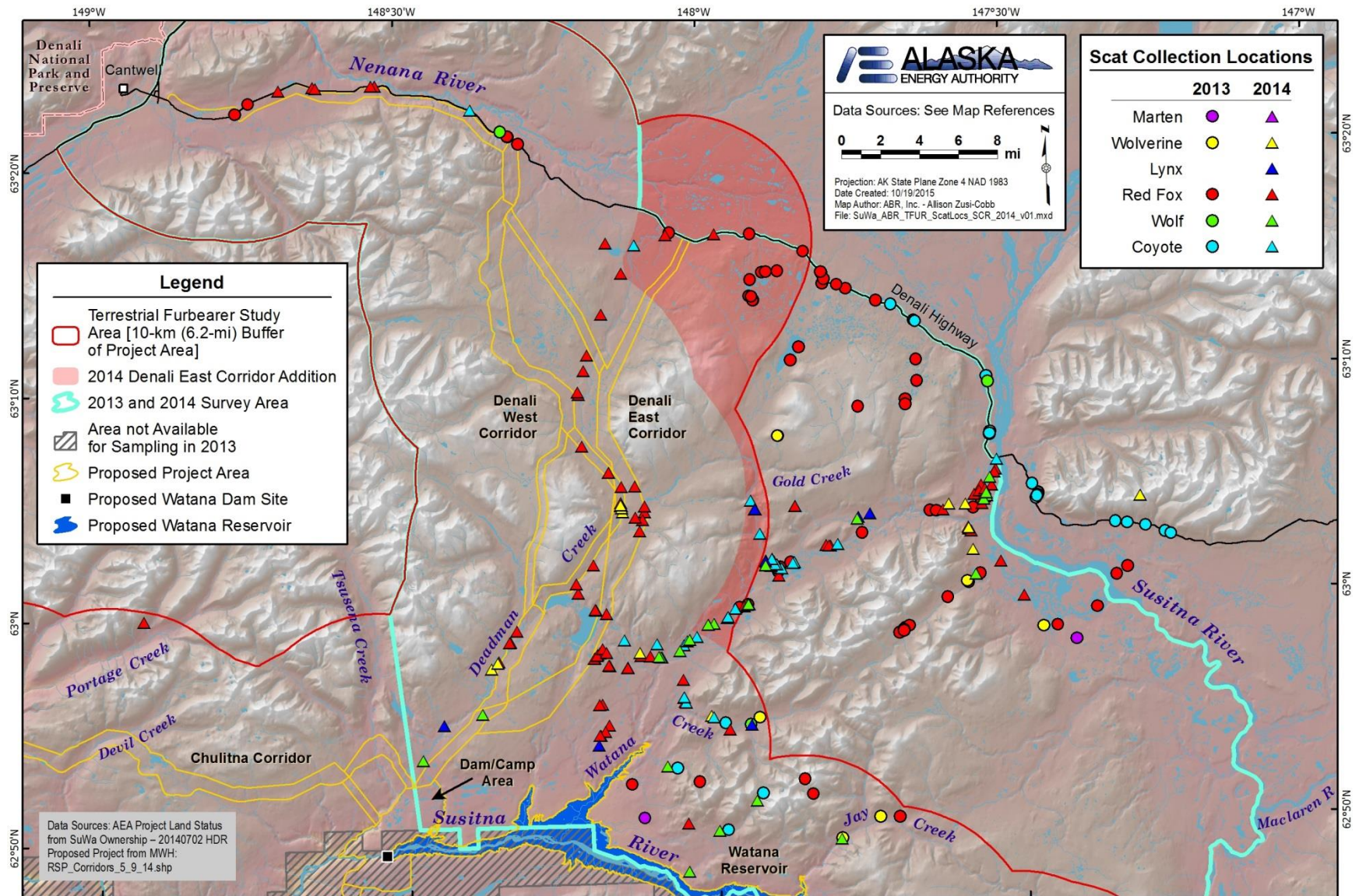
Figure 5.1-1. Scat Collection Locations for Terrestrial Furbearers in Winter 2013 and 2014.
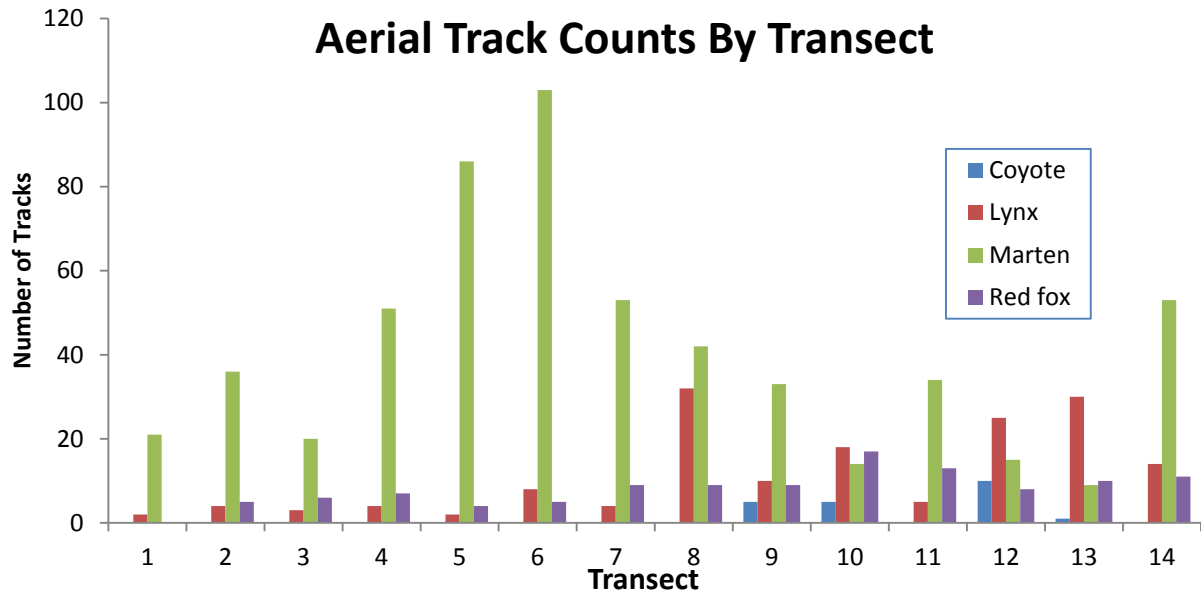
Figure 5.2-1. Track Counts of Terrestrial Furbearers along Each Aerial Survey Transect in Winter 2013 and 2014. Counts were summed across five surveys.
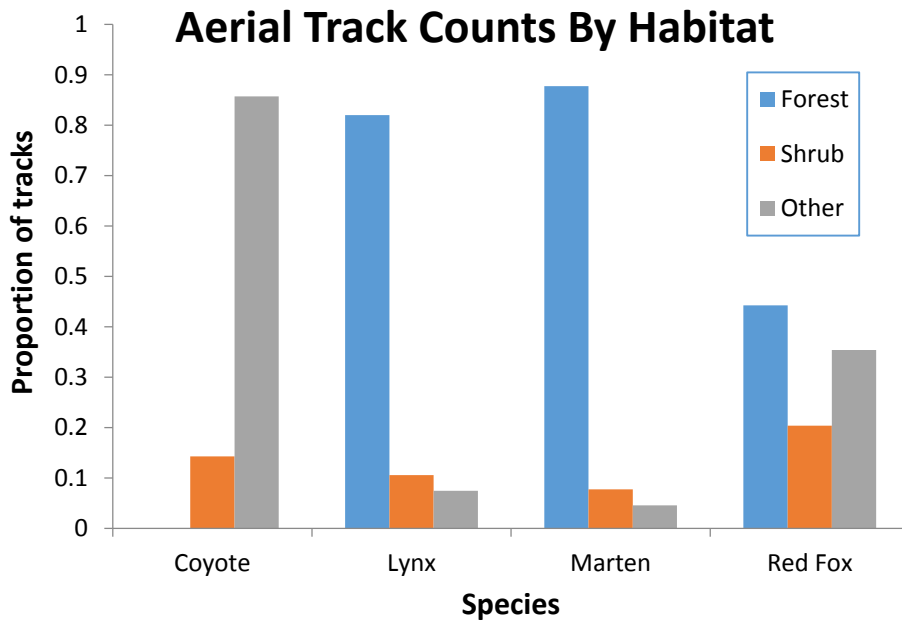
Figure 5.2-2. Proportion of Furbearer Tracks Counted Within Major Habitat Types During Aerial Transect Surveys in Winter 2013 and 2014. Counts were summed across five surveys.

Terestrial Furbearer Winter Detection Probability
2013-2014
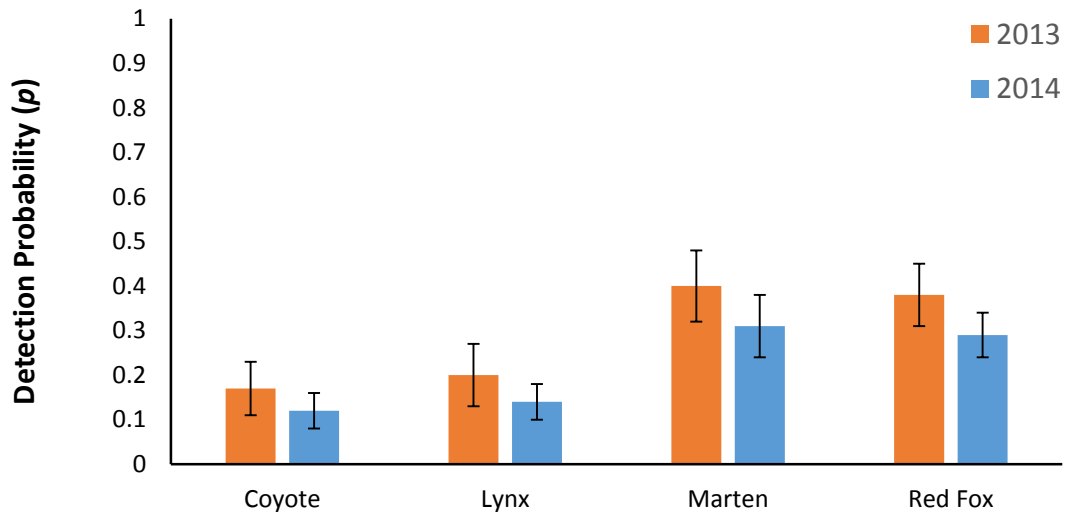
Figure 5.2-3. Detection Probabilities with Standard Errors for Terrestrial Furbearer Species in the Study Area, 2013–2014. Estimates are from the top-ranking occupancy model: $p(dist + dsls + species + method + year)$ $\psi(species*compaction)$. The continuous detection covariates 'survey distance' (dist) and 'days since last snowfall' (DSLS) were held constant at their mean values for these estimates.
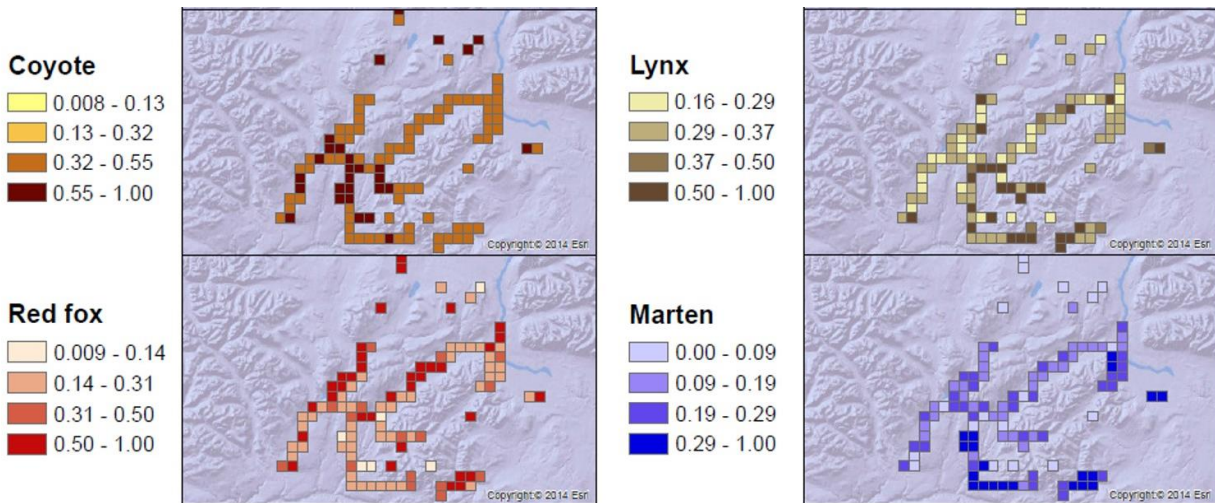

Figure 5.2-4. Cell-specific Maximum Occupancy Probabilities for Furbearers in the Study Area, Winter 2013–2014. Estimates are from model: $p(dist + dsls + species + method + year)$ $\psi(.)$. The continuous detection covariates 'survey distance' (dist) and 'days since last snowfall' (DSLS) were held constant at their mean values for these estimates.
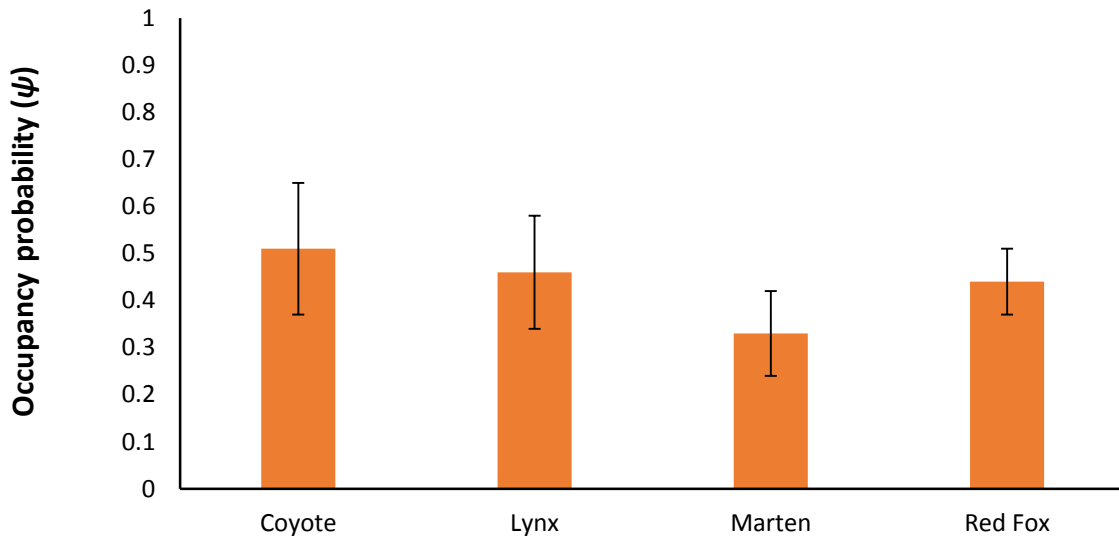
Figure 5.2-5. Occupancy Probabilities at Mean COMPACTION with Standard Errors for Tracks of Terrestrial Furbearer Species in the Study Area, 2013–2014. Estimates are from the top-ranking occupancy model: p(dist + dsls + species + method + year) ψ(species*compaction) where dist = survey distance, dsls = days since last snowfall, and method = square or linear track transect. Snow compaction values were held constant at their mean values for these estimates.
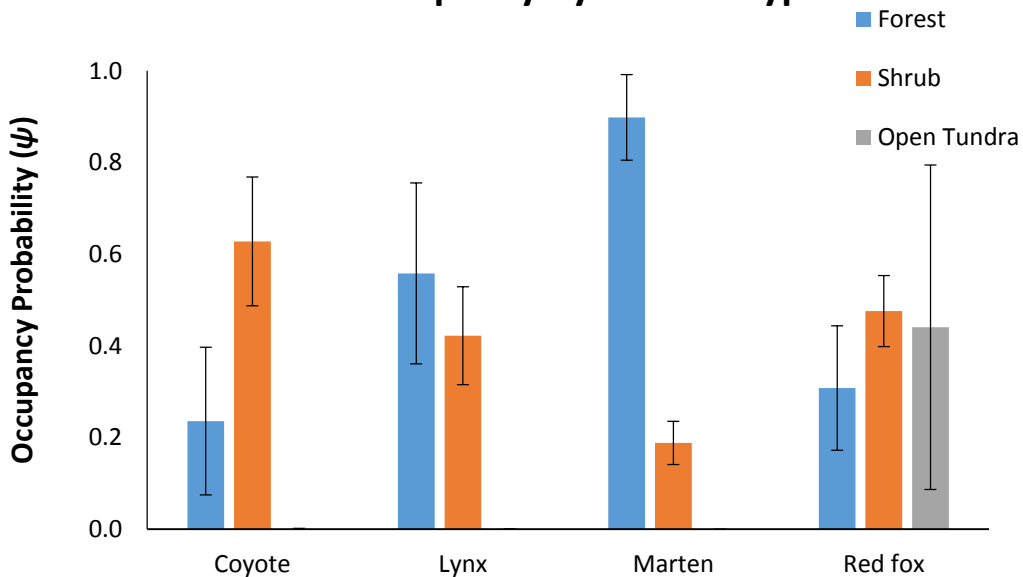

Figure 5.2-6. Occupancy Probabilities by Habitat Type with Standard Errors for Terrestrial Furbearer Species in the Study Area, 2013–2014. Estimates are from model: p(dist + dsls + species + method + year) ψ(species*habitat) where dist = survey distance, dsls = days since last snowfall, and method = square or linear track transect.
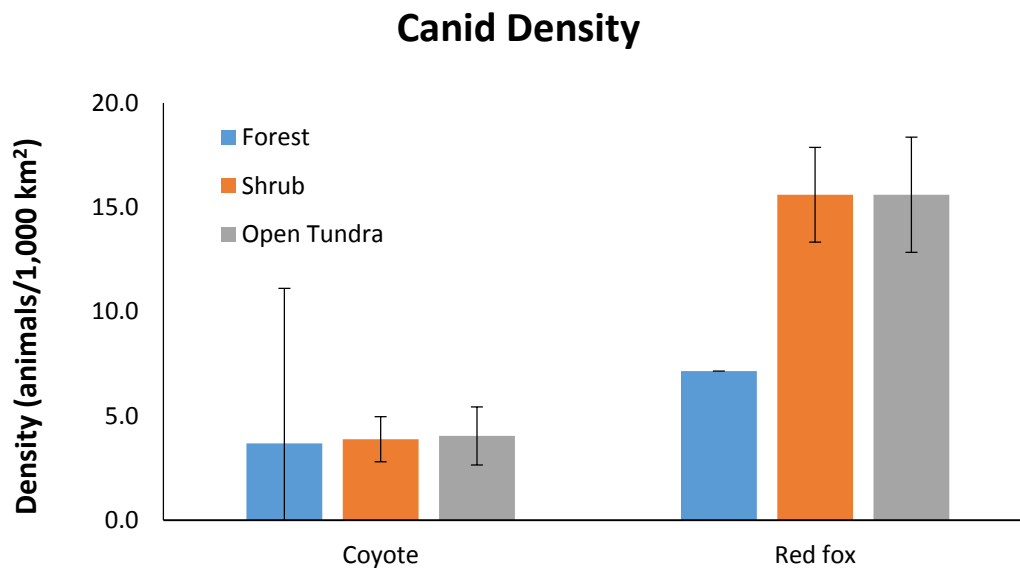
Canid Density

Figure 5.3-1. Model-averaged Density Estimates, with Standard Error, of Coyotes and Red Foxes during the 2014 Winter Survey Season in the Terrestrial Furbearer Study Area. 2013 and 2014 density estimates were nearly identical. Estimates are broken down by major habitat type. Variance estimates for red fox density in forest habitats were not estimable.