MONITORING OCCURRENCE AND HABITAT USE BY RIVER OTTERS, *LONTRA CANADENSIS*, ACROSS NEW YORK STATE

by

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A thesis
submitted in partial fulfillment
of the Master of Science Degree
State University of New York
College of Environmental Science and Forestry
Syracuse, New York
September 2018

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ACKNOWLEDGEMENTS

A Master of Science thesis is the result of effort from many sources and could not have been accomplished alone.

First and foremost, thank you to my major advisor, Dr. Jacqueline Frair, for countless rounds of edits, and always pushing me to improve my work in the name of good science. Bi-weekly meetings, potlucks, and lab outings help create a sense of community in the Frair lab that is essential for success in graduate school.

To the QSL, thank you for your invaluable feedback on my work, and patience with helping me learn new statistical techniques, but mostly, thank you for your friendship.

Chellby Kilheffer, Alison Devlin, Lisanne Petracca, Chee Peng Low, Lilian Bonjourne de Almeida, and Dina Matunkhina, thank you for your mentorship and setting a great example of a high-standard of research that I try to emulate. You are inspiring researchers and role-models.

Paul Picciano, thank you for your friendship, hunting guidance, and frequent references to The Office, the best show to ever grace television.

To Sam Peterson, the moose man, your friendship, shared venison, and moral support helped shaped my graduate studies and I am glad that you taught me the proper way to say Colo-rad-o.

Alex Barham and the NYS whitewater community, thank you so much for taking me under your wing and helping me learn to navigate the rivers of upstate NY, definitely a highlight of my time in central NYS.

Brandon Swart, thank you for making field season infinitely more enjoyable, with your sense of humor and positive attitude, and life-saving carpentry skills (that still come in handy!).

To my committee, and former and current members of the NYSDEC furbearer team, especially Scott Smith, Andrew MacDuff, and Paul Jensen, thank you for your guidance and feedback, shaping my thesis into the final product you see here.

To my co-workers, both at the Rhode Island Air National Guard and RI Dept. of Environmental Management, thank you for your support, helpful reviews of my work, and tolerance of my scheduling requirements as I coordinated my work and graduate school schedules these past few months, your assistance has been invaluable.

And finally, thank you to my parents, who remained supportive through the highs and lows, tolerated me skipping family events and eating dinner alone upstairs, all in the name of good science, a love of the river otter, and a stubborn determination to finish a task once I commit to it. I appreciate you more than you can know.

Table of Contents

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

ABSTRACT

K.M. Powers. Monitoring occurrence and habitat use by river otters, *Lontra canadensis*, across New York State, 80 pages, 6 tables, 9 figures, 2018. Journal of Wildlife Management style guide used.

Assessing the current distribution of river otter, *Lontra canadensis*, populations closed to harvest is a challenge faced by today's wildlife managers. I explored 2 non-invasive methods for assessing otter distribution in this thesis. First, I used a disparate set of otter occurrence records to model probability of otter occurrence across central and western NYS as a function of aquatic habitat and landscape disturbances. The model validated well (R^2 =0.90) using recent survey records and indicates approximately 20% of western NYS to contain high quality habitat for otters. Second, a pilot study using motion-sensitive cameras in conjunction with floating platforms proved successful in detecting aquatic wildlife but was not ideal for monitoring river otters. I provide recommendations that might increase the utility of motion-sensitive cameras for future river otter surveys. The results of this study provide valuable insights for managers researching non-invasive sampling methods for the elusive river otter.

Key words: river otter, *Lontra canadensis*, non-invasive surveys, opportunistic data, cameras

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PROLOGUE

Historically the North American river otter, *Lontra canadensis,* was widespread and abundant across the United States and Canada (Melquist et al. 2003). In the early 1900s, river otters became a species of special concern following large-scale habitat degradation and unregulated harvest across their range (Conner 1971, Polechla 1990, Burns 2014). River otters were significantly reduced in number across central New York State [NYS] by the 1930s. In response to this sharp decline, a 9-year harvest moratorium was imposed starting in 1936 (NYSDEC 2018). For the next half century, river otters in NYS persisted largely in isolated strongholds in the Adirondack and Catskills mountain ranges but remained functionally extirpated from central NYS (NYSDEC 2018). Over the course of the late twentieth century, 21 states, including NYS, implemented river otter reintroductions, either importing otters from external sources (e.g., other states, farms) or reintroducing otters from remnant populations within their respective state (Raesly 2001). The NYS Department of Environmental Conservation [NYSDEC] partnered with the non-profit New York River Otter Project Inc. to reintroduce river otters across central and western NYS between 1995-2000 (Burns 2014, NYSDEC 2018). Over the course of this 5-year period, 278 otters were trapped from the Catskills and Adirondack regions and transported to Cornell University for a complete physical and 2-week period of "fattening" prior to release at 22 sites across central & western NYS (Burns 2014, NYSDEC 2018). At the time of the initial translocation, NYSDEC had regional and site-specific follow-up studies planned (Spinola et al. 1999, NYSDEC Region 9) but had not yet established a comprehensive, long-term monitoring program.

Amongst states that have undertaken a river otter reintroduction, commonly used methods to monitor post-translocation population trends have included short-term telemetry studies (Erickson & McCullough 1987, Spinola et al. 1999) and winter sign surveys (NYSDEC Region 9). Telemetry surveys have demonstrated the high post-translocation survival rate of individual otters within the years directly following a translocation. However, the study-size of many of these surveys has been small (N <30) due to the intense manpower effort required to manually track each individual with an implanted VHF transmitter (Spinola et al. 1999, Spinola et al. 2008). Another issue with telemetry-based tracking efforts is that they are inherently short-term (~2.5 years) due to the lifespan of the radio-transmitters (Raesly 2001). Certain states (e.g., PA, NY) have implemented bridge-based track and sign surveys during winter as an alternative method to assess post-translocation trends (Serfass et al. 2003, NYSDEC Region 9 unpublished data). Many of the post-translocation sign surveys have been limited to short time periods (<10 years) and concentrated solely on a few sites within reintroduction areas (Serfass et al. 1993, Spinola et al. 1999, Raesly 2001). While these follow-up efforts have demonstrated localized, short-term success rates of translocated populations, there is a glaring lack of long-term studies assessing the post-translocation distribution of the species across the entirety of the translocation area (Serfass et al. 1993, Raesly 2001).

In NYS, efforts have been similar to those used by other states to monitor post-translocation populations of river otters – involving short-term, localized radio telemetry studies (Spinola et al. 1999, Spinola et al. 2008), annual bridge-based sign surveys in NYSDEC Region 9 (A. Rothrock, NYSDEC, unpublished data) with episodic surveys in other regions, intensive latrine site surveys in areas where translocated otters were previously released (Burns 2014), and recording incidental otter sightings across the state (NYSDEC, unpublished data). Collectively, these efforts indicate that otters have become widespread and increased in abundance over time (A. Rothrock, A. MacDuff, NYSDEC, personal communication; J. Frair, SUNY-ESF, unpublished data). However, these efforts have suffered from the same shortcomings as those implemented in other states; largely short-term (Serfass et al. 1993, Spinola et al. 1998, Spinola et al. 2008), localized (NYSDEC Region 9 bridge-surveys), and lacking a cohesive methodology that inhibits inference across the entirety of the recovery zone. Recently, the NYSDEC partnered with the State University of New York College of Environmental Forestry [SUNY-ESF] to assess the distribution of river otters across the entirety of the recovery zone via exploration of existing data, assessment of non-invasive survey methods, and bridge-based track and sign surveys extended to an occupancy modeling framework (J. Frair, SUNY-ESF, unpublished data). This thesis is a subset of that larger statewide research.

In this thesis, I examined 2 methods for assessing the distribution of otters across the river otter recovery zone. The first method focused on a data set of otter occurrence records across central and western NYS arising from 4 different sources (sign surveys, incidental harvests, road kills, opportunistic sightings) during the post-translocation period (2001-2012; Chapter One). I used ad hoc statistical corrections to address survey biases inherent in the data set, and the R package MAXLIKE (Royle & Chandler 2012) to develop a model for probability of otter occurrence across the otter recovery zone. I validated this model with an independent set of otter occurrence records collected by NYSDEC during their winter 2016-17 bridge-based track and sign surveys. This study (Chapter One), provides a baseline assessment of the potential distribution and relative abundance of otters across the recovery zone. The second method tested the feasibility of an experimental remote camera survey for river otters in aquatic habitat across a large study area. Whereas cameras deployed at latrine sites are a useful means of evaluating behavioral patterns and group sizes (Burns 2014), randomization of survey locations relative to known otter use sites is important for monitoring populations using occupancy-based analyses.

So, I tested the ability of randomly located cameras trained on a floating platform with lures to attract and detect river otter use of multiple types of aquatic habitat across central and northern NYS. This study (Chapter Two), provides a potentially efficient means of monitoring aquatic wildlife.

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CHAPTER 1: MAPPING SPECIES DISTRIBUTION FROM DISPARATE OCCURRENCE RECORDS: CASE STUDY OF RIVER OTTERS IN NEW YORK STATE

ABSTRACT Opportunistic records of animal occurrence may be problematic for inferring species distribution and habitat requirements due to unknown and uncontrolled sources of sampling variance. In this study, I used occurrence records for river otters, *Lontra canadensis*, derived from surveys, road kills, trapper by-catch, and opportunistic sightings (N = 185 records collected 2001-2012) to assess the potential distribution and habitat relationships of otters across central and western New York State. To mitigate for obvious observation biases, I equilibrated observation intensity across regions *a priori* and restricted inference to readily accessible areas (i.e., ≤700 m from the nearest road). Model selection, and the direction of covariate effects, proved robust to these sampling biases although effect sizes varied −7.1% to +48.0% after bias correction, with the coefficient for the proportion of available shoreline habitat being the most volatile. Ultimately, the top bias-corrected model proved a reliable index for otter probability of occurrence given a strong, positive, and linear relationship with a withheld set of standardized survey records for otters collected winter 2016-17 (N = 57; R^2 =0.90). This model indicated that ~20% of the study area represented high probability of otter occurrence at the time of this study. I demonstrated that reliable inference on the drivers and quality of wildlife habitat can be obtained from disparate records of animal occurrence provided data biases are known and effectively mitigated.

KEY WORDS occupancy, otters, *Lontra canadensis*, MAXLIKE, species distribution models

Knowledge of species distribution and habitat drivers is essential for guiding management actions (Shenk & Franklin 2001) such as protecting or enhancing habitat (Hooker et al. 1999, Jennings 2000, Hamazaki 2002, Martin et al. 2012), predicting changes in animal distributions (Murray et al. 2017), and improving survey designs (Reilly and Fidler 1994). For some groups of species, such as songbirds, standardized survey designs and formal frameworks for data-sharing promote robust data for modeling of species occurrence over broad spatio-temporal scales (e.g. Breeding Bird Survey; Robbins et al.1989, Sauer et al. 2017). For mammals however, such standardized, large-scale data collection is typically lacking (O'Connell et al. 2006). Innovations in noninvasive sampling methods, such as camera 'traps' (Gompper et al. 2006, O'Connell et al. 2006), passive genetic monitoring (O'Connell et al. 2006, Depue & Ben-david 2007, Burns 2014), and acoustic sampling (Rodhouse et al. 2011, Hansen 2013) have greatly increased available occurrence records for a wide range of elusive mammals. However, sampling designs using these methods remain an active area of study and have not yet yielded an effectively standardized approach across species and geographic regions (Long et al. 2008). As a result, broad-scale evaluations of mammal occurrence likely require drawing together disparate sources of animal occurrence data (Pittman et al. 2017) and mixing data from formal population surveys and opportunistic sightings (e.g., road kill locations), presenting challenges for rigorous statistical inference.

Species distribution models [SDMs] are used to detect statistical relationships between locations where species are known to occur and environmental and landscape covariates (Elith & Leathwick 2009). A major assumption of many SDMs is a constant detection probability over time and space; if that assumption is not met, heterogeneity in detection probability should be modelled. To account for sightability bias, occurrence records from standardized surveys might include a measure of effort (e.g., time spent surveying), local survey covariates known to influence detectability (e.g., proximity of roads, ambient noise levels), or other data related to spatio-temporal variation in detection probability (e.g., number of neighboring blocks in which the species was detected; MacKenzie et al. 2006). It is unlikely that any such information would be available with opportunistic records of animal occurrence, such as road kill locations, but it does not necessarily follow that opportunistic records are not informative with respect to species distributions. Lacking the ability to account for variation in species detection, and even when detection probability can be accounted for, one might rigorously validate model predictions using withheld, or out-of-sample, records of animal occurrences (Boyce et al. 2002). Validation enables an assessment of the conditions under which model predictions are valid and can help illuminate potential biases or uncertainties when making predictions. Another important assumption underlying SDMs is that the sample of occurrence records is derived from a random sampling process (Royle & Chandler 2012), so that each member of target population has a chance of being sampled. Faced with a substandard sampling design, robust inference might be gained by fitting separate models to the data collection and observation processes, (e.g. via Bayesian-hierarchical modeling; Carroll et al. 2010, Aing et al. 2011). However, doing so requires some understanding of the data collection process for each data source, which may not be available for disparate records of animal occurrence. Moreover, fitting models for the observation process becomes challenging given multiple data sources that have different underlying sampling processes, especially if data from any given source are scarce. Such formal statistical corrections involve a non-trivial level of complexity that may limit their accessibility to many conservation practitioners. As a result, uncertainty of the sampling process and the resulting effects on detection probability may cause potentially useful records of animal occurrence to be underutilized. Given that "all models are wrong, but some are useful (Box and Draper 1987)," one might instead attempt ad hoc corrections for obvious sources of bias in incidental occurrence records and apply a rigorous validation approach to evaluate the degree of utility of those bias corrections and model predictions.

The objective of this study was to evaluate the degree to which disparate records of river otter occurrence reliably reflect otter distribution and important habitat drivers across a large study area spanning central to western New York State [NYS]. The traditional means of tracking otter population trends in the northeastern United States is via monitoring annual trapping returns, but the study area has been closed to otter harvest since 1993 (Gotie 1991, Burns 2014). Otters were translocated into the region between 1995-2000 to enhance population recovery. Since 1998, verified records of otter occurrence in this region have been collected from opportunistic sightings reported by, and disparate sign surveys conducted by New York State Department of Environmental Conservation [NYSDEC] biologists, incidental harvests by licensed beaver trappers, and vehicle-related mortalities (S. Smith, NYSDEC, personal communication). Sign surveys involved searching 200-m transects of selected river banks once each winter and were carried out inconsistently among NYSDEC administrative regions (A. Rothrock, NYSDEC, personal communication). A greater number of otter sightings were recorded in western NYS, where translocations were conducted, compared to survey areas neighboring robust otter populations in the eastern portion of the state. In fact, very few incidental records of otter occurrence were reported in core otter range in the Adirondack and Catskill Mountains – a persistent problem with common species. In the study area relatively more ‘developed’ landscapes had a higher ‘apparent’ survey effort, which, if left uncorrected, may mask or bias statistical relationships between anthropogenic landscape features and otter distribution (Betts et al. 2007). Another obvious potential bias in these data stemmed from records being dependent to large degree on road accessibility (Keller & Scallan 1999, Betts et al.

2007). Even the standardized sign-based surveys were initiated at the intersection of rivers and roads, and thus road bias may be consistent across the data sources available for this study. I assumed these sampling biases to be important, but also to some degree correctable *a priori*. Herein, I tested the impacts of ad hoc bias corrections on (1) the identification of important variables for predicting the probability of otter occurrence, and (2), the direction and magnitude of the estimated covariate effects. I further assumed that 'survey' characteristics were optimized for otter detection (or at least constant across data types), and that site characteristics did not influence otter detection probability (Jeffress et al. 2011). As a result, I did not employ corrections for the probability of detection. Instead, I assessed model utility by employing a rigorous validation approach using an independent set of otter occurrence records stemming from a contemporary survey design implemented across the recovery zone. Ultimately, I mapped high, moderate, and low-quality habitats for otters with confidence across the recovery zone, aiding management planning for this iconic species in this region.

STUDY AREA

The study area encompassed ~53,300 km^2 of central and western NYS (Figure 1.1). The region was a heterogeneous mixture of forest (44% of the landscape) and agricultural lands (35%; USGS 2014). Forests included canopies dominated by deciduous (33%), conifer (4%), and mixed coniferous-deciduous (7%), with dominant canopy species in the region being maple (*Acer* spp.), beech (*Fagus grandifolia),* birch (*Betula* spp.), oak (*Quercus* spp.) and hickory (*Carya* spp.; Riemann et al. 2014, USDA 2015). Agricultural lands included both pasture/rangeland and cropland. Terrain conditions ranged from relatively flat in the Great Lakes Plain ecoregion (north-central; mean elevation = 238 m, +/- 119 m) to rugged conditions in the Catskill Mountains (southeast; mean elevation = 372 m, +/-159 m). Aquatic habitats for river otters included large lakes (~1.6% of the study area), such as the Finger Lakes (USGS 2013), open marshes and wetlands (~7,600 km^2), and numerous second+ order streams and rivers. The region is characterized by cold, snowy winters extending from October through March, and warm, humid summers with an average temperature of -5.4°C in January and 20.0°C in July (NOAA 2018).

METHODS

A total of 233 confirmed locations of river otters collected from 2001-2012 were available for this study. These data originated from 4 sources: bridge-based sign surveys conducted by NYSDEC staff (46%; surveys involved a single visit to a site/winter with 200 m of shoreline searched/visit), incidental reports from NYSDEC employees (37%), by-catch from trappers verified by NYSDEC staff (13%), and confirmed road kills (4%; A.J. MacDuff, NYSDEC, unpublished data). I restricted all sightings records to the winter season (November-March), the period providing the greatest number of records, to control for potential differences in seasonal detection probability. I assumed population closure for the purposes of modeling given that the winter period excluded the pup-recruitment season and the area was closed to otter harvest. The two sources of observation bias mitigated in this study involved: 1) a broad-scale effort bias, and 2) a fine-scale road bias. Within the raw data set, each region had a unique sightings density, indicating a difference in survey effort (Table 1.1). To mitigate the broad-scale bias, I equilibrated reporting rates *a priori* across administrative regions by randomly rarefying or augmenting observations to the average sighting density across all regions of 1 sighting/330 km^2 (Frair et al. 2004; Figure 1.1, Table 1.1). In an effort to mitigate survey effort bias by region via data augmentation, I necessarily incurred a violation of the independence assumption of occupancy analyses (MacKenzie et al. 2006), potentially over-fitting the model to certain points in the data set, which should be noted. This yielded a total of 185 observations available for further analysis. To mitigate the fine-scale bias, I sampled 'pseudo-absence' or background locations, those referenced in comparison to used locations, only within a 700-m wide buffer around all roads in the landscape (i.e., the maximum distance away from roads that a verified otter sighting occurred). Hansen et al. (2015) indicated that land cover types and terrain conditions sampled along roads differed from sample points placed randomly across the landscape by <2%. Moreover, less than 1% of the study area occurred at a distance >1 km from the nearest road. As a result, I expected the restricted sampling design to reasonably represent the range of habitat conditions available to otters across the entire study area.

Using the R package MAXLIKE (Royle & Chandler 2012), I estimated the probability of otter occurrence (ψ) via logistic regression. I expected the probability of river otter occurrence to be driven primarily by the amount of suitable aquatic habitat (Melquist & Hornocker 1983, Jeffress et al. 2011). I resampled all GIS layers used in this study to 250-m resolution prior to analysis. To quantify the availability of aquatic habitat, I first extracted linear features representing rivers, lakes, ponds and open marshes from National Wetlands Inventory data (USFWS 2009), and representing rivers ≥40 m wide from National Hydrography Data (USGS 2013). These linear features were combined and converted to a binary raster (shoreline = 1, no shoreline = 0), from which I derived the proportion of cells within radii of 1-, 5- and 10-km containing shoreline. These radii were chosen to encompass daily otter movement patterns (~0-1km/hour [Martin et al. 2010]; 1-12 km/day [Melquist & Hornocker 1983]; ~3.5 km/night [Wilson 2012]). Within each of these extents, I calculated the proportion forest cover (deciduous, conifer, mixed, or forested wetland) and proportion agricultural cover (pasture or cropland), derived from the 2011 National Land Cover Data product (USGS 2014; Homer et al. 2015). I further quantified road density (km/km^2) within each extent as an index to the level of human disturbance, based on NYS public roads data (Winters 2016). Lastly, local elevation (mean) and degree slope (rise/run) were derived for each cell from Digital Elevation Model data (USGS 1994). The value of each landscape metric was extracted at each retained otter ($y_i = 1$; N = 185) and pseudo-absence ($y_i = 0$) location. Landscape data were managed using ArcMap v10.2.2 (ESRI, Redlands, CA). All covariates were standardized using a z-transformation prior to model fitting (Schielzeth 2010, Jeffress et al. 2011).

Candidate models to estimate ψ included suites of uncorrelated variables (pairs of variables having Pearson $r < 0.7$ when $P < 0.05$; Dormann et al. 2013), as well as linear (x) and quadratic ($x+x^2$) covariate effects. Correlations among landscape variables were assessed *a priori* from 10,000 random points across the study region, and resulted in the exclusion of proportion forest cover from candidate models due to an inverse correlation with elevation ($r > 0.7$). A separate candidate model set including proportion forest cover was evaluated (Appendix C). I evaluated models using Akaike's Information Criterion corrected for small sample sizes [AIC_c], with AIC_c model weights ($\boldsymbol{\omega_i}$**;** Burnham & Anderson 2002) used to gauge relative support.

Bias assessment

Model selection was conducted using the fully-corrected occurrence (or presence) data, with the top 10 models refitted to partially corrected data (either effort or road bias correction applied) or the raw, uncorrected otter sightings to determine whether biases influenced model selection, the sign or magnitude of estimated coefficients, or the ranking of covariate influence. I refitted models to the original 233 incidental otter records without any adjustments to assess the effect of broad-scale effort bias, and selected background locations as described earlier to account for the fine-scale road bias. In contrast, to assess the effect of the fine-scale road bias, using the observations adjusted for broad-scale effort bias, I sampled background locations within the entirety of the study area rather than sampling only within areas ≤700 m from a road, adding an additional 3,207 km^2 of sampling area. Finally, I fit models to the data without any bias corrections. In each case, I ranked models by AIC$_c$ score to evaluate if the specific bias influenced model selection decisions. For the top model, I examined the direction of estimated coefficients and quantified for each the magnitude of change relative to the model fit to the fully corrected data.

Model validation and application

Using the most parsimonious model (based on the fully corrected data), I predicted ψ for each 250-m cell across the study area and extracted the predicted value at a withheld set of 57 otter observations acquired from a formalized, statewide, bridge-based sign survey conducted by NYSDEC staff during winter 2016-17. Sign surveys were conducted on second+ order streams (N = 1,362 survey sites), initiated at bridges and terminated up to 400-m from the road, and were undertaken when snow tracking conditions were favorable. Following Johnson et al. (2006), I compared the proportion of withheld sightings observed against the proportion expected within 10 bins of predicted ψ values. I calculated the expected utilization of each bin *i* as:

$$U(x_i) = {w(x_i)A(x_i)}\Big/{\sum_j w(x_j)A(x_j)}$$

where $w(x_i)$ was the midpoint value of ψ for bin *i* and $A(x_i)$ was the amount of area of the landscape corresponding to bin *i*. Next, I calculated the number of expected observations within each bin as $N_i = N \times U(x_i)$, given N = 57, and converting the result to a proportion (N_i/N). I considered the model valid, and its predictions useful, given a positive, linear relationship between N_i and the count of observed otter locations occurring within each *i* bin, and I measured the fit of that relationship by the coefficient of determination (R^2) from a linear model.

In practice, resource managers are likely to rank areas as habitat or non-habitat, or high versus low habitat quality, rather than manage habitat quality on a continuous scale. Therefore, following model validation I divided predicted ψ into categories representing high, moderate, and low habitat suitability for otter. To identify appropriate cutoffs between categories, I reorganized 10 bins of predicted ψ such that each bin corresponded to 10% of the areal extent of the study area following Boyce et al. (2002). Using this convention, by random chance alone one would expect 10% (p = 0.1) of withheld otter locations to correspond to each bin (i.e., use ≈ availability). By extension, bins having $p < 0.1$ (i.e., observed use disproportionately lower than expected at random) would indicate low habitat suitability and bins having $p > 0.1$ (i.e., use disproportionately greater than expected) would correspond to high habitat suitability.

RESULTS

The most parsimonious model for ψ received strong support (ω_i = 0.91; Table 1.2), with the important variables, in declining order of influence, being proportion shoreline cover (1-km radius), road density (5-km radius), local degree slope, and proportion agricultural land cover (5-km radius; Table 1.3). The top model indicated peak ψ where 9% of the surrounding landscape (within a 1-km radius) was covered by shoreline, with ψ declining towards zero in landscapes having ~0% and >20% shoreline while holding all other site covariates at their mean (Figure 1.2A). The model further indicated a negative relationship between ψ and proportion agriculture (5-km radius), degree slope, and road density (5-km radius; Figure 1.2B-D).

Bias assessment

Refitting the top 10 models to partially- or fully-uncorrected data yielded the same conclusions regarding the set of influential covariates (same top model received >90% of AIC_c model weight in each case; Table 1.2), effective covariate form (i.e., nonlinear fit to proportion shoreline detected in all cases), and the direction of coefficient effects (Table 1.2). However, failing to account for the broad-scale effort bias yielded somewhat greater confidence regarding the top model (from $\Delta AIC_c = 5.65$ and $\boldsymbol{\omega_i} = 0.91$ based on fully corrected data to $\Delta AIC_c = 8.19$ and $\boldsymbol{\omega_i} = 0.95$ given correction for road bias but not effort; Table 1.2), likely due to its influence on estimated coefficient values. In particular, correcting for regional differences in effort substantially modified coefficient values for proportion shoreline (Table 1.3; Figure 1.2). Overall, broad-scale effort bias contributed ~2.5 times more change in observed effect sizes (21.0% on average) as fine-scale road bias (8.4% change on average; Table 1.3). Failing to adjust for fine-scale road bias had the greatest effects on degree slope and proportion shoreline, with the least overall effect observed in the road density coefficient.

Model validation and application

The top model predicted the frequency of withheld otter observations well ($R^2 = 0.90$). Plotting withheld data against equal-area bins of $\widehat{\psi}$ indicated 20% of the recovery zone to be "highly suitable" for otters (bins 9, 10; corresponding to value of $\widehat{\psi} > 0.14$), an additional 30% of the landscape was deemed of intermediate/moderate suitability (bins 6-8; $0.04 > \widehat{\psi} \leq 0.14$), and 50% of the landscape was of low suitability (bins 1-5, $\widehat{\psi} \leq 0.04$; Figure 1.3).

DISCUSSION

Gaining robust inference on animal space use and important habitat drivers from disparate, opportunistic records of animal occurrence requires formal assessment and mitigation of potential sampling biases and proper validation of model predictions. Two common sources of potential biases with opportunistic sightings data – large-scale discrepancies in effort and small-scale sampling restrictions to road-accessible areas – were apparent in the river otter data set used in this study and were readily mitigated via ad hoc corrections.

Some inferences, such as which variables were important and the direction and form of their effects (quadratic vs. nonlinear), proved robust to the 2 biases identified, although effect sizes differed to varying degree depending upon the kind of bias corrected and the process of bias correction. Correcting for the fine-scale road bias involved eliminating ~5.7% of the study area (3,207 km^2) as being considered available to otters (specifically areas >700 m away from a road), which shifted the distribution of 'background' points in space without changing sample size. In contrast, in controlling for the larger-scale effort bias I adjusted sample size and, by extension, estimate precision and statistical power, while retaining the geographic position of reference points. Interestingly, coefficient bias resulting from road biased observations were not as one might anticipate. Correcting for road bias had the greatest effect on coefficients related to degree slope ($\%\Delta_\beta = +14.2$) and proportion shoreline ($\%\Delta_\beta = +12.7$), with the least change observed in the road density coefficient ($\%\Delta_\beta = +5.2$; Table 1.3). In contrast, adjustments to mitigate effort bias had an overwhelming effect ($\%\Delta_\beta = +31.3$ for x, -85.7 for x^2) on the coefficients for proportion shoreline, with minimal ($\%\Delta_\beta < |6|$) effect on the remaining covariates, perhaps due to geographic variation among administrative regions in the amount of aquatic habitat that influenced either observation or reporting rates.

An important assumption of the R program MAXLIKE, one that was not addressed directly in this study, is that sightings have a constant probability of detection (Royle & Chandler 2012). In all likelihood, detection probability varied over space and time both in the incidental records data used to fit models and in the out-of-sample surveys used to validate model predictions. The greatest proportion of the data used in this study (46%) came from bridge-based sign surveys. Bridge-based surveys involve sample controls both in the sites chosen (e.g., typically second+ order streams away from developments) and dynamic surveys conditions (e.g., survey on days meeting optimal snow conditions). Detection probabilities in otter sign-based surveys have been shown to vary with the length of shoreline searched but not with additional site or survey covariates (Jeffress et al. 2011). The sign surveys contributing data to this study were uniformly conducted along 400-m transects, and thus detection probability may have been relatively uniform. However, the remainder of the data (opportunistic reports from NYSDEC employees [37%], accidental by-catch from trappers verified by NYSDEC staff [13%], and confirmed road kills [4%]) each has its own unique detection probability, so the issue is combining these multiple data sets from various sources with inherently different detection probabilities. A major concern with variable probability of detection is a model biased towards sites of high probability of occurrence for river otters (Royle & Chandler 2012). I recommend that the predicted occurrence probability from the top model should be interpreted as a relative rather than true probability of occurrence.

The bias-corrected model indicated that shoreline habitat was the most influential factor for otter occurrence, and that otter response to shoreline availability was nonlinear—peaking at $\Psi = \sim 0.8$ when ~9% of the landscape within a 1-km radius consisted of shoreline habitat and declining thereafter (the maximum amount of shoreline within 1-km radius in the study area was 23%). Ninety percent of the available landscape represents 0 to 2.8% shoreline. The remaining 10% of the landscape ranges from 2.8% to 23%. The proportion of shoreline that corresponded with peak probability of otter occurrence (0.09) is characterized by freshwater forested and freshwater shrub wetlands on the peripheral edges of larges lakes and rivers. The nonlinearity in otter response to percentage shoreline likely reflects their dependence during winter on larger rivers, lakes and ponds that remain to some degree open during winter and enable feeding and ready land access (Melquist & Hornocker 1983). Smaller aquatic features, those yielding higher proportion shoreline values within 1-km radius, likely freeze over in winter and become unavailable to otters. In contrast, spring through summer, otters likely make greater use of marshlands to access amphibians and crustaceans (Day 2015), aquatic environments likely to have more complex edges compared to larger lakes and rivers and, as a result, have a higher percentage of shoreline within a 1-km^2 radius.

In line with the literature, the top model indicated that otters avoided areas of high road density (Robitaille & Laurence 2002, Gorman et al. 2006). Roads can be considered a proxy for direct human disturbances and risks such as potential mortality from vehicle collisions (Philcox et al. 1999). Roads have other negative effects in freshwater systems such as increased water pollution from run-off (Forman & Alexander 1998) that may decrease habitat suitability for species like the river otter. Areas with a high proportion of agriculture, which were avoided by otters, have similar issues to areas with high road density (increased runoff, high levels of pollution, and potential eutrophication issues from fertilizer runoff; Wang et al. 1997). The central portion of the study area was strongly influenced by agriculture (~35% of the total land cover was agriculture). As areas around waterways were developed with roads and agriculture, water quality suffered across NYS, leading to issues that may have affected the prey base for otters (Wang et al. 1997). Otters, a bioindicator species, were themselves directly affected by aquatic pollution throughout the state, as evidenced by mild to moderate heavy metal contamination (Hg, Cd, Pb) reported in samples of otter tissue at disparate points across the study area (Mayak 2012). So, although my model predicted 50% of the landscape to be of potentially moderate-high suitability for otters, the actual distribution and abundance of otters across this landscape will further depend upon factors not directly modeled in this study such as contaminant levels and prey availability. Importantly, discordance between areas my model predicts as highly suitable and survey returns might be used to identify areas where local site mitigations might be needed to realize otter habitat potential.

MANAGEMENT IMPLICATIONS

Useful insights into the distribution of river otters across NYS and its habitat drivers were obtained from low cost, opportunistic observation records, the kind of data that may often be distrusted, and thus underutilized, or overly trusted, and thus utilized improperly by resource managers (Yoccoz et al. 2001, Giraud et al. 2016). Moreover, the ad hoc data corrections employed in this study were straightforward and accessible, in contrast to the more sophisticated statistical methods typically used to model the detection process (e.g., Bayesian hierarchical models). As a result, the approaches applied here are readily transferable to other species and systems, although not all incidental sighting records may yield results similar to what I obtained for river otters in NYS. Importantly, any use of incidental records of animal occurrence in statistical models should carefully consider potential data biases and conduct formal validation to help guide management actions.

This study indicated that ~20% of the targeted recovery zone (~1,000 km^2) is of potentially high suitability for otters at present, and agreed with other models of otter occurrence that shoreline habitat and road density were primary drivers of otter space use. Continued collection of incidental sightings may be useful for informally monitoring otter populations but will not replace formal population survey data needed to track spatio-temporal changes in populations over relatively short time intervals. As a result, bridge-based sign surveys will likely remain the primary means of monitoring otters in areas closed to harvest, and the current study indicates those data are highly compatible with opportunistic sighting records. As a result, future sign surveys in NYS might be stratified based on the predictions of high, intermediate and low-quality habitat predicted in this study. Effective stratification can improve survey efficiency both in field-data collection and in the precision of statistical estimates. Future incidental sightings can in turn be used to validate models based on bridge-surveys given the high compatibility observed between incidental sightings and bridge-based survey observations in this study.

ACKNOWLEDGEMENTS

I would like to thank members of the NYSDEC, particularly the furbearer team, Region 8 biologist Scott Smith, and Region 6 manager Andrew MacDuff for their assistance in collecting the data and guidance in utility of a final project. Additionally, thank you to the NYSDEC field technicians who have helped collect data on river otters since the translocation period. I would also like to recognize SUNY-ESF faculty and members of the Quantitative Studies lab at SUNY-ESF for their help and guidance.

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Table 1.1. Data sources, distribution, and corrections for incidental sightings of otters across central and western New York State (2001-2012). The number of original otter locations (N_i) and percentage corresponding to each administrative region and observation source are indicated. Data sources included standardized sign surveys (survey), opportunistic sightings reported by New York State Department of Environmental Conservation [NYSDEC] personnel (report), incidental take by licensed trappers (harvest), and confirmed road-related mortalities (mort). Given gross differences in the density of observations by NYSDEC region, sighting density was equilibrated *a priori* to the mean (1 observation/330 km^2) by randomly augmenting or rarefying the number of records in each region as indicated (bias correction), yielding 44-50 occurrence records per region used in this modeling exercise (N_f).

DEC Region	Observation Source (%)				N_i	Area (km^2) per sighting	Bias correction	N_f
	Survey	Report	Harvest	Mort				
4	61%	24%	9%	7%	53	289	–7	46
7	98%	0%	0%	2%	90	164	–45	45
8	4%	58%	30%	8%	26	639	+24	50
9	64%	25%	11%	0%	64	227	–20	44
						$\bar{x} = 330$		

Table 1.2. Top 10 logistic regression models to predict occurrence probability (ψ) for river otters across central and western New York State as a function of the degree slope (S) and elevation (E) at each site as well as road density (RD), proportion agriculture (PAg), and proportion shoreline (PSh) quantified within circular buffers around each location (radii of 1-, 5-, or 10 km as indicated within parentheses). Quadratic responses are indicated by $x \pm x^2$. For each model, the model log-likelihood ($\log\mathcal{L}$), number of estimated parameters (K), difference in AIC_c value from the top model (ΔAIC_c), and AIC_c model weight (ω_i) are given. For the same models fit to either partially corrected or uncorrected data, only the difference in AIC_c value is reported to show if data biases led to model selection uncertainty.

		Corrected				Partially corrected		Uncorrected
Rank	Covariates	$\log\mathcal{L}$	K	ΔAIC_c	ω_i	$\Delta AIC_{ceffort}$	ΔAIC_{croads}	ΔAIC_c
1	$PSh_{(1)} \pm PSh_{(1)}^2$, $PAg_{(5)}$, S, $RD_{(5)}$	–1723.06	5	0.00	0.91	0.00	0.00	0.00
2	$PSh_{(1)}$, $PAg_{(5)}$, S, $RD_{(5)}$	–1726.94	4	5.65	0.05	5.59	8.19	8.30
3	$PSh_{(1)}$, $PAg_{(5)}$, S, $RD_{(5)}$, E	–1726.82	5	7.53	0.02	7.64	7.35	8.45
4	$PSh_{(1)}$, $PAg_{(10)}$, S, $RD_{(10)}$, E	–1727.45	5	8.79	0.01	7.16	8.64	6.84
5	$PSh_{(1)}$, S, $RD_{(5)}$	–1734.77	3	19.22	<0.01	16.81	26.63	23.71
6	$PSh_{(1)}$, $PAg_{(1)}$, S, $RD_{(5)}$, E	–1733.68	5	21.25	<0.01	19.34	28.49	26.66
7	$PSh_{(1)}$, $PAg_{(1)}$, S, $RD_{(10)}$, E	–1733.74	5	21.37	<0.01	18.72	28.60	24.71
8	$PSh_{(1)}$, $PAg_{(10)}$, S, $RD_{(1)}$, E	–1737.87	5	29.62	<0.01	26.87	42.75	39.01
9	$PSh_{(1)}$, $PAg_{(5)}$, $RD_{(5)}$	–1739.90	3	29.49	<0.01	28.53	51.23	49.91
10	$PSh_{(1)}$, $PAg_{(5)}$, $RD_{(5)}$, E	–1739.04	4	29.85	<0.01	29.16	44.13	43.67

Table 1.3. The AIC_c-selected top model predicting the probability of otter occurrence (ψ) across central and western New York State using data corrected *a priori* for broad-scale (regional) and fine-scale (road-related) observation biases. For each variable, I report the estimated coefficient (β) with standard error (SE). After refitting the same model to data uncorrected for one or both biases, I report the direction of the estimated coefficient (sign) along with the magnitude of change in effect size, reported as the percentage change in β value (Δ_β), compared to the model fit to corrected data. Coefficient estimates that were significantly different from zero given $P < 0.05$ are indicated by '*'.

	Corrected data Top model		Partially corrected data: Broad-scale (effort bias)		Partially corrected data: Fine-scale (road bias)		Uncorrected data	
Variable	β	SE	Sign	Δ_β	Sign	Δ_β	Sign	Δ_β
Intercept	−2.94	0.38*	−	+1.7%	−	+3.0%	−	+3.9%
Proportion shoreline	+1.51	0.27*	+	+31.3%	+	+12.7%	+	+48.0%
Proportion shoreline^{2}[a]	−0.13	0.03*	−	−85.7%	−	−8.3%	−	85.7%
Road density[b]	−1.28	0.29*	−	−5.8%	−	+5.2%	−	−0.8%
Degree slope[c]	−0.91	0.20*	−	0.0%	−	+14.2%	−	+13.3%
Proportion agriculture[b]	−0.61	0.15*	−	−1.7%	−	−7.0%	−	−8.9%
			$\lvert\bar{x}\rvert =$	21.0%		8.4%		26.8%

[a]Measured within a circular buffer having radius = 1 km; [b]Measured within a circular buffer having radius = 5 km; [c]Measured within each 90-m cell.

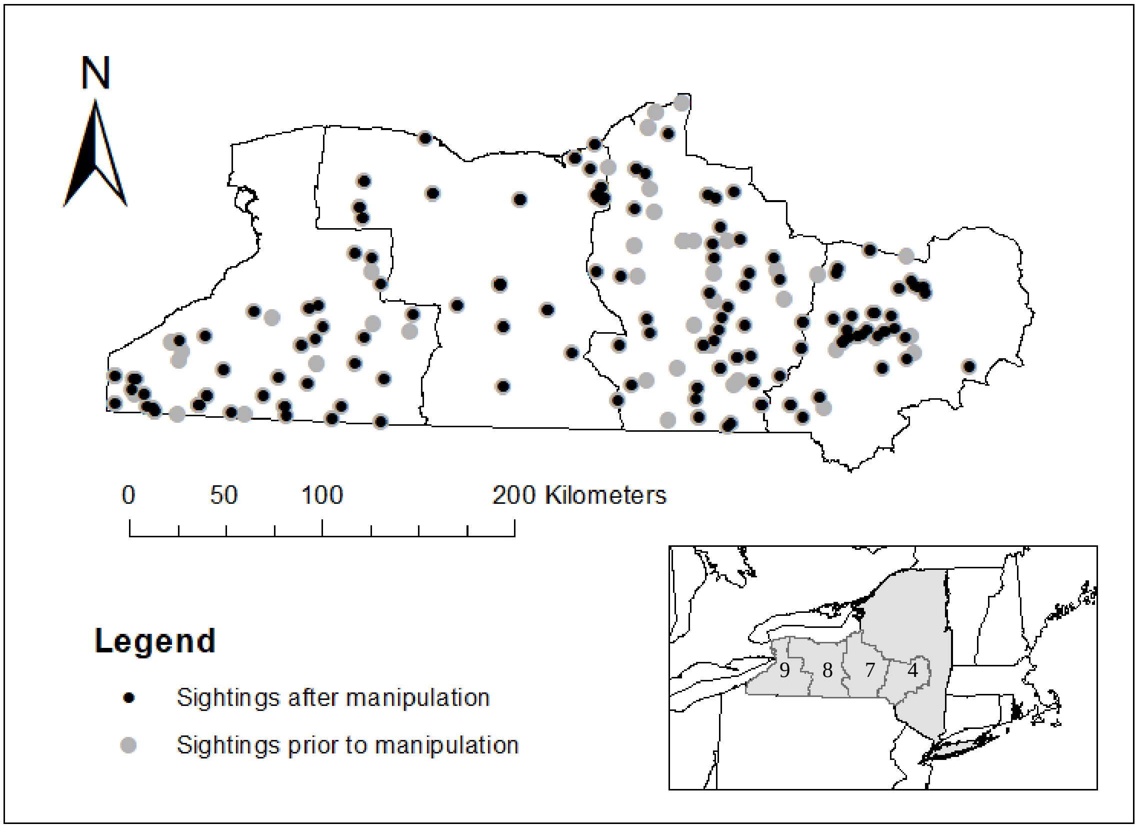

Figure 1.1. 53,300 km^2 study area (also referred to as "otter recovery zone") in central and western New York State. 235 verified otter observations prior to effort bias corrections (Methods) are shown as gray dots. The black dots are the final 185 observations used in the modeling processes. The 4 New York State Department of Environmental Conservation [NYSDEC] administration regions that contributed data to this study are shown in the inset map.

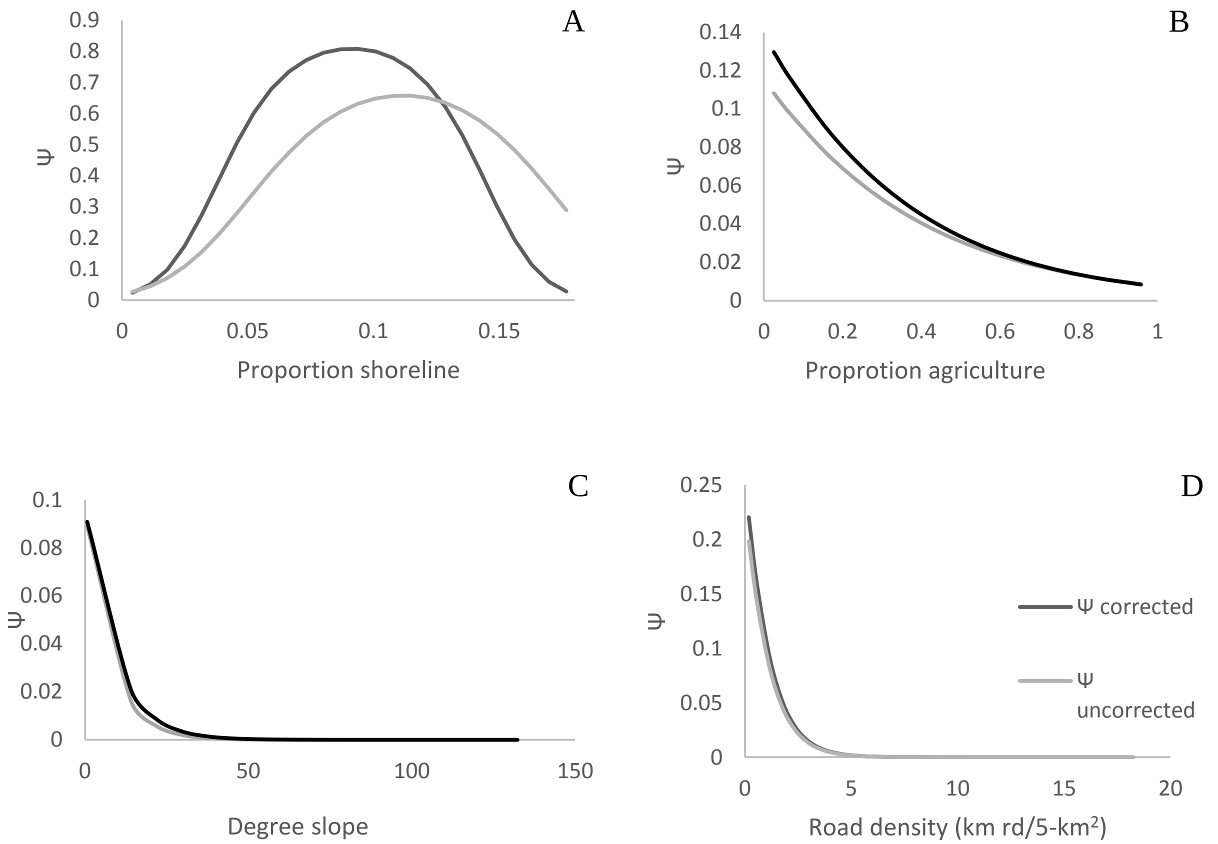

Figure 1.2. Partial slopes for the predicted probability of otter occurrence (ψ) given proportion shoreline cover (measured within a 1-km radius; A), proportion agricultural cover (5-km radius; B), percentage slope (C), or road density (5-km radius; D) while setting other site covariates to a value of zero. Lines indicate the relationships inferred from uncorrected sightings (gray) versus sightings corrected *a priori* to account for large- and small-scale observation biases (black).

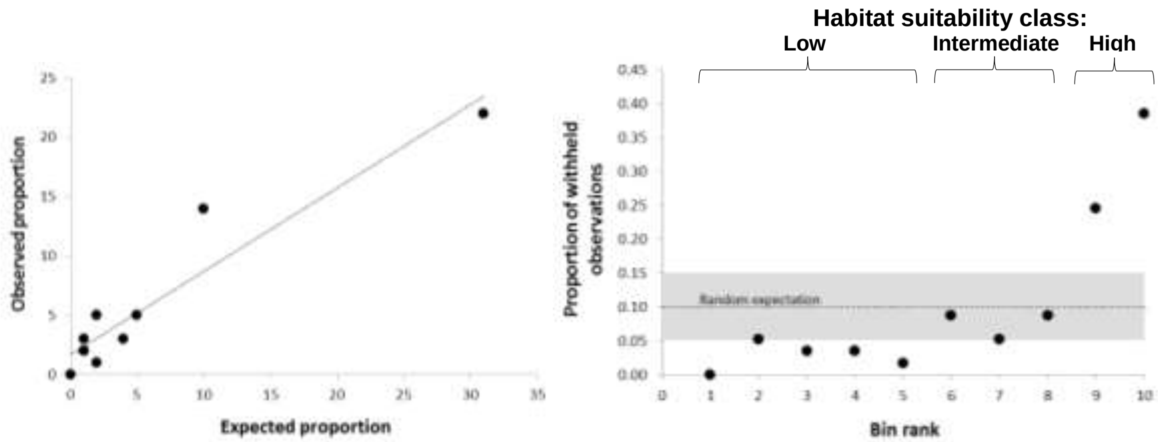

Figure 1.3. Model validation (A) and application to determine habitat suitability classes (B). In A, a model was fit between the expected and observed counts of withheld otter observations (N = 57) corresponding to 10 bins of predicted ψ based on the most parsimonious model. In B, bins of predicted ψ were reorganized such that each bin encompassed 10% of the areal extent of the study area, leading to the expectation that 10% of observed otter locations should correspond to each bin by random chance, with higher proportions reflecting higher habitat quality and lower proportions reflecting lower quality.

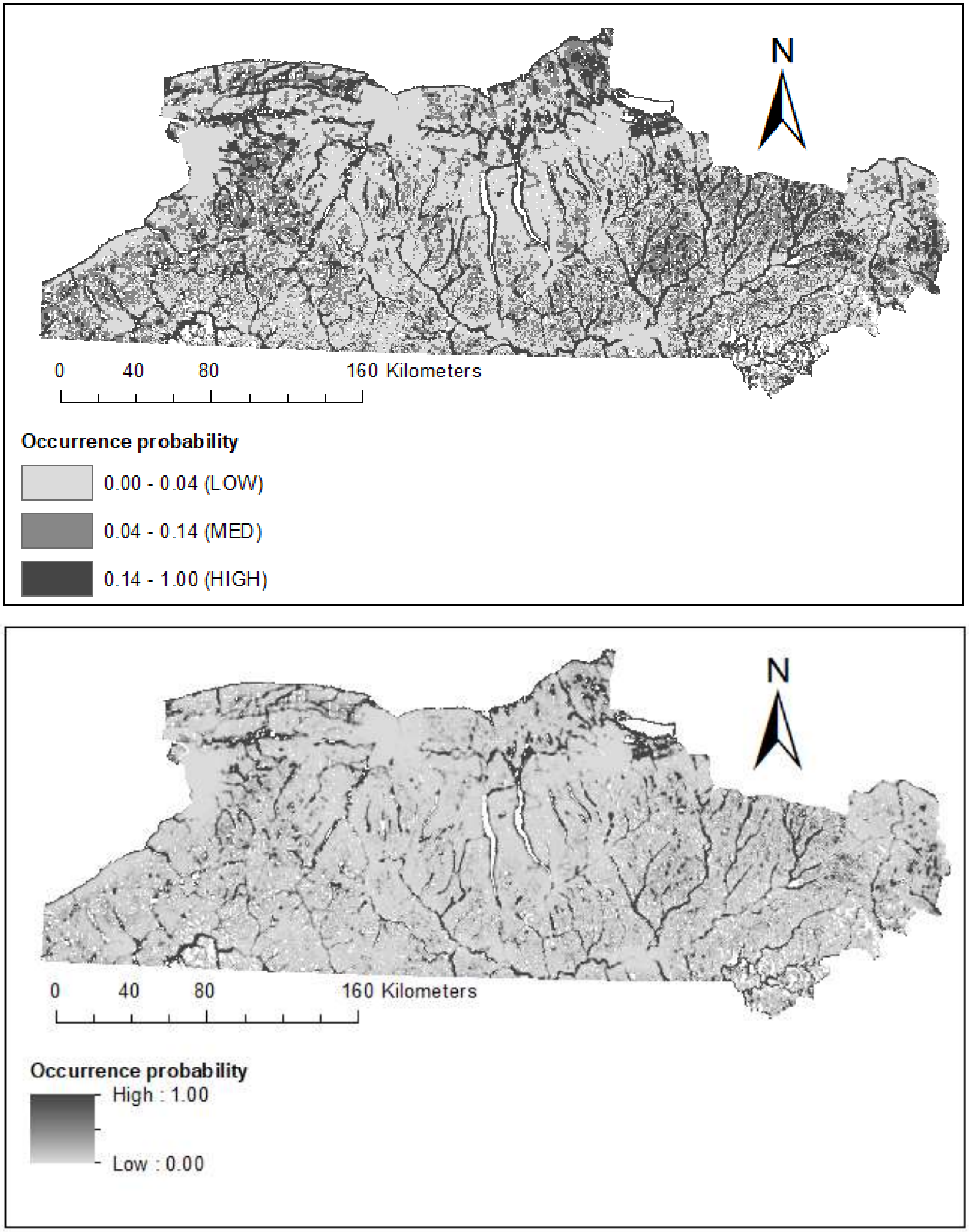

Figure 1.4. (A) Areas predicted to be of low (0.00 < Ψ < 0.04), medium (0.04 < Ψ < 0.14), and high (0.14 < Ψ < 1.00) suitability for river otters in central and western New York State, passed on top model for probability of otter occurrence. (B) Continuous map of probability of otter occurrence (0 ≤ Ψ ≤1).

Appendix A. River otter sightings from NYSDEC winter bridge-based track and sign surveys used as the "with-held model validation data" (J. Frair, SUNY-ESF, unpublished data). "FID" indicates the field identification number of the point (0-56), "bin" is the number of the model validation bin that points fell into (bins were numbered from 1-10, each containing ~10% of the land area for the study region, with bin 1 having the lowest occurrence probability and increasing through bin 10), "region" is the NYSDEC region that the observation was recorded in (each region conducted their own survey based on a pre-agreed upon study-design), "easting" and "northing" are the respective UTM coordinates for each observation, and "predicted ψ" is the occurrence probability predicted by the model at the location of each sighting.

FID	**Bin**	**Region**	**Easting**	**Northing**	**Predicted ψ**
10	2	9	261623	4742904	0.00727927
28	2	8	296729	4706216	0.000551888
29	2	4	512341	4702691	0.430413
8	3	4	538915	4744501	0.022457
47	3	8	311341	4675626	0.0101075
26	4	8	334053	4709270	0.0306228
35	4	8	287378	4693784	0.0275109
49	5	8	280852	4666827	0.0306027
6	6	6	442423	4761278	0.051477
18	6	4	505356	4726769	0.0750008
22	6	7	387742	4715576	0.0162333
43	6	8	338515	4683152	0.110378
45	6	9	262904	4678576	0.0429565
0	7	7	370441	4803300	0.0828742
51	7	4	514843	4663097	0.0898619
54	7	7	376355	4659246	0.00991141
4	8	8	333162	4774741	0.0754286
13	8	4	537175	4737573	0.0631964
19	8	7	394576	4724200	0.0405434
27	8	8	337816	4709143	0.0564703
31	8	8	297956	4701301	0.123127
1	9	8	253837	4794753	0.147321
3	9	8	275383	4774310	0.201472
7	9	9	249938	4748929	0.234341
11	9	4	511078	4737460	0.154775
14	9	8	267377	4735448	0.194106

20	9	9	238199	4726509	0.199958
21	9	4	556190	4723868	0.22902
32	9	7	445413	4697826	0.196327
33	9	8	307080	4698124	0.21685
40	9	9	165439	4684270	0.210031
41	9	7	433992	4683797	0.318889
42	9	8	328153	4683422	0.222835
44	9	9	221070	4679640	0.27814
52	9	7	444505	4663780	0.137193
2	10	8	232918	4778035	0.694337
5	10	6	427067	4771930	0.676626
9	10	8	294322	4740529	0.637544
12	10	4	528544	4737053	0.448393
15	10	8	319268	4730150	0.491484
16	10	8	294411	4731565	0.527917
17	10	4	480720	4727124	0.293653
23	10	9	205697	4713111	0.629554
24	10	4	515413	4710688	0.694482
25	10	7	425883	4709358	0.475997
30	10	8	296745	4703272	0.345954
34	10	8	328095	4698713	0.644517
36	10	8	357368	4693058	0.750734
37	10	8	347031	4692345	0.744645
38	10	8	359275	4694445	0.736908
39	10	8	346641	4694040	0.706167
46	10	7	455721	4672800	0.435043
48	10	8	266989	4670189	0.281909
50	10	9	204377	4666390	0.583187
53	10	9	106956	4663390	0.481801
55	10	8	332208	4657477	0.387038
56	10	7	382010	4654000	0.401916

Appendix B. Mean (μ)and standard deviation (σ) used to center and standardize (Schielzeth 2010) the environmental covariate layers using the raster calculator function in the spatial analyst toolbox in ArcMap v.10.2.2. All layers were centered and standardized using the mean and standard deviation from the road-bias corrected layer.

	Road-bias corrected		uncorrected	
Variable	μ	σ	μ	σ
Intercept	—	—	—	—
Proportion shoreline	0.01	0.01	0.01	0.01
Proportion shoreline^2	0.01	0.01	0.01	0.01
Road density	1.79	1.22	1.76	1.20
Degree slope	6.22	7.88	6.37	8.08
Proportion agriculture	0.36	0.20	0.36	0.20

Appendix C. Top 10 logistic regression models to predict occurrence probability (ψ) for river otters across central and western New York State as a function of the degree slope (S) at each site as well as road density (RD), proportion forest cover (PFor), and proportion shoreline (PSh) quantified within circular buffers around each location (radii of 1-, 5-, or 10 km as indicated within parentheses), for the data corrected for road and effort bias. Quadratic responses are indicated by $x \pm x^2$. For each model, the model log-likelihood ($\log\mathcal{L}$), number of estimated parameters (K), difference in AIC_c value from the top model (ΔAIC_c), and AIC_c model weight (ω_i) are given.

		Corrected			
Rank	Covariates	$\log\mathcal{L}$	K	ΔAIC_c	ω_i
1	$PSh_{(1)} \pm PSh_{(1)}^2$, $PFor_{(10)}$, S, $RD_{(5)}$	–1718.55	5	0.00	0.95
2	$PSh_{(1)}$, $PFor_{(10)}$, S, $RD_{(5)}$	–1722.78	4	6.35	0.03
3	$PSh_{(1)}$, $PFor_{(5)}$, S, $RD_{(5)}$	–1724.10	4	8.98	0.01
4	$PSh_{(1)}$, $PFor_{(10)}$, S	–1728.02	3	13.74	<0.01
5	$PSh_{(1)}$, S, $RD_{(5)}$	–1734.77	3	28.23	<0.01
6	$PSh_{(1)}$, $PFor_{(1)}$, S, $RD_{(10)}$	–1733.79	4	28.36	<0.01
7	$PSh_{(1)}$, $PFor_{(1)}$, S, $RD_{(5)}$	–1734.34	4	29.52	<0.01
8	$PSh_{(1)}$, $PFor_{(10)}$, $RD_{(5)}$	–1737.83	3	34.35	<0.01
9	$PSh_{(1)}$, $PFor_{(1)}$, S, $RD_{(1)}$	–1743.79	4	48.36	<0.01
10	$PSh_{(5)}$, $PFor_{(10)}$, $RD_{(5)}$, S	–1768.66	4	98.10	<0.01

Appendix D. Beta values (β), standard errors (SE), and confidence intervals for 4 model sets (bias corrected, effort-bias corrected, road-bias corrected, uncorrected).

		Bias-corrected			Effort-bias corrected			Road-bias corrected			Uncorrected		
Variable	Radius	β	SE	*P*	β	SE	*P*	β	SE	*P*	β	SE	*P*
Intercept	—	−2.94	0.38	<0.01	-2.99	0.37	<0.001	-3.03	0.35	<0.001	-3.06	0.34	<0.001
Proportion shoreline	1 km	+1.51	0.27	<0.01	+1.15	0.21	<0.001	+1.34	0.20	<0.001	+1.02	0.15	<0.001
Proportion shoreline^2	1 km	−0.13	0.03	<0.01	-0.07	0.02	<0.001	-0.12	0.03	<0.001	-0.07	0.01	<0.001
Road density	5 km	−1.28	0.29	<0.01	-1.21	0.29	<0.001	-1.35	0.26	<0.001	-1.27	0.25	<0.001
Degree slope	90 m	−0.91	0.20	<0.01	-0.91	0.20	<0.001	-1.06	0.18	<0.001	-1.05	0.18	<0.001
Proportion agriculture	5 km	−0.61	0.15	<0.01	-0.60	0.15	<0.001	-0.57	0.13	<0.001	-0.56	0.13	<0.0

CHAPTER 2: USING CAMERAS TO MONITOR RIVER OTTER USE OF LACUSTRINE AND PALUSTRINE WATER BODIES, NEW YORK STATE

ABSTRACT The North American river otter, *Lontra canadensis*, is a notoriously difficult species to detect due to low population densities and elusive behavior. Sign surveys are one of the most commonly employed methods for monitoring otter occurrence, but such surveys are time and labor intensive, and sign may be ambiguous, necessitating research into alternative methods to increase both survey efficiency and certainty. In this study, I tested an experimental method using a combination of motion-sensitive cameras and floating platforms to detect river otter use of different types of water bodies. From June 2016 to July 2017, 65 platforms were deployed at 8 independent sites across central and upstate New York State, with each site having a known history of river otter activity. Multiple camera-platform arrays were deployed at each site, ~100 -1000 m apart. Cameras detected 19 species but detected otters at 2 sites and only after 88 days of camera deployment. The low detection rate for otters precluded statistical analyses from this pilot study, but I provide recommendations to improve the efficiency of motion-sensitive cameras for future use in larger-scale otter surveys.

KEYWORDS river otters, *Lontra canadensis*, motion-sensitive cameras, New York State

The northeastern United States hosts an abundance of riverine, lacustrine, and palustrine environments that support river otters (*Lontra canadensis*) and other aquatic wildlife. Otters are highly vagile (Greer 1953, Melquist & Hornocker 1983, Spinola et al. 2008) and of generally low density in the Northeast (Roberts 2010), posing logistical challenges for population monitoring where regular monitoring data, such as that available from harvest records, may not exist. This is the case in central and western New York State [NYS], a region designated for otter population recovery in which harvest has been closed since 1993 (Gotie 1991, Burns 2014) and into which otters were translocated 1995-2000. Efforts to track recovery of the river otter population in the region have involved winter bridge-based sign surveys (A. Rothrock, NYS Department of Environmental Conservation [NYSDEC], unpublished data), summer collection of DNA (Burns 2014), monitoring of otters at latrine sites via remote cameras (Burns 2014), and incidental sightings (Chapter One). Whereas incidental sightings may provide information on habitat drivers influencing otter distribution (Chapter One), the infrequency of records within a given year combined with a lack of randomization and controls limits utility for formal modeling of changes in their distribution over space or time. DNA-based surveys of latrine sites proved effective in terms of identifying individual animals and their spatio-temporal use of known latrine sites (Burns 2014). But the effort extended to locate latrine sites was non-trivial, restricting the applicability of this approach to smaller geographic regions. Moreover, despite a reasonable number of detected latrine sites and recovered DNA samples from the pilot study in western NYS, the recapture rate for otters proved too low for a reliable population estimate (Burns 2014). As a result, bridge-based sign surveys are likely to remain the primary means of monitoring otters in the region.

Sign surveys are commonly employed for otters throughout their range (Reid et al. 1987, Sulkava 2007, Jeffress et al. 2011) and are used to provide either an index of otter abundance or more formal assessments of population status based on occupancy modeling. Yet, sign left by otters and other species may be ambiguous, which poses challenges to properly accounting for otter detection probability. Moreover, sign surveys are highly dependent on substrate for detection, seasonally limiting their utility to winter months (Reid et al. 1987, Sulkava 2007) that pose hazardous conditions to surveyors. Specific to central and western NYS, sign surveys carry a non-trivial burden in terms of the time involved in acquiring landowner permissions given that the great majority of land in NYS is privately-owned. Whereas sign surveys remain the primary means of monitoring otter populations in the region, alternative, more efficient and effective means of monitoring are desired.

Motion-sensitive cameras have recently increased our ability to more effectively monitor elusive and wide-ranging mammalian species (O'Connell et al. 2006, Kays & Slauson 2008, McCallum 2013). Camera studies involving river otters are a recent addition to survey methodology (see Stevens et al. 2004 for first targeted use of cameras and otters), and most surveys involving river otters have focused on latrine site usage (Stevens et al. 2004, Stevens & Serfass 2005, Burns 2014). Latrine sites are locations, often a hummock or peninsular outcropping, where multiple otters defecate and leaves spraints, feces, and anal glandular secretions (jellies) during repeat visitations to these same sites (Greer 1953, Melquist & Hornocker 1983, Ben-david et al. 1998). Studies incorporating cameras at latrine sites have refined camera technique in riparian habitat (Stevens et al. 2004), defined behavioral trends of otters (e.g., sliding behavior as play v. locomotion; Stevens & Serfass 2005), informed visitation rates and seasonal peaks in otter latrine site usage (Burns 2014), and documented minimum group size during otter latrine visits (Burns 2014). While informative, use of cameras at latrine sites has limitations as a survey method for otter distribution. Monitoring at latrine sites requires prior knowledge of where the latrines are located, which is a time-consuming effort (Burns 2014, Powers, personal observation). While otters maintain annual site fidelity to certain latrines (A. MacDuff, NYSDEC, personal communication), latrine use has been documented to change seasonally (Burns 2014), restricting potential camera deployment to certain windows of use.

Another issue posed by utilization of cameras at latrine sites is restricted analytical options for data acquired from this method. Otter presence at a site is verified upon identification of a latrine, so surveying only latrine sites violates the random-sampling assumption inherent for a variety of statistical techniques. O'Connell et al. (2006) recorded otters on randomly placed motion-sensitive cameras during a study in Cape Cod comparing the detection rates among 3 survey methods (cameras, cubby-holes, and hair snares). Of the methods, cameras had the highest detection probability across the 10 mammalian species recorded. Otters were only recorded by camera, albeit with a comparably low occupancy estimate (ψ) compared to the other species recorded. Despite the low ψ, the study demonstrated the ability of using randomly placed motion-sensitive cameras to record presence/non-detection of river otters for the purpose of occupancy monitoring (O'Connell et al. 2006). Each of the aforementioned studies demonstrated the potential for monitoring elusive aquatic mammalian species such as the river otter with motion-sensitive cameras, but acknowledged the need for further research to develop effective survey designs and refine technique.

In 2013, a USDA-APHIS study targeting the invasive nutria (*Myocastor coypus*) deployed camera traps in combination with floating platforms in the Chesapeake Bay area of Maryland and incidentally detected river otters utilizing the floating platforms (Kerr & Dawson 2013, B. Wilmouth, USDA APHIS, personal communication). Based on the reoccurring incidental sightings of river otters in the USDA APHIS study, I explored the utility of the floating platform technique to record presence/non-detection of river otters in central and western NYS. The objectives of my study were to 1) test motion-sensitive cameras linked to floating platforms as a means of detecting otter occurrence in lacustrine, palustrine, and riverine environments in New York State, and 2) evaluate the potential for extension of camera traps to a broad-scale study of otter occupancy.

STUDY AREA

For this pilot study I selected sections of rivers, lakes, ponds, and emergent marshes within 8 different sites across the state—the Huntington Wildlife Forest within the central Adirondack Park region, Tug Hill State Forest on the periphery of the Adirondacks, Northern Montezuma Wildlife Management Area [WMA] and High Tor WMA in the central Finger Lakes Region of the state, and Cicero Swamp WMA, Three Rivers WMA, Labrador Hollow Unique Area (UA), and Nelson Swamp UA within central NYS (Figure 2.1). At any given site, depending upon suitable shoreline area, camera arrays were established within multiple water bodies. Available aquatic habitat within the region consisted of freshwater lakes, rivers, marshes, and ponds (USGS 2013). Palustrine environments were sampled at High Tor WMA (Figure 2.2A), Tug Hill State Forest, Three Rivers WMA, and Northern Montezuma WMA; slow moving riverine environments were sampled at High Tor WMA (Figure 2.2B), Cicero Swamp WMA, and Nelson Swamp Unique Area; and lacustrine environments were sampled at Huntington Wildlife Forest and Labrador Hollow Unique Area. Water bodies having swift currents (i.e., wide, fast-moving rivers) were not included in this pilot study due to the potential for dynamic water levels affecting camera effectiveness given large potential movements in the floating platform. In 2016, sites ranged in size from 25-61 km^2 and were selected based on recent confirmed sightings of otters (tracks, latrine sites, or visual observations). In 2017, sites were concentrated in central NYS, ranged in size from 4-30 km^2, and were selected to contain medium to high probability otter habitat based on Powers 2018 (Chapter One).

METHODS

Platforms were constructed following the nutria hair-snare monitoring study (Kerr & Dawson 2013; B. Wilmouth, USDA APHIS, unpublished data; design modified by removing side rails and hair snare brushes) at a cost of ~$10.00 per platform. Platforms were constructed from 1-cm thick plywood sheets cut into a 0.6 × 0.6-m square under-mounted with a 2.5-cm thick sheet of housing insulation foam. Four fender washers and cabinet screws secured the foam to the plywood, with 4 pieces of scrap wood covering the sharp ends of the screws to prevent any injury to animals that may utilize the platforms (SUNY-ESF IACUC protocol #160401). A 4-cm diameter hole was drilled through the top center of the foam and wooden platforms to allow insertion of 3-cm diameter, 1.5-m section of PVC tube. The tube was inserted into the muddy substrate to secure the platform in place. A 3-cm diameter hole was drilled at the lower right corner of the platform to hold a second, 1.2-m long garden stake, which prevented changes in platform orientation that might trigger camera sensors. Platforms were secured in aquatic habitats ranging 0.3-1.3-m depth, within 5 m of the shoreline (Figure 2.3A).

A camera was setup 1 to 3 m from each platform and within 1 m of the water surface, focused on the platform, and secured using cable ties and cable locks to a tree or to heavy-duty metal stakes secured into the substrate (Figure 2.3A). Platforms were baited with a combination of 3 scent lures (Caven's Otter Lure Supreme, Caven's Gusto, and crayfish oil) mixed with petroleum jelly, with lure reapplied on a bi-weekly schedule (weather and schedule permitting). Likewise, cameras were checked bi-weekly, to ensure proper functioning and replace SD cards (8 GB) and batteries as needed (batteries replaced when >50% depleted). Deployed cameras were either Browning™ Spec Ops (Browning® Trail Cameras, Morgan, UT) or Reconyx

Hyperfire™ PC800 (RECONYX Inc., Holmen, WI), set to record a 4 or 5-round burst with a 5-second delay.

In 2016, a total of 29 arrays were deployed at 4 independent sites (4-11/site; Table 2.1) between 16 June and 9 November (see Appendix D for full site details). In 2017, a total of 36 camera arrays were deployed at 5 independent sites (4-10/site; Appendix E) between 18 April and 26 July. Within a site, I spaced cameras ~100-m apart depending upon available aquatic habitat (Figure 2.2), with the intention of later rarifying records to identify optimal camera spacing for future surveys. All photos were uploaded to Colorado Parks and Wildlife Photo Warehouse v.4 (Newkirk 2016). A single observer (K. Powers) identified all species.

RESULTS

Cameras were deployed from 62-145 days/site, summer-fall, in 2016 and 52-95 days/site in spring 2017 (Appendices D and E, respectively). During the spring survey, cameras recorded 503,078 photos, 94% of which were triggered by ambient motion of the platform or vegetation, leaving 6% (28,975) that contained photos of animals. Of the photos that contained an animal 7,542 (26%) were of a mammal, 21,433 (74%) were of an avian species, and 36 photos (<1%) were unidentifiable. Cameras detected a total of 19 species (including humans and dogs); 12 mammalian species and 7 avian species (Table 2.2; Appendix F).

River otters were detected at only one site in each season. In fall 2016, river otters were recorded by cameras within High Tor WMA, and within only one water body (upland pond area, not the lowland riverine area; Figure 2.2A and Figure 2.4). These otters were first detected 88 days following camera deployment (September 2016). After the initial detection of otters at High Tor WMA, 6 independent revisits of the site were recorded given 7 hours to 19 days between visits, and with the length of recorded visits ranging from 3 seconds to 25 minutes ($\bar{x} =$

00:04:24; Table 2). Visits were considered independent if they were separated by ≥30 minutes following Burns (2014) and Newkirk (2016). Over the course of these visits, otters were detected at a total of 4 out of 6 camera arrays, although during a given visit they were detected by only a single camera array. Photos recorded 1 to 4 individual otters per visit, with animals observed on and around the platform. Recorded activities of otters included traveling and feeding, with the longest recorded visit in 2016 (00:25:10) recording feeding behavior of 4 otters utilizing the platform as a base for eating frogs (*Rana* spp.; Figure 2.4A). In spring 2017, a single river otter was detected at 1 of 10 cameras at Three Rivers WMA, for a total of 4 recorded pictures on 28 April 2017 from 11:42:06-11:42:08. The otter swam by the platform but did not physically utilize the platform (Figure 2.5).

DISCUSSION

The information derived from the limited photos captured of river otters in this study supported past research with otters and cameras. Otter activities were largely crepuscular to nocturnal (Stevens & Serfass 2008, Burns 2014; Table 2.2), with 85.7% of visits taking place between 03:10:29 and 10:33:56 in 2016, and one outlier associated with feeding behavior (time = 14:03:38, duration = 00:25:10). The majority (87.5%) of recorded visits were ≥00:03:40, with the same outlier (t = 00:25:10) skewing the average visitation time to 00:04:24 (with outlier removed, $\bar{x}$ = 00:00:45). Group size of otters in the photos was also consistent with the literature, with otters roaming either individually or in small family groups (Melquist & Hornocker 1983, Burns 2014). In this study 42.9% of visits recorded a single otter, and minimum group size ranged between to 4 individual otters.

Camera-platform arrays were an efficient means for surveying wetland wildlife, requiring ~6 hours per site/bi-weekly for monitoring purposes, not including travel time. The bulk of the effort was related to the initial platform deployment, which took ~2 hours/platform due to the difficulty accessing and identifying suitable aquatic habitat for the platforms (e.g., appropriate water depth, proximity to trees or shallow areas to stake in posts for camera etc.), and carrying or canoeing cumbersome equipment to the sites, majority of which were only accessible by foot or boat. While I recorded a substantial number of wetland species, this approach did not prove fruitful for monitoring river otters, my target species, over the time period of this study. I failed to detect otters even in areas where sightings of animals or their sign had indicated recent use of the site by otters. The lack of otter detections in this study may be due to a variety of factors, including but not limited to, restricted movements of females and young-of-the year during the months directly post-parturition (spring, summer), large home-range sizes, lack of attraction to lures due to status as an active predator (P. Jensen, NYSDEC, personal communication), and seasonal landscape usage combined with low overall population density. Limited mobility of female otters has been documented from March to mid-August when females are giving birth (March – April; Hamilton & Eadie 1964) and young mothers are accompanied by nursing pups (Melquist & Hornocker 1983, Trani et al. 2007), indicating that my arrays may have been too geographically limited in their distribution. Females restrict their movements post-parturition to remain close to their young, and mother and pup groups may restrict movements to ≤10 km through October of the birthing year (Melquist et al. 2003). In addition, otter home ranges vary from 8–78 km (Melquist & Hornocker 1983). Large home-range size, compounded by the tendency of females and their young-of-the-year to restrict movements to within range of the den, makes intensive sampling of a portion of the home-range, such as what was covered by the WMAs, state forests, UAs, and research areas in this study, insufficient.

One goal of this study was over-saturation of aquatic habitat to allow later rarefication of platform spacing to develop an ideal study design for occupancy analysis. However, deployment of platforms was constrained by availability of aquatic habitat and suitable sampling conditions. Platforms require very specific environmental characteristics for deployment (e.g., accessible shoreline, water depth ≤1.3 m, minimal current), restricting the habitat suitable for deployment. Due to these restrictions, I may not have saturated a large enough area to detect otters. To increase the probability of detecting otters, cameras should be spaced out across areas the size of a minimum home-range for females (≤10 km; Melquist et al. 2003), as opposed to concentrated in specific aquatic habitat. Deploying cameras only on aquatic habitat with areas <10 km^2 in size is likely to incur the low rate of detections that this study observed. But sampling broader regions around WMA's involves accessing private lands and diminishes the efficiency of this approach as a surrogate for sign-based surveys. Nevertheless, otter detections by camera were far less subject to error in interpretation compared to assessment of otter sign.

Otters are a difficult species to lure to a site (Bohrman 2012). Otters are active predators, feeding on live prey such as seasonally available fish, crayfish, and amphibians (Melquist & Hornocker 1983, Day 2015, Paul Jensen, NYSDEC, personal communication). Due to their behavior as an active predator, it is difficult to attract otter using a lure (Paul Jensen, NYSDEC, personal communication), such as the scent lures used in this study. Bohrman (2012) demonstrated that there was no statistically significant difference between a lure and control in attracting captive river otters to a scent station. Each platform in this study was designed to lure otters in 2 ways; 1) as a physical stimulant to the natural curiosity of river otters, and 2) using scent, via baiting of the platforms with 3 common lures used by otter trappers. Based on the results of the Bohrman (2012) study, I hypothesized that lures would either have a null effect, or a positive effect, but not a negative effect on otter detection. The lack of photos recorded in this study, despite use of scent-lured platforms reinforced the results of Bohrman (2012) that lures have an overall null effect on otter behavior. Despite known otter presence on the landscape (during the 2016 season) even lure combined with a physical attractant (the platform) was not enough to attract the otters out of their pre-determined trajectory.

Otters' use of the landscape varies seasonally, with peak visitation to latrine sites in winter and late fall (February/March and October/November, Stevens & Serfass 2008; April and October, Burns 2014), and the lowest recorded visitation rates during summer months (Greer 1953, Stevens & Serfass 2008, Burns 2014). Likewise, my observations of otters occurred only in spring (late April) and fall (September through November) despite the large majority of the survey period taking place during the summer months (May-August). The suitable window for platform deployment, which requires ice free waters, runs exactly opposite to the trend in latrine site visitation rates for otters. It is likely that during ice-free periods, many more latrine sites are available to otters than those detected and surveyed as it is unlikely that their need for defecation is seasonally reduced. Activity at known latrine sites increases in winter and late fall likely as a result of restricted movement abilities due to icing. Based on low summer latrine site visitation rates reported in the literature, restricted movement of female and pup groups, and lack of detections during summer months over the course of this study, platform deployment is not recommended for May through August. I recommend deployment of camera arrays as early as possible in spring (late March through April, prior to parturition) or as late as possible into the fall (October through November). Fall may be the more favorable sampling period for future surveys, as spring poses additional challenges due to vegetation development and fluctuating water levels.

Spring surveys are necessarily more intensive than fall surveys using this design. Spring 2017 (April – May) had the highest incidences of camera interference (and resulting false detections) due to rapidly changing water levels in central NYS, from a combination of spring rains and final snowmelt. Moreover, rapid vegetative growth interfered with camera efficiency, obscuring the camera lenses, triggering the camera due to wind, and filling memory cards with empty photos. Stevens and Serfass (2008) recorded similar issues with false triggers from vegetative growth and reflection of light on the water during their study at latrine sites. In their study, regular site visits minimized the impact of vegetative growth, and angling the camera away from the backdrop of the water decreased incidences of light reflections. It is not possible to angle the camera away from water using the platform survey method, however incidences of photos triggered by sunlight were substantially less than those triggered by rapid vegetation growth. Increased site visitation during peak vegetative growth (1x per week), to readjust the camera or remove vegetation in front of the camera, is recommended for future studies to minimize false detections.

Inherent challenges associated with deploying cameras in wet marsh environments bear further consideration for application of this approach in the future. Maintaining functional equipment in an aquatic environment necessitates camera placement well above the water line, to accommodate changes in water levels from rain and drought. During a significant rain event, water levels would rise, endangering functionality of cameras and potentially submerging them. I placed cameras ~ 0.3 m above the water surface level and checked them directly prior to an anticipated significant rain event to relocate cameras to prevent water damage. Prolonged periods of drought significantly decreased water levels during both the summer-fall and spring seasons, negatively altering the angle of the camera towards the platform and causing a similar amount of manpower effort to correct camera angles in both fall and spring. If the water level changed +/− 0.3 m, it dramatically affected the camera angle (Figure 2.3), causing the platform to no longer remain within the camera focus and leading to the potential to miss animals utilizing the platforms. This was minimized through regular bi-weekly site visits to check camera functionally.

MANAGEMENT IMPLICATIONS

Based on the low detection of otters during this study, I do not recommend sole use of the camera and platform survey method in future studies designed to assess the distribution of otters across a large-scale landscape. I believe that the camera and platform method has utility in gaining insight into otter behavior, and perhaps in visitation rates to local water bodies. The platform approach will be most useful in areas otters are previously known to reliably visit. Platforms can be used to gather information on visitation rates of otters to a specific site, record minimum group sizes, and document behavior. Platforms themselves were not an efficient method in detecting otter presence on the landscape.

Larger-scale surveys, such as using cameras to assess otter population status across the central to western NYS recovery zone, would necessarily involve some modifications to initial survey design. First, it isn't clear from my study or the literature that either the platforms themselves or the scent lures were sufficient for attracting otters into the camera's view frame. Future studies might simply focus cameras on areas within 10 m of the shore, increasing the number of cameras surveying a given water body, and still reducing the over investment in effort required to set up and monitor cameras compared to the combination of cameras with floating platforms. Cameras may be better placed at strategic points on the landscape such as on beaver dams and lodges (occupied and abandoned; Melquist & Hornocker 1983, Swimley et al. 1998, LeBlanc et al. 2007) or at shallow stream crossings and peninsular outcroppings (Swimley et al. 1998, Paul Jensen, NYSDEC, personal communication) to increase the likelihood of detecting otters. Removing the floating platforms from the methodology eliminates a number of issues that I ran into during the course of this study, many of which required a 2-person survey team. One person could safely and efficiently deploy cameras at sites along the shoreline, greatly increasing the efficiency of the surveys while also ensuring the random sampling required for occupancy-based analyses (which is violated by deploying cameras only at known latrine sites). Based on the 8-km minimum home-range size reported by Melquist & Hornocker (1983), and the highly restricted range of females and pups post-parturition (≤10 km; Melquist et al. 2003), I recommend cameras be placed in a grid design across both shoreline and land, such as the type commonly used for occupancy analyses (MacKenzie et al. 2006), with a maximum diameter of 8 km. To ensure random sampling, a data set of beaver lodges (currently available for certain portions of the state, and not overly difficult to detect on the landscape if coordinates are not available) could be compiled, and a portion randomly selected to survey within the otter home-range.

In conclusion, while I was not to identify an optimal spacing for camera and platform arrays for otters, given that each visitation of a site involved detecting otters at only a single camera array, I believe the information gained from this effort was useful to inform future survey designs. Otters are a species of management interest, due to their status as a furbearer, making monitoring programs a necessity for state and federal agencies. A one-size fits all survey method is not practical for this species, given the wide range of ecosystems it inhabits. The current widely used sign survey method is reliable, yet time and labor intensive, making alternative non-invasive survey methods such as motion-sensitive cameras a worth-while endeavor for future research.

ACKNOWLEDGEMENTS

I would like to thank Bill Wilmoth of USDA APHIS for his help in development of the platform monitoring technique, as well as providing instructions for the platform construction and feedback on modifications for otters. Thank you to the NYSDEC furbearer team, and Region 6 manager Andrew MacDuff and Region 8 wildlife biologist Scott Smith for their help with identifying sites with active otter latrines, and NYSDEC Region 5 Manager Paul Jensen for his feedback on future study designs for river otters. Thank you to Stacy McNulty, Charlotte Demers, and summer 2016 student interns of the Adirondack Ecological Center for their assistance with platform deployment. Thank you to Brandon Swart, the field technician for the 2016 camera deployment, for his essential help in constructing and deploying platforms, as well as a handful of SUNY-ESF student volunteers who assisted during the 2017 field season. And thank you to Dennis Money for his help canoeing and collecting SD cards on one of the colder and windier days I have spent in the field.

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Table 2.1. Description of sites used in the pilot study, showing season and year of deployment, for what purpose each area is managed, the size of the managed area, and the total number of camera-platform arrays deployed.

Site	Season	Area managed for	Size (km^2)	Number camera arrays
Adirondack Ecological Center	Summer-fall 2016	research	61	4
High Tor WMA	Summer-fall 2016	wildlife	25	11
Tug Hill State Forest	Summer-fall 2016	forestry	50	4
Northern Montezuma WMA	Summer-fall 2016 Spring 2017	wildlife	30	10 8
Cicero Swamp WMA	Spring 2017	wildlife	20	4
Labrador Hollow UA	Spring 2017	recreation	6	7
Nelson Swamp UA	Spring 2017	recreation	4	7
Three Rivers WMA	Spring 2017	wildlife	15	10

Table 2.2. Species recorded by camera traps during 2017 field season (mammalian and avian) from initial deployment in April through final removal in July 2017. 503,078 photos were taken; 28,975 photos (6%) contained animals and 474,628 (94%) contained nothing. Of the photos containing animals, 7,542 (26%) contained mammals, 21,433 (74%) birds, and 36 (>1%) were unidentifiable animals.

Common name	Latin name	Photos	Mammalia	Aves
Beaver	*Castor canadensis*	102	X	
Canada goose	*Branta canadensis*	17,316		X
Coyote	*Canis latrans*	10	X	
Dog	*Canis lupus familiarus*	50	X	
Fisher	*Pekania pennanti*	77	X	
Gray squirrel	*Sciurus carolinensis*	11	X	
Great blue heron	*Ardea herodias*	940		X
Green heron	*Butorides virescens*	89		X
Human	*Homo sapiens*	4,690		
Mallard	*Anas platyrhynchos*	825		X
Muskrat	*Ondatra zibethicus*	333	X	
Mute swan	*Cygnus olor*	30		X
Owl	*Strigiformes* spp.	6		X
Raccoon	*Procyon lotor*	589		X
Red fox	*Vulpes*	30	X	
River otter	*Lontra canadensis*	4	X	
Turkey	*Meleagris gallopavo*	3		X
White-tailed deer	*Odocoileus virginianus*	1,618	X	
Wood duck	*Aix sponsa*	760		X
Woodchuck	*Marmota monax*	10	X	
Unknown bird	Aves spp.	1,425		X
Unknown mammal	Mammalia spp.	18	X	
Unknown	———	36		

Table 2.3. Cameras that captured images of river otters at High Tor WMA, from 13 September through 6 November 2016. Camera name is the Site (HI = High Tor WMA, Canandaigua, NY), Habitat (LO = riverine location, UP = pond location), and Number (1-6); see Figure 2.3 for camera placement). *rounded to nearest whole otter

Camera name	Visit date (2016)	Time (initial)	Total time elapsed	Group size (min.)
HIUP6	13 September	03:10:29	00:00:03	1
HIUP5	13 September	10:33:56	00:00:51	3
HIUP6	16 September	06:58:17	00:00:03	1
HIUP2	20 September	07:38:16	00:03:40	2
HIUP1	9 October	10:06:51	00:00:54	1
HIUP2	26 October	14:03:38	00:25:10	3
HIUP6	6 November	07:29:49	00:00:09	4
			$\bar{x}$ = 00:04:24	$\bar{x}$ = 2*

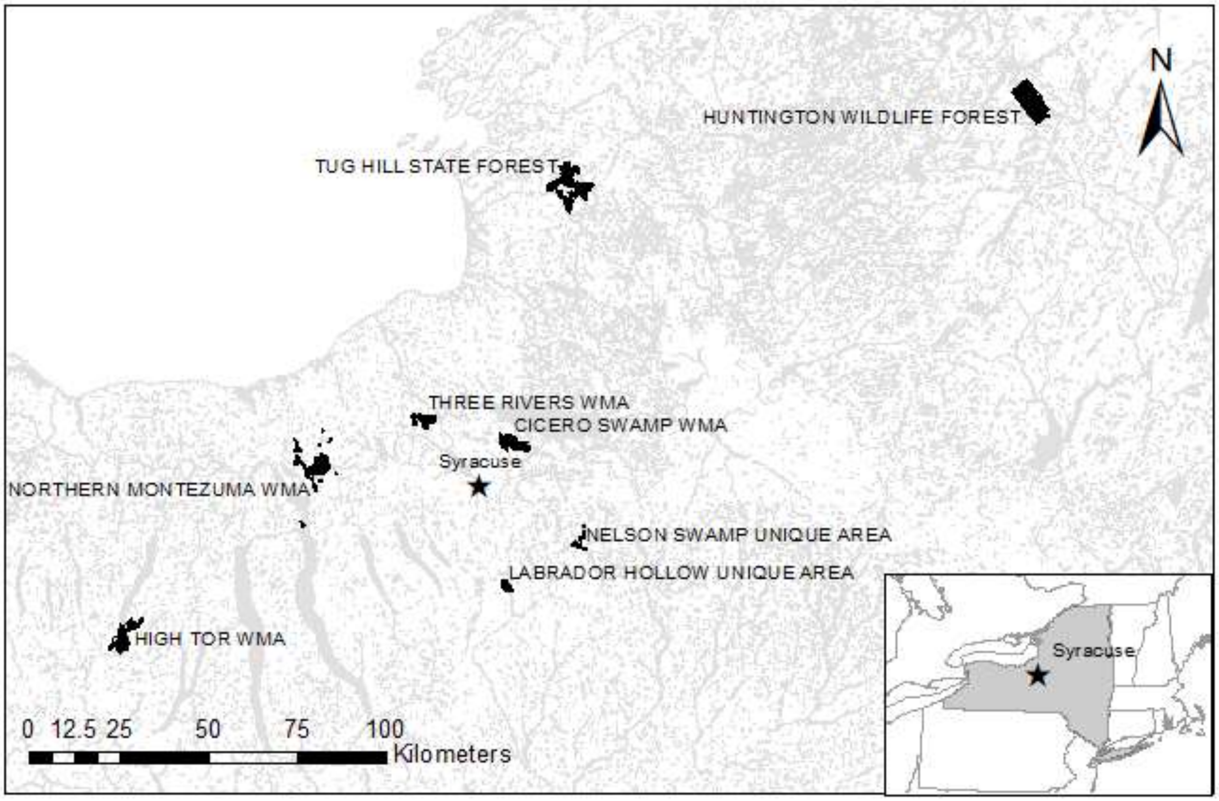


Figure 2.1. Location of 8 pilot study sites for deployment of camera-platform arrays to detect river otter 2016-17 in New York State. Each location for the 2016 study had verified sightings of river otters or their sign (scat, active latrine sites), and each location for the 2017 study had historical sightings of river otters, but sightings were not verifying via sign prior to deployment.

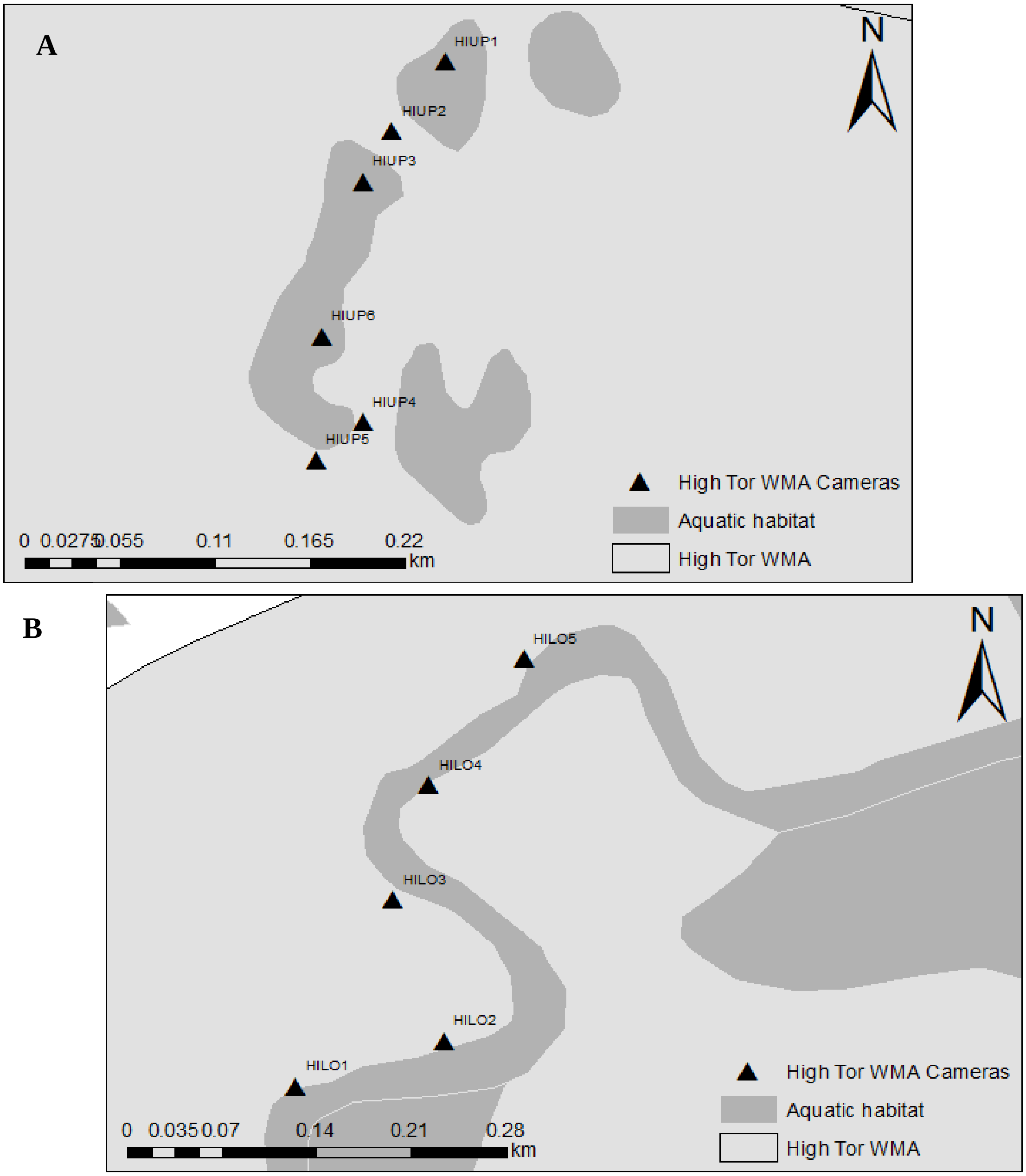


Figure 2.2. High Tor WMA map of the 2 different aquatic habitat type; palustrine (A; HIUP) and riverine (B; HILO). Otters were recorded by 4 out of 6 cameras at the palustrine site (A). Sites A and B are ~5-km apart, and separated by a state road, creating a potential barrier to river otters.

Figure 2.3. (A) Reconyx Hyperfire™ PC800 camera mounted ~0.2 m above the water surface on a metal fence post, secured with plastic zip ties and a cable lock. Camera was aimed at the 0.6 x 0.6-m floating wooden platform designed to attract river otters to a spot in front of the camera where they were in position to trigger the motion detector. (B) Desired view of the platform from the camera.

Figure 2.4. River otters recorded utilizing platforms at High Tor WMA, Naples, NY from 13 September through 6 November 2016.

Figure 2.5. A singular river otter, *Lontra canadensis*, swimming by a wooden platform on 28 April 2017 at Three Rivers WMA in Phoenix, NY. This is the third of a series of 4 photos of the same otter captured between 11:42:06 and 11:42:08 on a Reconyx™ PC800 trail camera set to 5-round burst. The otter was documented swimming by the platform but was not captured physically utilizing the platform.

Appendix E. 2016 pilot study camera trap and platform deployment data; site (Site: Northern Montezuma WMA (NOMO), High Tor WMA lowland riverine site/upland pond site (HILO/HIUP), Adirondack Ecological Center, Arbutus Lake (ARBU), Tug Hill State Forest (TUGH)), UTM coordinates, initial set date, final pull date, and total days deployed. Due to changing water levels some cameras were removed early. *Camera locations were neighboring, did not get separate GPS fix; cameras ≥100 m apart.

Camera UID	Site	Easting	Northing	Set	Pull	Days Deployed
NOMO1	N. Montezuma	363101.3716	4769658.069	06/23/16	9/23/2016	92
NOMO2	N. Montezuma	363152.0665	4769627.057	06/23/16	9/23/2016	92
NOMO3	N. Montezuma	363069.7157	4769675.813	06/23/16	9/23/2016	92
NOMO4	N. Montezuma	363161.4878	4769719.19	06/23/16	9/23/2016	92
NOMO5	N. Montezuma	362998.6228	4769790.336	06/23/16	9/23/2016	92
NOMO6	N. Montezuma	363804.4598	4770510.348	06/23/16	9/23/2016	92
NOMO7	N. Montezuma	361741.32	4772396.609	06/23/16	9/23/2016	92
NOMO8	N. Montezuma	361900.9616	4772330.828	07/01/16	9/23/2016	84
NOMO9	N. Montezuma	361770.7394	4771919.51	07/01/16	9/23/2016	84
NOMO10	N. Montezuma	409155.1827	4765271.262	07/06/16	9/23/2016	79
HILO1	High Tor	309849.671	4725812.868	06/17/16	11/9/2016	145
HILO2	High Tor	309959.1191	4725846.871	06/17/16	11/9/2016	145
HILO3	High Tor	309921.493	4725952.257	06/17/16	11/9/2016	145
HILO4	High Tor	309948.0418	4726037.99	06/17/16	11/9/2016	145
HILO5	High Tor	310018.4621	4726132.41	06/17/16	11/9/2016	145
HIUP1	High Tor	308118.0877	4721338.022	06/28/16	11/9/2016	134
HIUP2	High Tor	308087.5047	4721297.638	06/28/16	11/9/2016	134
HIUP3	High Tor	308070.8585	4721268.537	06/28/16	11/9/2016	134
HIUP4	High Tor	308071.0267	4721129.281	06/28/16	11/9/2016	134
HIUP5	High Tor	308043.8295	4721107.474	06/28/16	11/9/2016	134
HIUP6	High Tor	308046.957	4721178.957	06/28/16	11/9/2016	134
ARBU1	AEC	358608.2667	4770477.21	08/08/16	11/4/2016	88
ARBU2	AEC	560469.6475	4870994.138	08/08/16	11/4/2016	88
ARBU3	AEC	560761.3786	4871258.065	08/08/16	11/4/2016	88
ARBU4	AEC	560802.5324	4871020.849	08/08/16	11/4/2016	88
TUGH1	Tug Hill	434943.13	4847216.738	6/16/2016	8/17/2016	62
TUGH2	Tug Hill	435007.7265	4847246.431	6/16/2016	8/17/2016	62
TUGH3*	Tug Hill	435007.7265	4847246.431	6/16/2016	8/17/2016	62
TUGH4*	Tug Hill	435007.7264	4847246.431	6/16/2016	8/17/2016	62

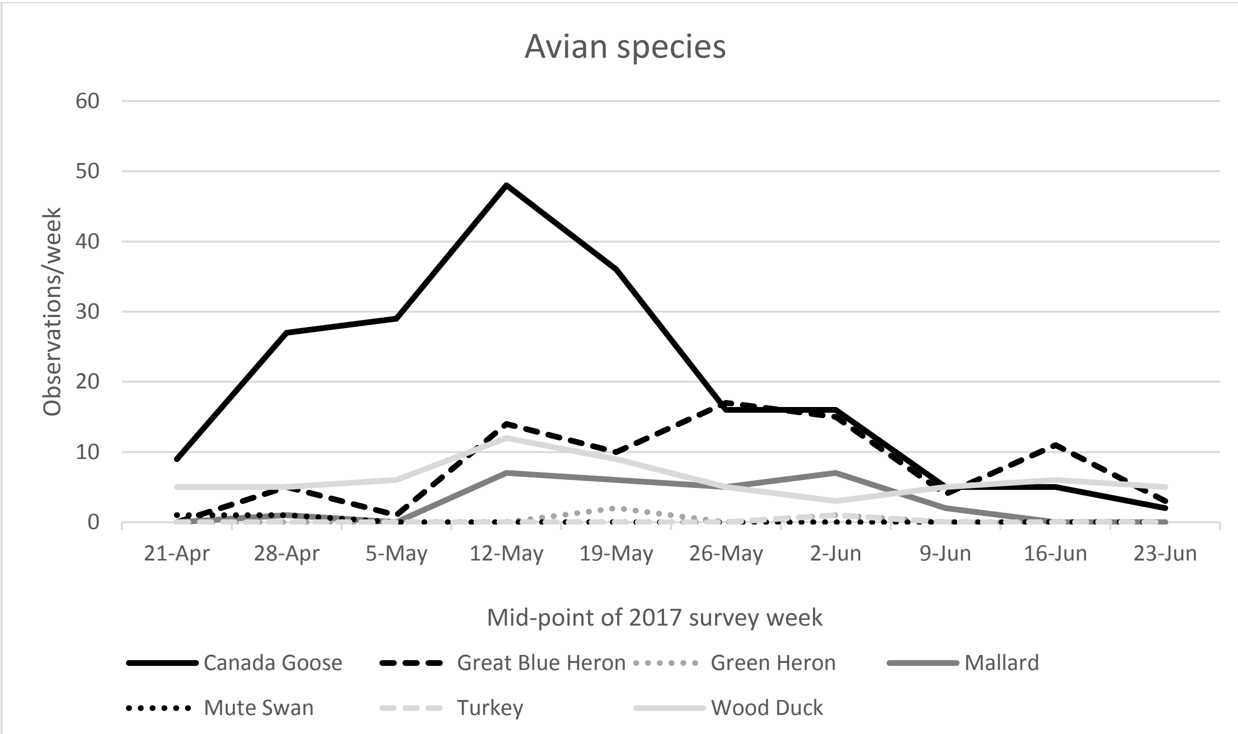
Appendix F. 2017 field study camera trap and platform deployment data; site (Cicero Swamp WMA, CICR; Nelson Swamp UA, NELS; Three Rivers WMA, THRI; Labrador Hollow UA, LABH), UTM coordinates, initial set date, final pull date, and total days deployed. Due to changing water levels some cameras were removed early. Cameras were deployed (set to pull) for a combined total of 2417 days. *Camera locations were neighboring, did not get separate GPS fix; cameras ≥100 m apart.

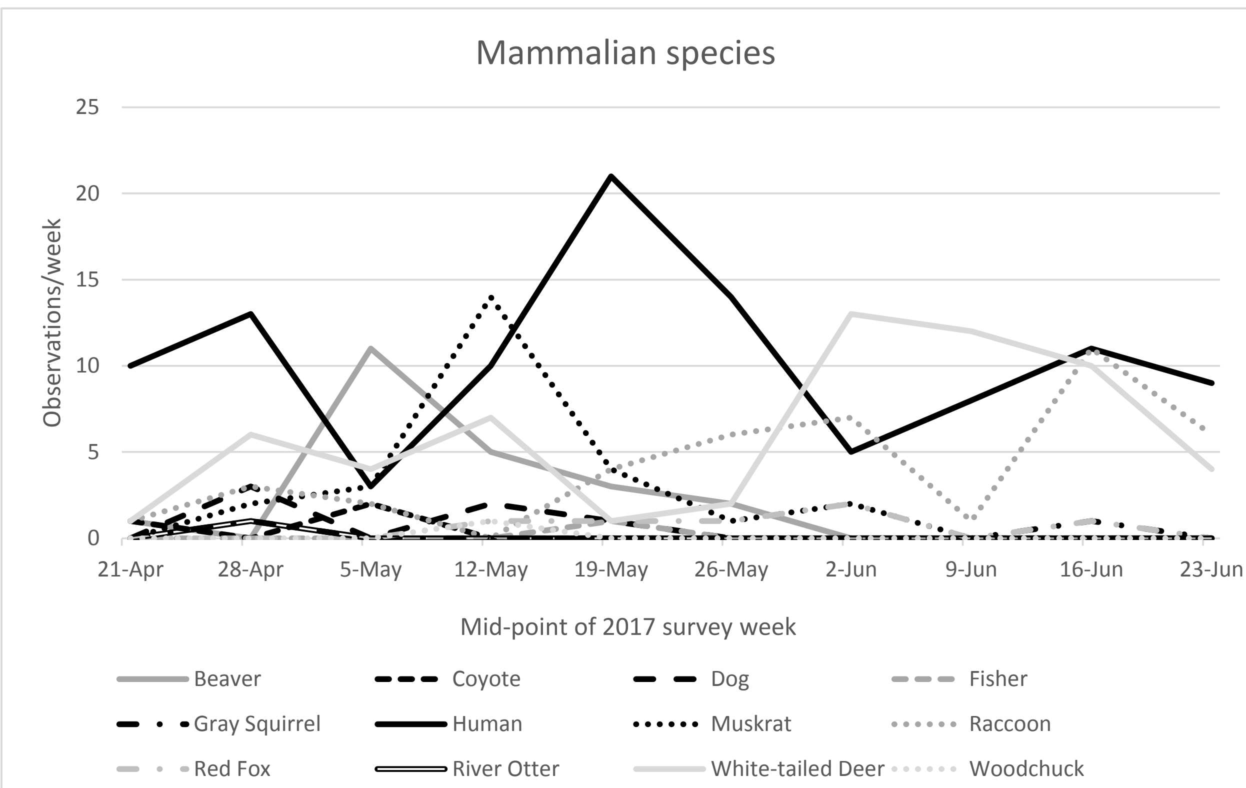
Camera UID	Site	Easting	Northing	Set	Pull	Days Deployed*
CICR01	Cicero Swamp	420295.6569	4776457.764	04/18/17	06/28/17	71
CICR02	Cicero Swamp WMA	420510.3797	4776379.953	04/18/17	06/28/17	71
CICR03	Cicero Swamp WMA	420416.5119	4776240.548	04/18/17	06/28/17	71
CICR04	Cicero Swamp WMA	420171.4221	4776052.49	04/18/17	06/28/17	71
NELS01	Nelson Swamp UA	434895.3789	4750062.178	04/21/17	06/25/17	65
NELS02	Nelson Swamp UA	434991.9221	4750126.344	04/21/17	05/16/17	25
NELS03	Nelson Swamp UA	434993.6399	4750247.491	04/21/17	07/10/17	80
NELS04	Nelson Swamp UA	434950.5843	4750399.713	04/21/17	07/10/17	80
NELS05	Nelson Swamp UA	392496.1273	4785019.225	04/24/17	07/10/17	77
NELS06	Nelson Swamp UA	433365.3044	4748892.64	04/24/17	06/25/17	62
NELS07	Nelson Swamp UA	435142.9205	4750449.87	04/24/17	06/25/17	62
THRI01	Three Rivers WMA	392684.9724	4785046.011	04/22/17	06/26/17	65
THRI02	Three Rivers WMA	392635.0316	4785181.542	04/22/17	06/26/17	65
THRI03	Three Rivers WMA	392743.828	4785125.171	04/22/17	06/26/17	65
THRI04	Three Rivers WMA	391998.4465	4784471.149	04/22/17	07/26/17	95
THRI05	Three Rivers WMA	391533.34	4784406.018	04/22/17	07/26/17	95
THRI06	Three Rivers WMA	391421.501	4784394.25	04/22/1	07/26/1	95

		6	1	7	7	
THRI07	Three Rivers WMA	392969.7831	4783602.567	04/22/17	06/13/17	52
THRI08	Three Rivers WMA	393334.7341	4783487.084	04/26/17	06/26/17	61
THRI09*	Three Rivers WMA	393186.5201	4783234.923	04/26/17	06/26/17	61
THRI10*	Three Rivers WMA	393186.5201	4783234.923	04/27/17	06/26/17	60
LABH01	Labrador Hollow UA	413445.1995	4738588.176	04/28/17	06/28/17	61
LABH02	Labrador Hollow UA	414275	4737894	04/28/17	06/28/17	61
LABH03	Labrador Hollow UA	414313.8045	4737742.803	04/28/17	06/28/17	61
LABH04	Labrador Hollow UA	414333	4737471	04/28/17	06/28/17	61
LABH05	Labrador Hollow UA	413804.0766	4737761.148	04/28/17	06/28/17	61
LABH06	Labrador Hollow UA	413825.876	4737941.131	04/28/17	06/28/17	61
LABH07	Labrador Hollow UA	413926.2716	4738120.244	04/28/17	06/28/17	61
NOMO01	N. Montezuma WMA	362999.95	4769800.04	05/05/17	07/14/17	70
NOMO02	N. Montezuma WMA	363082.2696	4769721	05/05/17	07/14/17	70
NOMO03	N. Montezuma WMA	363731.72	4770768.69	05/05/17	07/14/17	70
NOMO04	N. Montezuma WMA	363564.2276	4770886.653	05/05/17	07/14/17	70
NOMO05	N. Montezuma WMA	363917.13	4769667.92	05/09/17	07/14/17	66
NOMO06	N. Montezuma WMA	363873.2821	4769656.527	05/09/17	07/14/17	66
NOMO07	N. Montezuma WMA	364162	4769679.28	05/09/17	07/14/17	66
NOMO08	N. Montezuma WMA	361686.56	4772413.48	05/12/17	07/14/17	63

Appendix G. Graph of avian and mammalian species observations by survey week, 2017.

Observations were recorded as one independent observation at a trap per day.





EPILOGUE

This thesis research was part of a larger, on-going study on North American river otter, *Lontra canadensis*, distribution across New York State [NYS], conducted by the New York State Department of Environmental Conservation [NYSDEC] and Dr. Jacqueline Frair of the State University of New York College of Environmental Science and Forestry [SUNY-ESF]. The objectives of this thesis were 1) determine utility of opportunistic sightings of river otters for modeling probability of otter occurrence across the recovery zone (~53,000 km^2) in central and western NYS, 2) evaluate probability of otter occurrence models, identify important environmental covariates influencing otter occurrence, and validate the model based on corrected data using an independent data set, 3) implement and test an experimental non-invasive survey method based on a design by USDA-APHIS (Kerr & Dawson 2013) deployable across large-scale areas for river otters, and 4) recommend a study design based on the experimental study for future large-scale non-invasive surveys for river otters.

In Chapter One, I tested the impacts of ad hoc bias corrections on 1) the identification of important variables for predicting the probability of otter occurrence using the R package MAXLIKE (Royle & Chandler 2012), 2) the direction and magnitude of the estimated covariate effects, and 3) validated the model using an independent data set. In so doing, I demonstrated a readily-accessible approach for gaining reliable inference on broad-scale animal distributions from disparate records of animal occurrence. I found that opportunistic data from disparate sources could be used to generate a reliable model of probability of otter occurrence, given that potential biases inherent in the data set (such as unequal survey effort, bias from proximity to road) are identified and corrected prior to model development. The ad hoc corrections made to the data set affected the magnitude of the covariate effects (effect sizes varied −7.1% to +48.0%

after bias correction), but not the direction. The most influential variable affecting probability of otter occurrence in my top model was proportion of shoreline. The model was validated with an independent set of data collected during the winter 2016 NYSDEC bridge-based track and sign surveys following an approach modified from Johnson et al. (2006) and resulted in an R^2 of 0.90.

In Chapter Two, I 1) tested motion-sensitive cameras linked to floating platforms as a means of detecting otter occurrence in lacustrine and palustrine environments in NYS, and 2) evaluated the potential for extension of camera traps to a broad-scale study of otter occupancy. Platforms were an effective method to monitor for aquatic wildlife, but not for the target species, river otters. Otters were only recorded on camera at one site per year during 8 independent detection events over a 2-year study, not enough information to allow for determination of a study design for future surveys. I do not recommend platform deployment across large-scale areas going forward. I recommend continuing research with motion-sensitive cameras targeting habitat commonly used by otters (e.g., beaver dams, lodges, shallow streams, peninsular outcroppings; Melquist & Hornocker 1983, Swimley et al. 1998, LeBlanc et al. 2007, Paul Jensen, NYSDEC, personal communication), placed in a grid with a diameter no-greater than 8 km, based on minimum recorded home-range for river otters (Melquist & Hornocker 1983).

Moving forward with monitoring otter distributions in NYS, I recommend repeating the modeling procedure for probability of river otter occurrence using opportunistic records collected by NYSDEC every 5-years, to identify any drastic changes in the population. I additionally recommend research continue on use of cameras with river otters, taking into consideration the recommendations from this study.

LITERATURE CITED

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RESUME

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EDUCATION

SUNY Environmental Science and Forestry, Syracuse, NY
Master of Science in Fish and Wildlife Biology and Management
Research Foundation Project Assistant

The University of Georgia, Athens, GA Graduated Magna Cum Laude, December 2015
Warnell School of Forestry and Natural Resources GPA: 3.71
Bachelor of Science in Forest Resources, Wildlife Science
School of Public and International Affairs
Bachelor of Arts in International Affairs

Oxford University, Oxford, England Hillary Term, 2015
UGA School of Public and International Affairs Study Abroad GPA: 4.0

Community College of the Air Force, Montgomery, Alabama April 2015
Associate of Arts in Intelligence Studies and Technology

The Defense Language Institute, Monterey, CA Graduated with honors, March 2011
Associate of Arts in Modern Standard Arabic GPA: 3.9

RELAVENT EXPERIENCE

Rhode Island Department of Environmental Management (RIDEM) South Kingston, RI
Seasonal technician June 2018 – present

- Assistant to the regional deer biologist
- Plant identification for browse exclosure research
- Goose-banding
- Nuisance wildlife management
- New England cottontail rabbit data entry
- Preparing equipment for fall 2018 deer check stations and winter rabbit surveys
- Electro-shocking and seining to assess freshwater fish species
- Assistance with outreach and public education classes

Rhode Island Air National Guard — 143d AW, Quonset Point, RI
Staff Sergeant, Operations Squadron — January 2018 - present
Undergraduate Pilot Training (UPT) candidate

- Selected as a C130-J UPT candidate in 2017 UPT boards
- Currently completing UPT pre-requisites
- Assisting in Squadron Operations Intelligence while waiting to attend UPT

SUNY College of Environmental Science and Forestry — Syracuse, NY
Research Foundation Project Assistant — January 2016 – June 2018

- Worked with NYS-DEC furbearer team to estimate distribution of river otters in NYS
- Implemented experimental camera-trapping of aquatic habitats summer 2016 & 2017
- Hired, trained & managed technician & undergraduate volunteers for field work
- Mapped otter occurrence using presence-only data across central NYS

University of Georgia, Warnell School of Forestry — Sapelo Island, GA
Student Research Assistant — May-August 2015

- Used passive seining technique to collect biomass samples of tidal creek fish
- Collected seven repetitions of data from six sampling sites over the course of eight weeks
- Piloted Quadcopter UAV to obtain aerial imagery of tidal creek system
- Analyzed data using ArcGIS, Agisoft Photoscan, R, Excel
- Produced Senior Thesis and contributed to PhD candidate's thesis research

University of Georgia, Department of Ecology — Athens, GA
Dolphin Research Field Assistant — January-May 2014

- Collected data to build a dolphin population and migration database
- Monitored location of dolphins along transects
- Photographic documentation of dolphin fins for unique identification purposes
- Operated research vessel

Sandy Creek Nature Center — Athens, GA
Volunteer Trail Guide — August- December 2013

- Guided elementary school students on educational nature walks
- Facilitated hands-on learning environment to encourage interest in biological sciences

Warnell Natural Resource Core, UGA Costa Rica — San Luis, Costa Rica
Study Abroad Student — July 2013

- Visited conservation sites and studied the successful integration of local communities in conservation

- Designed and implemented a project comparing frequency of hummingbird visitation between two types of local plants and created maps incorporating the resulting GPS data using ArcMap v10.2.2.

Georgia Air National Guard Fort Gordon, GA
Staff Sergeant, Arabic Cryptolinguist October 2008-2014

- Language Analyst for Georgia Air National Guard
- Intelligence Oversight Monitor: ensured compliance with regulations regarding classified information
- Physical Training Leader
- Certified as a 3/3/2 in Modern Standard Arabic (MSA)
- Qualified as a 3/3+ in Spanish
- Received The Air Force Commendation Medal for Meritorious Service

CERTIFICATIONS AND WORKSHOPS

The Wildlife Society Associate Wildlife Biologist®
November 2016-present

- Completed certification requirements as an associate wildlife biologist for TWS

PADI Scuba Certification
September 2015-present

- Certified to dive up to 60ft

Trapper Certification (NY)
June 2016-present

Bowhunter's Safety Certification Course (NY)
March 2016-present

Hunter's Safety Certification Course (GA)
September 2014-present

Private Pilot
May 2009-present

- Licensed private pilot, single engine, 100+ flight hours, night and day current

Surf Rescue Lifeguard
May 2007-2014

- Certified as an open-water ocean lifeguard

First Aid/CPR/AED

- Completed Red Cross Trainings in First Aid/CPR/AED for the professional rescuer
  May 2007- 2014

WORKSHOPS

Safe Capture International Inc. Chemical Immobilization of Wildlife

February 2018-present

- Attended 16-hr certification course at UGA Veterinary Medical Learning Center

UAVs in Wildlife Workshop
April 14th, 2018

- NEAFWA workshop on application of UAVs to wildlife research

NYS-DEC Nuisance Beaver Trapping Workshop
September 2017

- Training by Scott Smith, NYS-DEC furbearer biologist on constructing and setting beaver traps

The Wildlife Society Intermediate R Workshop
October 2016

- Completed Intermediate R Workshop at TWS 23rd Annual Conference in Raleigh, NC

NYS-DEC Fur School
June 2016

- Hand-on training on trapping guidelines, how to properly set a trap, and preparing a hide

PROFESSIONAL SOCIETIES

The Wildlife Society National Chapter
active member

The Wildlife Society Northeast Chapter
active member

Xi Sigma Pi National Forestry Honor Society

SKILLS

Field: Capture and handling of wildlife, plant and animal identification, radio telemetry, camera-trapping, manual transmission 4x4, boat operation, electrofishing, prescribed burning, orienteering, piloting UAV, seining, biomass estimates, saltwater and freshwater fish identification, water and nutrient analysis, firearms training, bird-banding,

Languages: Modern Standard Arabic (fluent); Spanish (fluent); Italian (conversational)

Computer: Microsoft Office, ArcMap v10.5.1, Program R, Agisoft Photoscan, Google Earth