DETECTION, OCCUPANCY AND DISTRIBUTION OF ASIAN SWAMP EEL (*MONOPTERUS ALBUS*) IN GEORGIA

By

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Abstract: More than 100 species of exotic freshwater fishes have been introduced into the United States, threatening native species and causing an estimated economic impact of $1 billion annually. Currently, there are 4 known populations of Asian Swamp Eel (*Monopterus albus*) introduced into the continental U.S. The first population of Asian Swamp Eel was discovered in three ponds at a private nature facility in Northern Georgia in 1994. Research activity on this population has been limited due to challenging sampling conditions and characteristics that make Asian Swamp Eels difficult to capture. To account for this, we implemented an occupancy modeling approach with multiple sampling methods to garner information about detection, occupancy and distribution of this Asian Swamp Eel population. Leaf litter traps were used in the Chattahoochee River and backwater marsh areas within a 2-km radius of the presumed invasion point to estimate detection, and delineate the current invasion extent. Covariates likely to explain Asian Swamp Eel presence were used to model probabilities of occupancy from leaf litter trap sampling. The top model of detection probability included year, and temperature and depth at the time of sampling. Detection probability increased with sampling depth and temperature. The top model of occupancy included proportion of area vegetated and proportion of area silt substrate, and occupancy was positively associated with both. A distribution map was created by interpolating covariates for unsampled areas, and estimates of occupancy probability were backtransformed for the entire study area. Asian Swamp Eel distribution appears limited within the Chattahoochee River, and backwater marsh areas in the immediate vicinity of the presumed invasion point have the highest probabilities of occupancy. Adult Asian Swamp Eels were sampled within two ponds and the backwater marsh areas with canoe-based backpack electrofishing. Occupancy was unable to be modeled for adult Asian Swamp Eels, but detection was able to be estimated for the two ponds. Probable differences in detection along with persistence time indicate that there may be a more substantial marsh population than previously thought.

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CHAPTER I

DETECTION, OCCUPANCY AND DISTRIBUTION OF ASIAN SWAMP EEL (*MONOPTERUS ALBUS*) IN THE CHATTAHOOCHEE RIVER, GA

Introduction

More than 100 species of exotic freshwater fishes have been introduced into the United States, threatening native species and causing an estimated economic impact of $1 billion annually (Pimentel 2000). Introduced species pose threats to native fauna through competition for resources, predation, disease transmission, hybridization, loss of biodiversity, and a suite of indirect ecological effects (Davis 2009). Freshwater ecosystems have been recognized as particularly vulnerable to deleterious effects of exotic species introductions (Vander Zanden and Olden 2008); therefore, scientific interest of introduced species and their impact on freshwater ecosystems has grown exponentially in recent decades (Gozlan 2008).

Exotic species may be introduced accidentally or intentionally, stemming from a wide variety of vectors. Most plant and vertebrate introductions have been intentional, while most invertebrate and microbial introductions have been accidental (Pimentel 2000). Notable motivations for intentional freshwater fish species introductions are pest

control, sport, ornamental purposes, and food (Pimentel 2000). Of these sources, food production accounts for the majority of global fish introductions, and translocation of species continues today even with laws and regulations to control such actions (De Silva et al. 2009).

Currently known from four populations in the continental U.S. and one in Hawaii, Asian Swamp Eels (*Monopterus albus*, ASE herein) were likely introduced in hopes of propagating a food source (Collins et al. 2002). Native to temperate and tropical climates of southeast Asia, Indonesia, and Australia (Rosen and Greenwood 1976; Berra 2001), ASEs are considered a valuable food fish and can be purchased live in China, Japan, and the United States (Guan et al. 1996). Of the four populations in the continental U.S., three were discovered in Florida from 1997 to 1999, and one in northern Georgia in 1994 (Collins et al. 2002). The discovery of these populations drew interest due to a variety of invasive characteristics unique to ASEs that may increase their impact as an invasive species (Freeman et al. 2005; Shafland 2010).

In their native range, ASEs are considered strict carnivores and voracious predators, increasing concerns of direct impact on native fauna through predation (Liem 1998). As adults, ASEs are obligate air breathers and capable of overland migration, potentially increasing dispersal through nontraditional freshwater fish migration corridors (Liem 1967; Graham et al. 1995). Asian swamp eels exhibit a sequential protogynous hermaphroditic life history (Matsumoto et al. 2011), which may result in lower necessary propagule pressure to create viable populations. Asian Swamp Eels are equipped to endure periodic drought and desiccation through burrowing into the substrate, forming a

protective mucus layer and detoxifying endogenous ammonia, promoting their persistence in highly variable environmental conditions (Liem 1987; Ip et al. 2004).

In their native range, ASE populations are known to occur in swamps, ponds, and ricefields in southeast Asia (Liem 1961), fast running rivers and creeks in western Java (Liem 1961, 1963, 1967), and streamlets canals and estuaries on the Indian subcontinent (Day 1878 cited by Freeman et al. 2005). Through genetic analysis, Collins et al. (2002) found that the three Florida ASE populations were most closely related to specimens acquired from southern China through the Maylay Peninsula and Indochina, while the Georgia population was most closely related to specimens from a more northern climate in Japan or Korea. Additionally, ASEs likely represent three or more distinct species with mitochondrial DNA sequence divergence equivalent to that observed in some families of fish (Collins et al. 2002). As a result, Freeman et al. (2005) referred to the Georgia population as an undescribed species (*Monopterus* sp. *cf. M. albus*). Whether this taxonomic uncertainty reflects a difference in biology and ecology is unknown.

Research on the Georgia population of ASEs began with their initial discovery in 1994, at three ponds of the Chattahoochee Nature Center (CNC) near Atlanta (Figure 1) (Starnes et al. 1998). At the time of discovery, it was concluded that the population had been persisting for multiple years and reproducing successfully due to the range of sizes present (32 mm to > 225 mm TL), placing the original introduction prior to 1990 (Starnes et al. 1998; Freeman et al. 2005). Backpack electrofishing proved mildly effective for sampling adult ASEs within CNC ponds, but capture probability was extremely low (<1%) (Freeman et al. 2005). Leaf litter traps (LLTs), adapted from a method used to sample salamanders, proved to be the most effective way to capture juvenile ASEs

(detection probability ~ 11%) (J. Long, U.S. Geological Survey, unpublished data). Further investigation with LLTs found juvenile ASEs in a small (0.66 ha) backwater marsh area of the Chattahoochee River that is connected to the CNC ponds via a culvert (Figure 1) (Freeman et al. 2005; Long and LaFleur 2011). Although electrofishing surveys have been conducted, few adults have been found in the marsh or the river suggesting that reproduction may be constrained to the CNC ponds with juveniles present through emigration (Freeman et al. 2005).

Long and LaFleur (2011) used LLTs to capture juvenile ASEs in the marsh of the Chattahoochee River, and through age and growth analysis, estimated hatch dates from 13 June to 7 August, indicating that reproduction is occurring and at lower temperatures than other populations. Additionally, stable isotope and gut content analysis indicated populations in U.S. were low level predators, unlikely to exhibit significant direct predation pressure on other fishes (Freeman and Burgess 2000; Straight et al. 2005; Hill and Watson 2007). Although Georgia ASEs appear to be minimally piscivorous, and reproduction is potentially limited to a finite area, they may still pose threats to the ecosystem function through competition and indirect ecological effects. Furthermore, the inconsistency of traits among ASE populations emphasizes the importance of population specific research questions.

Although prior research has provided valuable biological knowledge about this population of ASEs, population level inference has been hindered by low capture probability and difficult sampling conditions (Freeman et al. 2005). Currently, the extent of the ASE invasion within the Chattahoochee River is unknown. Therefore, our research objective was to use an occupancy modeling approach using LLTs to evaluate

detection and occupancy probabilities of ASEs in the Chattahoochee River. Furthermore, we sought to develop a distribution map in relation to environmental covariates of occupancy to determine the extent of the ASE invasion in the Chattahoochee River.

Occupancy modeling is a relatively new approach to understanding imperfect detection and species occupancy through repeated sampling (MacKenzie 2006). A number of assumptions need to be met to effectively construct models and evaluate detection and occupancy. Briefly: (1) the population must be closed (no births, deaths, immigration or emigration may occur during a defined sampling season), (2) the species may not be falsely detected, and (3) sampling must be independent (detecting a species at a site has no influence on future detections at that site or any others) (MacKenzie et al. 2003). Occupancy modeling presented an ideal research and analyses framework because model assumptions were able to be met, and analysis allowed for interpretation of how heterogeneity in sampling and habitat conditions influenced detection and occupancy probabilities.

Methods

Study Area

The Chattahoochee River originates in northern Georgia flowing southwest across the state, where it turns south to form the boundary between Alabama and Georgia until its confluence with the Flint River at the Georgia and Florida line. The main stem Chattahoochee is highly altered by 14 dams, including Morgan Falls Dam which regulates peak flows from upstream Buford Dam (Georgia Power Company et al. 2006). Morgan Falls Dam impounds Bull Sluice Lake, a 272 ha surface area, shallow impoundment with backwater marsh areas where ASEs occur (Georgia Power and

Geosyntec Consultants 2005; Freeman et al. 2005; Long and LaFleur 2011). Two of the six ponds at the CNC are connected to the marsh of Bull Sluice Lake (Kingfisher and Frog), and previous records of ASE presence are confined to three ponds (Beaver, Kingfisher, and Frog), and the connected marsh (Figure 1) (Freeman et al. 2005; Long and LaFleur 2011).

Sampling

The shoreline of Bull Sluice Lake, as defined by the National Hydrography Dataset (NHD plus), within a 2-km radius of the CNC was broken into 5-meter segments within ArcMap 10.2.1 (ESRI, 2017) to designate sites for sampling with leaf litter traps. In 2015, 111 sites were randomly selected on the western shoreline (CNC side of the river), and 100 from the entire shoreline in 2016 for sampling (Figure 1). Transects consisted of a standardized array of 5 LLTs spanning the 5-meter transect (Figure 2). In both years, transects were sampled on 10 occasions from 06/24 – 08/09. Sampling-level information was recorded at every sampling event and site-level information was recorded at all transects once at the beginning of the sampling season to be included as covariates for occupancy modeling (Table 1).

Sampling-level information is used to describe the conditions at a site specific to each sampling event and included water depth [0.01m] at each LLT location and water temperature [0.1˚C] at the center of the transect during each sampling event. Sampling-level information is used to model detection probability under heterogeneous sampling conditions.

Site-level information was used to describe the conditions at a site over the entirety of each sampling season and were included as occupancy covariates. Submersed

and emergent aquatic vegetation, gravel, sand, silt, and organic substrate were calculated by placing a 1 x 1 m quadrat with 25 sub-units over the center of each LLT location within a transect, and recording the frequency of each habitat category (dominating sub-quadrat unit) within the quadrat (Krebs 1999) (Figure 2). Summing the number of sub-units representative of each habitat type allowed for the proportion of each habitat type to be calculated for each transect. Due to the low prevalence of several habitat metrics, our final set of model covariates recorded with the quadrat method was reduced to proportion of area vegetated and proportion of area silt substrate. Temperature variance and mean site depth was calculated for every transect from the 10 sampling occasions in each year. Distance from the CNC pond discharge (hypothesized invasion point) was measured to the center of each 5-m transect within ArcMap 10.2.1 (ESRI 2017). Additionally, year (2015 or 2016) was incorporated as a site-level categorical covariate to explore variation in occupancy probability between years.

Occupancy Model Analysis

Occupancy modeling was performed with package "unmarked" in program R (R Studio Team 2017), based on site and sampling level covariates. A likelihood ratio test was used to compare the null model of detection to the top detection model, and used to evaluate season as a site level covariate before inclusion in further occupancy models. Continuous site-level covariates were standardized and assed for multicollinearity with a Pearson's correlation matrix (R Studio Team 2017). A cut off of $r > |0.70|$ was used to eliminate the incorporation of two covariates in a model that explain the same variation (Guisan and Thuiller 2005). The most global model was assessed for fit with a chi-square parametric bootstrap with 10,000 simulations (Fiske and Chandler 2011). Model

selection was performed using AIC, and a threshold of 5 ΔAIC was established to identify plausible models due to the distribution of AIC weight among models (Burnham and Anderson 2002). All models < 5 ΔAIC were averaged and estimates of occupancy were back-transformed to specific values of covariates for every 5-m segment of the margin of Bull Sluice Lake (R Studio Team 2017).

Environmental Variable Interpolation

To estimate probability of occupancy throughout the entirety of the study area we used kriging to establish site-level covariates for unsampled areas. Kriging is a geostatistical method of interpolating spatial data using a covariance structure that represents the spatial relationship of the variable (Stein 2012). Four covariates (proportion of area vegetated, proportion of area silt substrate, mean site temperature, and temperature variance) were interpolated from all sampling locations for each year individually using the Geostatistical Wizard package within ArcMap 10.4 (ESRI 2017). Each covariate was first assessed for global trends in the dataset with a trend analysis. Covariance and semivariogram models were assessed for fit after being optimized with a cross validation procedure. The optimize function within the Geostatistical Wizard was used in kriging to assist in the fitting of semivariogram and covariance models to improve prediction error statistics for all covariates. Semivariogram and covariance models were assessed for anisotropy and the best model was chosen based on prediction error statistics. An example of model selection criteria for the suite of candidate models of a single covariate (proportion of area vegetated) from 2016 is shown in Table 2. A map of the distribution of ASEs was created with depth data from a bathymetry file, distance data from GIS, and habitat data from kriging.

Results

Sampling

Over the 10 sampling occasions, 31 ASEs were captured at 14 unique transects in 2015, and 36 ASEs at 10 unique transects in 2016. Mean size of captured ASEs ranged from 38.3mm to 77mm in 2015 and from 40.8mm to 165mm in 2016 (Table 3; Appendix A). ASEs were encountered during every sampling occasion with the exception of occasion 4 in 2015 and occasion 1 in 2016. In 2015, ASEs were detected as close as 32 meters from the presumed invasion point and as far away as 1,584 meters. In 2016, ASEs were detected from 154 meters to 860 meters from the presumed invasion point, including three transects on the opposite side of the river.

Analysis

The top model of detection probability was the global model, which included temperature (Temp), mean trap depth (Mtd) and year (Table 4). Year was identified as an important consideration in modeling detection probability with a likelihood ratio test between the null model and the global model of detection probability ($\chi^2 = 12.16$, DF = 3, $P<0.01$). At the mean values of sampling-level covariates, detection probability was 0.20 (SE = 0.04) in 2015 and 0.37 (SE = 0.13) in 2016, and was influenced by temperature (Temp) and mean trap depth (Mtd) and was included for all estimates of occupancy. In both years, detection probability increased with temperature and mean trap depth (Figures 3 and 4).

Year was not important in modeling occupancy, with a likelihood ratio test compared to the null model of occupancy ($\chi^2 = 0.36$, DF = 1, $P=0.55$). Multicolinearity never exceeded $r > |0.70|$ among any site-level covariates, therefore all covariates were

employed resulting in a suite of 60 candidate models of occupancy (Table 5 and Table 6). The global model of occupancy exhibited good fit ($\chi^2 = 0.64$), with no over-dispersion (c-hat = 1), (Burnham and Anderson 2002). The top model included proportion of area vegetated and proportion of area silt substrate within the transect and was included along with other covariates in all of the top 10 models. Occupancy probability increased with both vegetation and silt substrate (Figure 5). All six covariates appear within the top 19 models (<5 ΔAIC) indicating that all were valuable in modeling ASE occupancy.

Environmental Variable Interpolation

For interpolation, no covariates in either year exhibited a significant global trend to account for in the model selection procedure. A semivariogram model was used for interpolation of all but one covariate (2016 vegetation), and anisotropy improved the model fit for all covariates except temperature variance in both years and mean site temperature in 2016 (Table 7). Backwater marsh areas in the immediate vicinity of the CNC appeared to have the highest probabilities of occupancy within the study area (Figure 6), as well as a downstream backwater marsh area of Willeo Creek and the northern marsh on the opposite side of the river from the CNC. Furthermore, the study area was bounded upstream and downstream by low probabilities of occupancy.

Discussion

Juvenile ASE sampling with LLTs provided good estimates of detection probability, occupancy probability and current distribution within the Chattahoochee River system around the CNC ponds. Additionally, our sampling provided the furthest known detection of an ASE from the CNC and LLTs provided a scalable sampling methodology for population-level inference. Previous research concluded that low and

variable detection probability for this population was a factor limiting future study (Freeman et al. 2005). Our estimates of detection support these conclusions and highlight the importance of accounting for imperfect detection to inform management of this population.

It appears that the current distribution of ASEs in the Chattahoochee River is limited, with a variety of potential explanations. Although this population has persisted in the CNC ponds since at least 1994, it was first documented in the adjacent marsh in 2004 after LLTs, which are effective at documenting juveniles, were first used (Freeman et al. 2005). Considering the difference in persistence time, the population may be experiencing a lag in population growth or expansion, or the perceived lag may be a result imperfect detection prior to the use of LLTs.

Our detection modeling and distribution map can be used as valuable management tools for future control of ASEs in the Chattahoochee River. Specifically, this information can inform tactical sampling strategies for monitoring, suppression, or eradication of this invasive species. The detection model indicates that the best conditions for sampling ASEs with LLTs is during high, warm water conditions. This information can be used to save valuable field time and resources to maximize sampling efficiency for ASEs. Additionally, the distribution map can be used to target areas of highest probability of occupancy to increase sampling efficiency and garner the greatest impact on the population for suppression or eradication. Also, the distribution map informs areas of future invasion extent monitoring. The two small creeks that flow into the back of Willeo creek, and the northwest corner of the marsh adjacent to the CNC, for example, have never been sampled for ASEs, but their confluence marshes are predicted

as high occupancy probability areas. Thus, these tributaries may contain ASEs as this species is highly adaptable in their habitat use.

On a larger scale, this research can promote future investigations on other populations of ASEs. Currently, there are two populations in southern Florida in the proximity of Everglades National Park (Collins et al. 2002), which is ecologicaly sensitive but also has a prevalence of freshwater fish species introductions (Kline et al. 2014). Similar to the Georgia population, studies of ASEs in Florida have been plagued with sampling difficulties (Kline et al. 2014; Hill and Watson 2007), hindering management efforts. Leaf litter trap sampling with an occupancy modeling design may provide a valuable method for researchers of these populations due to its scalability.

Leaf litter trapping provided valuable inference about the population outside of the CNC at a scale that would not have been possible with any other method used for ASE capture to date. Considering our detection locations, it is highly unlikely that the marsh population is dependent upon the CNC ponds. ASEs are notoriously sedentary and the detections of juveniles on the opposite side of the river suggests that reproduction is occurring independent of the CNC. If there are reproducing adult ASEs outside of the CNC, they represent the most impactful life stage for suppression or eradication. To promote effective management of this population, future research should focus on detection of adults outside of the CNC.

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Tables and Figures

Table 1. All site-level and sampling-level covariates (excluding year), measure, model abbreviation, method of measurement or calculation, and summary statistics for Asian Swamp Eel sampling.

Covariate Level	Measure (Model Abreviation)	Method
Sampling	Temperature (Temp)	Measured at center of transect (YSI probe)
Sampling	Depth (Mtd)	Calculated mean of LLT (5) depths (0.10 M)
Site	Temperature Variation (Tvar)	Calculated temperature (ºC) variation from point data and continuous monitoring sites
Site	Mean Depth (Msd)	Calculated mean transect depth (0.10 M) from point data and continuous monitoring sites
Site	Submersed Aquatic Vegetation (Sub)	Calculated mean % area from quadrat sampling
Site	Emergent Aquatic Vegetation (Emg)	Calculated mean % area from quadrat sampling
Site	Boulder Substrate (Bldr)	Calculated mean % area from quadrat sampling
Site	Gravel Substrate (Grvl)	Calculated mean % area from quadrat sampling
Site	Sand Substrate (Snd)	Calculated mean % area from quadrat sampling
Site	Silt Substrate (Slt)	Calculated mean % area from quadrat sampling
Site	Organic Substrate (Org)	Calculated mean % area from quadrat sampling
Site	Distance (Dst)	Eucledian distance (M) measured within GIS
Site	Mean Temperature (Mst)	Calculated mean transect temperature (ºC) from point data and continuous monitoring sites

Table 2. Prediction error statistics for an interpolation model selection example from 2016 vegetation covariate. Model in bold was the selected model for this covariate from the 5 shown prediction error statistics.

Model	Mean Prediction Error[1]	Root Mean Squared Standardized Prediction Error[2]	Mean Standardized Prediction Error[3]	Root Mean Squared Error[4]	Average Standard Error[5]
Covariance	0.002	0.826	-0.006	1.066	0.769
Covariance Optimized	-0.015	0.808	0.001	0.910	0.850
Covariance with Anistropy	-0.021	0.811	0.399	1.013	0.809
Covariance Optimized with Anistropy	**-0.001**	**0.806**	**0.019**	**0.882**	**0.874**
Semivariogram	-0.009	0.809	-0.001	0.951	0.809
Semivariogram Optimized	-0.012	0.805	-0.010	0.960	0.800
Semivariogram with Anistropy	-0.003	0.805	-0.008	1.016	0.791
Semivariogram Optimized with Anistropy	-0.009	0.802	-0.002	0.946	0.810

1: Value should be close to zero
2: Value should be close to 1
3: Value should be close to zero
4: Value should be as small as possible and close to average standard error
5: Value should be as small as possible and close to root mean squared error

Table 3. Capture summary from 2015 and 2016 sampling with leaf litter traps for Asian Swamp Eels in the Chattahoochee River, Georgia.

2015										
Occasion	1	2	3	4	5	6	7	8	9	10
Date	06/30-07/05	07/06-07/09	07/10-07/18	07/19-07/21	07/22-07/24	07/25-07/28	07/31-08/02	08/03-08/05	08/06-08/07	08/08-08/09
# of Transects with detections	3	6	5	0	3	2	4	1	2	5
# of Eels Captured	6	9	7	0	5	4	5	1	3	7
Mean Length (mm)	38.3	43.3	46.9	NA	58.4	54.3	57.6	77	76.3	71.9
2016										
Occasion	1	2	3	4	5	6	7	8	9	10
Date	06/24-06/30	07/01-07/07	07/08-07/12	7/13-7/17	07/18-07/24	07/25-07/27	07/28-07/30	07/30-08/02	08/03-08/05	08/05-08/07
# of Transects with detections	0	2	2	3	3	4	4	3	3	2
# of Eels Captured	0	3	3	5	3	5	6	4	5	2
Mean length (mm)	N/A	165	46.3	40.8	54	65.4	63.5	51.8	51.6	70.5

Table 4. Candidate models of Asian Swamp Eel, leaf litter trap detection probability (p), with no covariates for occupancy (psi), K is the number of parameters in the model, AIC and derivatives are model selection criteria, see Table 1 for acronym definitions.

Model	K	AIC	ΔAIC	W
psi(.)p(Mtd+Temp+Year)	5	406.7	0.00	0.40
psi(.)p(Temp+Mtd)	4	407.3	0.59	0.30
psi(.)p(Mtd+Season)	4	407.7	0.98	0.24
psi(.)p(Mtd)	3	412.2	5.47	0.03
psi(.)p(.)	2	412.9	6.16	0.02
psi(.)p(Season)	3	414.4	7.71	0.01
psi(.)p(Temp)	3	414.7	8.00	0.01

Table 5. Correlation matrix (Pearson's r) of Asian Swamp Eel occupancy covariates from leaf litter trap sampling in the Chattahoochee River, Georgia.

	Dst	Veg	Slt	Msd	Mst	Tvar
Dst	1	-0.48	-0.51	0.01	-0.12	-0.37
Veg		1	0.66	-0.25	0.52	0.43
Slt			1	0.01	0.29	0.42
Msd				1	-0.47	0.14
Mst					1	0.24
Tvar						1

Table 6. Top models of Asian Swamp Eel occupancy probability (psi), top model covariates for detection probability not shown, K is the number of parameters in the model, AIC and derivatives are model selection criteria, see Table 1 for acronym definitions. Only models < 5 ΔAIC shown.

Model	K	AIC	ΔAIC	W
psi(Veg+Slt)	6	363.91	0.00	0.17
psi(Veg+Slt+Tvar)	7	364.84	0.92	0.11
psi(Dst+Veg+Slt)	7	364.93	1.02	0.10
psi(Veg+Slt+Msd)	7	365.85	1.94	0.06
psi(Veg+Slt+Mst)	7	365.88	1.97	0.06
psi(Dst+Veg+Slt+Tvar)	8	366.24	2.33	0.05
psi(Dst+Veg+Slt+Mst)	8	366.67	2.76	0.04
psi(Veg+Slt+Mst+Tvar)	8	366.75	2.83	0.04
psi(Veg+Slt+Msd+Tvar)	8	366.83	2.92	0.04
psi(Dst+Veg+Slt+Msd)	8	366.93	3.02	0.04
psi(Dst+Veg)	6	367.06	3.15	0.04
psi(Veg+Slt+Msd+Mst)	8	367.59	3.68	0.03
psi(Dst+Veg+Tvar)	7	368.05	4.14	0.02
psi(Dst+Veg+Slt+Msd+Tvar)	9	368.24	4.33	0.02
psi(Veg+Tvar)	6	368.35	4.43	0.02
psi(Dst+Veg+Slt+Msd+Mst)	9	368.37	4.46	0.02
psi(Dst+Veg+Mst)	7	368.40	4.49	0.02
psi(Veg)	5	368.55	4.63	0.02
psi(Veg+Slt+Msd+Mst+Tvar)	9	368.57	4.66	0.02

Table 7. Top models for interpolation of all covariates of Asian Swamp Eel occupancy in the Chattahoochee River for 2015 and 2016 and prediction error statistics which were used as model selection criteria. See Table 1 for covariate definitions and method of calculation or measurement.

Covariate	Model	Mean Prediction Error	Root Mean Squared Standardized	Mean Standardized Prediction Error	Root Mean Squared Error	Average Standard Error	Anistropy
				2015			
Veg	Semivariogram	0.002	1.095	0.010	0.213	0.209	anistropy
Slt	Semivariogram	0.001	1.048	0.006	0.246	0.239	anistropy
Tvar	Semivariogram	0.370	1.097	0.037	4.371	5.689	no anistropy
Mst	Semivariogram	0.228	0.935	0.082	1.998	1.986	no anistropy
				2016			
Veg	Covariance	0.001	0.882	0.019	0.806	0.875	anistropy
Slt	Semivariogram	0.084	0.954	0.086	0.818	0.841	anistropy
Tvar	Semivariogram	0.019	0.971	0.027	0.630	0.634	no anistropy
Mst	Semivariogram	0.037	0.933	0.017	0.080	0.844	anistropy

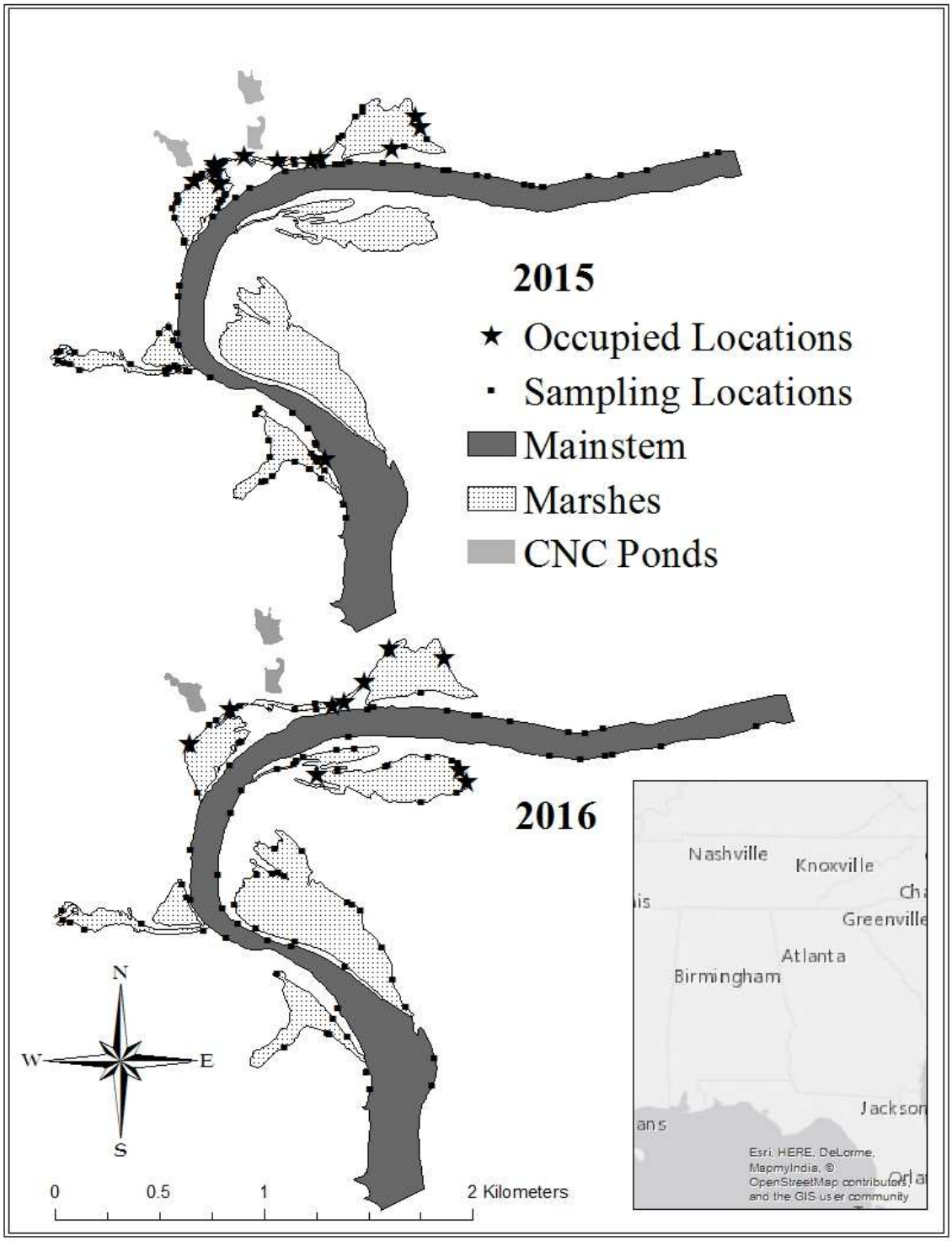


Figure 1. Study area extent and sampling locations for Asian Swamp Eels with leaf litter traps in 2015 and 2016 in the Chattahoochee River, adjacent to the Chattahoochee Nature Center (CNC) ponds, Georgia.

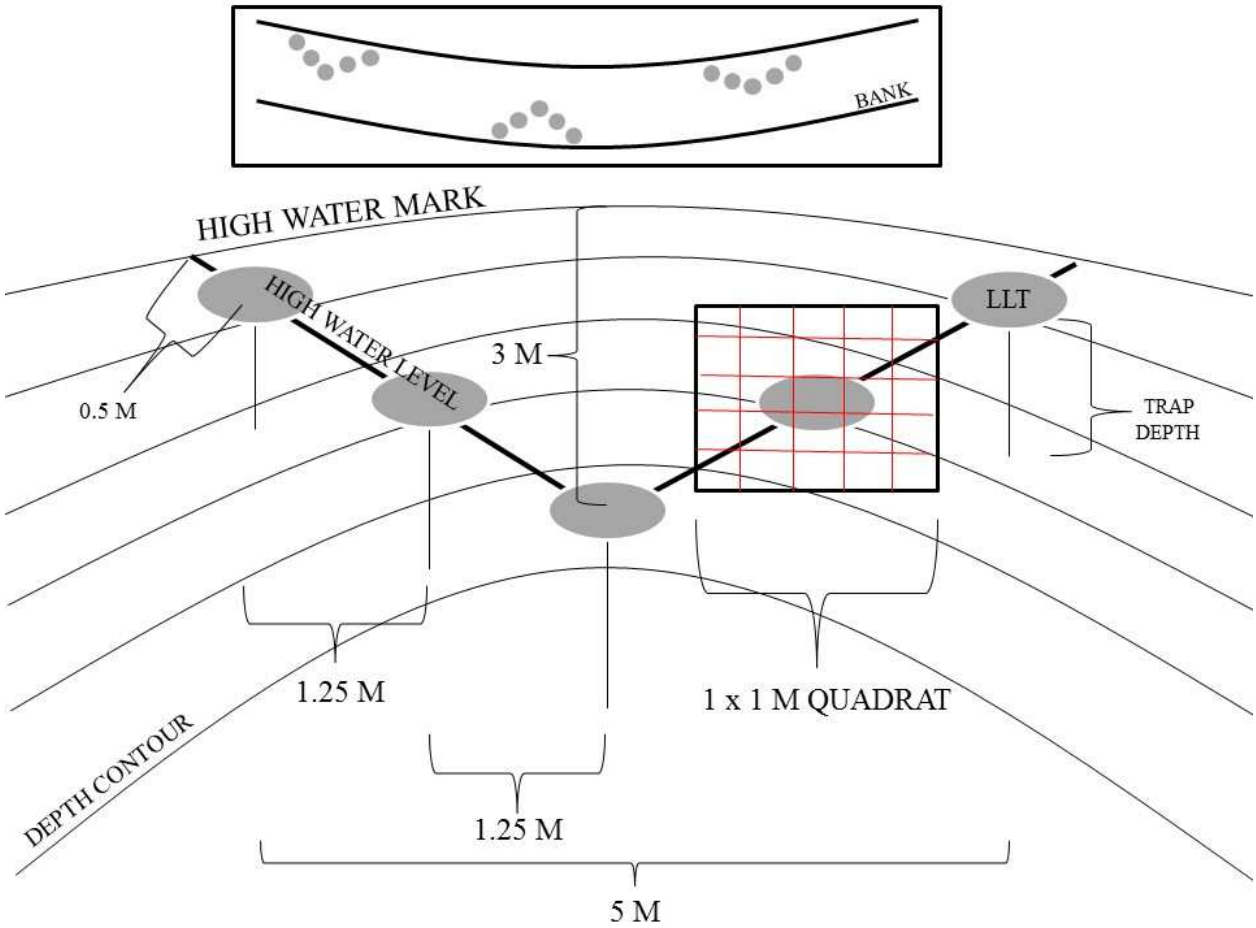


Figure 2. Distribution of leaf litter traps for Asian Swamp Eel sampling within a transect and example placement of quadrat for quantifying habitat variables.

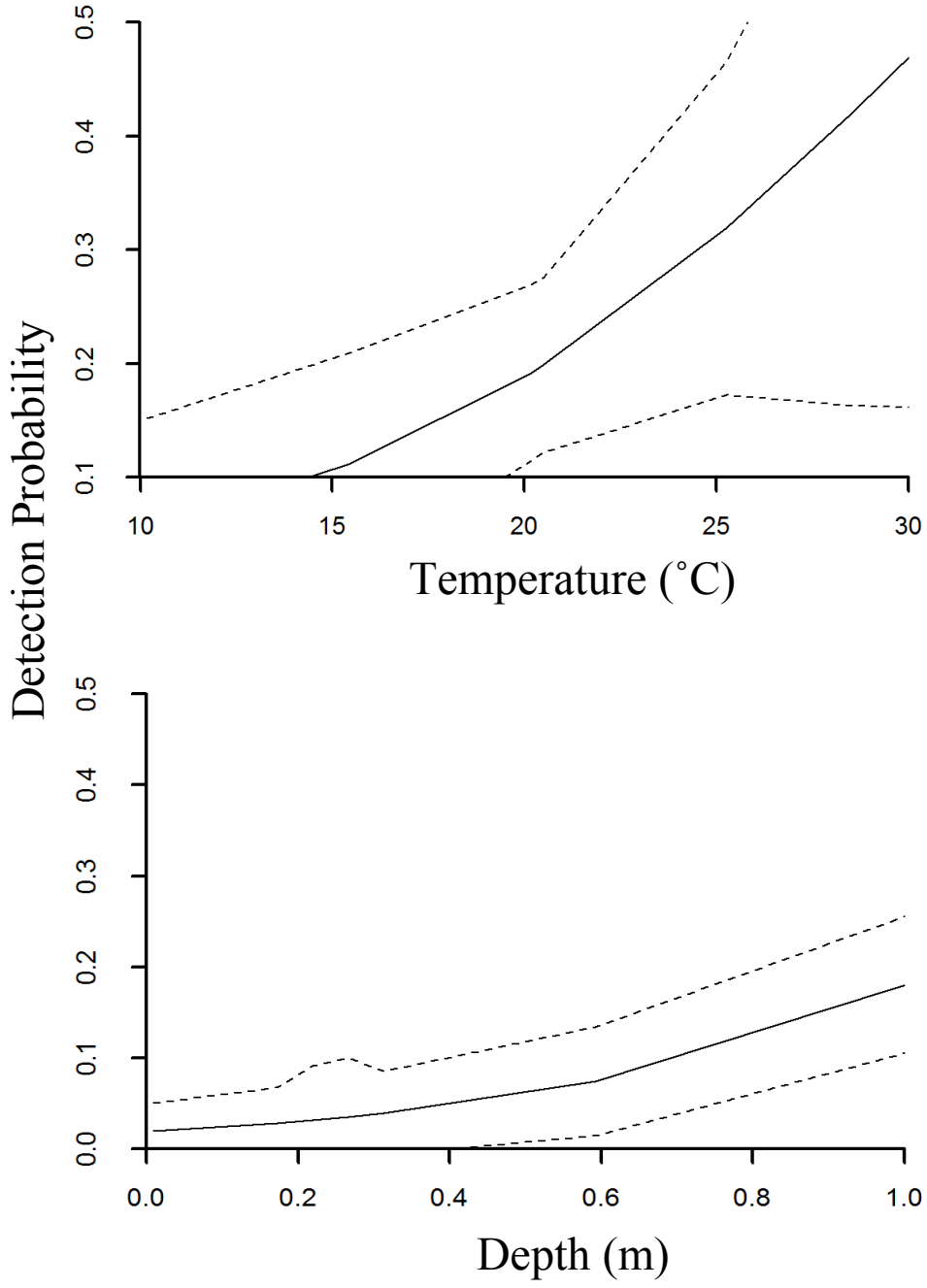


Figure 3. Detection probability vs top model covariates for 2015 from ASE LLT sampling in the Chattahoochee River, GA.

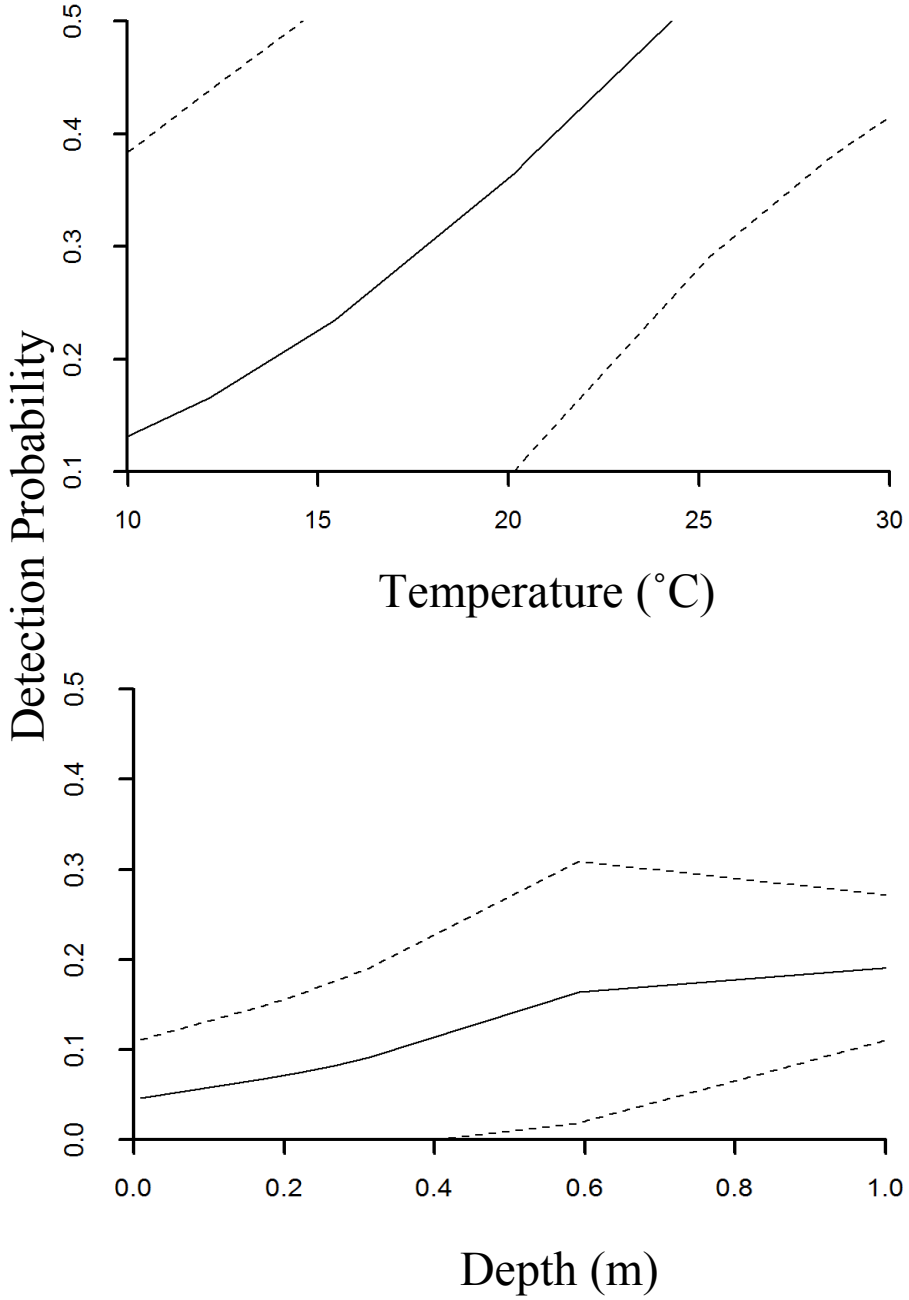


Figure 4. Detection probability vs top model covariates for 2016 from ASE LLT sampling in the Chattahoochee River, GA.

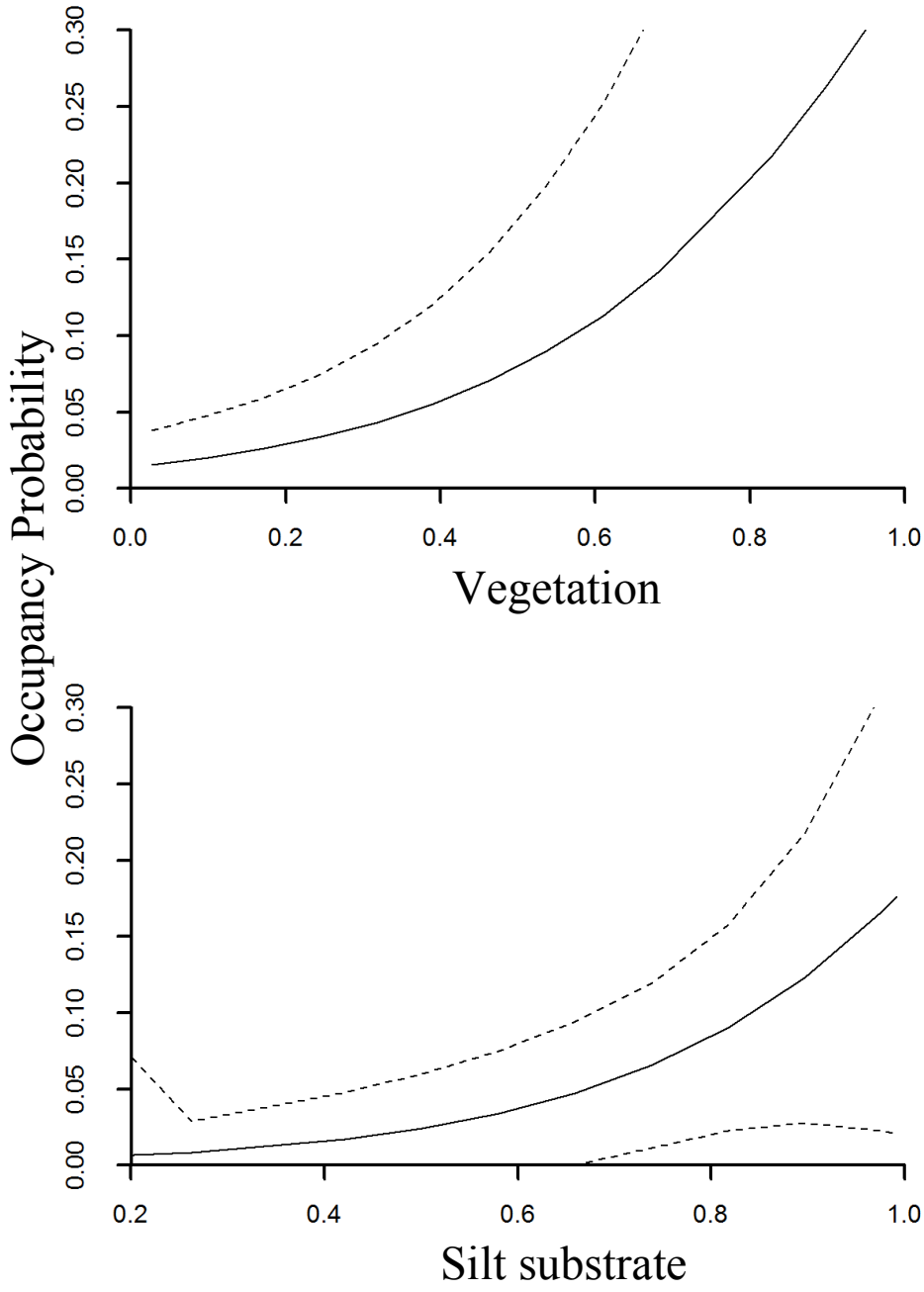


Figure 5. Occupancy probability vs top model covariates for Asian Swamp Eel leaf litter trap sampling in the Chattahoochee River, GA from 2015 and 2016.

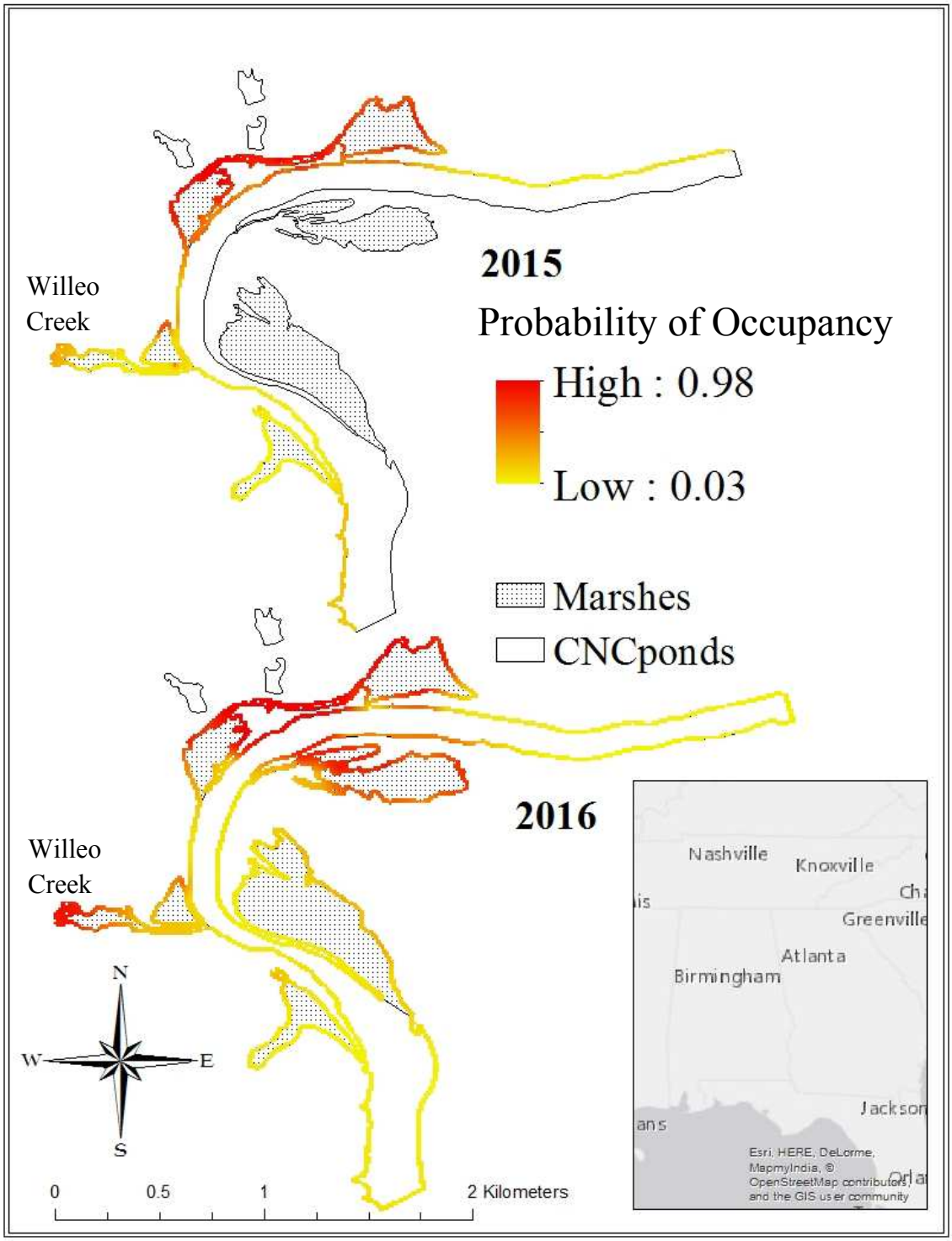
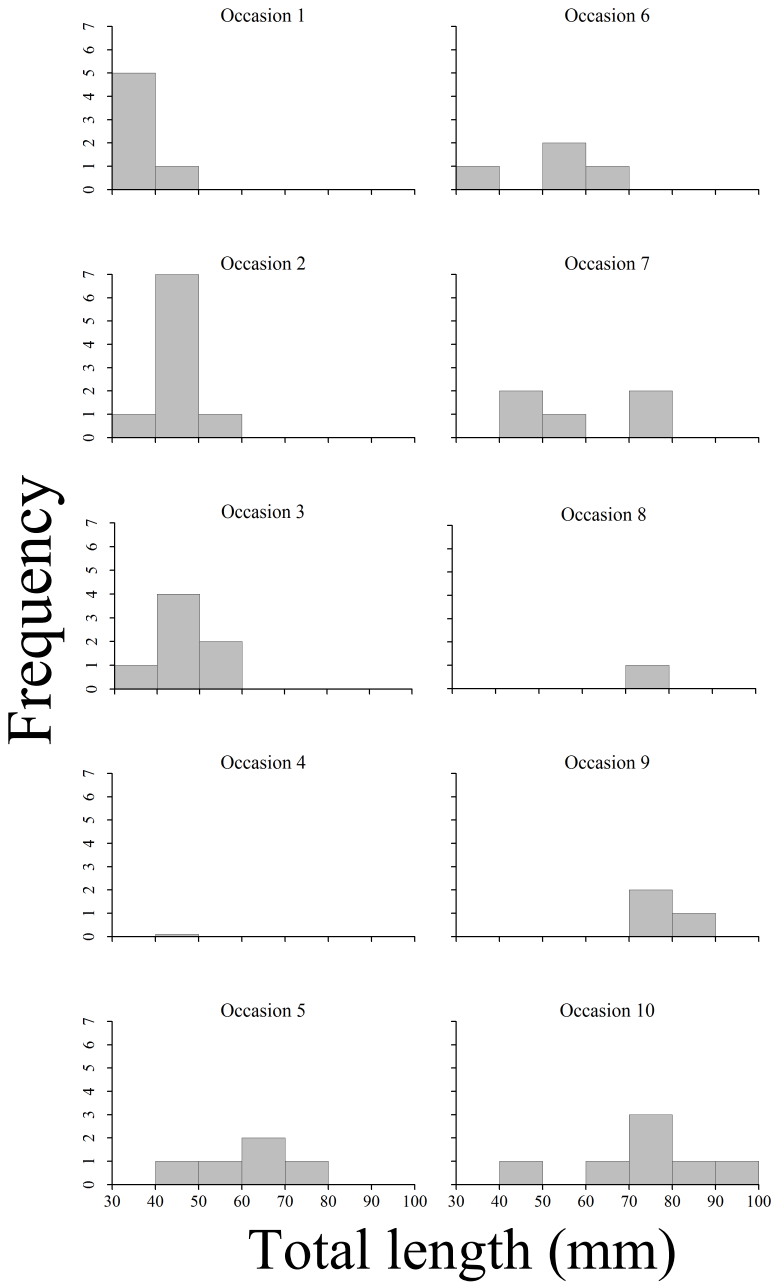


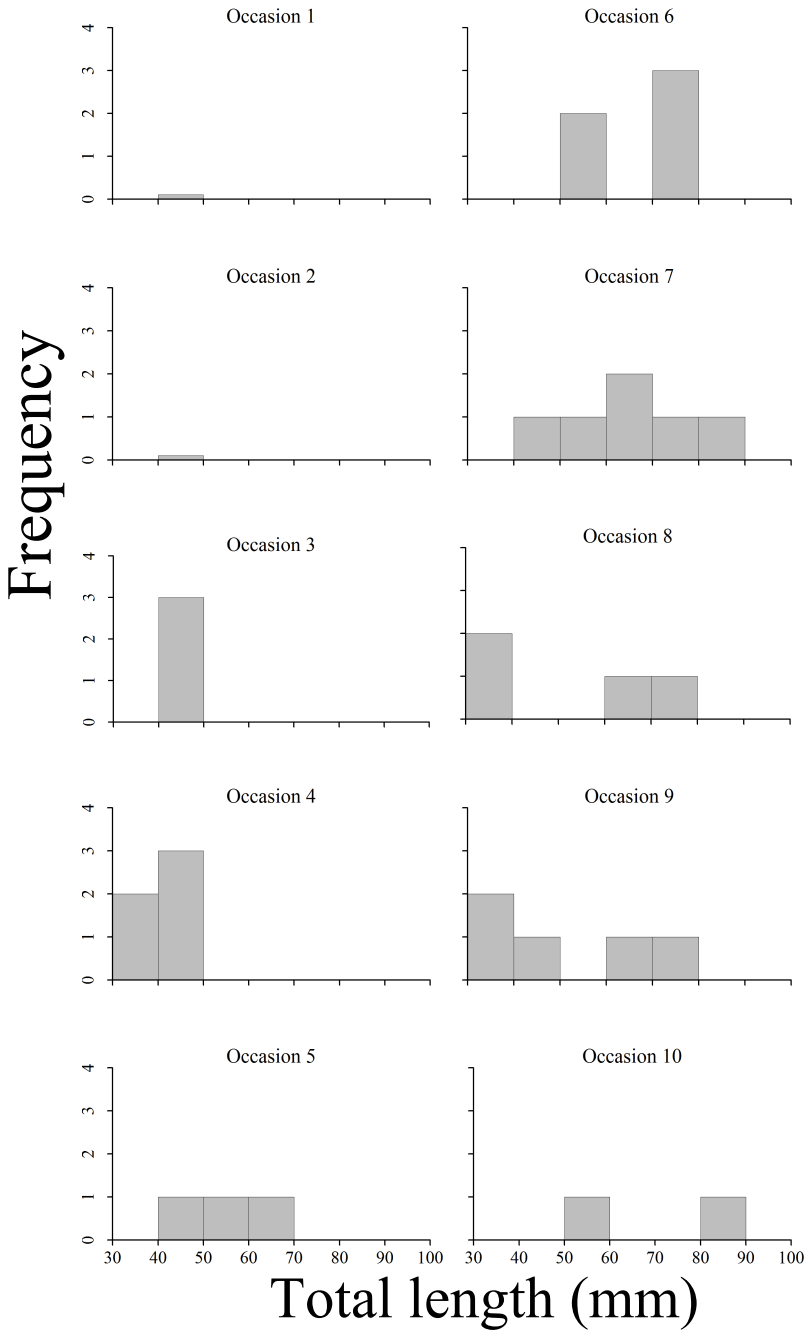
Figure 6. Asian swamp eel distribution map from backtransformed probabilities of occupancy to interpolated covariate values for the entirety of the study area in the Chattahoochee River, GA.

Chapter I Appendices

Appendix A1. Length frequency histogram of juvenile Asian Swamp Eels captured with leaf litter traps in the Chattahoochee River, Georgia 2015.



Appendix A2. Length frequency histogram of juvenile Asian Swamp Eels captured with leaf litter traps in the Chattahoochee River, Georgia 2016.



CHAPTER II

DETECTION PROBABILITY OF ADULT ASIAN SWAMP EEL (*MONOPTERUS ALBUS*) FROM MULTIPLE HABITATS IN GEORGIA

Asian swamp eels (*Monopterus albus*, ASE) are native to tropical and temperate climates of southeast Asia, Indonesia and Australia (Rosen and Greenwood 1976; Berra 2001), where they are found in a variety of habitats (Liem 1961,1963,1967), and have recently been documented in multiple locations in the United States. In their native range ASEs have been referred to as strict carnivores and voracious predators, emphasizing concern about their potential direct impact on native fauna through predation (Liem 1998; Freeman et al. 2005). Adult ASEs are obligate air breathers and capable of overland migration, which means they may transcend typical freshwater fish migration corridors (Liem 1967; Graham et al. 1995). Captive ASEs have lived as long as 15 years, during which they exhibit a sequential protogynous hermaphroditic life history (Matsumoto et al. 2011), which may result in lower necessary propagule pressure to create viable populations. Asian Swamp Eels are able to endure periodic drought and desiccation through burrowing into the substrate, forming a protective mucus layer and detoxifying endogenous ammonia, promoting their persistence in highly variable environmental conditions (Liem 1987; Ip et al. 2004). Currently ASEs are known to occur in four

locations in the continental United States (Collins et al. 2002), but little is known about their impact at those invaded sites.

Of the four populations, three were discovered in Florida from 1997 to 1999, and one in northern Georgia in 1994. These populations have garnered research interest due to a variety of characteristics that make ASEs unique and increase their potential ecological impact as an invasive species (Shafland 2010). From genetic analysis, Collins et al. (2002) concluded that the three ASE populations in Florida were most closely related to specimens acquired from southern China through the Maylay peninsula and Indochina, where the Georgia population was more similar to specimens from a significantly more northern source in Japan or Korea. Also, mitochondrial DNA sequence divergence equivalent to that observed in some families of fish suggested that ASEs in the U.S. likely represent three or more distinct species. As a result of this, Freeman et al. (2005) referred to the Georgia population as an undescribed species (*Monopterus* sp. *cf. M. albus*). Whether this taxonomic uncertainty reflects a difference in biology and ecology is uncertain.

After their initial discovery in 1994, various research endeavors took place on the Georgia population within three ponds of the Chattahoochee Nature Center (CNC) and an adjacent backwater marsh area of the Chattahoochee River near Atlanta (Starnes et al. 1998). In 1994, the population size structure (32 to > 225 mm TL) within the CNC ponds suggested that the original introduction was prior to 1990 and successful reproduction was taking place. (Starnes et al. 1998; Freeman et al. 2005). Further research investigated trophic level and gut contents, gonad analysis, age and growth, mark recapture, and evaluation of sampling techniques. Trophic analyses placed ASEs as

low level predators and gut content analysis supported this finding with the majority of items being aquatic invertebrates (Freeman and Burgess 2000; Freeman et al. 2005). Gonad analysis from 44 ASEs captured within the CNC ponds yielded a sex ratio of 5:1:1 (female to male to intersex) which led researchers to believe the ponds may contain an abnormal population structure and have a prolonged breeding season (Freeman et al. 2005). Age and growth analyses of juvenile ASEs estimated hatch dates from June 13th to August 7th (Long and LaFleur 2011), placing some boundaries on the length of the prolonged breeding season. Mark-recapture efforts estimated a population size from 53,000 to 116 million individuals, with the large variation due to low and variable capture and recapture probability (0.0001% and 0.82%, respectively), (Freeman et al. 2005). Multiple sampling techniques have been implemented (e.g. multiple electrofishing methods, barrel traps, and leaf litter traps) with varying degrees of success (Freeman et al. 2005). In general, however, Juvenile ASEs have been most successfully sampled with leaf litter traps (Chapter 1), while adult ASEs have been captured most effectively with backpack electrofishing (Freeman et al. 2005).

Although important research questions have been addressed through the study of juvenile ASEs, a robust population monitoring, control or eradication program should contain a method to effectively capture adults, and include descriptive information about the efficacy of the method. Therefore, our research objective was to evaluate three sampling methods and model detection and occupancy probabilities for adult ASEs under heterogeneous conditions.

Methods

Study Area

The Chattahoochee Nature Center is a private facility in the greater Atlanta area that provides tours, education programs, and other nature learning opportunities. There are multiple ponds on the property, in which ASEs are known to occur in three (Beaver, Kingfisher and Frog), as well as the adjacent backwater marsh area of the Chattahoochee River (Freeman and Burgess 2005) (Figure 1). We constrained our study site to two ponds (Beaver and Kingfisher) and the adjacent backwater marsh in the immediate vicinity of the CNC. Frog Pond was dry and prevented the use of our sampling methods, and adults have only been documented within the marsh at very limited distances from the CNC (<200 meters) (Freeman and Burgess 2005), but sampling effort has been low outside of this range.

Habitat

To evaluate the difference in habitat among the two ponds and the marsh, habitat within the littoral zone of each pond was quantified 1 meter from the shoreline at 10-meter intervals around the entire perimeter (Figure 2). A quadrat method was used to quantify submersed and emergent vegetation, open water area, and woody debris (Krebs 1999). The quadrat consisted of a 1 x 1 meter frame with 25 sub-units. The quadrat was placed at each 10-meter interval and the dominant habitat category (submersed vegetation, emergent vegetation, or open water area) was recorded for each sub-unit, from which we calculated proportions for each interval. The same method was used to quantify woody debris, but not exclusive to other habitat categories (e.g., woody debris

recorded as present or absent in each subunit, in addition to dominant habitat type). Additionally, at each 10-meter interval, depth was recorded, substrate was classified as hard or soft by probing with a dowel, and presence of leaf litter was recorded.

Sampling

At each transect, three methods were used for sampling. In each pond, 10 of the intervals were randomly selected for sampling. In the marsh 5, 10-meter transects were selected for sampling. Canoe-based backpack electrofishing (Smith-Root LR-24), and eel-pots (Gee's Minnow Trap/Eel Pot Trap #G40EP) baited with canned tuna and deployed overnight in two ways (floating or submerged) were employed from 05/27/2016 to 06/18/2016 in a temporally randomized sampling schedule to eliminate potential bias in sampling time. All three methods were used from 05/27/2016 to 06/06/2016, after which only canoe based electrofishing was implemented through 07/02/2016. Sampling transects were defined as a 10 x 2 meter area of the littoral zone, and site-level and sampling level information was recorded at each transect at the beginning of the sampling season (05/24/2016 – 05/26/2016) and at every sampling event respectively, to be included as covariates in an occupancy modeling approach (Table 1). Site-specific covariates were considered fixed and collected to investigate how habitat variables may influence adult ASE occupancy over the entire sampling season (MacKenzie 2006). Site-level information was collected in the same manner as habitat sampling but at the 2, 4, 6, and 8-meter intervals within the defined 10-meter transect to gain a finer scale measure of habitat within sampled areas (Figure 2). Additionally, temperature loggers (HOBO pendant) were placed at the center of all transects to provide a comprehensive thermal history. Sampling-level information was recorded at each transect during each sampling

event (conductivity [μs], and time of day) to model how variation in sampling conditions may influence detection of adult ASEs. Temperature loggers provided the temperature at the time of sampling for an additional sampling-level covariate.

Analysis

Occupancy modeling was performed with package "unmarked" in program R (R Studio Team 2017), based on site and sampling level covariates. Each area (Beaver Pond, Kingfisher Pond, and marsh) was analyzed separately due to likely differences in population level characteristics (i.e., time since introduction, abundance, behavior) and differences in habitat (Table 2). Model selection was performed with AIC criteria considering ΔAIC and dispersion of AIC weight among models (Burnham and Anderson 2002). For each area, the most global model was assessed for fit with a chi-square parametric bootstrap with 10,000 simulations (Fiske and Chandler 2011).

Results

Beaver and Kingfisher ponds, and the adjacent backwater marsh area of the Chattahoochee River represented three different habitats currently occupied by ASEs (Table 2). During the temporally randomized sampling schedule that incorporated all three methods from 05/27/2016 to 06/06/2016, no ASEs were captured with either method of eel pot sampling (submerged or floating) from 175 pot nights. Therefore, after 06/06/2016, only canoe-based backpack electrofishing was implemented and all transects were sampled on 10 occasions until 07/02/2016. Electrofishing effort at each transect was variable by time (mean 135 seconds ± 25 [SD]), but treated as equivalent effort. Over the 10 sampling occasions, 76 ASEs were captured from all 10 transects in Beaver

Pond ranging from 117mm to 624mm; 19 ASEs from 9 unique transects in Kingfisher pond ranging from 123mm to 655mm; and 5 ASEs from 3 unique transects in the marsh ranging from 208mm to 595mm (Table 3). ASEs were captured during every sampling event in Beaver Pond, 8 sampling events in Kingfisher Pond, and 3 sampling events in the marsh (Table 4). The number of eels captured during a sampling occasion ranged from 2 to 20 in Beaver Pond, 0 to 6 in Kingfisher Pond, and 0 to 2 in the marsh (Appendix A).

The marsh sampling supplied an inadequate amount of data to model detection or occupancy due to overdispersion (Mackenzie – Bailey goodness of fit test, c-hat =10.3). Although this prevents any statistical analysis of detection and occupancy in the marsh, the sampling provided valuable documentation of an adult ASE population outside of the CNC ponds.

Sampling data was adequate to model detection probability for Beaver and Kingfisher ponds (MacKenzie-Bailey goodness of fit test, c-hat 0.90 and 1.71 respectively), but the proportion of occupied sites prevented modeling and interpretation of occupancy probabilities. For Beaver and Kingfisher ponds, all unique combinations of sampling level covariates were included in detection models to create a set of 7 candidate models of detection probability (Table 5). The top model of AIC selection criteria for Beaver Pond included temperature alone and was significantly better than the null model (likelihood ratio test, $\chi^2 = 7.62$, DF = 1, $P < 0.005$). This indicates that in Beaver Pond, detection probability was influenced by the temperature at the time of sampling but not by conductivity or time of day within the range of our sampling conditions. In Beaver Pond, detection probability decreased as sampling temperature increased (Figure 3). The

top model from AIC selection criteria for Kingfisher Pond was the null model of detection probability. In Beaver Pond, detection probability from the top model at mean sampling temperature was 0.450 ± 0.052 (SE), while in Kingfisher Pond, the detection probability form the null model was 0.19 ± 0.39 (SE).

Discussion

Canoe-based backpack electrofishing was the only method we used that allowed for the successful capture of adult Asian Swamp Eels among the multiple habitats and sampling conditions found near the CNC. Detection of ASEs was imperfect, and with the sample size acquired, we were unable to form any strong conclusions about variables that influence detection. From the habitat with the largest sample size of detections (Beaver Pond), detection probability decreased with increased sampling temperature. There are a number of potential explanations for lower detection probability at higher sampling temperatures, but we speculate that ASEs are exhibiting a different behavior (e.g. more active and capable of evasion, burrowing in the substrate, seeking refuge in deeper water) under higher water temperatures. In general, our results support the conclusions of past research regarding the difficulty of capturing adult ASEs (Freeman and Burgess 2000; Freeman et al. 2005), and show there is unexplained variation or a lack of environmental relation with detection probability.

There was a difference in total captures among the two ponds, and a possible difference in detection probability, although questionable due to overlapping standard error. If this potential difference in detection probability was not the result of population characteristics (e.g., time since introduction, abundance and behavior), it could be

attributed to some natural variation between habitats. Considering this, it is possible that the total captures are a result of variable detection probability among habitats. However, the population may be limited if our detection results are attributed to abundance in each area (Royle et al. 2005; McCarthy et al. 2013). Beaver Pond would be considered having the largest population, followed by Kingfisher Pond and the adjacent backwater marsh. This may be the result of an abundance gradient in the population from an original introduction location (i.e., Beaver Pond). An abundance gradient from the introduction point could exist for this population of ASEs, because Beaver Pond is bounded on the northern edge by a residential area, while Kingfisher pond is less accessible due to being secluded in the CNC and fenced on the western, southern and eastern margins. Patterns in fish introductions have been shown to be commonly related to proximity to human activity (Leprieur et al. 2008), which would be supported if Beaver Pond was the original introduction location. This could mean that the pond with the highest hypothetical abundance is closest to an introduction source (i.e., an aquarium dump from the residential area). Additionally, if there is a relationship between ASE abundance among habitats, detection probability could be used in eradication or suppression efforts as part of an adaptive management program. Specifically, as abundance in a location decreased with removal, sampling effort (i.e., number of repeat occasions at a site) would increase commensurate with detection probability to ensure absence.

Past research of ASEs in Georgia has concluded it is unlikely there is a substantial population within the backwater marsh areas of the Chattahoochee River (Freeman and Burgess 2000; Freeman et al. 2005). Our results contradict this hypothesis for two reasons: (1) ASEs have been present in the adjacent backwater marsh areas of the

Chattahoochee River since 2000, and adults since 2004, (2) recent research documented juvenile ASEs on the opposite side of the river and as far as 1,584 meters from the CNC (Chapter 1). Considering this information, it is likely that there is a successfully reproducing population within the backwater marsh areas of the Chattahoochee River.

The persistence of ASEs within the backwater marsh areas since 2000 (Freeman and Burgess 2000; Freeman et al. 2005), indicates they are likely successfully established. Although the backwater marsh areas have seen limited sampling activity, ASEs have been documented in 2000, 2004, 2005, 2008, 2015, and 2016 (Freeman and Burgess 2000; Freeman et al. 2005, Long and LaFleur 2011; Chapter 1). Success or persistence of introduced species is not clearly defined within the literature due to the substantial natural differences that exist among species introductions (Sakai et al. 2001; García-Berthou 2007), but we believe the consistency in documented presence of ASEs in the backwater marsh areas provides strong evidence for a persisting population.

Recent research in backwater marsh areas within a larger vicinity of the CNC ponds documented juveniles as far as 1,584 meters away, and on the opposite side of the Chattahoochee River (Chapter 1). Taking into account the biology of the species, it is unlikely that the ASEs documented at these locations were a result of emigration from the CNC ponds. Adult male ASEs have been observed mouth brooding juveniles until about 37mm TL (Matsumoto and Iawata 1997). The length of the ASE captured 1,584 meters away was 38mm and the length of ASEs captured on the opposite side of the river ranged from 38mm TL to 43mm TL. Considering the estimated growth rate of 2mm per day for juvenile ASEs (Long and LaFleur 2011), it is extremely unlikely that reproduction occurred other than the immediate vicinity of where the juveniles were observed.

A robust management plan for this invasive species should therefore incorporate a method to monitor the distribution of the species on a large spatial scale, and target life stages that allow for the greatest impact on the population (i.e., reproducing adults). Initial introduction into Beaver Pond and subsequent abundance gradient and limited marsh population is possible, but we believe there is more evidence to support the hypothesis of a substantial population in the backwater marsh areas that is being masked by low and variable detection. Additionally, it appears that ASEs within the backwater marsh areas are more widespread than previously thought. Canoe-based backpack electrofishing appears to be the most useful method for capturing adult ASEs for suppression and control. Future research efforts should focus on sampling for adults at the extremity of their current extent where only juveniles have been documented, and potentially limit the expansion of the population. Locations of interest at the extremities of the current distribution include Willeo Creek downstream of the CNC, the marsh directly on the opposite side of the river from the CNC, and the creek that enters the northern marsh that is immediately adjacent to the CNC (Chapter 1). Control or eradication of adult ASEs in these locations would be a critical step towards limiting the future distribution and managing the population.

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Tables and Figures

Table 1. All site-level and sampling-level covariates, measure, model abbreviation, method of measurement or calculation from Asian Swamp Eel sampling in Beaver and Kingfisher ponds and the adjacent backwater marsh area of the Chattahoochee River, Roswell, Georgia 2016.

Covariate Level	Measure (Model Abreviation)	Method
Sampling	Temperature (Temp)	Measured at center of transect (ºC) (YSI probe)
Sampling	Time of Day (Time)	Recorded at time of sampling
Sampling	Conductivity	Measured at center of transect (µs) (YSI probe)
Site	Temperature (Temp)	Calculated mean temperature (ºC) from continuous monitoring data loggers
Site	Mean Depth (Msd)	Calculated mean transect depth (meters) from 6 points within transect (0,2,4,6,8 and 10 meters)
Site	Submersed Aquatic Vegetation (Sub)	Calculated mean % area from quadrat sampling
Site	Emergent Aquatic Vegetation (Emg)	Calculated mean % area from quadrat sampling
Site	Woody Debris (Wdy)	Calculated mean % area from quadrat sampling
Site	Open Water Area (Open)	Calculated mean % area from quadrat sampling
Site - Categorical	Hard or Soft Substrate (Substr)	Calculated mean % hard substrate present from 6 points within transect (0,2,4,6,8 and 10 meters)
Site - Categorical	Leaf Litter Presence or Absence (Lfltr)	Calculated mean % leaf litter present from 6 points within transect (0,2,4,6,8 and 10 meters)

Table 2. Summary of habitat variables collected at 10-meter intervals in Beaver and Kingfisher ponds within the Chattahoochee Nature Center. Habitat variables from the marsh were from the fine scale measurements taken at the end points and 2,4,6, and 8 meter intervals form the 5 transects in the adjacent backwater marsh area of the Chattahoochee River, Roswell, Georgia 2016. From Table 1, % Veg is the combination of Emg and Sub, % Woody is Wdy, and % Soft is soft Substr.

Variable	Beaver	Kingfisher	Marsh
% Veg	12	2	62
% Woody	11	9	11
% Soft	88	67	100
Mean Depth (m) (SD)	0.43 (0.18)	0.39 (0.19)	0.28 (0.20)
Mean Temp (SD)	28.32 (1.23)	28.25 (1.49)	24.88 (4.75)

Table 3. Capture summary for adult Asian Swamp Eels with canoe based electrofishing sampling in Beaver and Kingfisher ponds and the adjacent backwater marsh area of the Chattahoochee River, Roswell, Georgia 2016.

Location	Beaver	Kingfisher	Marsh
# of Occupied Transects	10	9	2
Total # Captured	76	19	5
Mean Length (mm) - (Range)	321(117-624)	273(123-655)	405(208-595)

Table 4. Capture summary for adult Asian Swamp Eels from individual sampling occasions with canoe based electrofishing sampling in Beaver and Kingfisher ponds and the adjacent backwater marsh area of the Chattahoochee River, Roswell, Georgia 2016.

Beaver										
Occasion	1	2	3	4	5	6	7	8	9	10
Date	7-Jun	8-Jun	9-Jun	10-Jun	12-Jun	13-Jun	14-Jun	15-Jun	16-Jun	17-Jun
# of Transects with detections	7	5	7	4	4	2	4	8	2	6
# of eels captured	20	8	8	5	5	2	6	12	3	7
Mean length (mm)	328	316	290	456	330	322	353	285	266	298
Kingfisher										
Occasion	1	2	3	4	5	6	7	8	9	10
Date	7-Jun	8-Jun	9-Jun	10-Jun	12-Jun	13-Jun	14-Jun	15-Jun	16-Jun	17-Jun
# of Transects with detections	2	2	1	0	0	2	6	3	1	2
# of eels captured	2	2	1	0	0	2	6	3	1	2
Mean length (mm)	351	361	215	NA	N/A	290	197	370	169	255
Marsh										
Occasion	1	2	3	4	5	6	7	8	9	10
Date	1-Jun	8-Jun	9-Jun	10-Jun	12-Jun	13-Jun	13-Jun	17-Jun	21-Jun	2-Jul
# of Transects with detections	0	1	0	0	0	0	1	2	0	0
# of eels captured	0	2	0	0	0	0	1	2	0	0
Mean length (mm)	N/A	535	N/A	N/A	N/A	N/A	331	312	N/A	N/A

Table 5. Detection models and selection criteria from canoe-based backpack electrofishing in Beaver and Kingfisher ponds for Asian Swamp Eels at the Chattahoochee Nature Center, Roswell, Georgia 2016.

Model	K	AIC	Δ AIC	W
		Beaver		
psi(.)p(Tmp)	3	135.62	0	0.3794
psi(.)p(Tmp+Time)	4	136.83	1.21	0.2072
psi(.)p(Tmp+Cond)	4	137.53	1.91	0.1458
psi(.)p(Cond)	3	138.33	2.71	0.0978
psi(.)p(Tmp+Time+Cond)	5	138.6	2.98	0.0853
psi(.)p(Time+Cond)	4	139.55	3.93	0.0532
psi(.)p(.)	2	141.24	5.62	0.0229
psi(.)p(Time)	3	143.23	7.61	0.0084
		Kingfisher		
psi(.)p(.)	2	101.25	0	0.246
psi(.)p(Tmp)	3	101.33	0.087	0.235
psi(.)p(Cond)	3	102.43	1.18	0.136
psi(.)p(Time)	3	102.71	1.468	0.118
psi(.)p(Tmp+Cond)	4	103.27	2.02	0.089
psi(.)p(Tmp+Time)	4	103.3	2.054	0.088
psi(.)p(Time+Cond)	4	104.28	3.035	0.054
psi(.)p(Tmp+Time+Cond)	5	105.2	3.957	0.034

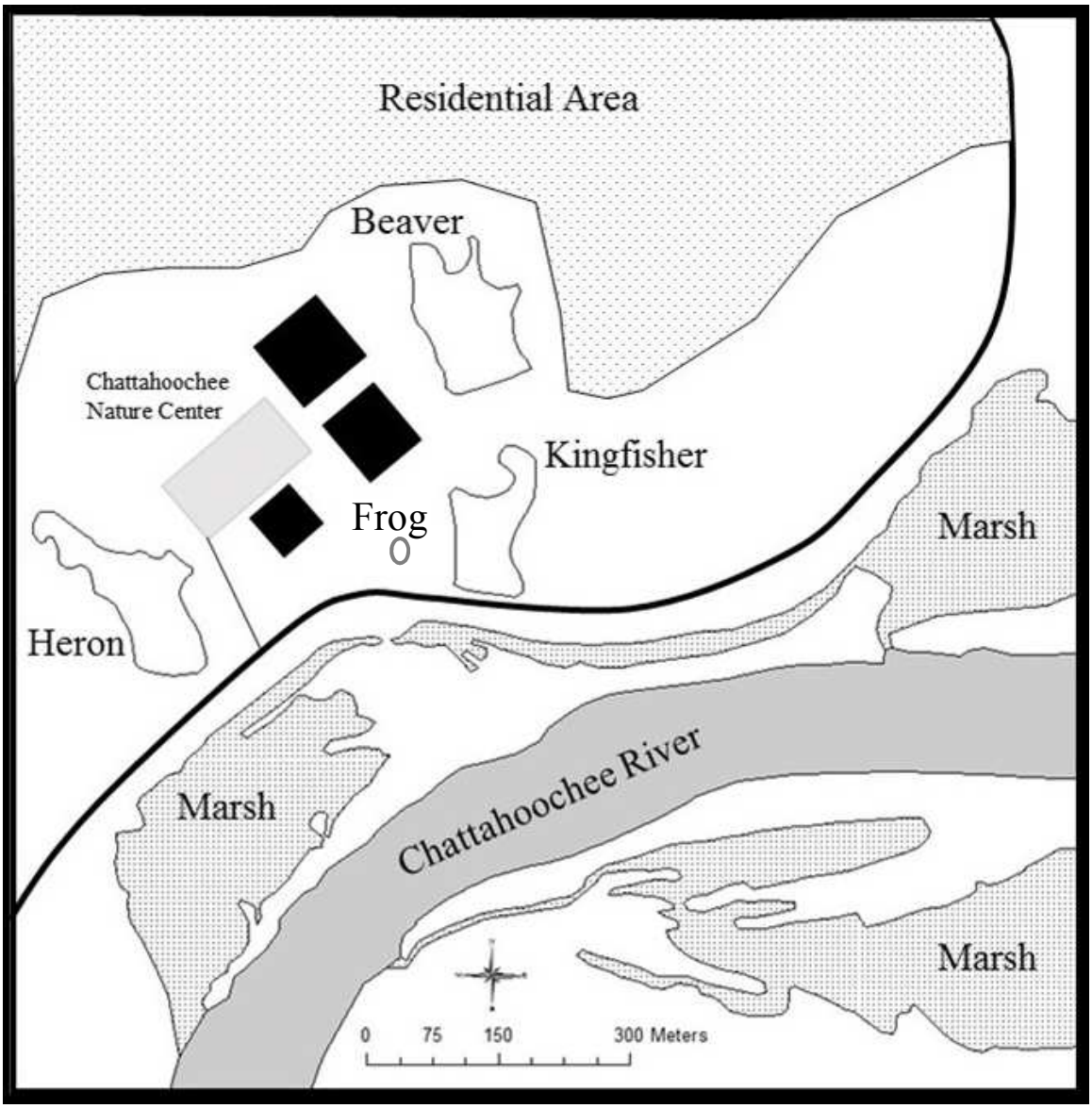


Figure 1. Study area showing sampled ponds (Beaver and Kingfisher), and adjacent backwater marsh area for adult Asian Swamp Eel sampling, Roswell, Georgia.

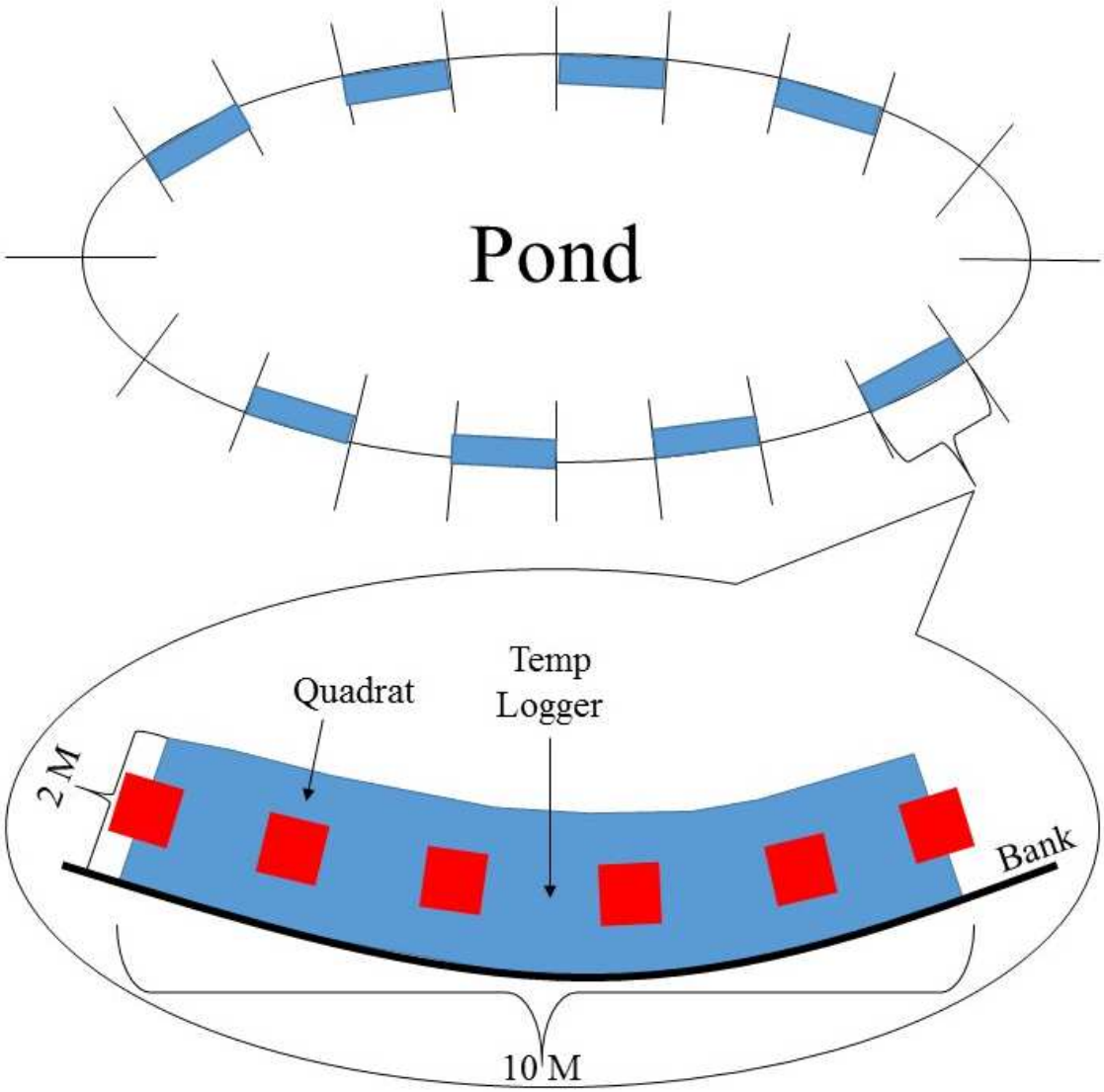


Figure 2. Example transect layout for habitat quantification and sampling for adult Asian Swamp Eel within the Chattahoochee Nature Center ponds, Roswell, Georgia.

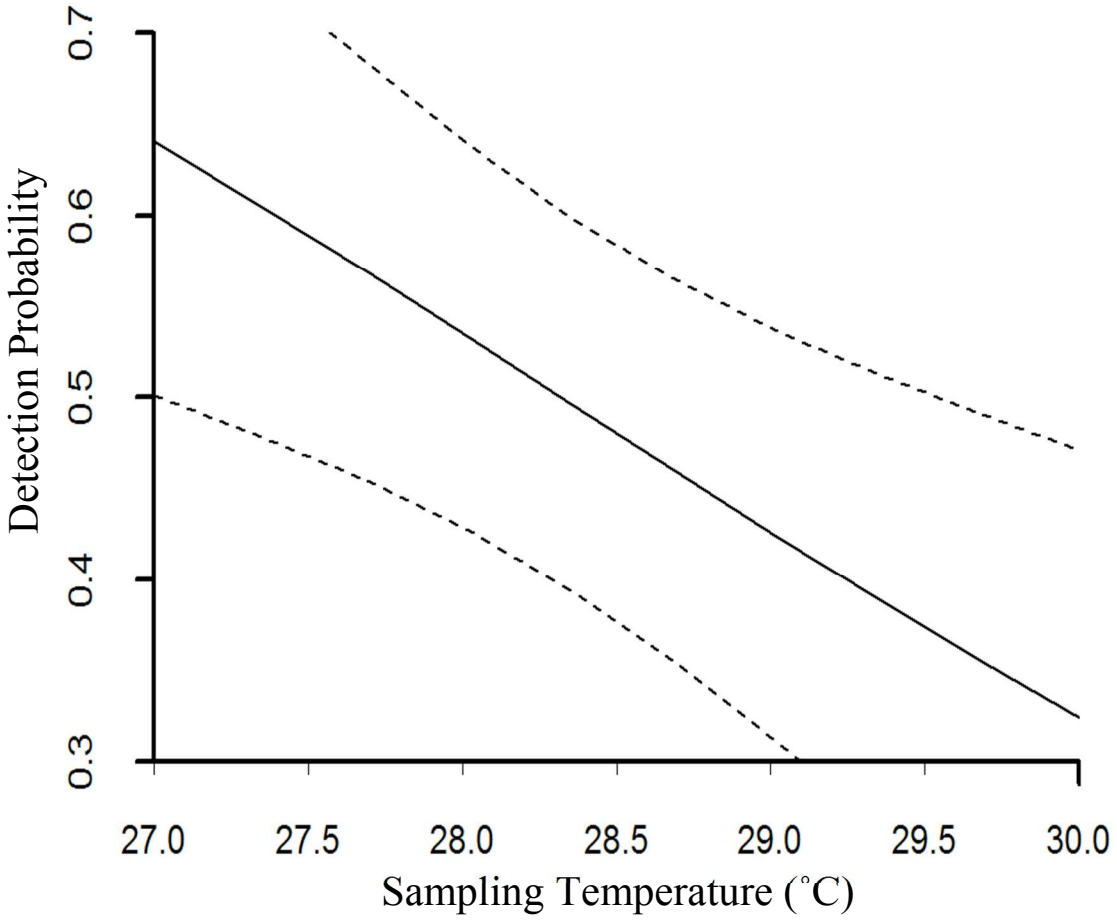
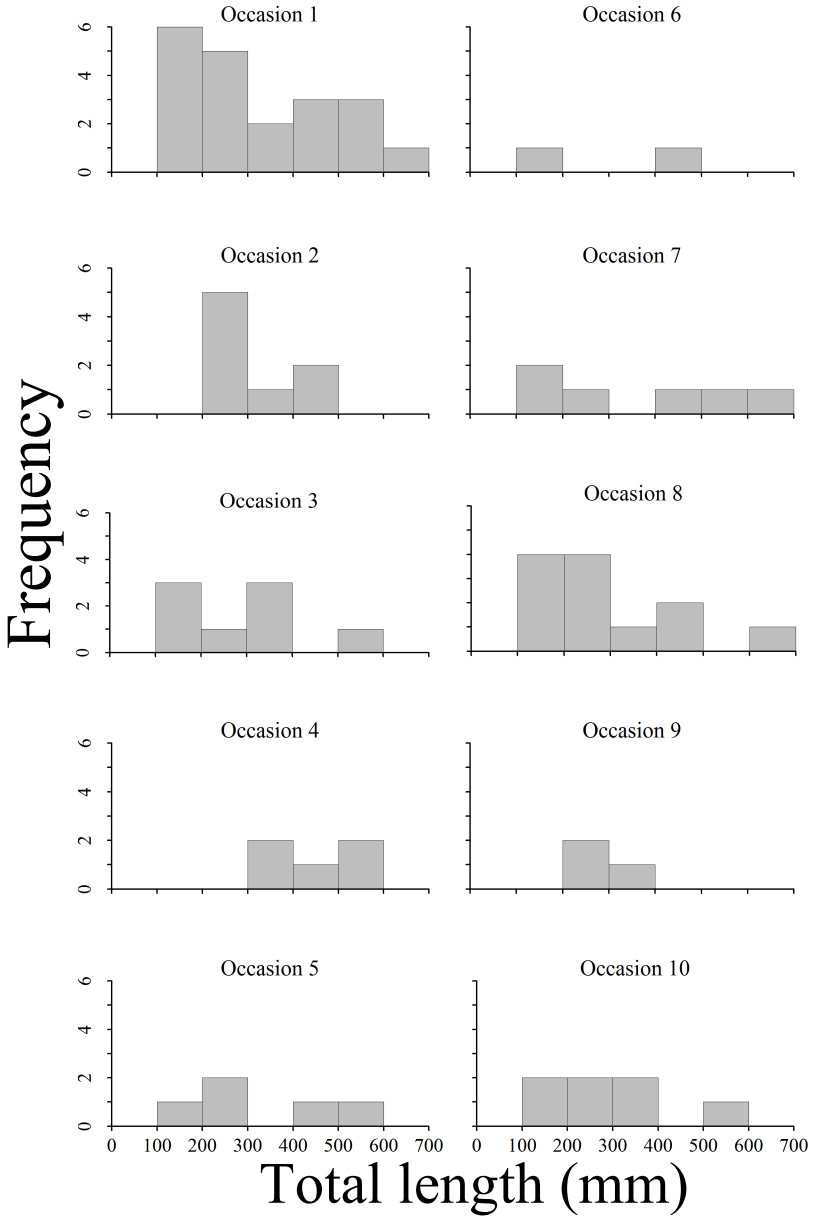


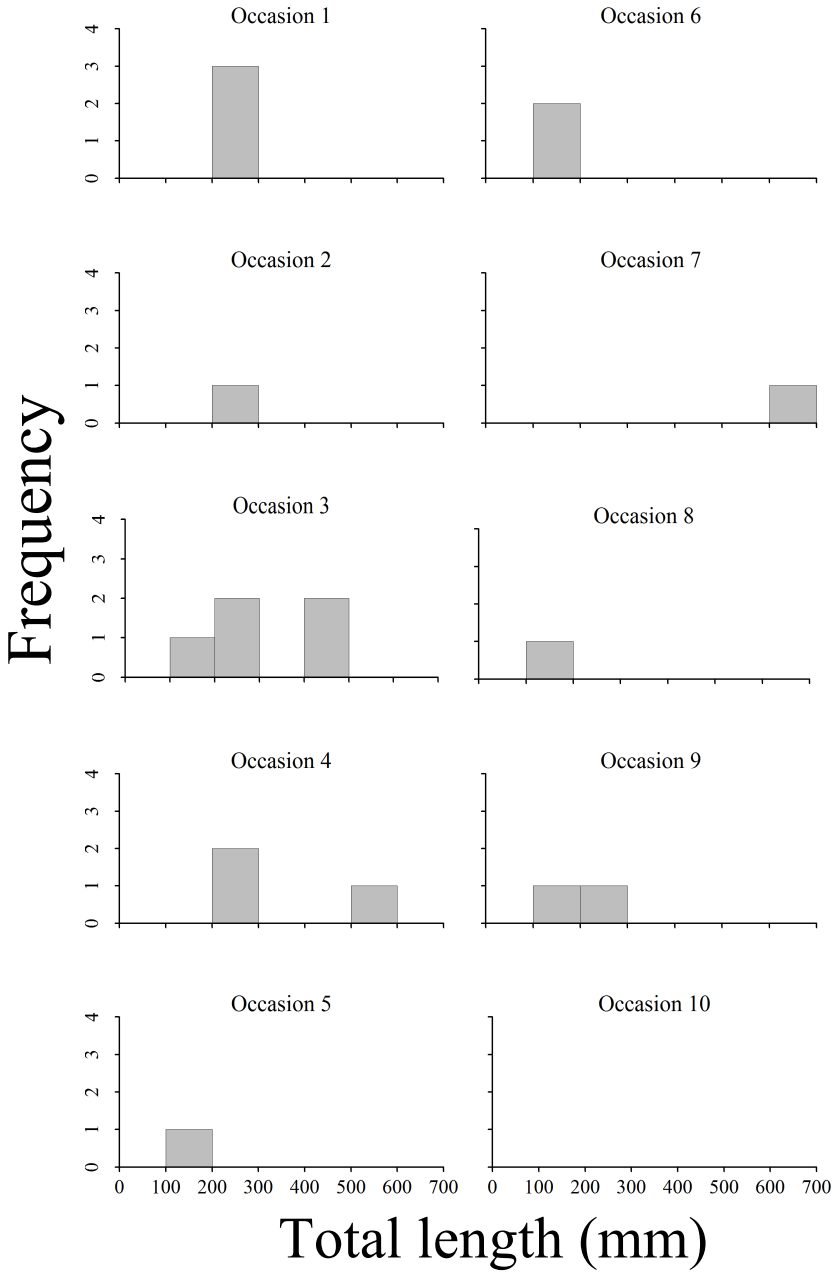
Figure 3. Detection probability at sampling temperature for canoe-based backpack electrofishing for Asian Swamp Eels in Beaver Pond at the Chattahoochee Nature Center, Roswell, Georgia, 2016.

Chapter II Appendices

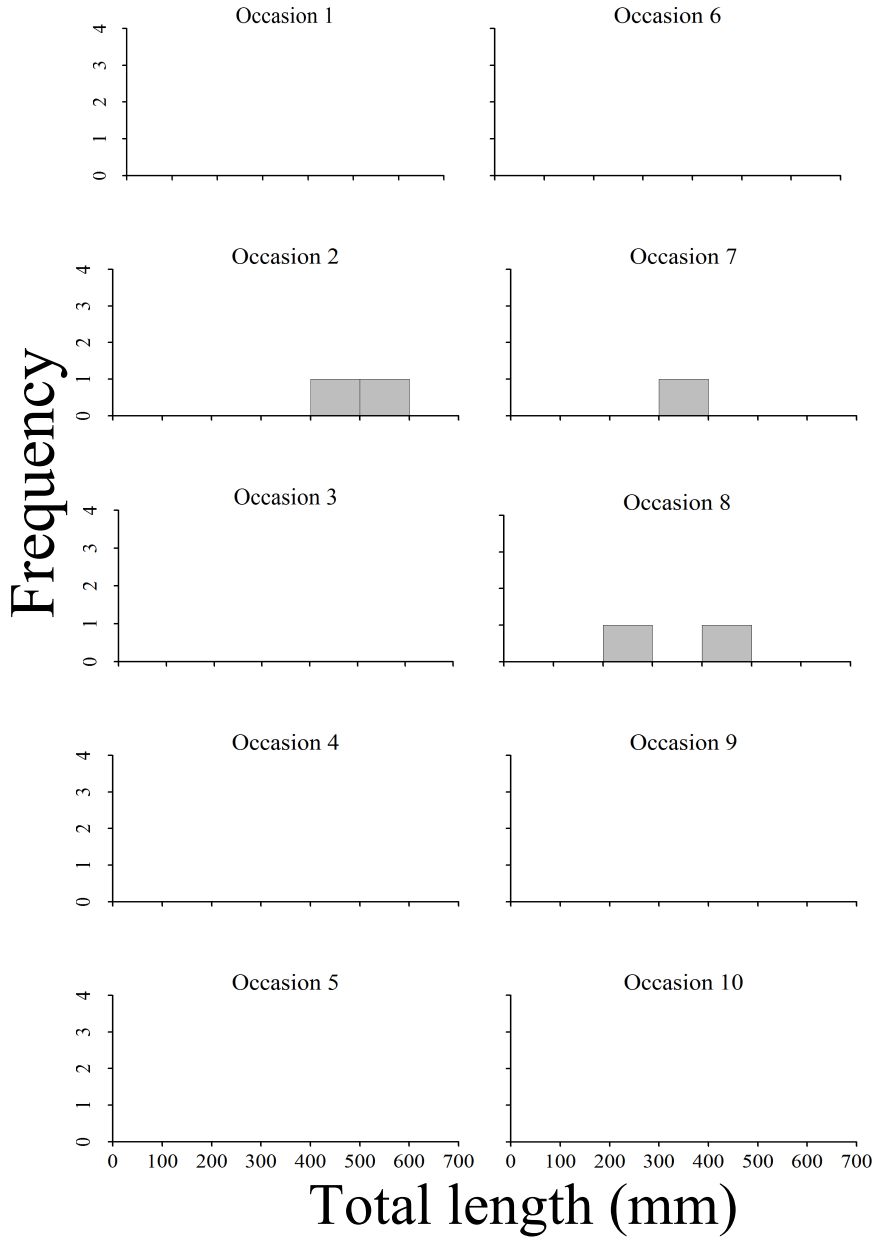
Appendix A1. Length-frequency histogram of adult Asian Swamp Eels captured from Beaver Pond using canoe-based backpack electrofishing within the Chattahoochee Nature Center, Roswell, Georgia, 2016.



Appendix A2. Length-frequency histogram of adult Asian Swamp Eels captured from Kingfisher Pond using canoe-based backpack electrofishing within the Chattahoochee Nature Center, Roswell, Georgia, 2016.



Appendix A3. Length-frequency histogram of adult Asian Swamp Eels captured from the backwater marsh area adjacent to the Chattahoochee Nature Center using canoe-based backpack electrofishing, Roswell, Georgia, 2016.



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