AMPHIBIAN DIVERSITY IN THE GEORGIA SEA ISLANDS:
LAND-USE HISTORY AND LANDSCAPE CONTEXT

by

Nancy Kay O'Hare

(Under the direction of Marguerite Madden)

ABSTRACT

Amphibian declines have been noted worldwide, but recent inventories are lacking for many state and federal conservation lands. How can future changes be addressed if present-day species diversity is unknown? The Georgia Sea Islands is a region of islands isolated by tidal salt marshes and rivers; it includes 35 conservation units, ranging from small parcels to entire islands. Most islands have an extensive land-use history and limited freshwater resources to support amphibian breeding. This study integrated a herpetofaunal inventory from a previously unstudied site (Wormsloe, Isle of Hope) with recent amphibian trapping data for four federal parks in the region and literature records of amphibian diversity for seven other islands (12 islands total). Landscape characteristics, including microtopographic features influencing freshwater resources, were also determined. The islands supported 23 of the 26 amphibians occurring on the adjacent mainland, but individual islands supported between 5 and 19 species. Species occurring across more islands tended to have greater total reproductive output (longer life span, greater number of eggs) and a shorter tadpole/larval stage (<60 days). Larger islands supported more species; however, the relationship between island size and species diversity was more idiosyncratic on smaller islands. Island size, geological age, and habitat (gamma) diversity were correlated, so it was not possible to separate their effects. Dispersal was limited more by the matrix of freshwater rivers and tidal marshes separating islands rather than by distance. Canals at Wormsloe, a land-use legacy feature common across many other islands, allowed tidal influx inland into seasonal freshwater ponds approximately 50 days per year. Consequently, the ability of some species to persist at Wormsloe is uncertain if inland freshwater resources continue to be degraded by tidal influx along canals. Resources management for conservation areas should minimally include annual monitoring of amphibian breeding, establishment of surface water gages, especially on islands more likely to be influenced by adjacent development or water-intensive land-uses. Freshwater quality should be protected by preventing tidal influx along canals and quantity increased by slowing rainwater percolation by lining depressions, after the residual salts in the soil have been leached downward.

Index words: Biogeography, Seasonal Ponds, Herpetology, Lidar

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By

Nancy Kay O'Hare

B.Sc., Northern Arizona University, 1986

M.Sc., Florida International University, 1989

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Nancy Kay O’Hare

Major Professor:	Marguerite Madden
Committee:	C. Ronald Carroll Thomas R. Jordan Lan Mu

Electronic Version Approved:

Suzanne Barbour
Dean of the Graduate School
The University of Georgia
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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 INTRODUCTION AND LITERATURE REVIEW

This study investigates biogeography of amphibians in the Georgia Sea Islands region, relating field data to remotely sensed characteristics of habitat structure, including those indicative of local land-use history.

Amphibian declines have been noted in the USA and worldwide (Wake 1991).Local declines were noted more than 100 years ago but could usually be related to specific activities such as hunting or habitat destruction (Dickerson 1906). Starting in the 1960s, reports of declines became more common, but it wasn't until 1989 at the First World Congress of Herpetology in Canterbury, England that scientists began to recognize that the problem was not isolated, but worldwide (Collins and Crump 2009). Habitat loss and degradation, pesticides, disease, or climate change are frequently cited as causes, but cumulative impact of factors is becoming more important (Blaustein and Kiesecker 2002, Blaustein et al. 2011, Wake 2012). Therefore, declines need to be investigated with regard to local causes.

The small size, secretive nature, and temporal variability in detection of amphibians means that they are easily overlooked. Amphibians occupy low to mid-trophic levels; most juveniles (tadpoles) forage upon diatoms and detritus and most adults prey upon insects and arthropods. Amphibians, in turn are preyed upon by snakes, birds, and mammals. As an ectotherm, amphibians tend to have biomass and energy flow thru to higher trophic levels can be as high as 50%, compared to less than 10% for birds and mammals. Amphibians link wetland and upland ecosystems in ways that many birds and mammals cannot.

In most regions, including coastal Georgia, the seasonal freshwater wetlands that amphibians rely on for breeding are a limiting resource. Human alteration of the landscape and climate variability may independently or cumulatively alter the size, spatial distribution, and biophysical characteristics of the seasonal ponds that amphibians, rely upon for breeding (Brooks 2004, McMenamin, Elizabeth and Wright 2008).

The southeastern coastal plain of the United States of America (USA) has been suggested as an unrecognized hotspot of floral and faunal biodiversity (Noss et al. 2014). Yet basic knowledge is still lacking of present-day species diversity and natural population fluctuations in many conservation areas within the region. Consequently, the impacts of future anthropogenic influences and climate change on species distribution are difficult to determine. How can potential future changes be addressed if present-day species diversity is unknown?

REMOTE SENSING FOR WILDLIFE: PROBLEMS OF SCALE

Most optical satellite and airborne remotely sensed data used for earth observations passively sense electromagnetic energy in the ultraviolet (400 nm) to near infrared wavelength (700 to 1100 nm) These sensors have limitations in directly characterizing habitat structure at spatial scales relevant to smaller animals, such as amphibians. Medium-resolution satellite imagery with 30-m pixel size (e.g. Landsat and ASTER) can be used to assess wildlife habitat type and health over broad areas that are square kilometers in size. The landcover types derived from coarse and medium resolution imagery will include finer scale microhabitats, but the microhabitats themselves are not directly captured. Higher resolution optical imagery (e.g. National Aerial Inventory Program (NAIP), National High Altitude Photography (NHAP), Quickbird) can be used to directly detect some finer scale aspects of habitat, but are limited for use in detecting features beneath the forest canopy. Applications in vegetation assessment using images of high spatial resolution rely upon spectral differences of canopy plants. Habitat characteristics evident on the ground are not necessarily imaged from above. In forested areas, optical sensors have limited ability to penetrate forest canopy, yet forest understory vegetation, ground, cover, soil moisture and small waterbodies in the forest may be critical to separating habitat types.

Airborne light detection and ranging systems (lidar) systems, unlike optical sensors, are active sensors that emit pulses of narrow bandwidths of electromagnetic energy “light” at 1064 nm (infra-red band)[1] that travel through the vegetation canopy and are reflected back to the lidar sensor (Renslow 2012). The timing of the pulse emission and return capture is used to calculate the x,y, and z positions of the object reflecting the pulse. Since some portion of the emitted pulse often travels between leaves and branches of trees, lidar data can assess forest vertical structure more than optical sensors. Moreover, pulses that are reflected back from the ground can be used to generate digital elevation models (DEM), typically with vertical resolutions of 15 to 50 cm. The high resolution lidar derived DEM may present issues in areas with relatively low topographic relief since some microtopographic features may have dimensions that approach the DEM resolution in either the horizontal or vertical dimensions.

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[1] Bathymetric lidar tends to use 532 nm (green band) which, unlike the 1064 nm wavelength, will penetrate water.

The wildlife functions derived from representations of vegetation and physical structure obtained from remotely sensed optical and lidar data will be influenced by the spatial extent of the study area and the minimum mapping unit (MMU) relative to the smallest feature of interest. For example, National Wetlands Inventory (NWI) maps have a MMU of 0.5 to 1.5 ha, depending upon region, but smaller features can be included at discretion. A MMU of 0.5 to 1.5 ha may be adequate when applied over larger spatial extents which, by virtue of scale, would tend to include a range of finer scale features. However, at smaller spatial extents, the range of finer scale features may not be adequately represented. Because of their small size, seasonal freshwater ponds are frequently not directly captured in NWI maps, regardless of whether the ponds are in open vegetation (Martin, Kirkman and Hepinstall-Cymerman 2012)or closed canopy forest (Burne and Lathrop 2008).

LAND-USE HISTORY OF STUDY AREA

The Sea Islands region (Fenneman 1946) of the Georgia coast consists of larger outer barrier islands (>2000 ha) separated from the mainland over 1 to 3 km by a series of smaller back-barrier islands (<1500 ha; Figure 1-1) (Zeigler 1959). The back-barrier islands are isolated from the mainland and from each other by freshwater rivers, tidal rivers, and tidal marshes. The physical geography of the region creates complex landscape linkages that may be a natural analog for linkages within the “habitat islands” created by human influence.

The study area has more than 35 conservation areas currently in private, state or federal conservation protection. Collectively, these conservation areas protect approximately 66,300 ha, of which 33,200 ha (50%) are tidal salt marsh, 23,350 ha (35%) is upland island habitat, 7,150 ha (11%) are tidal brackish rivers, 1,450 ha (2%) of freshwater wetland habitat on islands, and 1150 ha (2%) developed. It includes widely recognized conservation areas such as Cumberland National Seashore and Jekyll Island State Park, as well as smaller, regionally known areas such as Wormsloe State Historic Site, Skidaway Island Observatory, Skidaway Island State Park, Ossabaw Island and Wassaw National Wildlife Refuge.

The region also has a long land-use history: Native Americans prior to European colonization, plantations during the colonial and antebellum periods, a shift to private retreats in the late 1800s, and then to permanent conservation lands, especially during 1960s to 1970s (Vanstory 1970). Native Americans lived in the area prior to European colonization, albeit at lower population levels. Native Americans in general used fire as land management tool, but there are significant regional differences (Frost 1998). Intentional use of fires in the southeastern coastal plain is thought to be minimal due to high natural fire frequency from lightening and the gentle topography and extensive piney flatwoods that facilitated wildfire spread once ignited. During the colonial and antebellum periods, the upland maritime forests[2] and long-leaf pine forests were either logged, had free-roaming livestock, or were completely cleared for agriculture; freshwater wetlands were hydrologically manipulated by construction

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[2] The *Quercus virginiana* (live oak) of the islands were important naval stores. The wood has natural bends, facilitating ship building. During the early history of the USA, defensive positions tended to rely upon ships for coastal patrol or travel along major rivers inlands. Blackbeard Island was acquired by the Department of the Navy in 1800 to supply timber. Other islands had similar commercial timber value, but went thru periods of private ownership prior to being transferred to public ownership.

of canals or drained. While secondary mixed hardwood and pine forests have regrown in upland areas, species composition has changed; the most noticeable is the replacement of long-leaf pine with either slash or loblolly pine. The present impact of canals on freshwater wetland resources is unclear. All conservation areas in the Georgia Sea Islands experienced similar general land use history of logging, agriculture, and ditching, although the intensity on any particular island likely varied.

Drainage canals may cause occasional tidal inundation into historically freshwater ponds. Therefore, the ability of some ponds to support breeding amphibian populations may be compromised since few amphibians tolerate brackish (2 to 18 parts per thousand (ppt) dissolved salts) or saltwater (18 to 30 ppt). Some amphibians are able to persist if the population successfully reproduces once or twice every decade (Taylor, Scott and Gibbons 2006). However, increasing summer temperatures and changing precipitation patterns have been linked to declining amphibian populations and local extirpation (McMenamin et al. 2008). In coastal Georgia, the cumulative impacts of changing climatic patterns, regional development (Savannah GA) and industry (timber mills in Brunswick GA) may cumulatively decrease freshwater availability to natural areas.

GOALS AND ORGANIZATION OF CHAPTERS

Inventory and monitoring of many conservation lands is experiencing revived interest, in part to address concerns of climate variability on species distribution and abundance. Yet addressing such concerns requires recent data on amphibian distribution and natural variation of abundances, as well as linking the physical habitat structure with hydrological processes.

Inventory and monitoring of conservation areas should be assessed in a regional context, including consideration of the relative isolation of conservation areas either by natural or anthropogenic characteristics of the landscape. For amphibians, regardless of geographical area, the detection and hydrological assessment of the seasonal ponds used for breeding is critical. The study area presents challenges for remotely detecting seasonal ponds in that it is semi- deciduous hardwood forest with low topographic relief. Moreover, hydrological function may be influenced by the canals and the changing regional context of development altering groundwater recharge.

The over-arching aim of this study was an investigation of the biogeography of amphibians in the Georgia Sea Islands region and the geographical characteristics likely to influence amphibian distribution. The three broad objectives were an inventory of Wormsloe State Historic Site (Wormsloe) on the Isle of Hope, placing amphibian diversity at Wormsloe within the landscape context of the Georgia Sea Islands, and evaluation of freshwater resources critical to amphibian breeding success specifically at Wormsloe, recognizing that these results would be relevant across the Georgia Sea Islands region. Chapters 2 and 3 focus on the biological aspects of the study (inventory of Wormsloe then a comparison to the region); Chapter 4 focuses on the physical topography necessary to support amphibians, using Wormsloe as a representative study site. Each of these three broad objectives is a chapter, with specific sub-objectives outlined below.

Chapter 2, Herpetofaunal Inventory of Wormsloe State Historic Site: Landscape Context From Past To Present, contributes to regional biogeography with a herpetofaunal inventory from an unstudied island, the Isle of Hope. The goal was to document actual species occurrences, relate species information to past and present landscape context, and contribute to the literature for future testing of hypotheses regarding body size and island biogeography.

Chapter 3, Amphibian Diversity in the Georgia Sea Islands and Implications for Long-term Conservation, places the current Isle of Hope amphibian fauna into a regional context.
We summarized amphibian distribution from trapping and museum collections for 12 coastal Georgia islands held in conservation at either the state or federal level, and correlated distribution with species life history traits and landscape characteristics.

Chapter 4, Lidar Detection of Microtopographic Features in a Forested Landscape With Low Topographic Relief And Extensive Land Use History, describes detection accuracy of microtopographic features related to freshwater resources of import to amphibian breeding within Wormsloe State Historic Site, Isle of Hope. The microtopographic features included natural seasonal ponds along with anthropogenic canals and deep pits dug to access freshwater for liquor distillation (e.g. still wells). Tidal influences on inland seasonal depressions along canals are also investigated.

Chapter 5, Conclusions, integrates the results and resources management recommendations, as well as addresses broader significance.

SIGNIFICANCE OF STUDY

Conservation areas within the Georgia Sea Islands each have some degree of land-use history and most can be affected hydrologically by regional water use. Freshwater wetlands on islands surrounded by tidal marsh represent approximately 2% of the area, yet it is these habitats that are necessary to sustain amphibian populations on individual islands. Freshwater wetlands were frequently manipulated in the past to suit human needs. This including storing water (e.g. stacking water) during wet periods for use during dry periods or draining wetlands for either agriculture or mosquito-transmitted disease prevention (malaria, yellow fever). The canals used to manipulate hydrology still exist. Changes in freshwater resources may limit the ability of conservation lands to support common amphibian species today and in the future.

While all species, including amphibians, adapt to some level of spatio-temporal variability in resource availability and climate, human influences may have disrupted the pathways along which animals would move to adapt to variability. While amphibians may be limited in dispersal, a range of wetlands habitats within their dispersal ability facilitates adaptation to spatio-temporal variability (Compton et al. 2007).

The legal status as “conservation” land implies that these lands will conserve native flora and fauna in perpetuity. There have been a few recent surveys of amphibians in the Georgia Sea Islands in the past 15 years, yet these studies have been limited to federal lands (Dodd and Barichivich 2007, Byrne et al. 2010a, Byrne et al. 2010b, Byrne et al. 2011, Tuberville et al. 2005) or do not consider landscape ecology aspects of the spatial distribution of species or of the spatial distribution of seasonal ponds (Laerm et al. 2000, Shoop and Ruckdeshcel 2003). A landscape level perspective will inform individual conservation units by identifying what might be expected for the size of their lands, and identifying resources that might be limiting. Kushlan (1987) warned that external threats to public lands need to be considered, even when seemingly large areas have been protected. This warning is relevant to the Georgia sea island region since protected areas are downstream of regional land uses (development, timber mills) that affect the limited freshwater resources on island surrounded by tidal marshes.

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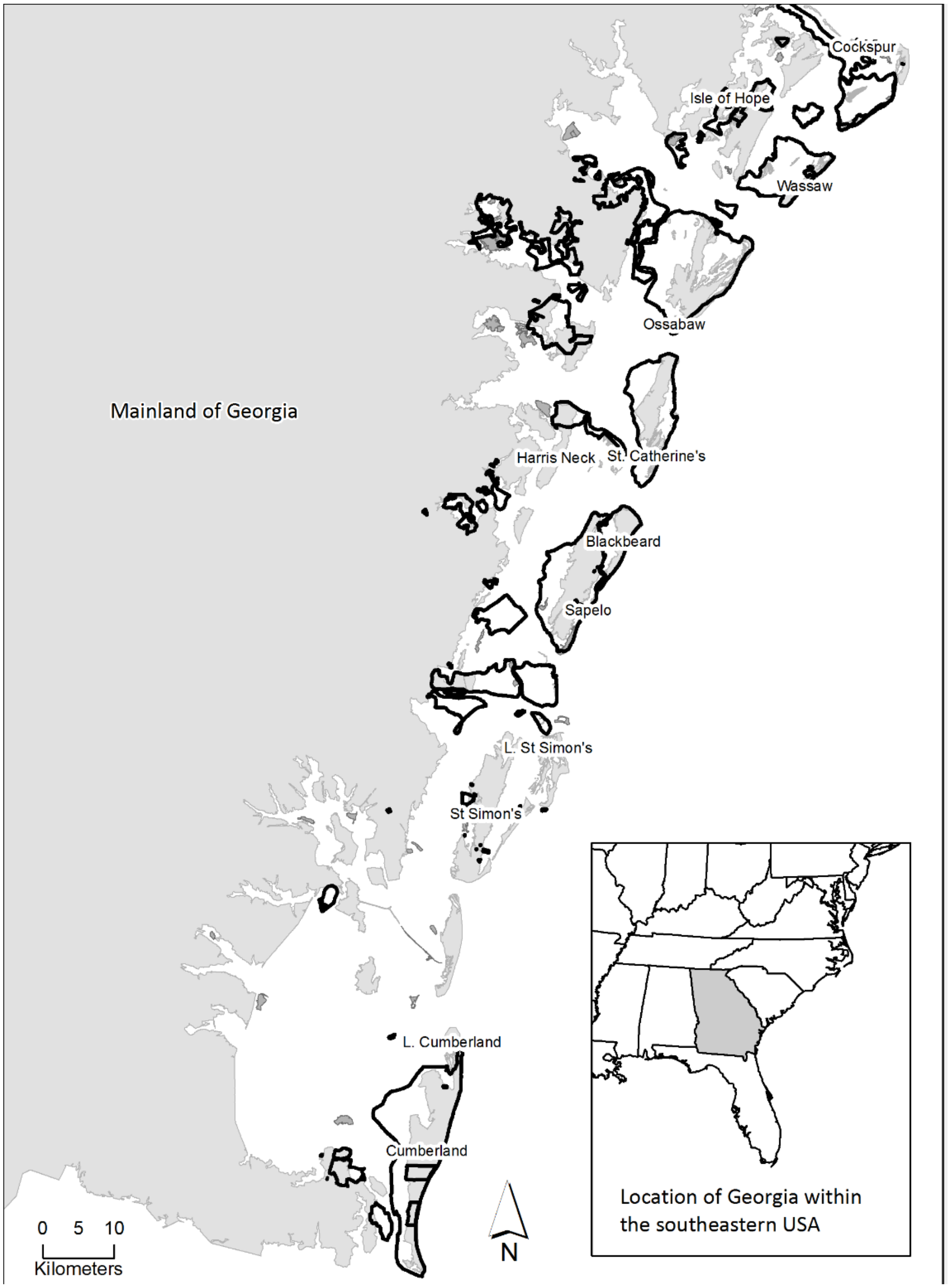

Figure 1-1. Georgia Sea Islands region, with conservation areas included in this study identified.

Wormsloe, a major focal site, is located on the Isle of Hope.

CHAPTER 2 HERPETOFAUNAL INVENTORY OF WORMSLOE STATE HISTORIC SITE: LANDSCAPE CONTEXT FROM PAST TO PRESENT[3]

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[3] O'Hare, N.K., M.M. Madden, and C.R. Carroll. To be submitted to *Southeastern Naturalist*

ABSTRACT

Inventory and monitoring of many conservation lands is experiencing revived interest, in part to address concerns of the effect of climate variability on animal distribution and abundance. We inventoried a previously understudied conservation area, Wormsloe State Historic Site located on the Isle of Hope, an inner barrier island near Savannah, GA, USA. Twenty-one herptile species were directly observed or trapped (8 anurans, 1 salamander, 3 turtles, 5 lizards, and 4 snakes) during the three years of the field trapping (2011, 2012, 2013). Chao estimates to extrapolate trap data predicted that <1 additional amphibian species and 3 to 4 additional reptile species might occur with further trapping, suggesting that the inventory detected the majority of species. Most species had low abundance, with fewer than 10 individuals for 15 of the 21 species. Species richness on Wormsloe (9 amphibians and 12 reptiles) was about half of that occurring on the adjacent mainland (26 amphibians and 50 reptiles). Some species may be absent because of lack of suitable habitat (e.g. long hydroperiod wetlands, running streams). Others may be absent because of land-use legacies from farming (e.g. upland salamanders) or the changing landscape context. While all sites are wooded today, 4 of the 5 trap sites were fields in 1908 and 1 site remained open field until at least 1937. More importantly, today each of the depressions has a drainage ditch connecting it to salt water tides. Species richness is only one indicator of ecosystem function. Body size of individuals is another metric which may indicate population structure as well as susceptibility to extirpation. Body size of adults of the common Southern toad (*Anaxyrus terrestris*) was less than on the mainland (48.7 cm vs 61.2 cm, respectively), and had limited variance. This may indicate either limited resources or that most individuals were of similar age, due to limited annual reproduction. Two species of toads were observed amplectant on several occasions, but neither egg masses nor tadpoles of any species were noted in natural depressions in 2011, 2012 or 2013. While 21 herptile species were detected, their ability to persist at Wormsloe in the face of future changes is uncertain. To support breeding by the remaining, common amphibians, depressions need to have at least 60 days of continuous surface freshwater during spring and summer at least once every few years. Resources management can improve amphibian breeding habitat simply by eliminating tidal influxes along drainage ditches. Maintaining existing populations of common species should be a priority. Wormsloe has similar land-use legacies as many of the other 42 conservation units in the Georgia Sea Islands, and resources management may be regionally applicable.

INTRODUCTION

Inventory and monitoring of many conservation lands is experiencing revived interest, in part to address concerns of climate variability on species distribution and abundance. Yet addressing such concerns requires recent data on distribution and natural variation of abundances.

The Georgia Sea Islands region has ~76,000 ha of land and tidal marshes protected in 42 discrete units either privately (6), at the county (1), state (24) or federal level (11) (Figure 2-1). Much of the knowledge of herpetofauna in the region comes from museum specimens (Williamson and Moulis 1979, Williamson and Moulis 1994) and incidental observations summarized as species lists, with many of the observations occurring more than 25 years ago.

Moreover, knowledge of species distribution is unequal, with more information for the larger, outer islands, and less information for the inner, smaller islands. Overall, recent herpetofaunal distributions and abundances in many of the conservation areas in the Georgia Sea Islands are incomplete or outdated.

Two previous studies integrated available information on herpetofaunal distribution to investigate biogeography of the Georgia Sea Islands. Both Laerm et al. (2000) and Shoop and Ruckdeshcel (2003) perspectives' on amphibian diversity overlooked significant physical geographical factors, land-use history, and species specific life-history traits likely to influence amphibian diversity on islands in the region.

Recent initiatives of amphibian inventory or monitoring on federal lands have added data for four conservation areas (Harris Neck (Dodd and Barichivich 2007), Fort Frederica on St. Simon's (Byrne et al. 2010b), Fort Pulaski on Cockspur (Byrne et al. 2011), and Cumberland Island National Seashore (Byrne et al. 2010a)). Similar efforts for inventory and monitoring of herpetofauna in many state or private conservation lands, however, are mostly lacking and much remains unknown about current species distributions. Consequently, efforts to predict effects of climate change on herpetofauna in the region is stymied by lack of information.

Recent information on species distribution is imperative because a number of conservation areas in the region may have experienced indirect hydrological impacts in the past 25 years. Impacts include an increase in impervious surface area along the Georgia coast and subsequent altered ground water recharge (O'Driscoll et al. 2010, Stringfield 1964), and increased industrial water usage, e.g. paper mills in Brunswick, Georgia affecting Sapelo Island.

Aside from the regional land-use and resources management issues affecting the species distribution, the area has the potential to test biogeographical hypotheses regarding body size on islands. Body size may be important in predicting species vulnerable to extirpation or invasive species likely to successfully established in changing landscapes (Allen, Forys and Holling 2010).

Chapter 2 contributes a herpetofaunal inventory from an unstudied island, the Isle of Hope, related to historical land-use change also contributes to further understanding of amphibian responses to human impacts. Our goal was to document actual species occurrences, relate species information to past and present landscape context, and contribute to the literature for future testing of hypotheses regarding body size and island biogeography.

Chapter 3, Amphibian Diversity in the Georgia Sea Islands and Implications for Long-term Conservation, places the current Isle of Hope amphibian fauna into a regional context.

METHODS

Study Area

Wormsloe State Historic Site (Wormsloe) was established 1979. It is located on the Isle of Hope, ~8 km southeast of Savannah, Georgia, and is presently managed by the Georgia Department of Natural Resources (Figure 2-1). Continuously owned by the same Jones/DeRenne/Barrow family since an original Land Grant from King George in 1738 (Kelso 1979), land conservation of the Isle of Hope under the aegis of the Barrow family predates establishment of the state historic site and extends back more than 100 years (Bragg 1999).

Wormsloe encompasses 806 ha, of which 371 ha are uplands split among three islands. The largest upland portion (306 ha) is located on the southern end of the Isle of Hope (629 ha), with the northern end of the island being residential, primarily single-family homes at density of 2 to 5 units per acre. Neither Long (49 ha upland) nor Pigeon Islands (16 ha upland) have any extant development, paved roads, or public access. The remaining 425 ha of Wormsloe are the tidal marshes and creeks separating the Isle of Hope from the mainland, as well as the islands themselves. Ground elevation on the islands range between sea level and 4.5 m (NAVD 1988). All freshwater sources are seasonal, and dependent upon rainfall; the sitehas no freshwater streams or dredged ponds (Georgia 2010).

Field Methods for Species Inventory

Due to limited access to the other islands of Wormsloe, trapping occurred on the 306 ha of Wormsloe on the Isle of Hope. We selected five sites in wooded areas with each of the sites being within 30 m of a depression having field indicators of seasonal freshwater such as freshwater wetlands plants, water marks, or saturated soils (Figure 2-2).

Drift fence arrays set up at each of the 5 trapping sites had three arms, each 10 m long, with a single pitfall trap at the center (Enge 2001). Funnel traps were made of 3 mm (1/8”) hardware cloth and ~1 m long by 0.35 m in diameter. Traps were place midway along each arm on either side, and at the end of each arm (nine traps total per array). Traps placed midway along the arm had funnels at both ends of the trap; traps at the end of each arm had two funnels at one end of the trap, placed so that a funnel opened on either side of the drift fence (Figure 2-3). The pitfall trap was 18.9 liter (5 gallon) bucket buried so that the rim of the bucket was at ground level. Drift fence arrays were operated between May 2011 – August 2011 (29 check days), early March 2012 (3 check days), May 2012 – June 2012 (7 check days), mid March 2013 (4 check days) and late July/early August 2013 (3 check days). Time between check days varied from 1 to 3 days; UGA permit protocols required 1 day check period during 2013 as part of standard permit requirements. Traps were not operated during cooler winter periods when amphibians are less active and only for short periods during the summer when high day time temperature (>30° C) increased mortality. Providing shallow water or moist sponges decreased direct mortality, but also resulted in other animals vandalizing traps to get at the fresh water.

Individuals were measured (snout-vent length[4] and total length) to the nearest 0.1 mm using digital calipers, weighed to the nearest gram, gender determined if possible, and released near the capture site.

PVC pipe refugia sampled arboreal treefrogs (Boughton, Staiger and Franz 2000) near the same five depressions sampled by drift fence arrays. PVC pipes were cut to ~75 cm length and fitted with a single end cap. At each site, we deployed 25 1” diameter pipe and 25 1 ½” diameter PVC, with the bottom of the pipe being ~2 m above ground surface. PVC refugia were checked quarterly (March, June, September, December) from March 2011 to December 2014.

Summaries of opportunistic encounters of *Terrapene carolina carolina* (Eastern box turtle), occasionally aided by a specially trained hunting dog, are also included. Surveys were

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[4] Snout-vent length is standard measurement for anuran amphibians. It is the distance between the tip of the nose and anus (vent). Since anurans lack tails, snout-vent length is for all intents and purposes the total length.

random and irregular, depending upon the dog's availability and weather. As noted by others, turtles disappeared within a few minutes into the deep leaf cover once released. Visual encounter surveys and drift fence trapping detected 3 turtles, and the trained hunting dog detected the other 19, including 10 in approximately 3 hours, suggesting that alternate methods may be needed in areas with deep leaf litter.

Comparison data on body size of *Anaxyrus terrestris* (Southern toad) was obtained from the University of Georgia Savannah River Ecology Lab (http://srel.uga.edu/), located 200 km to the northwest on mainland Georgia.

Historical and Recent Land-Use Maps and Digital Elevation Models

Historical maps of land use and land cover of the Isle of Hope from 1897, 1908 and 1927 were available from the DeRenne Family Collection housed at the University of Georgia Hargrett Library. Digital images of the historical maps were geo-rectified in ArcGIS 10.2, with estimated horizontal accuracy of ~5 to 10 m. Land cover from these maps were digitized for use in subsequent analyses.

Georgia Land Use Trends (GLUT) land cover maps from 1978 and 2008 were downloaded from Georgia GIS Clearinghouse (http://data.geospatial.org). Both of these land cover maps were derived from Landsat satellite image data and National Land Cover Data sets and used a classification scheme similar to the U.S. Geological Survey (USGS) Anderson Level II (Anderson 1976). The 1978 map had a pixel size of 60 m, whereas the 2008 had 30 m pixel size. The percentage of area classified as developed in a 1 km, 2.5 km, and 5 km buffer around Wormsloe was summarized.

A digital elevation model (DEM) of the region was obtained from Chatham County, Georgia as part of the Coastal Georgia Elevation Project. The DEM was derived from lidar data collected in 2010 at nominal ground spacing of 1 point per meter, with up to four returns per pulse. The resultant DEM had 95% confidence interval for vertical accuracy not to exceed 36 cm (2 ft) and horizontal accuracy not to exceed 72 cm (4 ft). Depressions were identified on the DEM and field indicators of hydrology used to select trapping sites. Drainage ditches were also evident on the DEM; a line file of the drainage was created using heads-up digitizing in ArcGIS 10.2, and verified in the field with notes on ditch width, depth, and steepness of sides (see Chapter 4 Lidar Detection of Microtopographic Features in a Forested Landscape with Low Relief and Extensive Land-Use History).

Rainfall

Daily rainfall data was obtained from station at Savannah Interational Airport, approximately 20 km north northwest from the study area. This station has the longest period of record in the region, since 1948.

Software and Statistical Analyses

ArcGIS 10. 2 was used for all mapping and geospatial figures. Statistica 12 (StatSoft 2014) was used for statistical analyses, and for creating histograms, scatterplots, and box and whiskers plots. The box and whiskers plots of mean (point), standard error (box), standard error multipled by 1.96 (whiskers) can be visually interpreted for statistical significance; if the whiskers of the plots do not overlap, then the means were significantly different. Non-parametric analyses of variance (ANOVA) and Kruskall-Wallis tests were used to determine significant differences between means. Spearman R was used to test for correlations between variables. Estimation of total species richness from the sample data were made using Chao's non-parametric estimates based upon low numbers in most classes (Chao 1984) using the freeware EstimateS (Colwell and Elsensohn 2014). Extrapolation randomized capture data for 50 iterations to minimize the effect of order of captures, and used Chao original formula.

RESULTS

Trapping at Wormsloe

A cumulative total of 1,135 individuals of 19 species were trapped in drift fences (Table 2-1). Fourteen of the 19 species were trapped on 10 or fewer of the total number of check days; the number of individuals for each of these species was also less than ten, indicating no pulse of high number of individuals on a few trap dates for the majority of species. Two amphibian species accounted for 86% (n=997) of all individuals trapped, with nearly half of these (443) captured on a single night. Species accumulation curves and Chao estimate of species richness were calculated for all captures and excluding the single night of high abundances (Figure 2-4).

Table 2-1. Summary of captures in five drift fence traps arrays, PVC pipe refugia, and incidental observations.

Scientific Name	Common Name	Sites	Dates	Individuals
Drift Fencing Trapping				
Acris gryllus	Southern cricket frog	3	2	4
Anaxyrus terrestris	Southern toad	5	39	613
Gastrophryne carolinensis	Eastern narrowmouth toad	4	13	45
Hyla cinerea	Green treefrog	4	3	4
Hyla squirella	Squirrel treefrog	2	4	6
Rana sphenocephala	Southern leopard frog	1	1	1
Scaphiopus holbrookii	Eastern spadefoot	5	14	364
Plethodon grobmani	Southeastern slimy salamander	3	3	7
Kinosternon subrubrum	Eastern mud turtle	2	2	2
Terrapene carolina carolina	Eastern box turtle	1	1	1
Anolis carolinensis	Green anole	3	8	9
Eumeces fasciatus	Common five-lined skink	1	1	1
Eumeces inexpectatus	Southeastern five-lined skink	1	1	1
Eumeces laticeps	Broadheaded skink	5	17	33
Scincella lateralis	Little brown skink	5	12	26
Agkistrodon contortrix	Copperhead	2	5	5
Cemophora coccinea	Scarlet snake	1	1	1
Coluber constrictor	Black racer	4	8	10
Thamnophis sirtalis	Eastern garter snake	1	2	2
Amphibians	8 species	5	41	1044
Reptiles	11 species	5	11	91
PVC Pipe Refugia				
Hyla cinerea	Green treefrog		9	12
Hyla femoralis	Pinewoods treefrog		11	36
Hyla squirella	Squirrel treefrog		12	77
Totals	3 species		12	125
Incidental Observations				
Chelydra serpentina	Common snapping turtle		1	1
Cumulative Totals All Methods		Number of Species		
Amphibians		9		
Reptiles		12		
All Herptiles		21		

Maximum species richness was extrapolated to 2,500 individuals separately for all capture dates and then excluding the single night in which 40% of all individuals were captured to determine how this activity pulse affected calculations. Maximum predicted species richness at 2,500 individuals was 22.2 for all dates, compared to 23.0 if the single night was excluded. Separate curves for amphibians (Figure 2-4 middle) and reptiles (Figures 2-4 bottom) suggested that any new species trapped would more likely be reptiles (~4 species if 500 individuals trapped) rather than amphibians (add less than 1 species if 2,500 individuals trapped).

Species richness and captures were unequal between sites (Table 2-2). Four of the 19 species occurred at all trap sites; 6 species, all reptiles, occurred at only one site. As noted above, there was a pulse of 443 individuals on March 4, 2012. The pulse of individuals was restricted to two species, *A. terrestris* (total n=191) and *Scaphiopus holbrooki* (total n=215), with each dominant at a different site. *A. terrestris* was more abundant at Birdy (n=128) while *S. holbrooki* was more common at BentTree (n=201). Each of these species was captured at the

Table 2-2. Captures in drift fence traps by site. Captures of *Anaxyrus terrestris* and *Scaphiopus holbrooki* on the night of March 4, 2012 given in parentheses.

	Amphibians			Reptiles		
Site	Dates	Species	Individuals	Dates	Species	Individuals
BentTree	27	5	329 (215)	16	8	23
Birdy	26	5	425 (191)	10	5	13
Palmglade	20	7	64 (5)	12	4	16
VP3	25	3	147 (35)	12	6	20
Yanxi	16	7	79 (0)	13	3	19
Total	41	8	1044 (443)	32	11	91

sites on other nights, and were among the four species captured at all sites. The following night was also very rainy, but only 15 individuals were captured.

PVC pipe refugia were checked 12 times from July 2011 – December 2013. A total of 125 individuals of 3 species were recorded (Table 2-3). Each of the three species was found during the first check, and Chao's estimates of species richness extrapolated to 500 individuals did not predict additional species (Figure 2-5). Each of the three species had a high frequency of occurrence (75% to 100% of check dates), but abundances of the species was more variable than frequency of occurrence.

Table 2-3. Treefrogs in PVC pipe refugia.

Scientific Name	Indivdiuals	Number of Dates
Hyla cinerea	12	9
Hyla femoralis	36	11
Hyla squirella	77	12
Totals	125	12

Both sampling methods and incidental observations detected 21 species of herptiles at Wormsloe: 8 toads/frogs, 1 salamander, 3 turtles, 5 lizards, and 4 snakes. Only two species, the treefrog *Hyla femoralis* and the turtle *Chelydra serpentina*, were not sampled by drift fence arrays. The treefrog *H. femoralis* was the most common species detected in PVC pipe refugia. Incidental observations added *C. serpentina*. Amphibian call surveys did not detect any additional species.

Anaxyrus terrestris and *Scaphiopus holbrookii* were the only amphibians observed amplectant (attempting to breed). On two separate dates, multiple amplectant pairs were observed in the water, hopping in and out of the water. Egg masses were not laid during more than one hour observation periods, nor were egg masses found the following day. Salinity of water was tested at this point and was 5 ppt, prompting more regular testing of water salinity as well as soil salinity areas were dry (see Chapter 4). More importantly, no egg masses of any species were found in natural depressions at Wormsloe in 2011, 2012 or 2013, suggesting that water salinity from tidal intrusion along drainage ditches was limiting egg laying (verified by salinity measurements; see Chapter 4). *Anaxyrus terrestris* did breed annually in an artificial water garden.

Metrics of body size of species with more than five individuals captured were summarized (Table 2-4). The gender of the most common species, *A. terrestris*, could also be reliable determined. Average size of *A. terrestris* was 47.2 ± 0.6 cm s.e., with no significant

Table 2-4. Metrics of snout-vent length for species with more than five captures. Data for *Anaxyrus terrestris* from SREL provided by David Scott, SREL.

			Total length (mm)			
Species		N	Mean	Min	Max	Std.Dev.
Wormsloe (Island)						
Anaxyurus terrestris	All	273	47.0	23	80	7.7
	Males	168	47.0	27	74	5.9
	Females	97	47.2	23	80	10.2
Gastrophryne carolinensis		22	31.9	20	65	12.2
Scaphiophus holbrolli		299	50.6	24	69	6.1
Hyla squirrela		21	33.5	25	42	5.1
Hyla femoralis		8	35.5	27	46	5.6
Hyla cinerea		9	52.0	41	59	5.3
SREL (Mainland)						
Anaxyurus terrestris	All	154	61.2	46	84	7.6
	Males	57	54.9	46	61	2.9
	Females	97	65.6	50	84	6.7

differences in size between females (47.2 ± 0.5 cm) and males (46.9 ± 0.3 cm; F (1, 272) = 0.130, p = 0.72). On the mainland, average size was 61.6 ± 0.6 cm, and females (65.6 ± 0.6 cm) were larger than males (54.9 ± 0.7 cm; F (1, 152) = 131.8, p<0.001). For comparison between island and mainland, body size of genders were lumped together to test difference in average size of species. Average size of *A. terrestris* at Wormsloe was significantly smaller than on the mainland (F =1, 516 = 516.5, p <0.001; Figure 2-7). Body size was normally distributed for both sites, yet kurtosis in body size was more on the mainland, indicating greater variance in body size, compared to the insular population (-0.13 ± 0.4 on mainland vs 1.2 ± 0.2 s.e. on island; Figure 2-8).

Only one *Terrapene carolina carolina* (Eastern box turtle) was captured during drift fence trapping. Other opportunistic searches turtles found 21 turtles; 19 of these were found

Table 2-5. Body size metrics for *Terrapene carolina carolina* at Wormsloe.

Gender	N	Mean (cm)	Min	Max	Std.Dev.
Males					
Length	15	13.3	12.2	14.7	0.72
Width	8	10.3	9.0	12.3	1.13
Weight	15	462.0	330.0	630.0	62.84
Females					
Length	7	12.6	11.0	13.8	1.02
Width	3	9.0	8.3	10.0	0.91
Weight	7	464.3	370.0	570.0	81.11
All turtles					
Length	22	13.1	11.0	14.7	0.87
Width	11	9.9	8.3	12.3	1.20
Weight	22	462.7	330.0	630.0	67.18

using a trained hunting dog. Males and females were of similar length (Table 2-5; ANOVA F (3, 7) = 2.611, p = 0.133) and weight (ANOVA F (2, 18) = 2.734, p = 0.092). There was wide variation in the color of both carpace (Figure 2-8) and plastron. Plastron color ranged from light to pure black. The smallest turtle captured was a male measuring 12.2 cm (at 11 o'clock in Figure 2-8). A second male of similar size and color was found within 20 m, but escaped[5]. Males were more common than females (15 versus 7 individuals). All except one turtle had more than 20 annuli on scutes, indicating age exceeding 20 years old (Budischak et al. 2006).

Landscape Context

The general context surrounding each depression was summarized from available historical maps (Table 2-6). Although all sites are wooded today, four of the five sites were fields in 1908 (Figure 2-9) and one site remained open field until at least 1937. In 1908, distance from depression to wooded area ranged from 0 to 425 m. Distance from a depression to nearest neighbor ranged between 75 and 275 m. More importantly, today each of the depressions has a drainage ditch connecting it to tide-influenced waters. The presence of ditches was only included on the 1897 map, which only showed the northern portion of the

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[5] A turtle "escapes" when Merlot is hot and bored waiting for me to process the first turtle he brought me. So, he wades into the adjacent creek and frolics, still holding the second turtle in his mouth. I asked Merlot to come, which he did, but without the turtle. While I spotted the place in the water where he dropped the turtle, the frolicking muddied the water to the extent that visibility was near zero. Even with shuffling my feet, I could not locate the turtle. Merlot was happy and cooler.

island, but in 1897, the three depressions included on the map each had a ditch connecting it to tidal marsh.

Table 2-6. Historical landscape of areas surrounding drift fence trap sites, along with first evidence of drainage ditch connecting the depression to tidal marsh, and current impact of tidal influx. Habitat types were standardized across time; HW = hardwood and Mixed = mix of hardwood and pine.

Site	1897	1908	1927	1937	1972	1988	2014	Ditch Evidence		Current Tidal
BentTree	HW	HW	HW	HW	Mix	Mix	HW	1897	Wide	Monthly
Birdy	Field?	Field	HW	HW	Mix	Mix	HW	?	Narrow	Monthly
Palmglade	Field	Field	HW	HW	Mix	Mix	HW	1897	Wide	Biweekly
VP3	Field?	Field	HW	HW	Mix	Mix	HW	?	Narrow	No
Yanxi	Field	Field	Field	Field	Mix	Mix	Mixed	1897	Wide	Monthly

The landscape context of Wormsloe has also changed, although we are only able to quantify changes from satellite imagery starting in 1974 (Table 2-7). The percentage of developed land has increased in the 1 km, 2.5 km, and 5 km buffers around the site. In the early

Table 2-7. Maximum distance (m) to forest in recent past and to other seasonal ponds.

Site	Land Use History	Maximum Distance (m) Forest	Other Pond
BentTree	Forest since at least 1897	0	80
Birdy	Forest at least since 1908	20	135
Palmglade	Forest regrowth after 1908, before 1927	50	80
VP3	Forest regrowth after 1908, before 1927	425	275
Yanxi	Forest regrowth after 1937, before 1972	110	75

1970s, construction of the Diamond Causeway, directly linking Skidaway Island to the mainland, altered tidal flux between the southern end of the Isle of Hope and Pigeon Island[6]. The resultant sedimentation added land between the Isle of Hope and Long Island to the east; this land is presently dominated by pine. It also increased tidal marsh in the area, at the expense of open tidal river. Percentage of area in residential and commercial development within a 5-km buffer surrounding Wormsloe has increased from 10% in 1974 to 23% in 2008 (Table 2-7; Figure 2-10). There has been modest in-filling development on the northern portion of the Isle of Hope. The majority of surrounding development occurred on the mainland to the west and Skidaway Island to the east. Development occurred primarily on forested lands.

Rainfall

Average annual rainfall for the study area is 130 cm. There is seasonal variation, with the highest rainfall occurring during July and August (Figure 2-11). During the trapping years of 2011, 2012, and 2013, annual rainfall levels were lower than average (85, 90, and 110 cm respectively).

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[6] Prior to construction of the Diamond Causeway, road access to Skidaway Island was via road traversing the middle of the Isle of Hope, and partly thru Wormsloe. It is still evident at Bell Point on Wormsloe, and on Long Island.

Table 2-8. Landscape context of Wormsloe State Historic Site in 1974 and 2008.

	Area in hectares		Percentage	
Land Cover	1974	2008	1974	2008
1 Km				
Other	137	58	7%	3%
Developed	142	287	8%	15%
Forest	499	359	26%	19%
Open Water	369	314	20%	17%
Tidal Marsh	740	869	39%	46%
Grand Total	1887	1886	100%	100%
2.5 Km				
Other	514	252	9%	4%
Developed	502	1391	9%	24%
Forest	1976	1207	35%	21%
Open Water	759	668	13%	12%
Tidal Marsh	1950	2189	34%	38%
Grand Total	5702	5706	100%	100%
5 km				
Other	997	356	7%	2%
Developed	1494	3562	10%	23%
Forest	4358	2674	29%	18%
Open Water	2048	2072	14%	14%
Tidal Marsh	6248	6499	41%	43%
Grand Total	15144	15163	100%	100%

DISCUSSION

Twenty-one herpetofaunal species occur at Wormlsoe (9 amphibians and 12 reptiles). In comparison, 74 species occur on the mainland (24 amphibians and 50 reptiles). Given the smaller land area and habitat diversity (e.g. gamma diversity) compared to the mainland, lower species diversity relative to mainland is not unexpected. Extrapolation of trapping data to predict maximum species richness suggested that 3 to 4 additional reptile species and 1 amphibian species would be likely be added to the species list given further trapping

Some species present on the mainland or other islands are likely absent from Wormsloe since it lacks certain micro-habitats (e.g. longer hydroperiods, streams). This would include *Siren*, *Amphiuma*, most *Rana*. Other species may be absent because of land-use legacies from farming or the changing landscape context. For example, because farming cleared old growth forests and disturbed soils, salamanders relying upon upland forested habitats may have been extirpated. While forests regrew once farming ceased, the successional forests provided altered habitat conditions and developing landscape context may have hindered recolonization. Woodland salamanders avoid traversing grasslands (Rittenhouse and Semlitsch 2006), and even small, first order streams effects movement (Marsh et al. 2007). Consequently the salt marsh grasslands and tidal streams present significant dispersal barriers for salamanders.

Unexpectedly, most species had low abundance with 10 or more individuals captured for only 6 of the 21 species. Amphibian individuals were an order of magnitude more abundant than reptiles. Even so, abundances were skewed to a few, generalist species of mesic forests with inclusions of shorter-hydroperiod micro-habitats. Amphibians can have explosive population changes, so it is difficult to determine if skewed abundances reflected the lower rainfall amounts over several consecutive years coinciding with the trapping period or longer term trends. Regardless, lower rainfall is more likely to affect abundances and breeding of amphibians, rather than species presence or absence.

The changing land-use context of Wormsloe affects not only meta-population dynamics, but also seasonal hydrology. Increase in both impervious surface area and anthropogenic water changes ground water re-charge pattern and the freshwater lens in the soil surface that allows rainwater to seasonally pond on the surface. Although there are no hydrologic gages of either freshwater lens or surficial water levels in the study area, anecdotal evidence for decrease in both exist. There were numerous small open pit wells associated with bootlegging still sites located throughout the Isle of Hope that were operational in the 1920s to 1930s era. These wells would have held freshwater, yet surface water was not observed in these sites during this study. Limited sampling at Wormsloe found lack of freshwater lens within 2 m of the surface (unpublished data collected in November 2013). A freshwater lens may exist deeper below the surface; but the soil between the surface and 2 m was saline. Other evidence for decreased hydrology is presence of trees that require longer hydroperiod than observed. One particular site at Wormsloe has several adult *Taxodium distichum* (bald cypress), but no saplings or seedlings. Indeed, the site did not support freshwater hydroperiod longer than ~45 days during 2010 to 2014. The freshwater lens may have been depleted by lower than normal rainfall for four consecutive years; years of higher rainfall may natural recharge the freshwater lens.

Of concern is how the changing landscape context of Wormsloe may affect the ability of the remaining 21 herpetofaunal species to persist. The natural island habitats combined with surrounding development that reduces habitat and likely changes groundwater recharge, is likely to increasingly limit already scare freshwater resources. Body size may predict persistence in changing landscapes (Allen et al. 2010), so that metrics on body size of the herpetofaunal community may be useful to guide resources management.

Average body size of adult *Anaxyrus terrestris* was 20% (13.5 mm) smaller on the Isle of Hope (48.7 mm) than on the mainland (61.2 mm), and less than reported for other islands in the Atlantic Coastal Plain (70 to 100 mm on Ossabaw Islands; John Crawford, University of Georgia Marine Extension, Skidaway Island, personal communication). Biogeographically, the reduced number of predators, including snakes, at Wormsloe would have predicted that body size would increase. However, predator-prey interactions are not the only determinants of population size structure. The low variation in body size combined with the observed limited breeding in multiple years suggests that body size at Wormsloe may be a reflection of pulsed breeding, and limited annual recruitment under sub-optimal hydrologic conditions. Sandy substrates, common at Wormsloe, has also been correlated with decreased adult body size in some areas (Marangoni, Tejedo and Gomez-Mestre 2008)

Biogeographical principles governing body size should be inferred only if a consistent directional shift in body size occurs within a guild of species of similar body size. Directional shifts in body size may not hold true across guilds since selective pressures likely differ as body size changes. Consequently, some species on an island may become larger, while other smaller, depending upon their niche. Although not conclusive, we did have limited comparative information on body size for the turtle, *Terrapene carolina carolina.* Size was not sexually dimorphic at Wormsloe, unlike populations in West Virginia (Weiss 2009) and the Florida Keys (Verdon and Donnelly 2005); this may be an artifact of relatively low sample size at Wormsloe (15 males, 7 females) . Overall, turtles at Wormsloe were larger than reported for *T. carolina carolina* in West Virginia, similar to population on Egmont Key in central Florida (Dodd 1997) but smaller than for *T. carolina bauri* Big on Pine Key in Florida Keys (Verdon and Donnelly 2005) This supports a biogeographical trend for increasing body size of as move closer to the equator (e.g. Bergman's Rule relating to latitude; (Ashton and Feldman 2003)).

CONCLUSIONS

This study met its objectives to inventory herpetofauna of Wormsloe State Historic Site. As such, it contributes to regional biogeographical studies. It also collected data on body size of a common toad; in the future, this data can be compared to body size of this species on other islands to investigate conflicting reports of both insular dwarfism and gigantism of this species.

The ability of some species to persist at Wormsloe in the face of future changes is uncertain if sufficient freshwater resources to support breeding of common toads and frogs requiring less than 60 days occurs only every few years. Recolonization or "rescue" of populations at Wormsloe from adjacent areas is unlikely since majority of adjacent mainland and islands have been fragmented by development, increasing the isolation of Wormsloe. Eliminating tidal influxes inland along anthropogenic canals to historically freshwater ponds may improve amphibian breeding habitat. Sustaining existing populations of common species should be a priority, and also keep options to address future issues related to changing climate.

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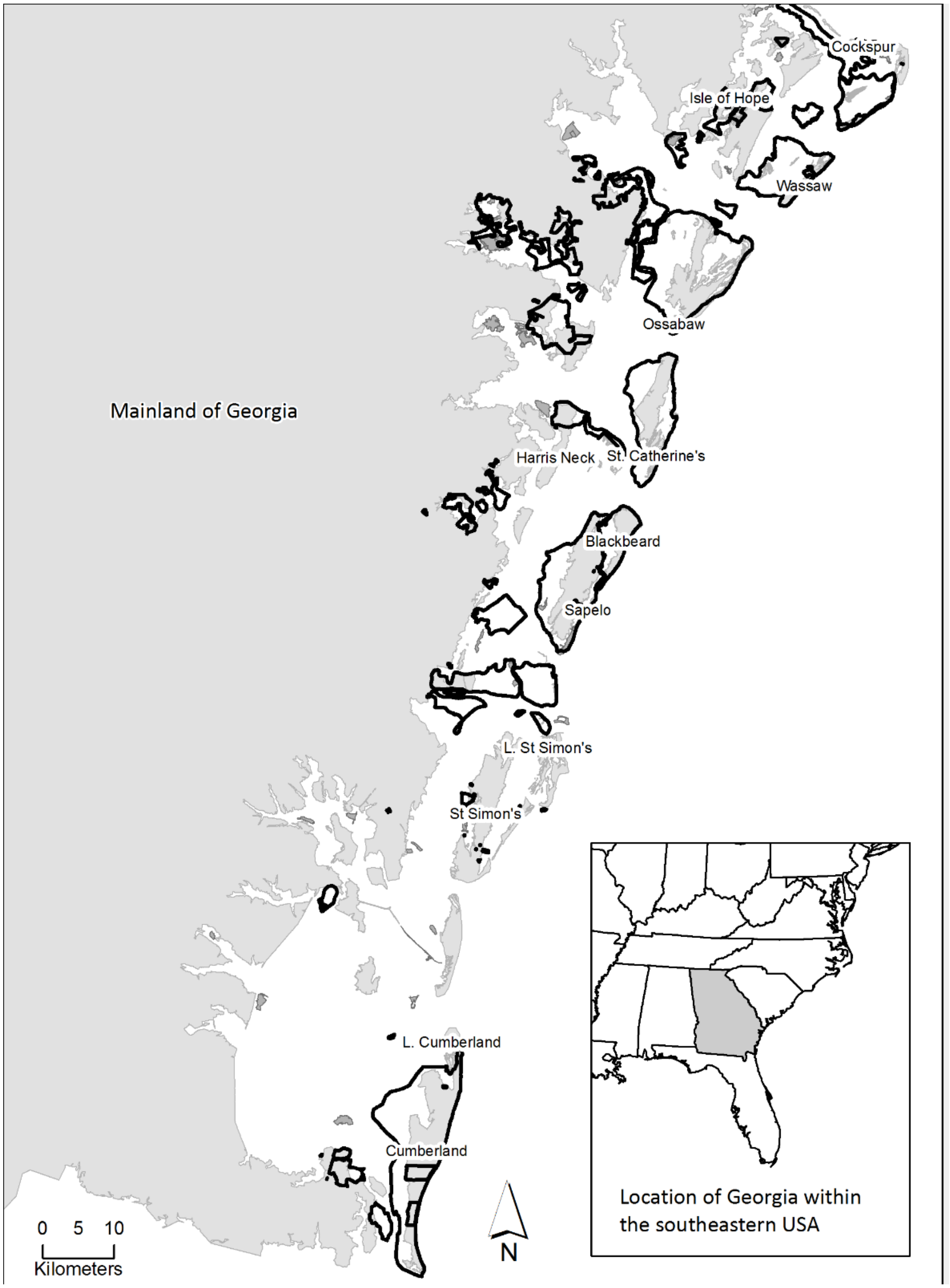


Figure 2-1. Location of the Georgia Sea Islands Region, with major islands identified. Conservation areas are also identified.

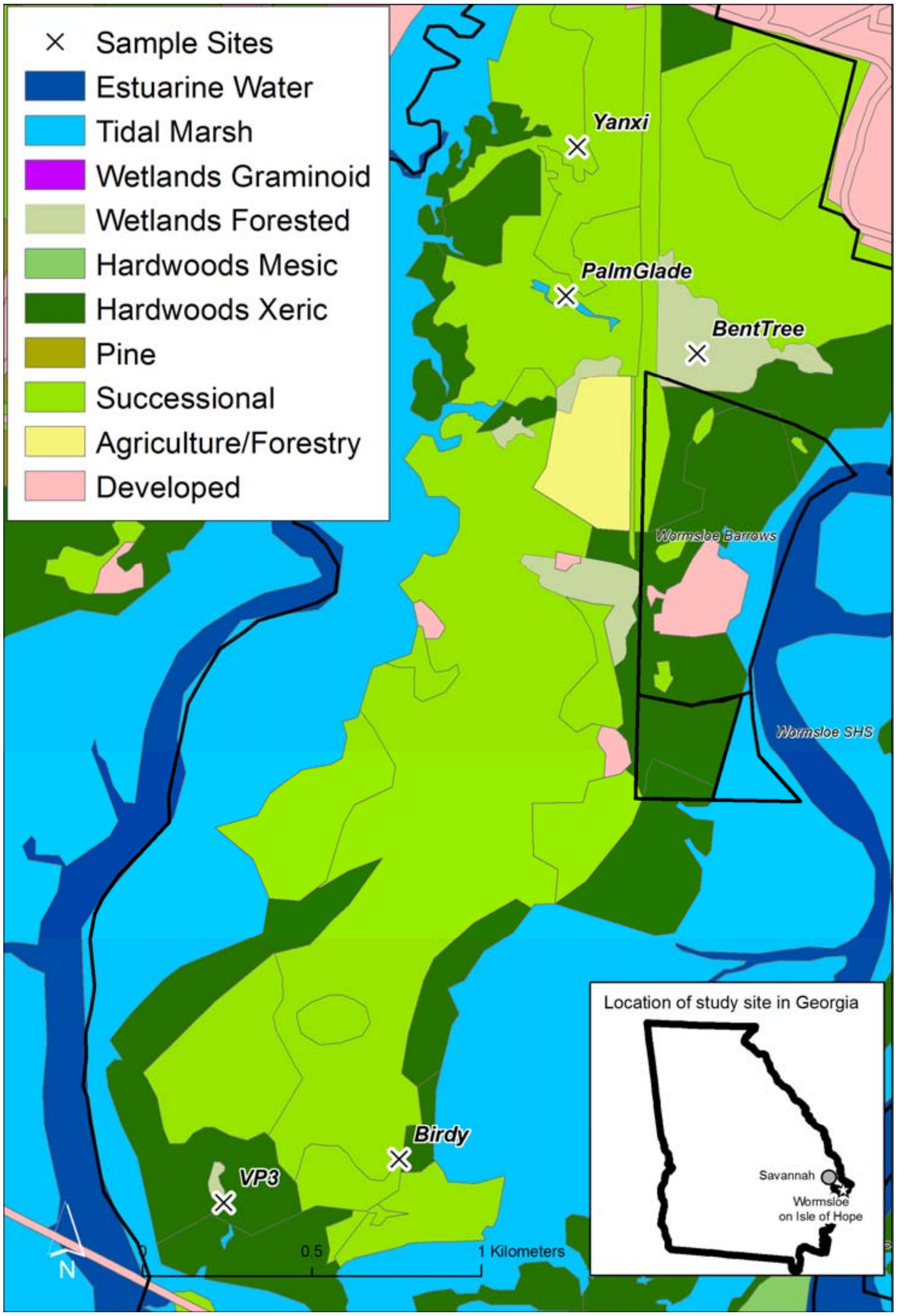


Figure 2-2. Location of Wormsloe State Historic Site, near Savannah, Georgia, and of trap sites within Wormsloe. General land cover types, from the Georgia DNR vegetation 2011 vegetation map, included.

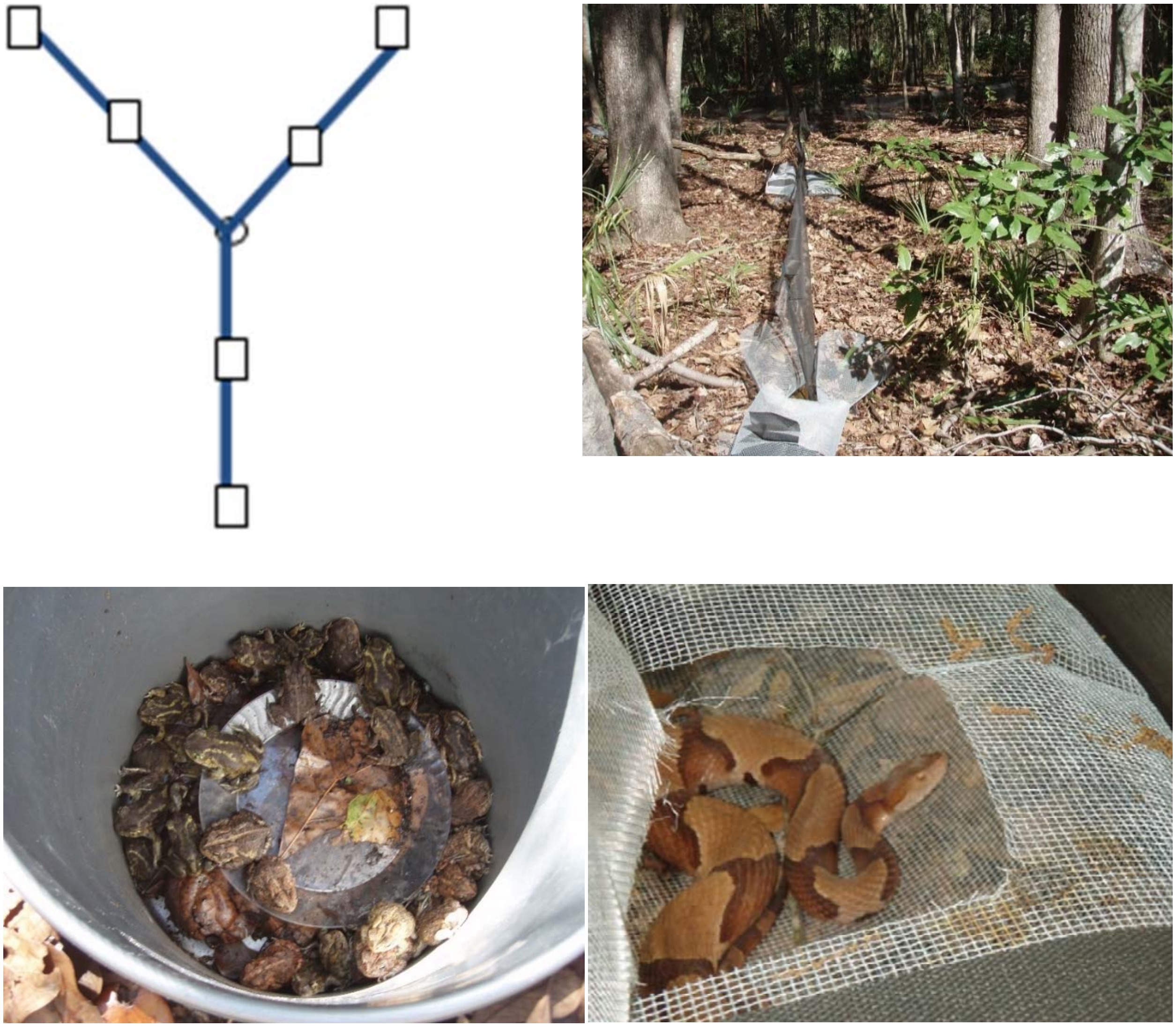


Figure 2-3. Views of drift fence trap arrays and types of traps used. Top Left: Schematic of overhead view of drift fence trap array. Each arm is 10 m long; there is a center pit fall trap and funnel traps (rectangles) placed along the fencing material. Top Right: Ground view of drift fence trap array. Bottom Left: Pitfall trap with mix of toads. Bottom Right: Funnel trap with Copperhead.

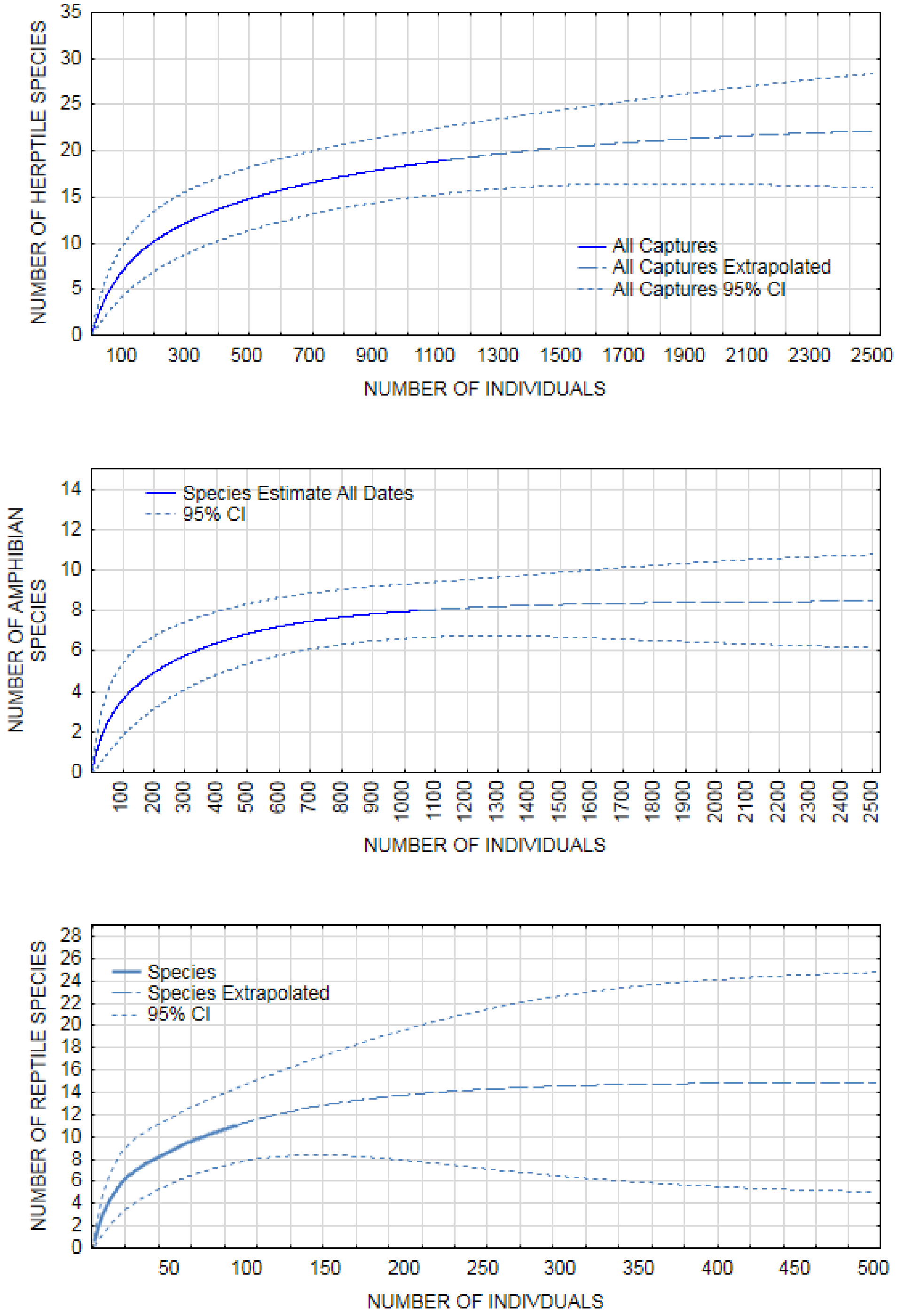


Figure 2-4. Chao estimates of species richness extrapolated from drift fence trap data. Top: All amphibians and reptiles. Middle: Amphibians only. Bottom: Reptiles only.

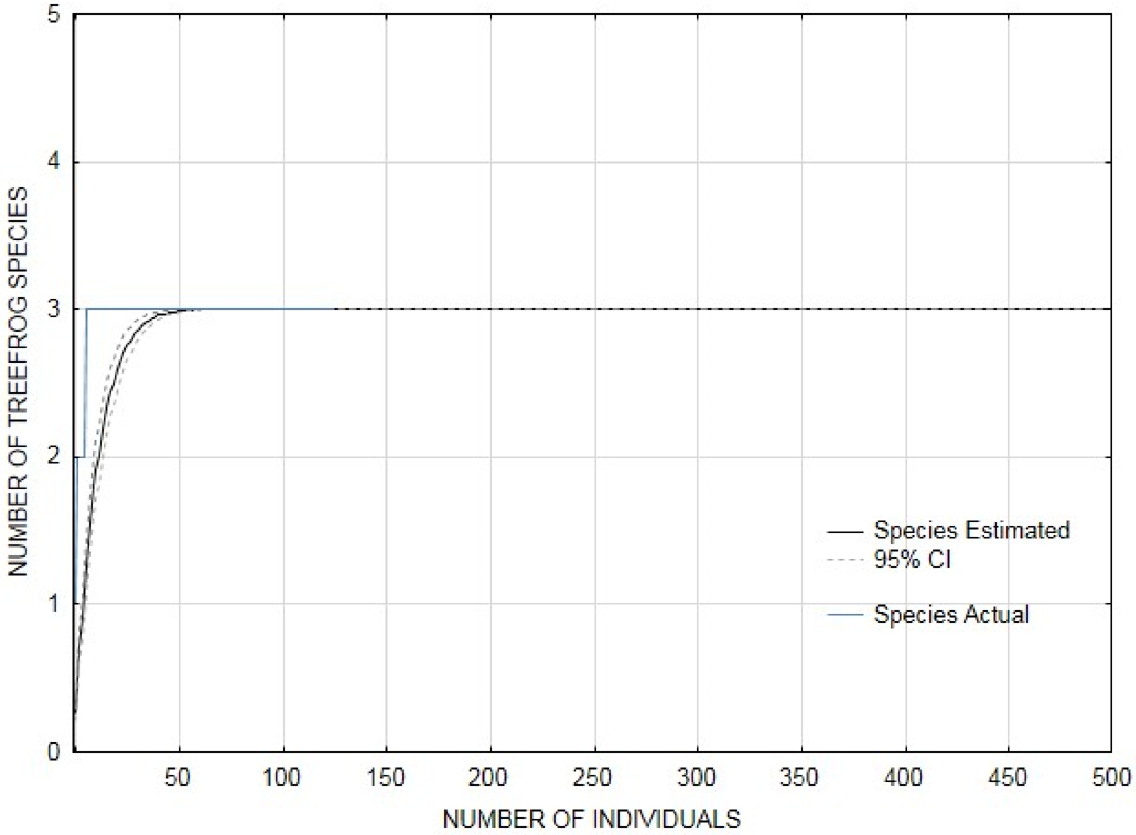


Figure 2-5. Chao estimates extrapolated from PVC pipe refugia captures of tree frogs.

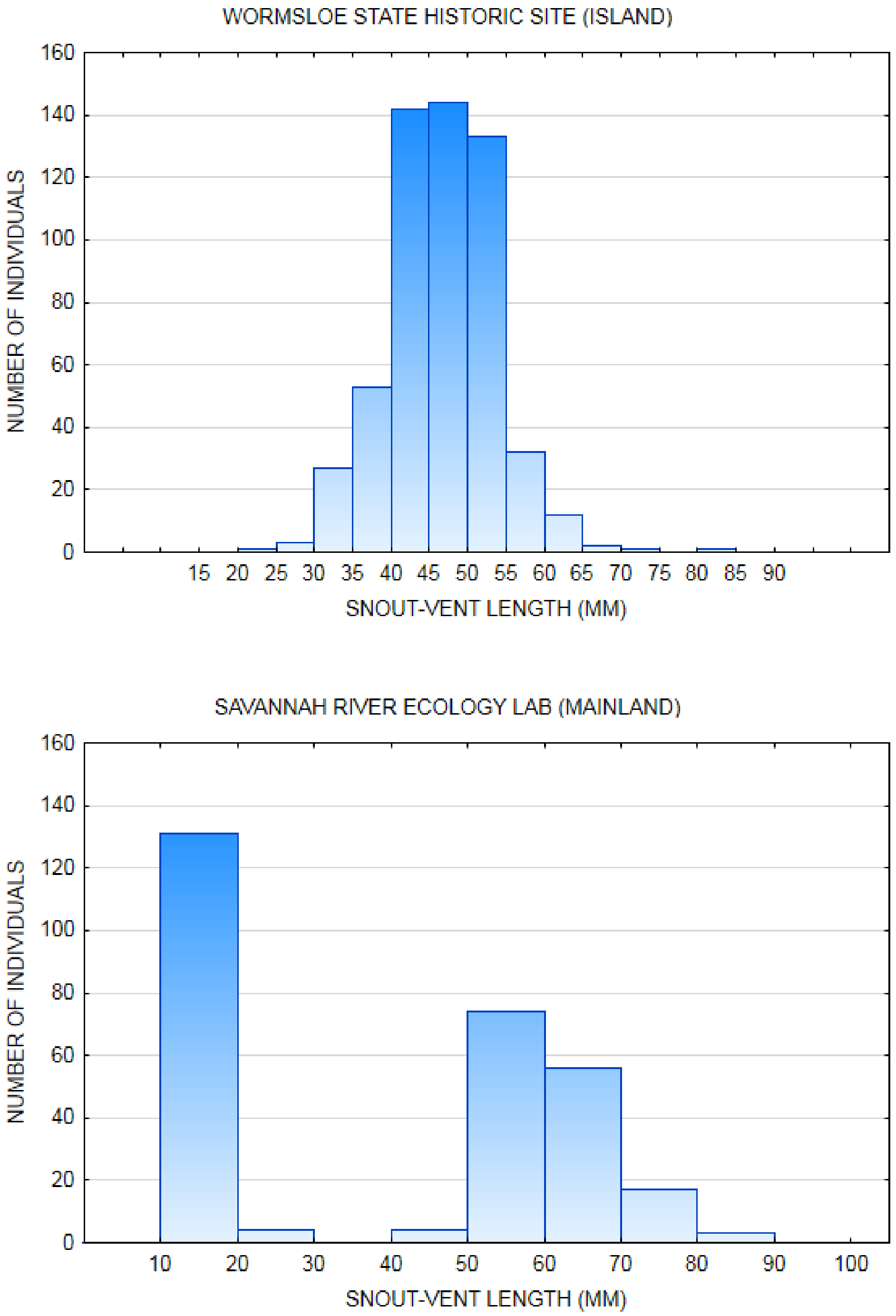


Figure 2-6. Histograms comparing body size of *Anaxyurus terrestris* between island and mainland.

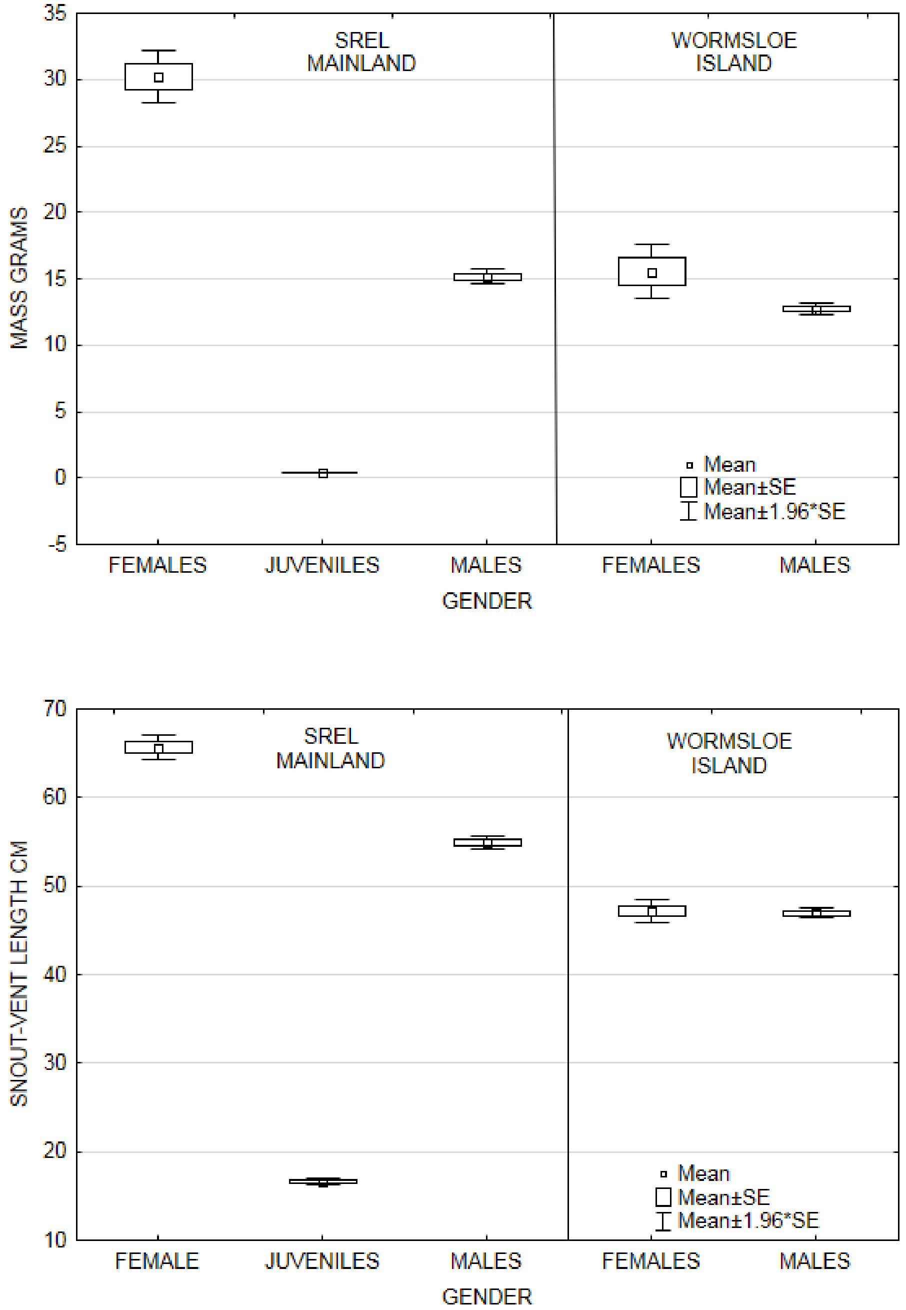


Figure 2-7. Box and whiskers plots of mass and length of *Anaxyrus terrestris* by gender, by site. No metamorphic juveniles were captured at Wormsloe. If error bars do not overlap, then means were significantly different.

Figure 2-8. Photograph of 9 *Terrapene carollina carolina* (Eastern box turtles) captured on a single day. Note diversity in color and pattern.

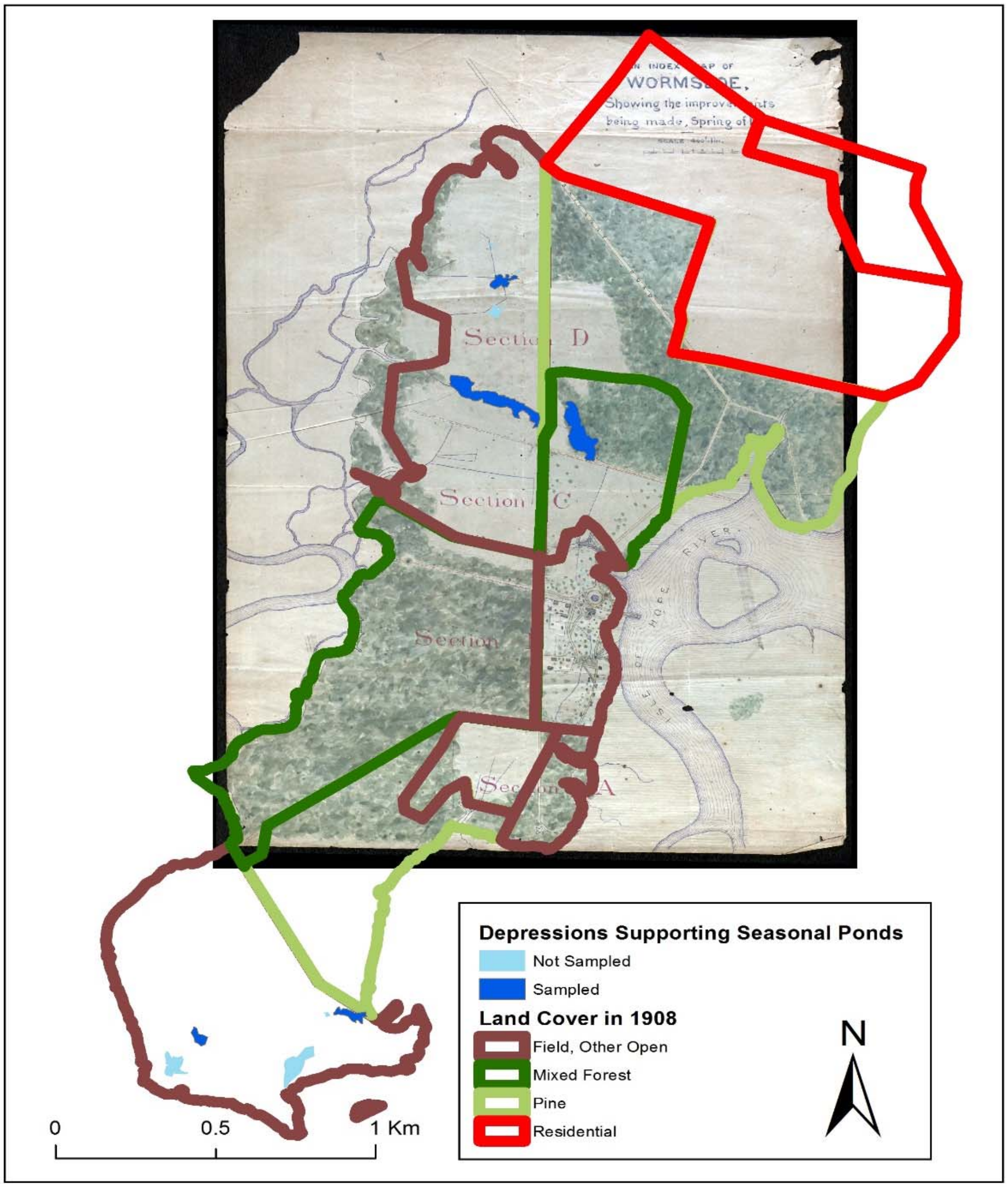


Figure 2-9. Background image of 1897 map, with outlines of land cover in 1908 and overlay of present-day areas of with potential to support seasonal freshwater ponds. The five areas included in present-day sampling identified depicted as blue polygons.

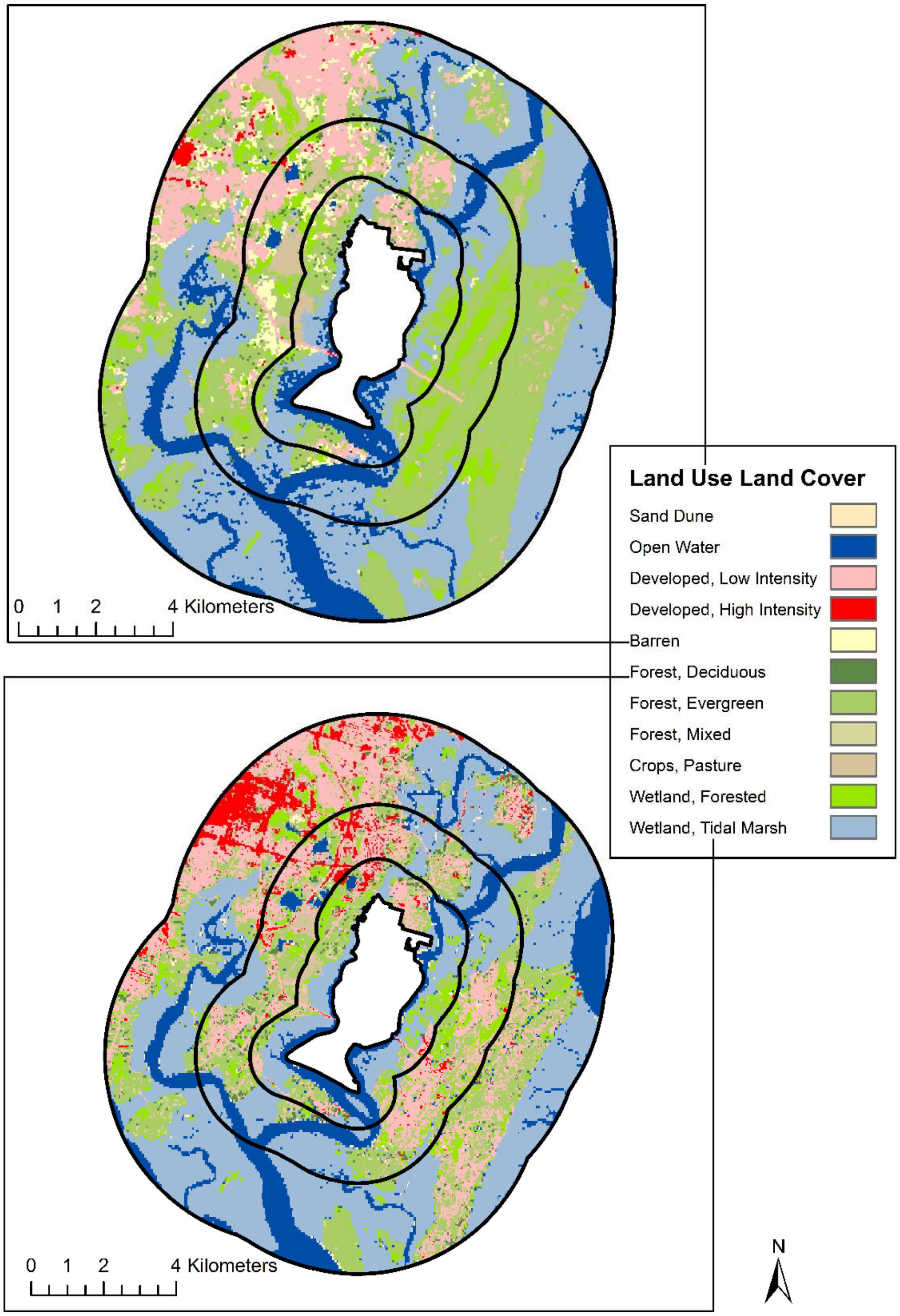
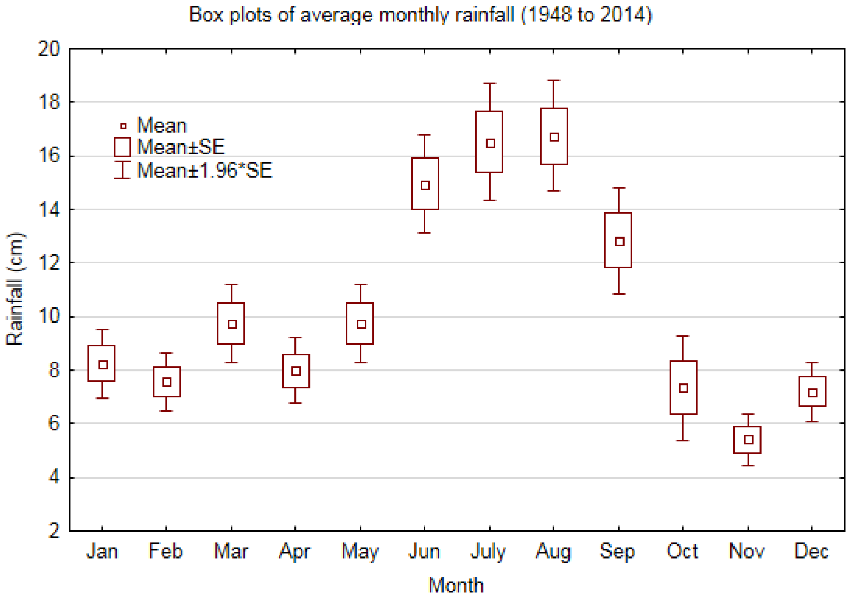


Figure 2-10. Changing landscape context of Wormsloe State Historic Site. Surrounding land uses land cover in 1974 (top) and 2008 (bottom). Dark rings are at distances of 1 km, 2.5 km, and 5 km.



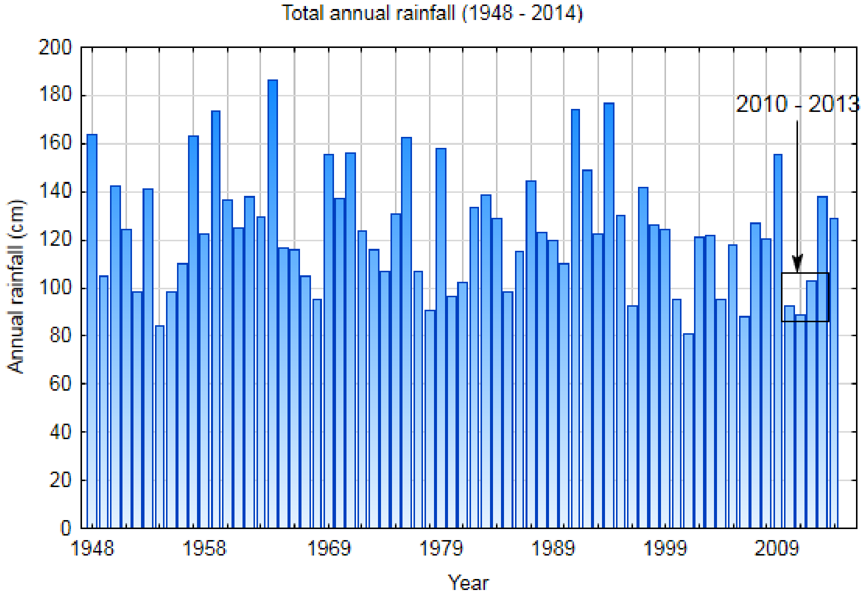


Figure 2-11. Variation in rainfall by month (top) and by year (bottom). Most trapping occurred between 2011 – 2013.

CHAPTER 3 AMPHIBIAN DIVERSITY IN THE GEORGIA SEA ISLANDS AND IMPLICATIONS FOR LONG-TERM CONSERVATION[7]

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[7] O'Hare, N.K., M. Madden, and C.R. Carroll. To be submitted to *Journal of Biogeography*

ABSTRACT

Amphibians as a group are thought to be sensitive to changing environs, in part due to: 1) their complex life history requiring both wetlands and uplands; and 2) their limited ability to move long distances. We summarized amphibian distribution from trapping and museum collections for 12 coastal Georgia islands held in conservation at either the state or federal level, and correlated distribution with species life history traits and landscape characteristics. Species occurring on more islands tended to have greater total reproductive output (i.e., life span >4 yrs, and annual egg production >1,000 eggs) and toleration of brackish water by adults. Larger islands had greater area of freshwater wetlands, particularly suitable for short hydroperiod communities. Species tied to long hydroperiod wetlands (>6 months) were more restricted in their distribution across islands. The younger Holocene islands had weaker correlation between island size and species richness, but overall larger islands supported more species. While Euclidean (e.g. straight line) distance between islands does not necessarily preclude inter-island dispersal, the matrix of tidal marshes, tidal creeks and wide rivers suggests that inter-island dispersal is very limited along the Georgia coast since most amphibians avoid salty environments. The paucity of recent occurrence data for amphibians in the region, let alone standardized annual monitoring data, hinders efforts to model species' vulnerability to changing environs or assess ecosystem resilience, despite state and federal directives to do so. While not replacing actual population data, consideration of life-history traits and breeding habitat availability are easier to obtain and can be used to direct management to support long-term species persistence.

INTRODUCTION

The southeastern coastal plain of the United States of America (USA) has been suggested as an unrecognized hotspot of floral and faunal biodiversity (Noss et al. 2014). Yet basic knowledge is still lacking of present-day species diversity and natural population fluctuations in many conservation areas within the region. Consequently, the impacts of future anthropogenic influences or climate change on species distribution are difficult to determine. How can potential future changes be addressed if present-day species diversity is unknown? Determining species diversity for areas already in conservation should be a priority. These areas are less likely to have future direct anthropogenic changes and will therefore serve as refuges critical to supporting existing levels of species diversity.

Amphibian declines have been noted from well-studied sites worldwide (Wake 1991) (Collins and Crump 2009). Habitat loss and degradation, pesticides, disease, or climate change are frequently cited as causes for amphibian declines, but the cumulative impact of multiple factors is recognized as more important (Blaustein and Kiesecker 2002, Blaustein et al. 2011, Wake 2012). Therefore, declines need to be investigated with regard to local causes, across multiple temporal and spatial scales. In the Georgia Sea Islands region (Fenneman 1946), local causes of amphibian decline are likely related to physical geography and land-use history. However, these environmental stressors may have differential impacts among individual species with different life-history traits. The physical geography of the Georgia Sea Islands related to species diversity and life history traits of amphibians of the Georgia coast were, therefore, identified as the focus of this study.

Geographically, the Georgia Sea Islands region occurs in the Georgia Bight of the Atlantic Coast (Zeigler 1959), paralleling the mainland for 185 km and varying in distance from the mainland between 3 and 15 km (Figure 3-1). The region includes more than 950 islands greater than 0.5 ha, of which 68 are named and 35 are greater than 100 ha in size. The outer islands are the largest and the geologically oldest, with a Pleistocene core area augmented by additional land area formed primarily by deposition of sand during the Holocene Era. These outer islands are isolated from each other by four main freshwater-tidal rivers (i.e., the Savannah, Ogeechee, Altamaha and Satilla Rivers), ranging in width from ~2 to 8 km wide at the outer barrier island edge. The back-barrier islands are isolated from each other and the mainland by the upstream of these freshwater-tidal rivers, as well as numerous smaller tidal channels, tidal marshes, and the dredged Intra-Coastal Waterway (ICW). The freshwater river channels have migrated thru time, most recently ~1,500 years before present (Chowns et al. 2008), as have tidal channels. Consequently, the landscape connections among islands have shifted through time.

Species' life history traits are also important in considering their geographical distribution. There are several life history traits of amphibians relevant to their distribution in the Georgia Sea Islands region. For example, the ability of a species to persist in an area is likely linked to both life span and reproductive effort. The life span of local amphibians varies greatly from approximately 2 to 15 years (Snider and Bowler 1992), while reproductive effort varies from a <50 eggs to thousands (Wright 1932). Second, the relatively thin, non-keratinized, permeable skin of amphibians makes them prone to desiccation. They desiccate in dry conditions or when exposed to brackish or salty water since water is pulled from their bodies by osmosis. In coastal regions, amphibians may adapt behaviorally or physiologically to reduce desiccation (Neill 1958, Rogell et al. 2011). Adults able to tolerate brackish water are more likely to successfully disperse across brackish areas between islands, as well as tolerate occasional tidal water mixing with inland freshwater ponds. Thirdly, the Class Amphibia has two subclasses in the region, Anurans (toads and frogs) and Urodeles (salamanders), which have divergent life histories. There are 17 native Anurans[8] in the coastal plain adjacent to the region; all have tadpoles which develop in freshwater and the adults have varying degrees of reliance upon freshwater which separates them into different functional groups. Adults of three species live in wetlands; adults of four species are arboreal near the natal wetlands in which they breed; and adults of ten species live in xeric to mesic uplands but typically return to natal wetlands to breed. The eight Urodeles in the region have more variable relationships with freshwater wetlands for breeding and adult habitat. Three genera live and reproduce entirely in mesic uplands (*Desmognathus*, *Eurycea* (two species), *Plethodon*), without need for freshwater ponds. The adults of one genus (*Ambystoma*) live in mesic uplands, but require freshwater for breeding. Two genera are tied to wetlands, but survive dry-downs by a state of dormancy known as aestivation (*Siren*, *Amphiuma*). There is one genus, *Notophthalmus*, with environmental plasticity in life-history within the species. In some regions, *Notophthalmus* adults live and reproduce in wetlands; in other regions, adults live in wetlands, but there is a juvenile (i.e., eft) stage that lives in mesic uplands and returns to wetlands after two to three

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[8] The non-native *Eleutherodactylus planirostris* (greenhouse frog) was recently documented at Fort Pulaski on Cockspur Islands, and likely occurs in the nearby areas of Savannah, Georgia. This species was most likely introduced in potted plants of the landscape nurseries trade. It has direct development (e.g. no tadpole stage).

years to mature and reproduce. The life-history of *Notophthalmus* in the Georgia Sea Island region, however, has not been described. In addition to variable life histories, most amphibians are able to persist for years in changing environs in part because of their lower energetic requirements as ectotherms (i.e., animals able to reduce basal metabolic rate but not maintaining constant body temperature), and also because of their explosive breeding during favorable conditions (Vitt and Caldwell 2014)

The combination of physical geography and amphibian life history traits creates complex, species-specific landscape linkages for dispersal between islands. Most anurans have a home range of up 1 km, with dispersal of up 10 km (Smith and Green 2005) while pond-breeding caudates tend to dispersal less than 1 km from natal ponds (Pittman and Semlitsch 2013). Dispersal between most islands may be limited by combination of distance and the habitat separating islands. Habitats separating island are a matrix of salty tidal marshes, tidal channels varying in both depth and width as result of 2 - 3 m diurnal tides, and the five major freshwater rivers of the Georgia coast (Savannah on border with South Carolina, Ogeechee, Altamaha, Satilla, and St. Mary's on border with Florida). Dispersal barriers vary within the Georgia Sea Islands region in both type and permeability. The barriers between most inner islands are temporally semi-permeable barriers of tidal marshes and tidal channels with diurnal variability in width. The five large freshwater rivers divide the region into east-west sub-basins, with one or two outer barrier islands in each sub-basin. The hydrological barriers between outer islands and their associated back-barrier islands are probably more permeable than hydrological barriers between the outer islands separated by the wide freshwater rivers.

Dispersal barriers between islands are not the only factors influencing current species diversity. The availability of seasonal freshwater ponds and land cover on the islands and the adjacent mainland has been altered by extensive land use during the past 250 years following colonization by Europeans (Vanstory 1970, Bragg 1999, Coulter 1955). Native Americans lived in the area prior to European colonization, albeit at lower population levels. Native Americans in general used fire as land management tool, but there are significant regional differences (Frost 1998). Intentional use of fires in the southeastern coastal plain is thought to be minimal due to high natural fire frequency from lightening and the gentle topography and extensive piney flatwoods that facilitated wildfire spread once ignited. A few of the large islands have been completely developed (Sea Island, Wilmington), while many have been partially developed (Tybee, Jekyll, Isle of Hope, Sapelo, Cumberland, Skidaway). Other large islands have minimal development today (Ossabaw, Wassaw, St. Catherine's); vegetation may be native and look undisturbed but the sites still have a land-use history. For instance, most of the original forest on all islands was either heavily logged or completely cleared for agriculture; consequently, the forests of today are mostly secondary successional forests. During the colonial period, ditches were created to mark property boundaries in the area. In addition, the subsequent agriculture and desire for disease control (malaria, yellow fever) resulted in the drainage of wetlands by ditches. Although these ditches are often still evident today, their water control structures are no longer in place. Without water control structures, salt water may intrude inland due to tidal flow along the ditches (O'Hare, Jordan, Madden, and Mu, in prep, see Chapter 4). Land-use history may have extirpated (e.g. local loss of a species) some species while facilitating dispersal of others. It is impossible to determine pre-European amphibian species diversity to estimate what has changed on the islands due to more recent human influence. However, an investigation of amphibian life histories combined with physical characteristics of coastal islands may explain observed differences in current amphibian species diversity

It should be noted that determining current species diversity has its own challenges. Recent summaries of the biogeography of amphibians rely upon museum collections (Williamson and Moulis 1979, Williamson and Moulis 1994) and many of the museum records pre-date extensive regional and on-island developments in the last 30 years. This is significant since the highly variable population abundances (i.e., r-selected life history) and metapopulation[9] dynamics may allow species to persist for decades in recently urbanized environments (Gagne and Fahrig 2010). However, metapopulation dynamics, which rescue local populations from extirpation, may eventually collapse as distances between remaining local populations increase beyond normal dispersal distances. There have been a few recent short-term amphibian trapping efforts on the Georgia coast (Dodd and Barichivich 2007, Tuberville et al. 2005) as well as species lists generated by incidental observations or short-term surveys. The National Park Service Southeast Coast Monitoring Network recently established amphibian monitoring protocols for the three federal park service units in the region (Byrne et al. 2010a, Byrne et al. 2010b, Byrne et al. 2011). However similar efforts are lacking for the majority of the islands including those managed by the State of Georgia.

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[9] Spatially separated populations of the same species interacting at minimal level through long-distance dispersal

Two previous studies considered herpetofaunal biogeography of the Georgia Sea Islands. Laerm et al. (2000) related species diversity of selected islands to geological age (Pleistocene versus Holocene), island size, distance to the mainland, and qualitative assessments of sampling effort and development. They found significant correlation of species diversity with geological age, island size, and sampling effort. Distance to mainland was not significant and attributed to relative short distances (<10 km). However, they did not test whether island age was related to island size, nor did they include quantitative habitat variables such as the wetland area which may influence amphibian diversity. Shoop and Ruckdeshcel (2003) proposed an alternative causal factor related to meta-population dynamics. The outer islands with a Pleistocene core were presumed to have similar fauna as the mainland before their isolation by sea level rise during the warming Pleistocene-Holocene transition 10,000 to 15,000 years before present. Their present day species fauna were assumed to have resulted from island-specific extinction events. The Holocene islands were formed by sedimentation in the past 10,000 years from freshwater river discharge between the mainland and the now isolated outer ridge forming present-day Pleistocene-core islands. Species present on these newly formed Holocene island represent successful colonization. While prescient, Shoop and Ruckdeshcel (2003) did not investigate whether there were patterns to extinction or colonization or offer any geographic analyses. The perspectives of both Laerm et al. (2000) and Shoop and Ruckdeshcel (2003) on amphibian diversity overlooked physical geographical factors, land-use history, and species specific life-history traits likely to influence amphibian diversity on islands in the region.

This study contributes to the existing knowledge of the amphibian biogeography of the Georgia Sea Islands that is relevant to a wider geographical area by evaluating existing species diversity related to species life history traits and landscape characteristics. We predict: 1) species occurring across more islands have greater total reproductive output (longer life span, greater number of eggs laid per year), shorter tadpole/larval stage, and adults that tolerate brackish or salty water; and 2) larger islands, regardless of geological age, have greater area of wetland habitats and consequently support higher species richness. We also predict species tied to either longer hydroperiod habitats (6 to 12 months) (e.g., Anurans of genus *Rana* and Caudata of genera *Siren* or *Amphiuma*) have restricted present-day distribution stemming from land-use history legacies that altered seasonal freshwater availability. Finally, dispersal between outer islands is expected to be limited by unfavorable habitat (e.g., wide rivers) rather than the physical Euclidean distance between islands.

METHODS

Study Area

The study area consists of 12 islands of coastal Georgia, USA for which information on amphibian diversity exists and at least part of the island is in conservation at either privately, state or federal level (see Figure 3-1; Table 3-1). It includes widely recognized conservation areas such as Cumberland National Seashore, as well as smaller, regionally known areas such as Wormsloe State Historic Site, Ossabaw Island and Wassaw National Wildlife Refuge. Most, if not all, of these 12 islands were former antebellum cotton or rice plantations, along with subsistence agriculture and livestock, and timber. Upland areas were either logged or cleared for agriculture and freshwater wetlands were drained or hydrologically manipulated by construction of drainage ditches. Secondary forests have regrown in island upland areas, although forest structure and soils have residual influences.

Table 3-1. List of islands and the conservation area on the island. Only two islands (Isle of Hope and St Simon's) are partially in conservation. GA DNR = Georgia Department of Natural Resources. NWR = National Wildlife Refuge. NERR = National Estuarine Research Reserve. NM = National Monument. NS = National Seashore. SHS = State Historic Site.

Age	Island	Site Name	Agency	Area (Ha)	Percent Natural	Percent Conserve
Holocene						
	Cockspur	Fort Pulaski NM	NPS	121	94	100
	Wassaw	NWR	USFWS	760	99	100
	L. St Simon's		Private	1239	100	100
	Blackbeard	NWR	USFWS	1427	97	100
	L. Cumber	NS	NPS	531	98	100
	Isle of Hope	Wormsloe SHS	DNR	630	51	51
	Harris Neck	NWR	USFWS	1644	90	
Pleistocene Core						
	St Catherine's		Private	2674	98	100
	Ossabaw		Private	4021	97	100
	Sapelo	NERR/Private	NOAA, DNR, Private	4083	94	
	St Simon's	Fort Frederica NM	NPS	4816	37	
	Cumberland	NS	NPS	6371	96	

Amphibian Species Diversity and Life History Traits

Amphibian species diversity at Wormsloe Historic Site on the Isle of Hope was measured in this study by field trapping over a three-year period (Chapter 2). For sites other than Wormsloe, we relied upon the literature to determine species presence/absence for each island (Williamson and Moulis 1979, Williamson and Moulis 1994, Laerm et al. 2000, Shoop and Ruckdeshcel 2000, Shoop and Ruckdeshcel 2003, Tuberville et al. 2005, Byrne et al. 2010a, Byrne et al. 2010b, Byrne et al. 2011). If a museum record or occurrence was disputed and was the only record of occurrence of a species on the island, then we did not include the species.

Life history traits were also determined from the literature. Traits included life span (Snider and Bowler 1992), breeding season, clutch size and length of larval stage (Wright 1932, Dorcas and Gibbons 2008), and tolerance for brackish water (Neill 1958, Christman 1974). Life history traits analyzed to determine correlation with landscape variables.

Landscape Characterization and Linkages

The number and location of islands were derived from the Georgia Department of National Resources (GA DNR) digital vector vegetation maps at 1:3,000 scale and derived from 2009 and 2010 aerial imagery (available http://data.geospatial.org). Vegetation communities were classified by both the National Vegetation Classification System (NVCS) (Jennings et al. 2009) and the State Wildlife Action Plan (SWAP) classification for Georgia (Association 2012) with a crosswalk between classification schemes. The SWAP categories were more readily related to wildlife and also included anthropogenic land covers; consequently we relied primarily upon SWAP classification scheme. We created a geographic information system (GIS) layer of island outline by querying the GA DNR vegetation maps for non-estuarine or brackish land covers. Contiguous polygons were dissolved to obtain island outlines, with the transportation linkages between islands clipped at the island edges. The SWAP categories were simplified even further to “wetland”, “upland”, and “developed”. The SWAP “wetland” habitats were characterized as short (<6 months) or long (>6 months) hydroperiod based upon dominant vegetation. SWAP also classifies some upland communities as successional which we used as indicator of substantial land-use history (e.g., clear cutting).

We created a database of amphibian species presence/absence by island, species life history traits, and landscape characteristics. Life history traits included tolerance of salt-water, hydroperiod required for larval development, and life span of adults. Landscape characteristics included land cover by SWAP vegetation community, island area, developed area, area of successional vegetation classes, and area of wetland habitats.

To investigate linkages between islands, two rasters were created at 10-m pixel size resolution for the entire Georgia Sea Islands region (180 km north-south and 10 to 20 km east-west). The first was simply a Euclidean distance (nearest neighbor) matrix from the outline island to each other island. The second raster was derived from the GA DNR habitat map by suitability to create a cost resistance surface (Adriaensen et al. 2003). The habitat map was first converted to raster format with a 1-m cell size so that the smaller tidal creeks separating islands would be represented in the cost resistance surface. Natural freshwater and upland cells were assigned a relative cost of 1, development, pine plantations and other agriculture a cost of 2, tidal brackish and salt marshes a cost of 3, tidal creeks and rivers a cost of 4. The 1 m raster was up sampled to 10 m raster, with resultant cost distance raster cells assigned the sum of the 1 m raster input values. Both the Eucliean distance and cost resistance surfaces were symbolized using values related to amphibian travel distances derived from the literature (e.g., <1 km, 1 to 2.5 km, 2.5 to 5 km, 5 to 10 km, 10 to 25 km, and more than 25 km). Less than 1 km would be typical within population migration between adult and breeding habitat for many species. Less than 5 km covers both within and between population dispersal by most species. Higher values represent decreasing likelihood of dispersal, with anything beyond 25 km rare and random chance.

Software and Statistical Analyses

We used ArcGIS 10.2 (ESRI, Inc.) for most GIS analyses (e.g., feature extraction from existing layers, spatial joining of layers), the licensed version of LAStools (LAStools, "Efficient LiDAR Processing Software", version 141017, academic, obtained from http://rapidlasso.com/LAStools), and Quick Terrain (QT) Modeler 8.0 (Applied Imagery) for digital surface models. Species accumulation curve and estimates of species richness were produced using EstimateS 9.1.0 for Windows (Colwell and Elsensohn 2014). Scatterplots and statistical analyses were performed in Statistica 12 (StatSoft 2014). Correlation between diversity, life-history traits, and landscape variables were test using the non-parametric Spearman R for correlation; significance was interpreted for p values less 0.05.

RESULTS

Amphibian Species Diversity

Results of field trapping at Wormlsoe on the Isle of Hope (Chapter 2) and literature review for 11 additional Georgia coastal islands documented a total of 23 amphibian species (16 Anurans, 7 Caudata; Tables 3-2, 3-3, and 3-4). The highest species diversity (19) was recorded from Cumberland Island. Four species occurred on all 12 islands, with a fifth species occurring on 11 of the 12 islands. The majority of species (10) occurred on three or fewer

Table 3-2. Species occurrence (+++) on islands formed during the Holocene periods. Islands are arranged from smallest size to largest size.

	Holocene						
Species	Cockspur	L. Cumber	Isle of Hope	Wassaw	L. St Simon's	Black-beard	Harris Neck
Anurans							
Acris crepitans							
Acris gryllus	+++		+++				+++
Bufo quercicus							
Bufo terrestris	+++	+++	+++	+++	+++	+++	+++
Gastrophryne carolinensis	+++	+++	+++		+++	+++	+++
Hyla chrysoscelis	+++						
Hyla cinerea	+++	+++	+++	+++	+++	+++	+++
Hyla femoralis			+++				+++
Hyla gratiosa		+++					+++
Hyla squirella	+++	+++	+++	+++	+++	+++	+++
Limnaeodus ocularis	+++						+++
Pseudacris crucifer							+++
Pseudacris nigrita							
Rana catesbiana							+++
Rana grylio							+++
Rana sphenocephala	+++	+++	+++	+++	+++	+++	+++
Scaphiopus holbrookii		+++	+++				
Eleutherodactylus planirostris							
Caudates							
Ambystoma talpoideum							
Amphiuma means							
Desmognathus auriculatus							
Eurycea cirrigera							
Eurycea quadridigitata	Disputed						
Notophthalmus viridescens							+++
Siren lacertina							
Plethodon grobmani			+++				
Grand Total	8	7	9	4	5	5	13
Anurans	8	7	8	4	5	5	12
Caudates	---	---	1	---	---	---	1
Disputed	---	---	---	---	---	---	---

islands. Species persisting on Pleistocene islands also tended to colonize a larger number of Holocene islands (Spearman R = 0.837, $p < 0.01$; Figure3-2). No species were endemic to the islands. There were three additional species (*Acris crepitans*, *Bufo quercicus*, and *Eurycea cirrigera*) present on the mainland but their occurrence on the islands is disputed by Shoop and Ruckdeschel (2003).

Table 3-3. Species occurrence (+++) on islands with a Pleistocene core. Islands are arranged from smallest size to largest size.

	Pleistocene Core				
Species	St Catherine’s	St Simon's	Sapelo	Ossabaw	Cumberland
Anurans					
Acris crepitans					
Acris gryllus		+++			+++
Bufo quercicus					Disputed
Bufo terrestris	+++	+++	+++	+++	+++
Gastrophryne carolinensis	+++	+++	+++	+++	+++
Hyla chrysoscelis					+++
Hyla cinerea	+++	+++	+++	+++	+++
Hyla femoralis	+++	+++	+++		+++
Hyla gratiosa		+++		+++	+++
Hyla squirella	+++	+++	+++	+++	+++
Limnaeodus ocularis	+++	+++	+++	+++	+++
Pseudacris crucifer		+++	+++		+++
Pseudacris nigrita			Disputed		+++
Rana catesbiana				+++	
Rana grylio			+++	+++	+++
Rana sphenocephala	+++	+++	+++	+++	+++
Scaphiopus holbrookii	+++	+++	+++	+++	+++
Eleutherodactylus planirostris		+++			
Caudates					
Ambystoma talpoideum		+++			+++
Amphiuma means	+++		+++		+++
Desmognathus auriculatus					+++
Eurycea cirrigera				Disputed	
Eurycea quadridigitata	+++			+++	+++
Notophthalmus viridescens	+++		+++	+++	+++
Siren lacertina				+++	
Plethodon grobmani	+++			+++	
Grand Total	12	13	12	14	19
Anurans	8	12	10	10	14
Caudates	4	1	2	4	5
Disputed	---	---	1	1	1

Table 3-4. Summary of number of islands species occurred on by geological age.

	Species	# Islands	Holocene Colonized	Pleistocene Core Extirpated
Anurans				
	Acris crepitans	Disputed		
	Acris gryllus	5	3	2
	Bufo quercicus	Disputed	0	5
	Bufo terrestris	12	7	0
	Gastrophryne carolinensis	11	6	0
	Hyla chrysoscelis	2	0	4
	Hyla cinerea	12	7	0
	Hyla femoralis	6	2	1
	Hyla gratiosa	5	2	2
	Hyla squirella	12	7	0
	Limnaeodus ocularis	7	2	0
	Pseudacris crucifer	4	2	2
	Pseudacris nigrita	1	0	4
	Rana catesbiana	2	1	4
	Rana grylio	4	1	2
	Rana sphenocephala	12	7	0
	Scaphiopus holbrookii	7	2	0
	Eleutherodactylus planirostris	1		
Caudates				
	Ambystoma talpoideum	2	0	3
	Amphiuma means	3	0	2
	Desmognathus auriculatus	1	0	4
	Eurycea cirrigera	Disputed	0	5
	Eurycea quadridigitata	3	0	2
	Notophthalmus viridescens	5	1	1
	Siren lacertina	1	0	4
	Plethodon grobmani	3	1	3
Grand Total		23		
	Anurans	16		
	Caudates	7		
	Disputed	3		

Correlates of Species Diversity: Island Size, Landscape Metrics, and Life History Traits

The size of islands ranged between 121 ha and 6,371 ha, with an average size of 4,287 ha. Five of the 12 islands formed during the Pleistocene with additional area accreted during the Holocene; the remaining seven were entirely formed during the Holocene. Geologically older islands were larger (Figure 3-3). The average size of Holocene islands ($n = 7$; mean 1065 ± 18 standard error (SE)) was smaller than Pleistocene (mean 4,326 ± 62.4 se; ANOVA, $df = 1, 9$, $F = 16.4$; $p < 0.001$). There was no overlap in size; the largest Holocene island (Harris Neck, 1,643 ha) covered ~1,000 ha less than the smallest Pleistocene island (St. Catherine's, 2,673 ha). Larger islands supported greater species richness (Spearman $R = 0.822$; Table 3-5, Figure 3-4). Correlation was less significant if analyzed separately by geological age, and the correlation was relatively higher for geologically older islands (Spearman $r = 0.648$ for Pleistocene and 0.171 for Holocene islands).

For 10 of the 12 islands, the entire island was in conservation, and development was less than 10% of island area. The first exception was the Isle of Hope; Wormsloe State Historic Site occupies the southern portion of the island and the northern 48% of the island is residential development, primarily single family homes with density of ~3 houses per acre. The second exception was Fort Frederica on St Simon's Island, which represents ~1% of the island, with the remaining portion developed as residential or urban areas.

The percentage of GA DNR SWAP communities representing wetland habitats ranged between 0% (Cockspur Island) and 21.0% (Blackbeard Island), with an average of 6.7% wetland habitat per island. Area of wetland habitat was positively correlated with island size (Spearman $R = 0.773$). Shorter hydroperiod wetlands were more common than longer hydroperiod wetlands (average of 5% vs 1% of area, respectively). Area of shorter hydroperiod wetlands was also more strongly correlated with total island area (Spearman R = 0.776) than area of longer hydroperiod wetlands (Spearman R = 0.585).

All islands had secondary successional forest communities, an indication of past agriculture or logging. The percentage ranged from <1% (Little St. Simon's Island and Little Cumberland Island, both Holocene islands) to 52% (Cockspur Island, also Holocene), with an average of 18.8% per island. There was no correlation between percentage successional forest communities and amphibian diversity (Spearman R = 0.025).

Table 3-5. Spearman rank order correlations between species richness, area, wetland area, and island age (Pleistocene or Holocene). Marked correlations (*) were significant at p<0.05.

			Wetland Area				
	Richness	Island Area	*Total*	*Long*	*Short*	% Successional	Island Age
Richness	---	0.822*	0.589*	0.494*	0.659*	0.025	0.825*
Island Area		---				0.145*	0.864*
Total Wetland		0.773*				-0.156*	0.469*
Long		0.585*	0.577*			-0.211*	0.382*
Short		0.776*	0.971*			-0.089*	0.529*

Species that laid greater number of eggs occurred on more islands (Spearman R = 0.594; Table 3-6). There was no significant correlation between species distribution and adult longevity, breeding season (winter versus spring-summer), and minimum hydroperiod required for egg and tadpole development. However, scatterplots suggest that species living longer,

Table 3-6. Spearman Rank correlations of life history traits with occurrence on islands (12 maximum). Breeding season was either winter, spring/summer, or summer/ fall. Tadpole duration was minimum number of days from egg laying to tadpole hatching. Salinity was ability of adults to tolerate brackish water for short periods, but not breeding in brackish water. Marked correlations (*) were significant at $p < 0.05$.

	Longevity	Number Eggs (log)	Breeding Season	Tadpole Duration	Salinity
Number Islands	0.105	0.594*	0.415	-0.033	0.675*
Longevity		0.115	0.150	-0.044	0.262

having shorter tadpole stage (<90 days), and with adults with some salinity tolerance tend to be more widely distributed (Figures 3-6 and 3-7).

The Euclidean distance between islands were within the range of long-distance amphibian dispersal (Figure 3-8). The average Euclidean distance of the five nearest neighbors was calculated for each of the study islands (Table 3-8). Euclidean distance for first nearest neighbor ranged between 25 m (Isle of Hope to mainland) and 2.8 km (Wassaw to Flora Hammock). There was no correlation between island area and Euclidean distance to any nearest neighbor (Spearman R = 0.130 for 1st, -0.217 for 2nd, 0.035, for 3rd, -0.025 for 4th, and 0.018 5th nearest neighbor, respectively) nor between diversity and Euclidean distance to any nearest neighbor (Spearman R = -0.080 for 1st, -0.181 for 2nd, -0.140, for 3rd, -0.113 for 4th, and -1.163 5th nearest neighbor respectively). Overall, Euclidean distance suggested that most outer Pleistocene islands were theoretically connected to each and to the mainland thru inner Holocene islands. The exception was Cumberland Island, which was isolated from all by Little Cumberland Island (Holocene addition to the Pleistocene core-Cumberland) and the mainland.

Table 3-7. Euclidean distance between each of study islands and its nearest neighbor.

Age	Island Name	Area Ha	Diversity	Euclidean Distance (meters) to NN					Distance (meters)		
				1st	2nd	3rd	4th	5th	Mean	Max	Min
Holocene											
	Cockspur	121	8	746	1219	1980	3505	4398	2370	4398	746
	L. Cumber	531	8	112	3948	4776	9568	13749	6431	13749	112
	Isle Hope	630	9	15	103	783	892	1900	739	1900	15
	Wassaw	747	4	2757	3188	4793	5258	5314	4262	5314	2757
	L. St Sim	1146	5	611	1054	1906	3287	9197	3211	9197	611
	Blackbeard	1427	5	198	534	2907	5567	5666	2975	5666	198
	Harris Neck	1644	13	196	571	688	2689	3235	1476	3235	196
Pleistocene											
	St Cath	2637	13	2328	2907	4703	5705	6677	4464	6677	2328
	Ossabaw	4021	15	2328	2907	4703	5705	6677	4464	6677	2328
	Sapelo	3960	14	257	2328	4971	5258	5754	3714	5754	257
	St Sim	4383		198	788	3425	4991		2350	4991	198
	Cumber	4816	20	230	337	1755	1906	4375	1721	4375	230
Mean				831	1657	3116	4528	6086			
Maximum				2757	3948	4971	9568	13749			
Minimum				15	103	688	892	1900			

Tidal rivers, having both current and salinity, are major dispersal barriers for amphibians. The widest divisions were north-south, creating nine basins defined by river mouth. River mouths varied in width between 2 and 5 km (Figure 3-9). Five of these nine basins had an outer Pleistocene core island (Ossabaw, St Catherine’s, Sapelo, St Simon’s, Jekyll, Cumberland). Other basins had outer islands formed during the Holocene (Tybee, Wassaw, Wolf). There were smaller east-west divisions demarcated by tidally influenced rivers and creeks of varying widths and depths.

Cost resistance surface incorporated the increased resistance to amphibian travel posed by developed, salt marshes and tidal rivers. Cost resistance surface indicates that four of the nine basins identified above are actually isolated from the mainland and that all outer islands are isolated from each other (Figures 3-10 thru 3-13). Island groups falling between the major rivers may have connectivity to others within the same sub-basin.

DISCUSSION

A total of 23 amphibian species occurred across the 12 conservation areas included in this study of the Georgia Sea Islands. However, no single island supported all species; species richness on individual islands ranged between 5 and 18. Species occurring on more islands tended to have greater total reproductive output (longer life span >4 yrs, and annual egg production >1,000 eggs) and adults tended to tolerate brackish, salty water. Larger islands had more wetland area, particularly short hydroperiod wetlands. Species tied to long hydroperiod wetlands (>6 months) were more restricted in their distribution. Overall, larger islands, regardless of geological age, supported greater species richness; the correlation was stronger for geologically older islands.

This study added the first occurrence of the woodland salamander *Plethodon glutinosus* (slimy salamander complex) to a Holocene island, at Wormsloe State Historic Site on the Isle of Hope. Also of interest, occurrence records of *P. glutinosus* exist for only two of the five Pleistocene-core islands (St Catherine’s and Ossabaw). This species is likely to occur on other Holocene and Pleistocene islands; lack of confirmed records is probably due to lack of data. Also, the first occurrence of the non-native *Eleutherodactylus planirostris* (Byrne et al. 2011) was recently documented for any Georgia Sea Island. Regular inventory and monitoring of species is also valuable for detecting establishment of non-native species, thereby likely to decrease their impact or management costs.

Three species present on the mainland but missing from the islands have disputed occurrence on Pleistocene islands (*Acris crepitans*, *Bufo quercicus*, and *Eurycea cirrigera*). The southern edge of the natural range of *Acris crepitans* occurs around the Savannah River, the boundary between Georgia and South Carolina, and few undeveloped islands occur in this region. Both *Bufo quercicus* and *Eurycea cirrigera* occur throughout coastal Georgia. *Eurycea cirrigera* tends to favor woodland streams, which are absent from the islands. Suitable habitat for *Bufo quercicus* exists on the islands. While it has a short tadpole stage (30 to 45 days), its overall total reproductive output (300 to 500 eggs and life span of ~4 years is relatively less than other species common across the islands), although it is similar to that of *Acris gryllus,* which has been verified on three islands by recent trapping. Overall, species occurrence on Pleistocene islands was more similar to the mainland, while occurrences on Holocene islands were more idiosyncratic, with less clear patterns.

Laerm et al. (2000) suggested that herpetofaunal species diversity in the Georgia Sea Islands region was partly related to island age. Shoop and Ruckdeshcel (2003) suggested that current patterns resulted from differential extirpation on Pleistocene islands whereas patterns on Holocene islands resulted from differential colonization. Higher correlation of species richness with island size on Pleistocene versus Holocene islands is congruent with both of these previous studies. Both persistence and colonization should be higher on larger islands, especially if the island is closer to a source population on either the mainland or a Pleistocene-core island. The largest Holocene island, Harris Neck, had the highest species diversity (13) of Holocene islands, and was similar in diversity to 4 of the 5 Pleistocene islands even though it was ~1,000 ha less in size than the smallest Pleistocene island. Species that tended to persist on the older, larger Pleistocene islands also colonized a greater number of Holocene islands. Both the mainland and the Pleistocene islands could serve as source for colonization of Holocene islands. However small sample sizes precluded quantitative, statistical analyses.

On small islands, diversity-area relations may shift, in part due to balance between local extirpation and recolonization from adjacent areas (Whitehead and Jones 1969, Barrett, Wait and Anderson 2003, Triantis and Sfenthourakis 2012, Triantis et al. 2006). However, the definition of “small island” differs even within a taxonomic group (Santos et al. 2010, Triantis and Sfenthourakis 2012, Triantis et al. 2006). Each of the islands in the study area exceeded literature definitions of “small”. However, many amphibians rely upon micro-habitat features for breeding. Micro-habitat features critical for breeding, e.g. seasonal ponds, are still under-represented in broad scale vegetation mapping, in part because of difficulty of detection and pre-defined minimum mapping units. For example, USFWS National Wetlands Inventory maps typically have a minimum mapping unit of about one half hectare, purposively avoid errors of commission (i.e., the error of including non-wetlands as wetlands) and recognize that small wetlands in forested areas are likely unmapped in forested areas of the eastern USA (Tiner 1997). Regional studies tended to rely upon higher resolution aerial photography, but imagery was frequently acquired during the growing season, so that seasonal ponds in forested areas were difficult to detect. Even the finer scale habitat maps used in this study did not map any wetland habitat on one island for which eight seasonal pond-breeding amphibian species were recently confirmed (Byrne et al. 2011). Consequently, the area of a critical micro-habitat may be as influential as total land area.

The ability to detect the structure of microhabitat wetland features via remote sensing is improving (Burne and Lathrop 2008, Carpenter, Stone and Griffin 2011). In the past decade, airborne light ranging and detection (lidar) with point density on the order of one point per square meter has effectively detected small, seasonal freshwater ponds even in forested habitats (also see Chapter 4). While techniques to detect and characterize the structure of seasonal ponds are improving, the ability to tie structure to seasonal hydrology lags behind. The hydrology of seasonal ponds, especially the depth and duration of inundation, relies upon precipitation which naturally varies temporally and is still difficult to determine without local measurement of water depth and duration (Kish, Milla and Means 2012).

Few United States Geological Survey (USGS) gages monitoring surface water occur on coastal islands (approximately 18) (Peck, Painter and Leeth 2009), and most are clustered in developed areas (Savannah, Jekyll Island) or areas with high industrial water use (Brunswick). Estimated water table map for coastal Georgia described water table as nearly coincident with the ground surface along the coast (Peck and Payne 2003), creating a freshwater lens. However, anecdotal evidence suggests that surface freshwater is decreasing on several of the Georgia coast islands. At Wormsloe, for example, there is one particular area which exhibits evidence of a previous longer hydroperiod than was observed in the past four years of this study. The small depression currently has <5 mature canopy *Taxodium distichum (*bald cypress), numerous cypress knees, yet no seedling or saplings. This species prefers more than a 6-month hydroperiod and seedlings fail to germinate if soil moisture is too low. Between 2010 and 2014, this pond within Wormsloe was never observed to have more than a few continuous weeks of surface water. Even then, the surface water from rainfall was influenced monthly by tidal influx along canals (see Chapter 4). Separating water inputs into freshwater rainfall versus saltwater tidal influx would was not possible.

Regional development continues and may indirectly alter freshwater resources on conservation lands. Given temporal delay in amphibian population change related to their low energetic requirements as r-selected ectotherms, the results of regional development and freshwater withdrawals may take decades to result in local species extirpations (Gagne and Fahrig 2010). Wetland desiccation has been tied to a decrease in four common amphibians in Yellowstone National Park (McMenamin, Elizabeth and Wright 2008). Coastal Georgia seems to be experiencing both changes in the surficial, freshwater lens as well as changes in the deeper Floridan aquifer. Artesian wells, supplied primarily by the deep Floridan aquifer, no longer flow on Wormsloe, Sapelo , St Catherine's (Nelson et al. 2012)and Ossabaw Islands, and municipal water supplies tapping the Floridan aquifer now require pumping. Artesian wells on the islands likely supplied surficial freshwater wetlands which have become ephemeral as artesian flow decreased (Vance et al. 2010). Since hydrology is major abiotic factor with regional controls, resources managers of individual conservation units need to consider the external threats to their internal management (Kushlan 1987). Areas with moderate species diversity which are in conservation, such as Wormsloe, should consider resources management to improve breeding habitat to maintain current species diversity.

Annual monitoring of amphibian breeding would catalog species persistence, and facilitate correlation between population dynamics and changing environs (Ray et al. 2014). Narrower time-windows of either extirpation or colonization would also be possible. Annual amphibian monitoring has recently been implemented at the three federal parks in the region.

These surveys included the first record of the non-native *Eleutherodactylus planirostris* on the islands, as well as confirmed the occurrence of *Hyla gratiosa* (Byrne et al., 2009). State conservation lands would benefit from adopting a similar monitoring protocol. In this study, species richness on 7 of the 12 islands was derived from museum vouchers and surveys from 30+ years prior yet it remains the best available information.

While a total of 23 amphibian species occurred on the 12 Georgia coastal islands in this study area, no island supported all 23 species. Species which persisted on Pleistocene islands were also more likely to colonize Holocene islands, suggesting that the life history traits for persistence thru time were similar to life history traits facilitating colonization of more recently formed Holocene islands. Body size has also been shown to influence extirpation and colonization in transforming landscapes (Allen, Forys and Holling 2010). Yet without data on body size in our study area, we were unable to further investigate this relationship. It is unclear if Bergman's Rule describing shift in body size on islands applies to amphibians in the Georgia Sea Islands, although shift in body size has been documented in some amphibian populations on islands in North America and Europe (Olalla-Tárraga and Rodríguez 2007). Until local data on body size are available, testing hypotheses relating species extirpations and colonizations to body size discontinuities cannot be performed in this region.

Resources management of regional amphibian populations is challenged by jurisdictional boundaries, as well as hydrological boundaries. The conservation areas included in this study occur in five of the six coastal Georgia counties. Four are privately owned. Two are owned by the State of Georgia and managed by Department of National Resources. Six are in federal ownership, managed by either the National Park Service (three units) or the Fish and

Wildlife Service as National Wildlife Refuges (three units). The areas also cross hydrological units (HUC) defined by USGS. Major rivers tend to be central feature of a HUC basin (HUC 6). However, these rivers are major dispersal barriers for amphibians and consequently were the edges between dispersal basins for amphibians. The study area spans 4 basins (HUC6). Smaller rivers or tidal channels further sub-divide basins into sub-basins (HUC8) or watersheds (HUC10). There were 11 watersheds, and watershed boundaries bisected 9 of the 12 islands.

The Georgia Sea Islands developed under fluctuating sea levels during the Pleistocene-Holocene transition (Hoyt 1967). Predictions for future sea level rise suggest that tidal marshes between the islands are likely to convert to more open water (Craft et al. 2009), further isolating the islands from an amphibian's perspective. The outer islands are the least likely to be completely inundated by future sea level rises since they tend to be higher in elevation and larger in area. Establishing baseline amphibian population data now will facilitate future correlations to changes island area, connectivity, and in the timing or amount of precipitation. Moreover, existing high resolution topographic models derived from improved remote sensing data would permit modeling of seasonal hydrology if data from surface water monitoring gages could be correlated with wetland hydrology. Any species extirpated from an island is unlikely to re-establish from natural dispersal given that remaining populations tend to be separated by large rivers which are dispersal barriers. Biologically, isolation can be stimulus for speciation, but in cases of smaller areas may also lead to increased risk of local extirpation.

CONCLUSIONS

We confirmed that species occurring across more islands tended to have greater total reproductive output (longer life span, greater number of eggs), had a shorter tadpole/larval stage, and adults had some tolerance for brackish water, and thereby more likely to disperse across saltwater marshes. Larger islands, regardless of geological age, supported more species; the relationship between geological age and species diversity was more idiosyncratic on smaller islands. Larger islands also had larger area of wetland areas, particularly short-hydroperiod wetlands. Consequently, we were unable to separate the effect of island size, geological age, and habitat (gamma) diversity on amphibian diversity. Finally, dispersal between outer islands was limited more by between island matrix of freshwater rivers and tidal marshes than by Euclidean distances between islands.

While the Georgia Sea Islands have no endemic species, individual conservation areas should maintain current species diversity by supporting existing populations. On some islands, the scarcity of freshwater wetlands to support breeding appears to be a limiting factor. Resources management actions should minimally include annual monitoring of anuran breeding, establishment of surface water gages on island, especially those more likely to be influenced by adjacent development or water-intensive land-uses, as well as actions to protect freshwater quality by preventing tidal influx along canals connecting the depressions to adjacent salt marshes. If the surficial freshwater lens has been permanently depleted, then measures to retard percolation of rainfall into soil, such as lining natural depressions, may be useful.

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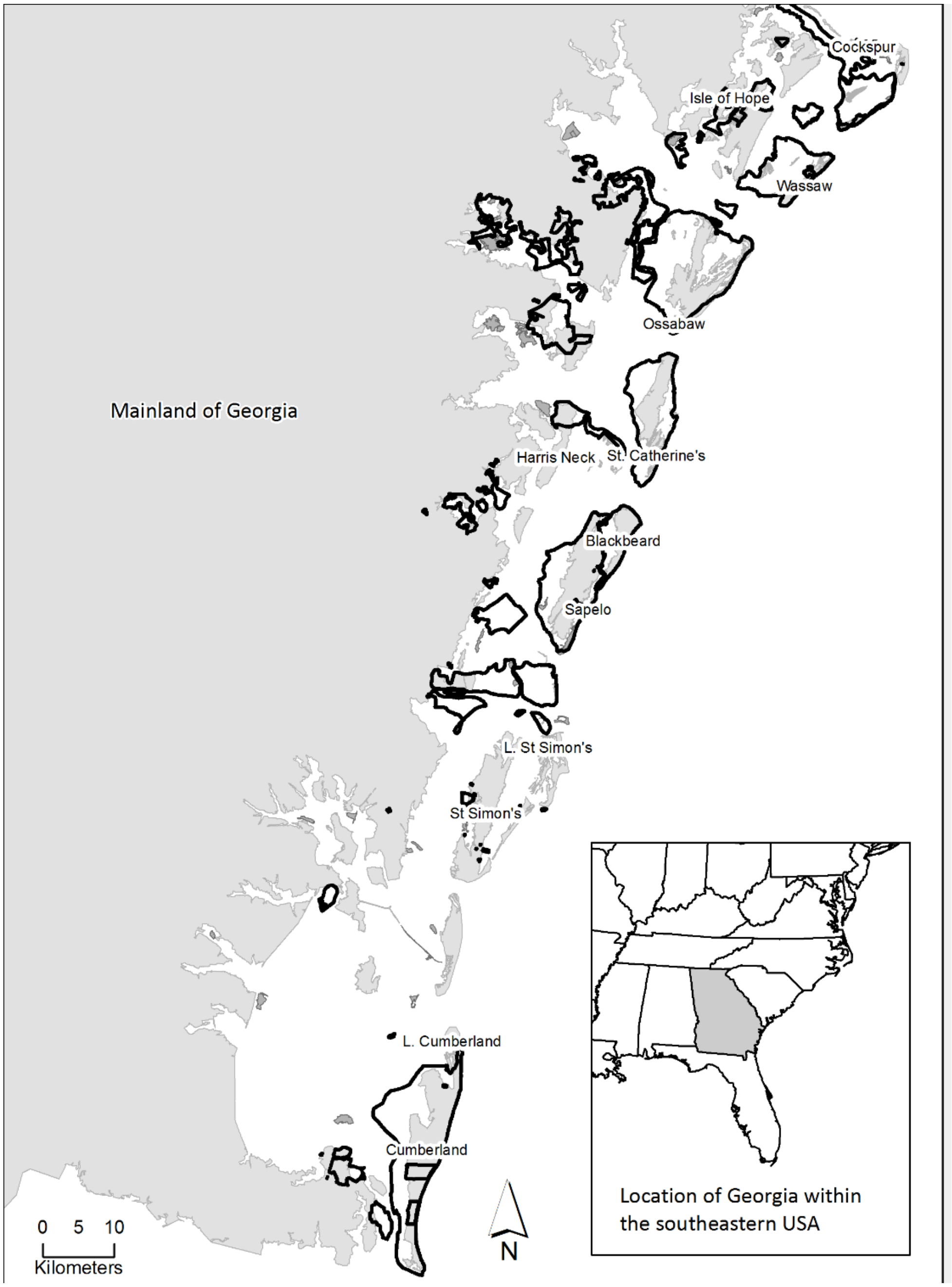

Figure 3-1. Study area location. Conservation areas included in this study are outlined in black. Other conservation areas are included for reference and outlined in gray.

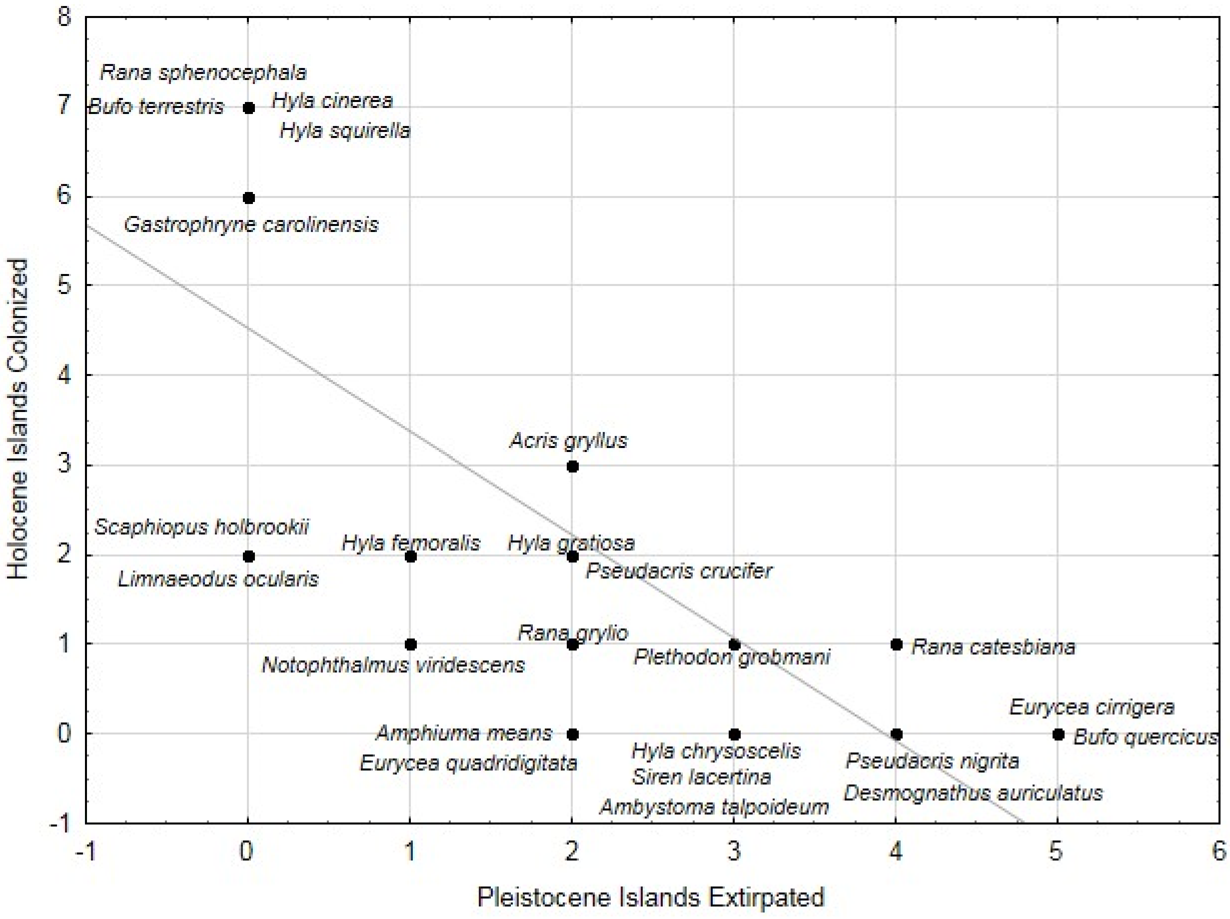

Figure 3-2. Scatterplot of number of Pleistocene islands species extirpated from (maximum of 5) versus number of Holocene islands species colonized (maximum of 7).

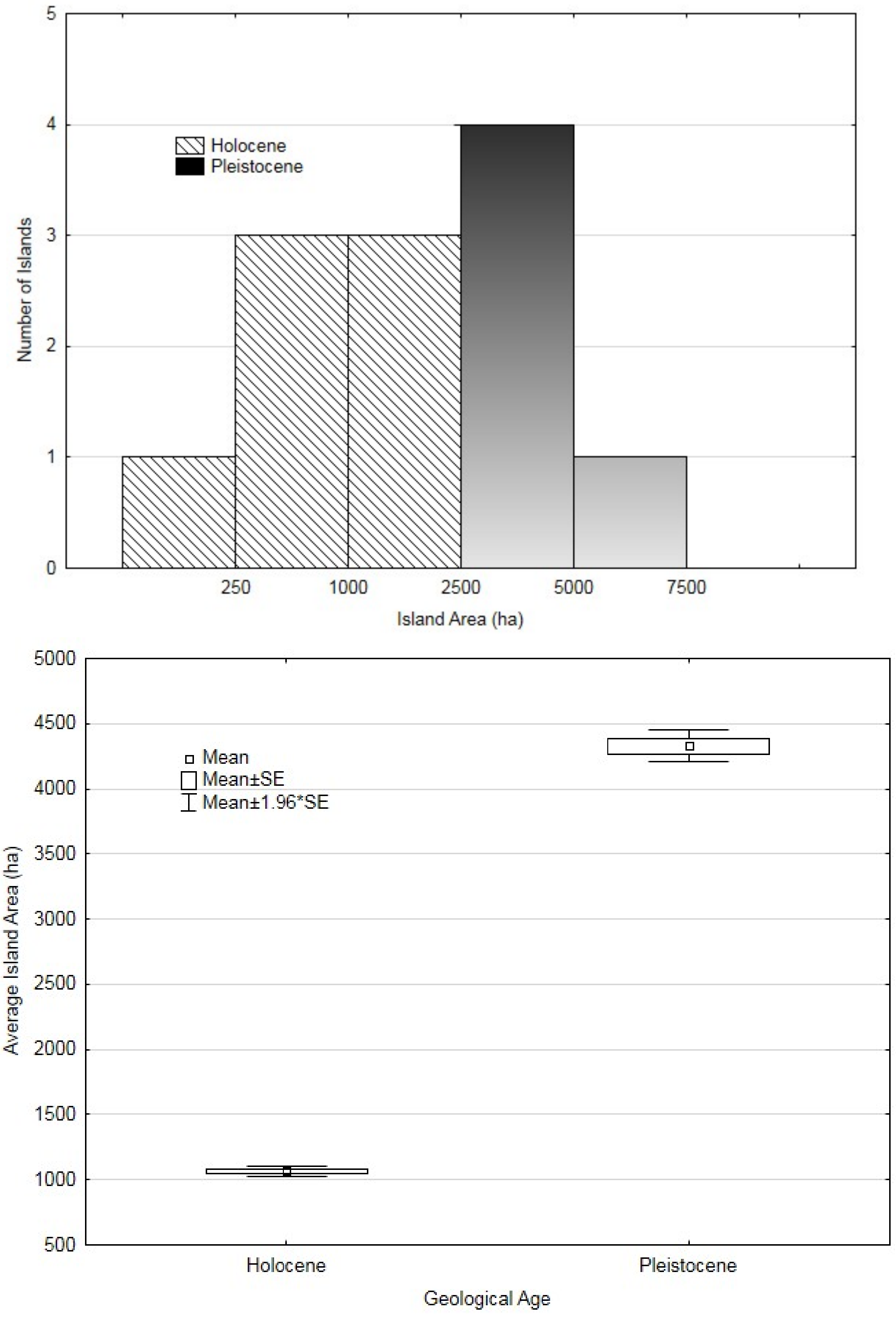

Figure 3-3. Histogram (top) and box and whisker plot (bottom) of island area (hectares) categorized by geological age (Holocene or Pleistocene). Holocene islands formed by sedimentation within the past 10,000 years before present. Pleistocene islands were once connected to the mainland, but isolated 10,000 to 15,000 years before present by rising seas.

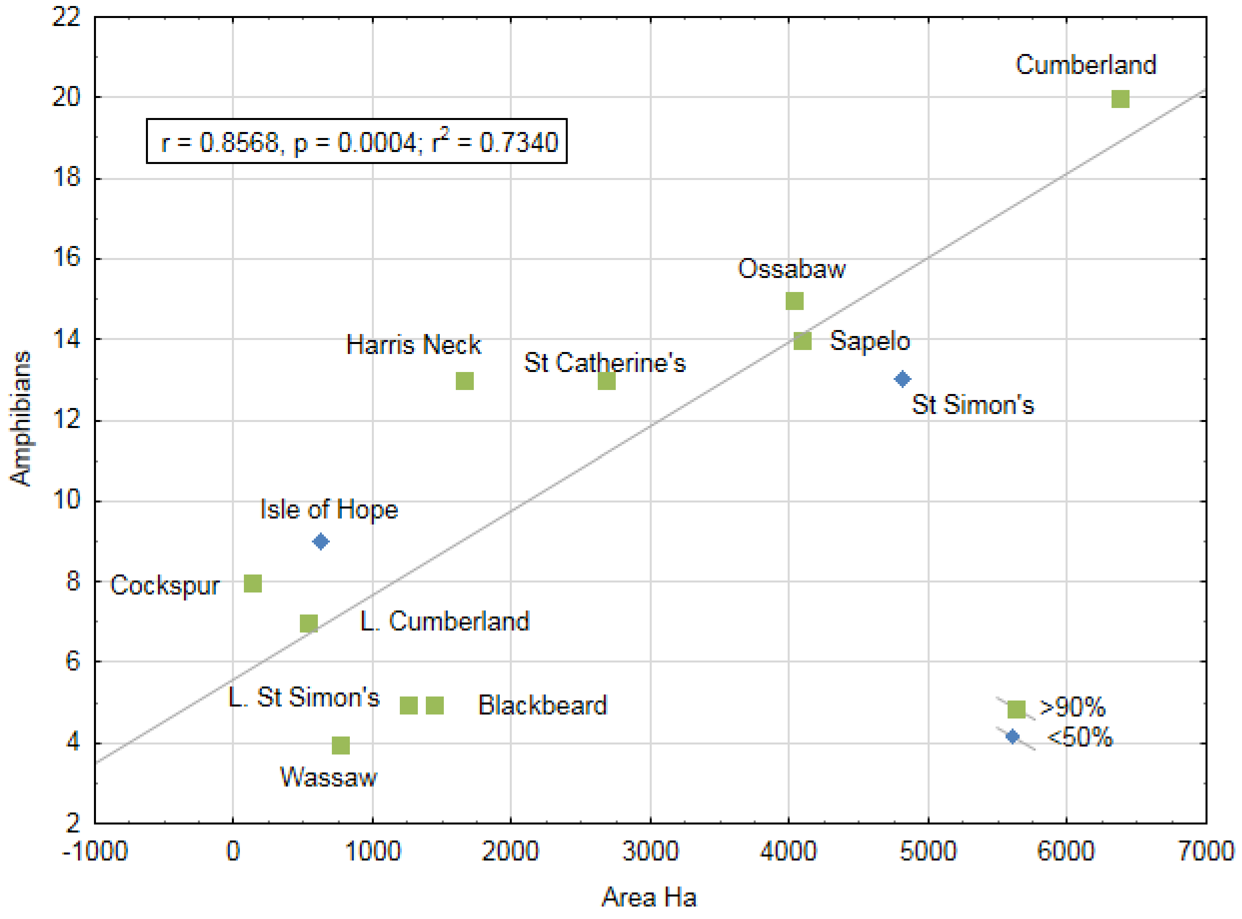

Figure 3-4. Scatterplots of island size and species richness, coded by percent of island that is currently undeveloped. Harris Neck is the largest Holocene island, and St Catherine's is the smallest Pleistocene-core island. Development tends to be low to moderate density housing (2 to 5 units per acre).

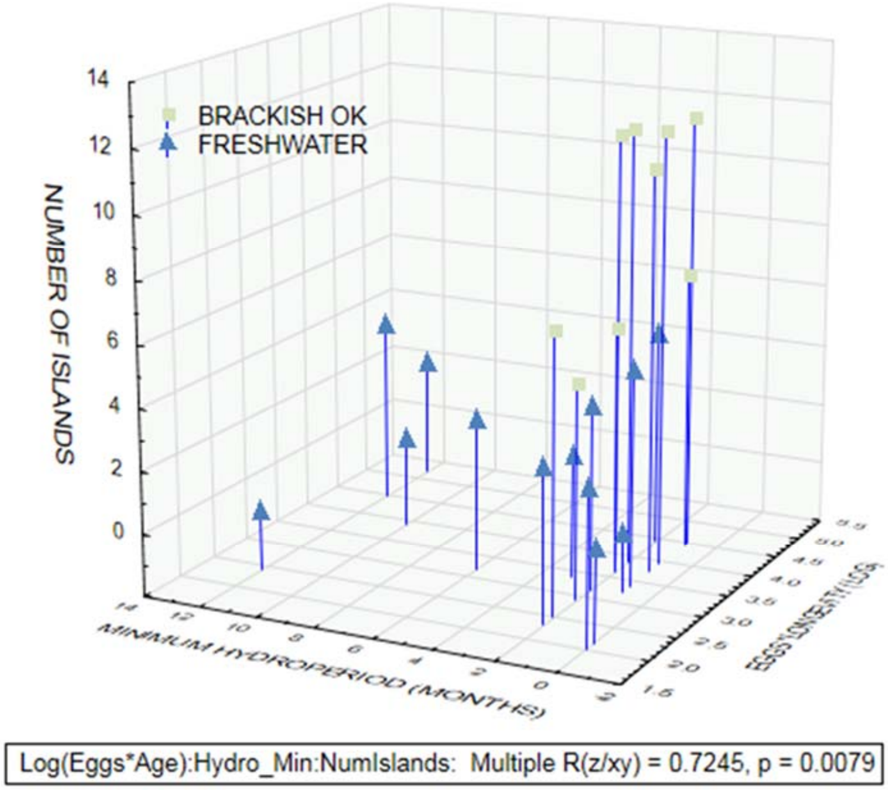

Figure 3-5. Three dimensional scatter plot of minimum hydroperiod (months; left axis), reproductive output (# eggs laid * longevity in years; right axis), and number of islands species occurred upon (up axis). Points classified by ability of adults to tolerate brackish water.

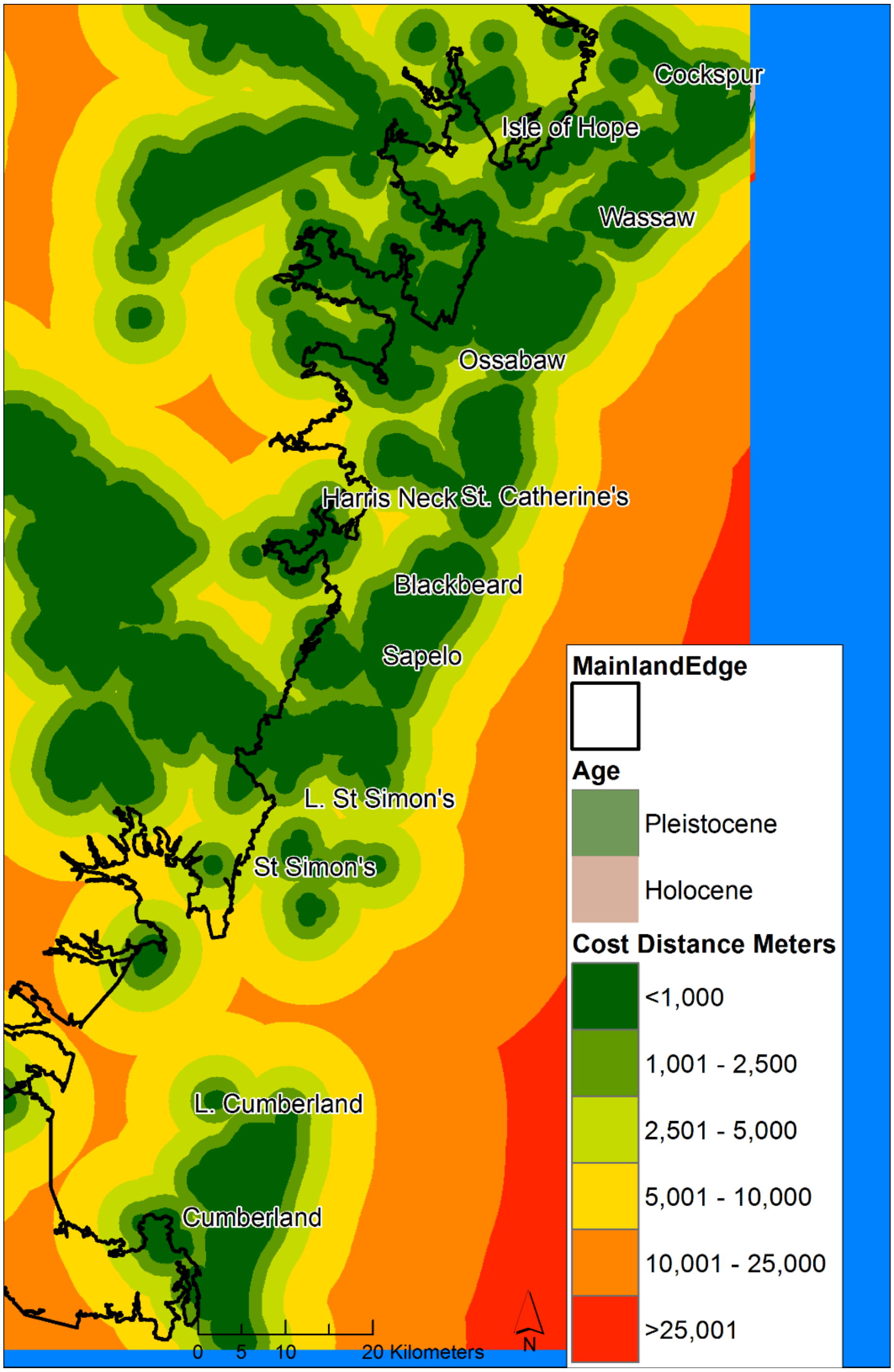

Figure 3-6. Euclidean (nearest neighbor) distance between islands.

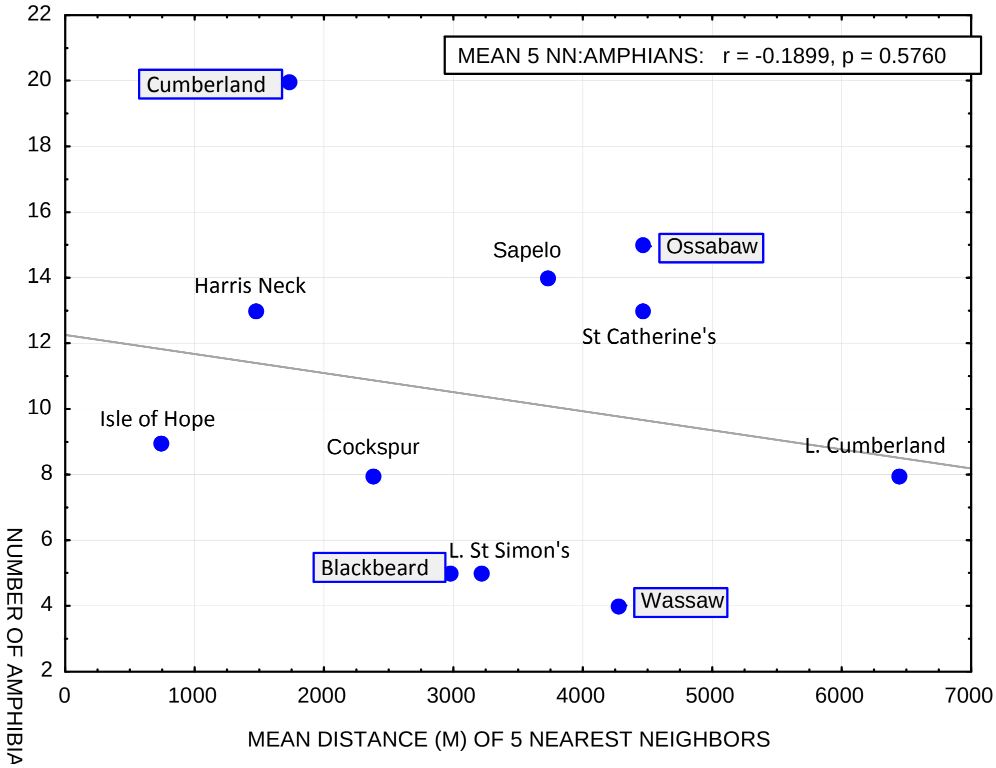

Figure 3-7. Scatterplot of mean distance between five nearest neighbors and amphibian diversity.

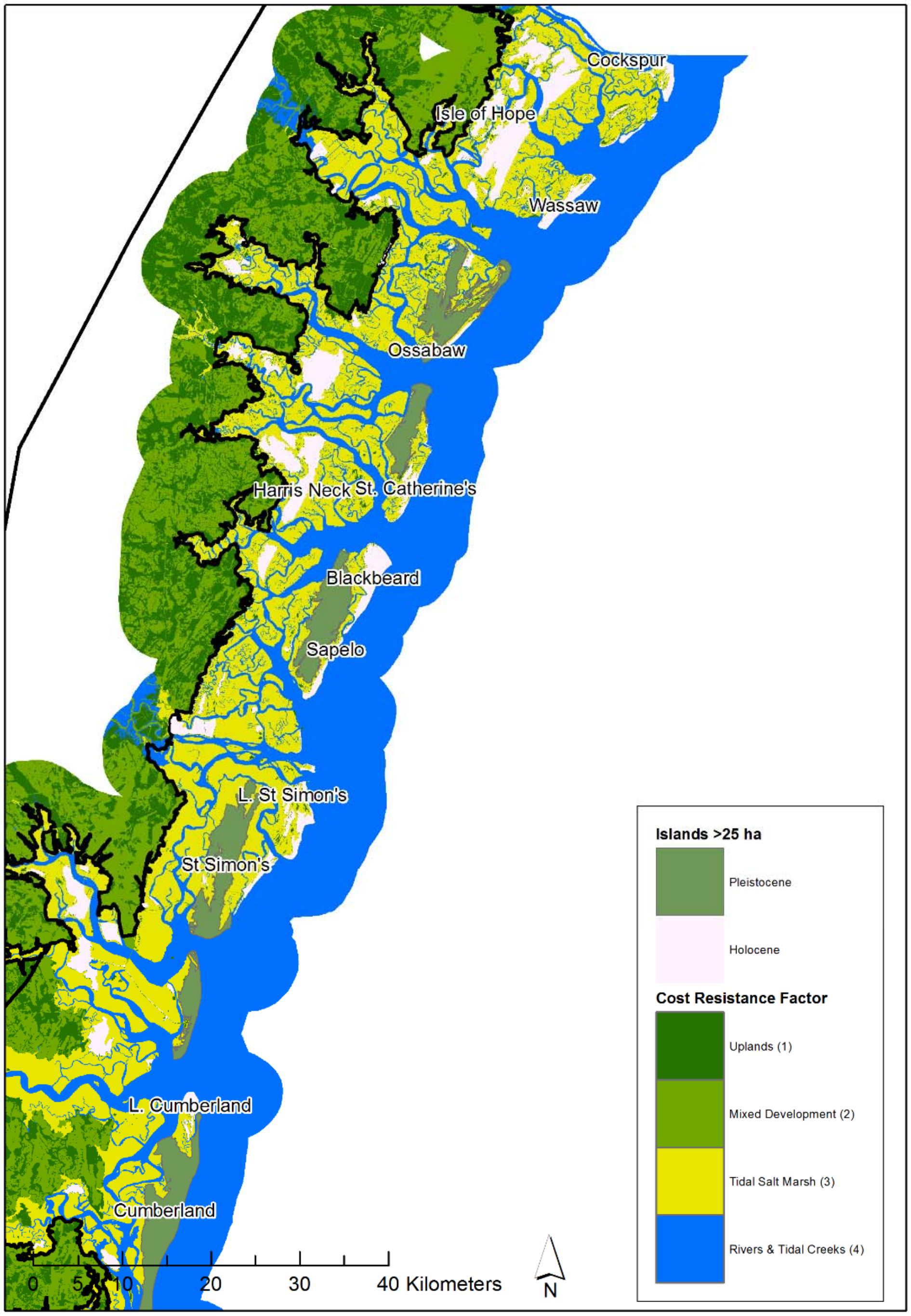

Figure 3-8. Cost resistance factors assigned to habitat types. Natural habitats were given a value of 1, representing no resistance. Maximum cost of 4 was assigned to permanent rivers.

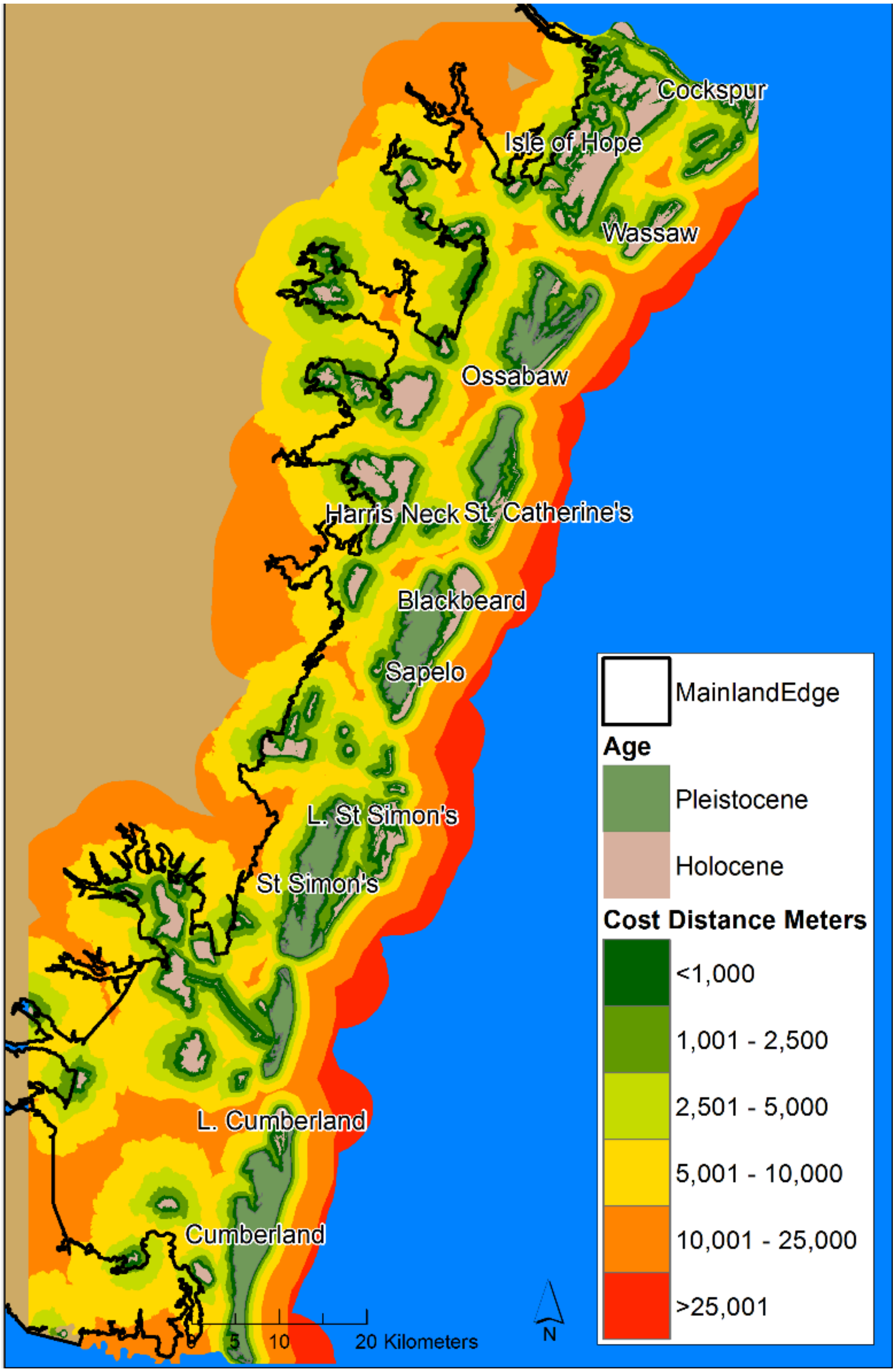

Figure 3-9. Distance factored by cost to travel across habitat types for the Georgia Sea Islands region.

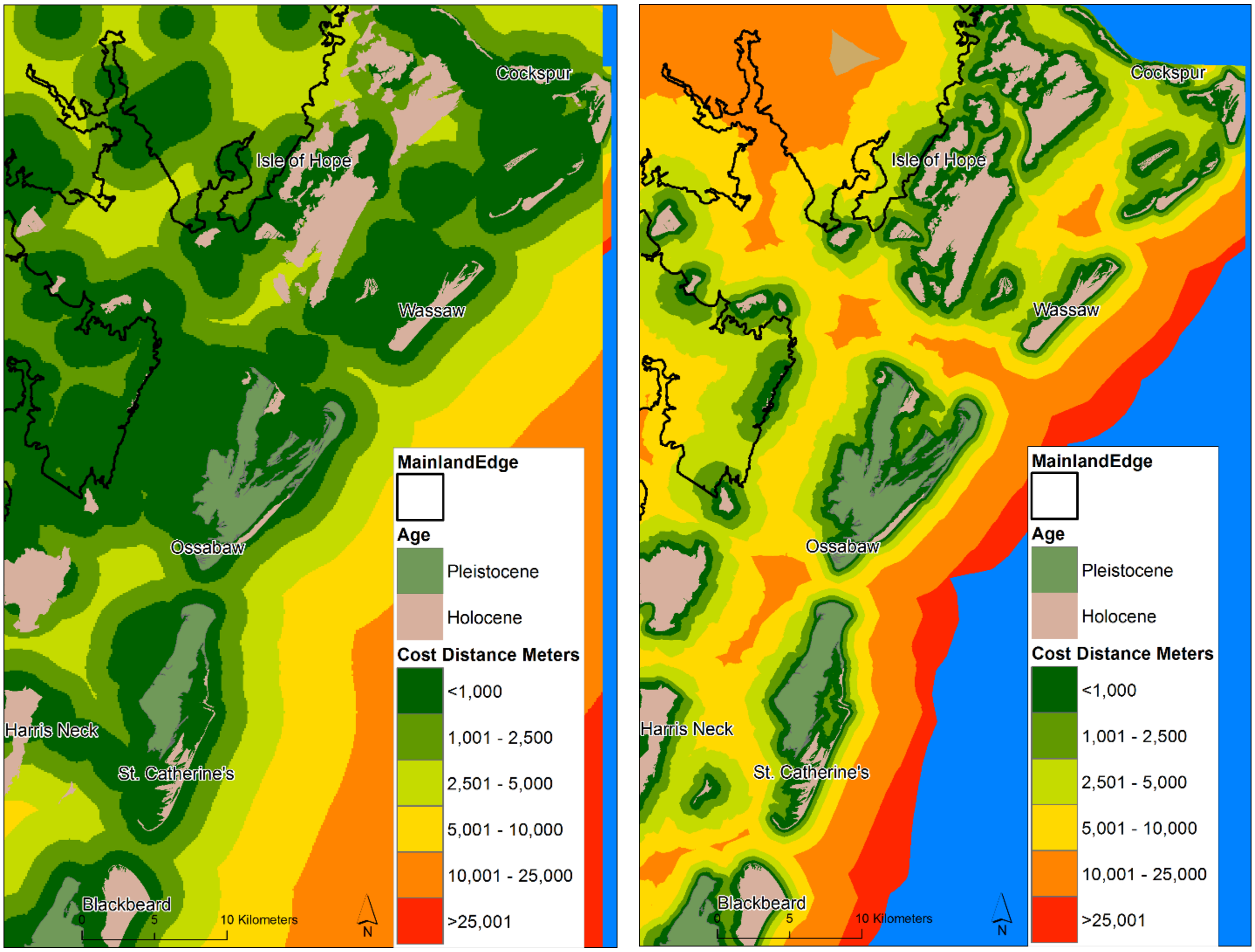

Figure 3-10. Comparison of Euclidean distance (left) and distance factored by cost to travel across habitat types (right) for the northern portion of the Georgia Sea Islands region (Cockspur to St Catherine's).

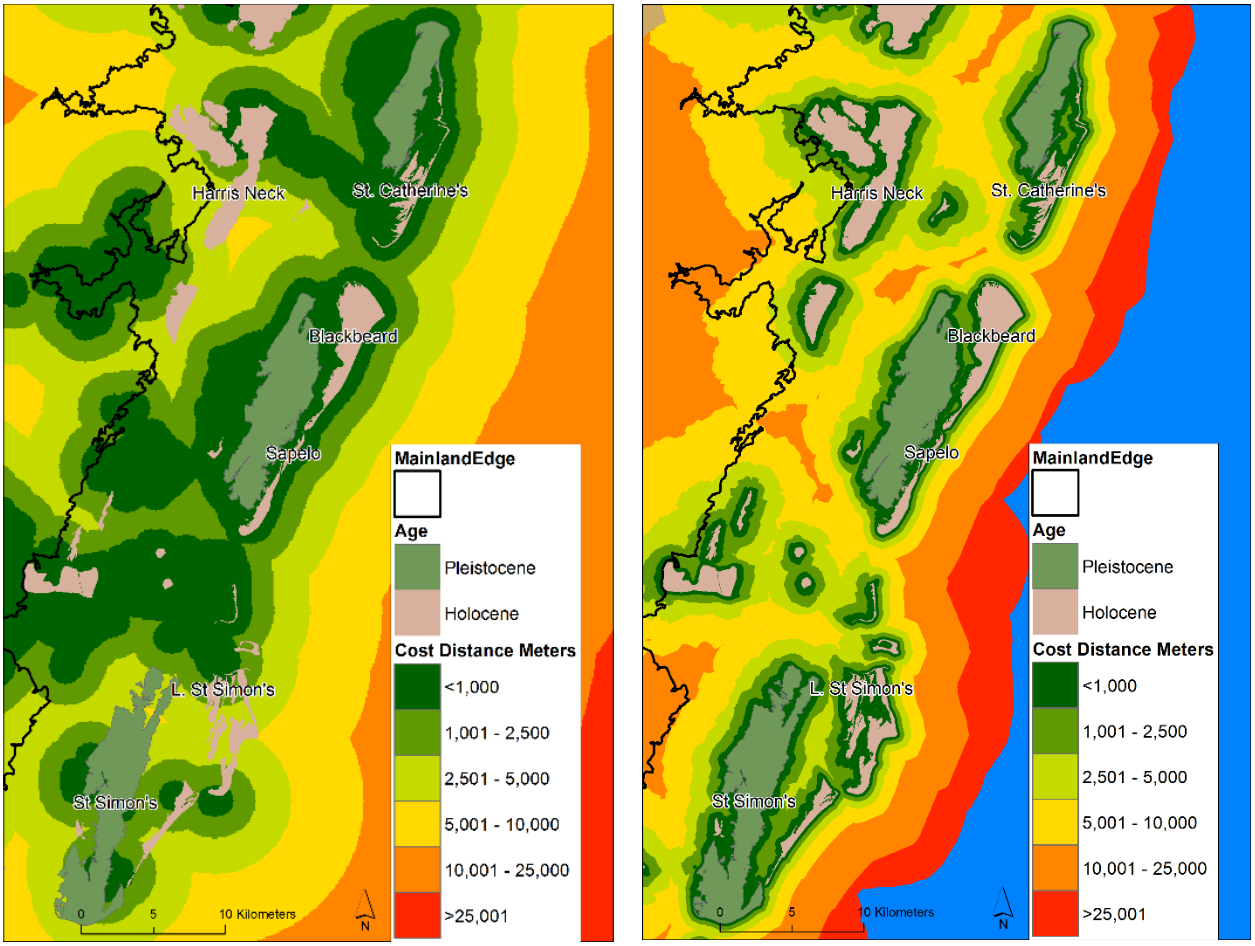

Figure 3-11. Comparison of Euclidean distance (left) and distance factored by cost to travel across habitat types (right) the central portion of the Georgia Sea Islands region (St Catherine's to St Simon's).

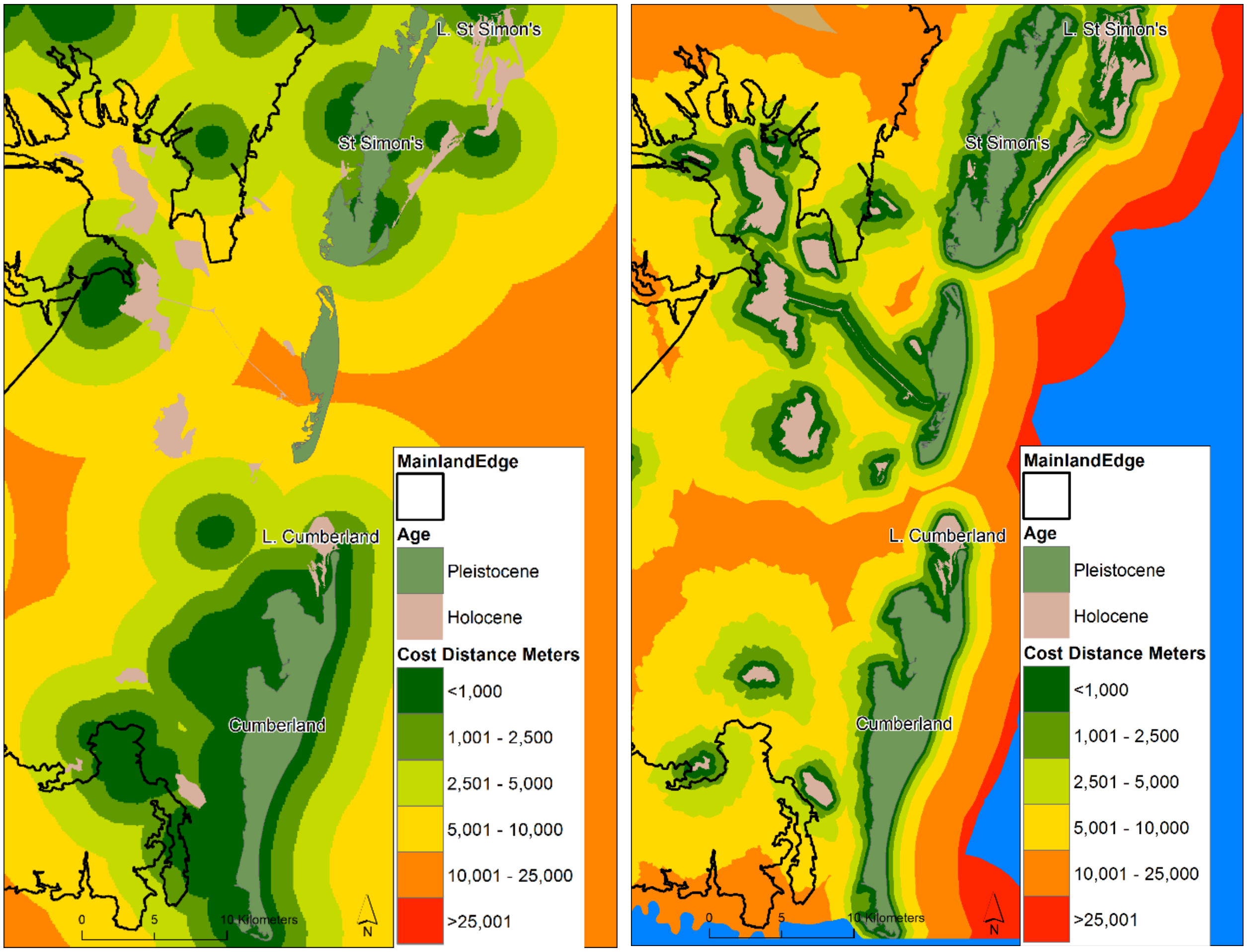

Figure 3-12. Comparison of Euclidean distance (left) and distance factored by cost to travel across habitat types (right) southern portion of the Georgia Sea Islands region (St Simon's to Cumberland).

CHAPTER 4 LIDAR DETECTION OF MICROTOPOGRAPHIC FEATURES IN A FORESTED LANDSCAPE WITH LOW RELIEF AND EXTENSIVE LAND-USE HISTORY[10]

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[10] O'Hare, N.K., T.R. Jordan, M. Madden, and L. Mu. To be submitted to *Frontiers in Ecology and the Environment*

ABSTRACT

This paper describes detection accuracy of microtopographic features from lidar data collected over a semi-deciduous hardwood forest with low topographic relief. Three of the four microtopographic features of interest in the study area were characterized by lack of vegetation growing within them, although trees branches or palm leaves may overhang. Canals that were >1 m deep could visually be reliably detected and classified. We were able to detect small canals and unpaved roads having relative elevation difference (0.16 m and 0.15 m, respectively) less than the reported vertical error of the DEM (0.24 m). However, the overall accuracy of individual point values extracted from the DEM had known error in 45% of the canal points (e.g. canals at same elevation or higher than the surrounding ground surface). While we were able to characterize ground structure, there was still difficulty in tying hydrological processes to structure. The difficulty stemmed from lack of surficial ponding of rainwater seasonally during three years of study, the small number (18) and clumped distribution of surficial water gages along the Georgia coastline, and the confounding effect of tidal saltwater influx. Tidal influx was related tidal stage at nearby by gage; analyses indicated that tidal influx would occur approximately 25 to 50 times per year, coinciding with the biweekly high tides. Overall, the lidar-derived DEM captured linear microtopographic features and depressions in forest areas, but tying structure to hydrological processes with not possible.

INTRODUCTION

Digital elevation models (DEM) of ground surface topography are fundamental to diverse applications (Sensing 2007). In the United States of America (USA), a national, seamless elevation data set has been produced by the United States Geological Survey (USGS) since the early 1990s, and updated regularly to reflect both changing land surface and technological advances which improve DEM accuracy (Davidson and Miglarese 2003, Gesch, Oimoen and Evans 2014). The development of light ranging and detection (lidar) over the past 20 years was a technological shift in generating DEMs, allowing production of DEMs with vertical resolution ranging from 15 cm to 1 m rather than 3–10 m. Lidar derived DEMs are being integrated into the USA national DEM as data become available . The ability to detect finer scale, microtopographic features using freely distributed lidar offers new opportunities for the diverse applications of DEMs.

Wide scale lidar DEMs initially were restricted to coastal areas, where flooding and sea level rise were joint concerns. Coastal Georgia has a 2.5 to 3.0 m diurnal tidal range over a relatively flat topography. Consequently, there are extensive tidal marshes and creeks surrounding islands that tend to be less than 4 m above mean sea level (MSL) and generally less than 9 m NAVD 1988[11]. In coastal marshes typical of Georgia, dense graminoids (e.g. grass like plants) <2 m are the dominant species and the uppermost vegetation stratum. Dense graminoids typical of these coastal marshes limit penetration of lidar (Hladik and Alber 2012).

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[11] Tidal elevations and ground elevations reference different "0" datums. Tidal elevations reference tidal stages, usually Mean Lower Low Water. Ground elevations, including NAVD 1988 and NGVD 1927 datums, tend to be related to mean sea level, which is recalculated every 18.6 (19) years in the USA. In the Georgia Sea Islands, the offset between Mean Lower Low Water datum used to report tidal stage and NAVD 1988 is approximately 1.2 m. Fort Pulaksi is used since it has longer period of record, and produces verified records of stage, not just predictions of stage.

Lidar DEMs still have issues in characterizing small creeks (Chassereau, Bell and Torres 2011). In forested areas, the forest canopy has received more attention, but vegetation <2 m also presents challenges. Custom algorithms (Schmid, Hadley and Wijekoon 2011) or species specific correction factors (Hladik and Alber 2012) can improve the resultant DEM. But many end-users may lack the technological ability or location-specific vegetation data to produce custom DEMs.

The original focus was detection and delineation of small (<0.5 ha) natural depressions that would support seasonal ponding of freshwater, and consequently provide amphibian breeding habitat. Seasonal ponding would occur either by rainwater or groundwater in-inflow from surrounding upland. Generally, small seasonal wetlands are not captured in LandSat (30 m pixel size) derived National Wetland Inventory (NWI) maps or in non-lidar DEMs, such as the 3 m and 10 m National Elevation Dataset (NED) (Martin, Kirkman and Hepinstall-Cymerman 2012), and can be difficult to detect from air photos (Carpenter, Stone and Griffin 2011). However, lidar-derived DEMs have been successfully used in other areas to detect small, seasonal ponds in other forested areas (Burne and Lathrop 2008 Wang, 2005 #2587). But unlike these other areas, the Coastal Plain physiographic region of the southeastern USA has lower overall topographic relief (0 to 4 m) and no true leaf off season since many hardwood trees are evergreen or semi-deciduous.

More importantly, it became evident that description of depressions from the DEM was inadequate to determine present-day hydrological function. Some sites had adult trees favoring more than six months of surficial freshwater, indicating a longer hydrology in the recent past than occurs today. Surficial water was observed, and amplectant (mating) pairs of toads were observed in two of the seasonal ponds in March 2012, but pairs moved in and out of the water, an atypical behavior, and no egg masses were laid. Water salinity from tidal influx along canals was then investigated as a potential issue limiting amphibian breeding. Therefore, the original scope of study was expanded to include canals.

The DEM signature of some canals and one-lane unpaved roads sometimes overlapped. Consequently, unpaved roads were included as ancillary features to improve detection of canals rather than as a feature with a direct hydrological impact. Initial field surveys indicated that each of these features had a vertical difference of 0.5 to 2 m from the surrounding areas. None of these features would be evident on non-lidar derived DEM or imagery.

We predicted that seasonal ponds would be evident on the DEM, evident by sharpness in elevation change from the surrounding uplands. Canals having >1 m elevation difference from surrounding areas would have higher detection accuracy on the DEM than features with <1 m difference in elevation from surrounding areas, regardless of amount of overstory vegetation. Shallow, narrow canals could be separated from unpaved one-lane roads by ground elevation. Still sites would not be evident on the DEM despite their depth since their small size would be smoothed out in processing DEM for general purposes.

METHODS

Study Area and Features of Interest

The study area was the portion of Wormsloe State Historic Site located on the Isle of Hope, in Chatham County, Georgia, near Savannah (Figure 4-1). The boundaries of the historic site spans three island (southern half of the Isle of Hope, Long and Pigeon Islands in their entirety), as well as the intervening tidal marshes. The portion of Wormsloe on the Isle of Hope was the only site easily accessible, and is the largest.

There were four microtopographic features of interest; one was natural (seasonal ponds) and the other three anthropogenic (still sites, canals, and unpaved roads). In this area, seasonal ponds occur in shallow natural depressions in upland forests with a mixed canopy of dominated by deciduous (*Liquadumbar styraciflua*, *Liriodendron tulipifera*), tardily or semi-deciduous (*Q. alba*, *Q. virginiana*), and evergreen (*Magnolia virginiana*, *Pinus elliottii*) trees and an understory including many evergreen palms (*Serenoa* and *Sabal*), and shrubs. Seasonal ponds were evident in the field by a shift to more open canopy of mesic to hydric trees (*Sabal palmetto, Liquadumbar*, *Acer rubrum*, *Taxodium distichum*) or had complete lack of trees. The transition zone from upland to wetland could be distinct (occurring over <1 m; Figure 4-2) or more gradual (occurring over several meters). Field determination of anthropogenic features (e.g. still, canals, and unpaved roads) was relatively simple, without ambiguity. Still sites were characterized by a deep pit, with numerous large, rusting metal barrels, an assortment of older glass bottles strewn around, and sometimes remains of brick or concrete which would have been used to contain the heat source (e.g. fire) required for distillation (see Figure 4-2). Canals in the region present as either deep (up to 1.5 m) and wide (1 to 2 m) with well defined, steep sides, or as shallower (0.5 to 1 m) and narrower (<1 m) with sloping sides (Figure 4-3). Roads were unpaved, 2–3 m wide, and obvious by lack of vegetation (see Figure 4-3).

Lidar Point Cloud and DEM

We acquired processed lidar point cloud data in ASPRS 1.1 LAS format and the associated DEM directly from Chatham County, Georgia GIS in October 2011. The lidar data were collected February 21 – March 22, 2009 by Sanborn Map Company under contract to the Coastal Georgia Elevation Project, with the purpose to generate a high resolution DEM for coastal flood modeling. The sensor was a Leica ALS-50, with a wavelength of 1064 nm, scan frequency of 44.5 khz, up to 4 returns recorded per pulse, scan angle of ±20° off nadir, and flight altitude of approximately 1,100 m above ground level. The wavelength of the sensor, 1064 nm, does not penetrate water, which was a major concern for data collection over low-lying coastal areas of Chatham County. Consequently, flight times over coastal marshes were restricted to ±2 hours of low tide to minimize the effect of tidal inundation. Each point had an associated x, y, z, and intensity value. The coordinates were in State Plane Georgia East FIPS 1001, referencing the NAD 83 horizontal datum and GRS vertical datum; units for x, y, and z coordinates were in US survey feet. Raw lidar data were post-processed by Sanborn Map Company using Leica ALS post-processing software, proprietary filters, and visual inspection as necessary to categorize each point as unclassified (e.g. vegetation, building), ground, noise or water.

The DEM of ground surface elevation was a raster of 1 m (3.048 ft) pixel size (Figure 4-4). The original x, y coordinates were in State Plane Georgia FIPS 1001, with x, y, and z units in US survey feet, like the lidar point cloud. To maintain accuracy of the original data set, analyses were performed without geographic transformation between coordinate systems and unit. Consequently, units are given first in feet, followed by metric equivalent.

Feature Delineation from GPS and from DEM

Features were delineated using both global position system (GPS) and heads-up editing and, or digitizing using the DEM as a reference. The DEM was used to guide field work to assure that all seasonal ponds and canals apparent on the DEM were verified on the ground. To assure detection of features that may not be evident on the DEM, the entire area of Wormsloe was walked on a regular 100 m grid both in north-south and east-west direction, as well as more than 10 random walks.

GPS data were collected using either a Trimble GeoXh or Trimble 6000 (both with 2 to 5 m accuracy), with location data collected only if maximum positional dilution of precision (PDOP) was lower than 6. For depressional features, the ecotone between wetland and upland vegetation based upon field indicators of hydrology was walked with GPS location collected every 3 seconds as streaming line feature. Separate line features were collected for sharp (<1 m) versus gradual ecotones (several meters); for gradual ecotone, the approximate center was walked. Separately, the extent of depression features, evident on the DEM as a distinct change in elevation, were digitized in ArcGIS to compare field determination to the DEM indicators. All The lines representing depression edges were converted to polygons and polygon area determined in ArcGIS. For still wells, a point feature was collected by the averaging 30 positions taken while standing stationary in the center of still well pit. Maximum dimensions and depth measured to the nearest 10 cm. For canals, GPS data were collected as line vertices near the centerline either approximately every 100 m or at a bend, whichever occurred first. Canals were categorized as large (more than 1 m wide and more than 1 m deep) or small (less than 1 m wide or less than 1 m deep regardless of width). Unpaved roads were collected as streaming line features while driving at a slow speed (<10 mph) in a golf cart, with the unit positioned so that it was near the center line of the road. A GPS location was recorded every 1 second. While GPS location data was collected with the intent to post-process to improve accuracy, base station data was missing for some dates. Consequently, GPS data were not post-processed, and resultant GPS location accuracy was 2 to 5 meters.

Heads up digitizing in ArcGIS was used to make minor corrections to GPS delineation of canals, and roads (±3 m) so that the feature coincided with the DEM signature of the feature. Editing and digitizing was done at a scale between 1:500 and 1:1,000. Finer scales (<1:500) made it difficult to separate feature pattern from pixel level variation, while broader scales (>1:1,000) were likely to increase digitizing error. The linear nature of roads and canals in particular facilitated extrapolation across pixilation in the DEM surface. We felt confident making these manual edits for two reasons. Firstly, the GPS data and the DEM each had some error in horizontal location (2–5 m for GPS and <50 cm for DEM). Secondly, and more relevant, the purpose was to determine ground surface elevation of features, not GPS accuracy in delineating features. Edits were not made to GPS of seasonal pond edges' because there was a wider discrepancy, sometimes more than 5 m, in the edge delineated in the field and the DEM signature. Consequently, heads-up digitizing of perceived pond edge from the DEM was digitized based upon slope changes and compared to extent determined by GPS in the field.

Determination of Ground Elevation of Features

Both the absolute ground elevation and elevation relative to the surrounding land surface were calculated from points placed along the edited GPS data delineating features. Using the ArcGIS tool to create points, points was placed every 10 m along line features; for features less than 10 m, 1 point was randomly placed. The elevation for each point was then extracted from the DEM (1 m grid cell size; Figure 4-5). Average elevations by feature type were calculated from the point data to determine if features occurred within an elevation range. To determine difference between feature elevations and surrounding ground surface elevations, first the ArcGIS tool to create parallel was used to create lines parallel to the linear features at a distance of 10 m, and then ArcGIS tool to create points every 10 m along these parallel lines was used. Elevation of these parallel points was extracted from the DEM (see Figure 4-5). Relative elevation was calculated by subtracting point elevation of the point within the feature from the elevation of the two points on either side.

Accuracy Assessment

To calculate errors of omission in canals and road feature detection from the DEM, the points placed every 10 m along the linear feature (canal or road) were categorized as either visible (DEM accurate) or not visible (DEM error) at scale of 1:500 to 1:1,000.

Tidal Inundation and Affects on Soil Salinity

Direct observation of tidal influx were made to determine the approximate tidal stage recorded at Fort Pulaski National Monument (NOAA gage 8670870; data of observed stage downloaded from http://tidesandcurrents.noaa.gov) that resulted in tidal flow inland along canals at Wormsloe. The number of days each year that tidal stage exceeded the stage correlated with tidal inundation was calculated.

Separately, salinity of the five depressions near sites of herpetofaunal trapping were measured (see Figure 4-1). If there was surface water, salinity was measured using Extech RF20 salinity meter. If site was dry, soil samples were taken, samples were rehydrated using 1 part soil to 5 parts distilled water, stirred for 30 seconds, then allowed to settled until water fraction was clear (usually ~15 minutes). The salinity of the water fraction was then measured using Extech RF20 salinity meter.

RESULTS

Features were delineated first and accuracy *a posteriori* related to characteristics of the lidar point cloud. However, in presenting results, characteristics of the point cloud density are addressed first, since it may have affected feature delineation and subsequent elevation attributes.

Lidar Point Cloud Attributes

Within the boundaries of the study area, point density of all returns was 0.63 points per square foot (0.06 points per square meter), with spacing of 1.26 ft (0.38 m). Point density for ground returns only was 0.23 per square foot (0.02 per square meter), with spacing of 2.10 feet (0.64 m). RMSE was determined from 20 survey control points in flat open terrain distributed throughout Chatham County. Horizontal accuracy was processed to meet 1 m accuracy; RMSExy was not calculated since it varied with point density, which was influenced by scan angle and laser beam divergence. RMSEz with 95% confidence of 0.78 ft (0.24 m).

Feature Detection and Verification

All deep ditches and roads were delineated in the field; some shallow ditches may have been missed in areas with dense understory. There were 5.8 km of deep ditches in 19 segments, 8.3 km of shallow ditches in 43 segments, and 18.3 km of roads delineated (Figure 4-6). Deep canals were longer than shallow canals (mean 308 m ± 66.1 s.e. and 188 m ± 39.3 s.e., respectively). Seventeen of the 19 deep canals connected directly to tide or fed directly into another deep canal that connected to tide; the two segments that lacked connection to tide or other canals were associated with the embankments built for automobile races in the early 1910s. Shallow canals connected to a either a deeper canal (23), another shallow canal (7), directly to tidal marsh (13), or into a natural depression (1). Deep canals had a lower average elevation than shallow canals and unpaved roads (Figure 4-7). However, histograms of the point elevations showed significant overlap, suggesting that elevation alone was not sufficient to categorize a feature (Figure 4-8). For the deep canals that connected directly to tidal marsh, ground elevation was plotted versus distance from the marsh. The deep canals increased in elevation as they moved away from marsh, but then decreased again in elevation as moved inland toward a natural depression (Figure 4-9). Traditional ground level survey confirmed this pattern, but actual height values and degree of slope differed between the DEM and ground level surveys (Figure 4-10).

All features showed a significant change in elevation from the surrounding ground surface, but relative change in elevation was different for each feature type (Figure 4-11). Wider, deeper canals showed greatest change, followed by shallow ditches, and then roads. Segments of features that were not evident on the DEM had smaller average difference between ground elevation in the feature and ground elevation in 10 m buffer (Figure 4-12). However, average calculated differences were less than actual differences measured in the field (Figure 4-13). For example, all large canals were at least 0.75 to 1.0 m lower than adjacent ground surface, and occasionally more than 1.25 m lower, yet average calculated difference was 0.37 m ± 0.01 s.e.).

Fifeteen depressions were evident on the DEM, but only five showed field indicators of hydrology, such as change in vegetation or water marks on trees (Figure 4-14). Some of these others had sparser tree canopy, but dominant species was similar. All seasonal ponds were connected to tidal salt marsh via canal, usually a deep wide canal (Table 4-1).

Table 4-1. Characteristics of five depressional features. “Canal” indicates size of canal and length to saltwater tide. Large canals were 1.5 to 2 m wide and 1 to 1.5 m deep. Small canals were < 1 m wide and 50 to 75 cm deep.

Site	Area (ha))	Canal; distance to tidal marsh	Tidal Influx
Yanxi	0.03	Large; 275 m	Monthly
Bent Tree	0.14	Large; 450 m	Monthly
Palmglade	1.19	Large; 200 m	Weekly
Birdy	0.08	Small; tidal inlet prior to 1970s?; S70 m	Monthly
VP3	0.24	Small; 55 m	Not observed

Seventeen still sites with dug-out wells were located; there were additional sites with barrels only. Still sites were concentrated in the southern end (Figure 4-15). Of the 17 still wells, field measurements of depth relative to ground surface and dimensions were recorded at 8. Depth ranged between 75 cm and 170 cm, with an average depth 114.3 ± 11.3 s.e.. Maximum dimensions ranged between 210 cm and 380 cm, with an average of 283 cm ± 20.6 s.e., and minimum dimensions of 165 to 340 cm, with an average of 233 cm ± 20.5 s.e.. The GPS locations of still sites were tied to the lowest elevation pixels within 2 to 3 m (corresponding with estimate of GPS and DEM error). The average elevation of the bottom of still wells from the DEM was 1.86 m ± 0.13 s.e., with a range between 1.22 to 2.44 m. The average number of pixels on the DEM correlating with still sites was 2.2 ± 0.78, with a range between 2 to 7 pixels).

Tidal Inundation

In August 18 and 19, 2013 (higher high tides of 5.18 and 5.31 ft, NAVD 1988), November 4 and 5 2013 (higher high tides of 4.93 and 5.21 ft) , 2013, and Feb 28 –March 1, 2014 , tide flowing along canals into depressions at three inland depressions (BentTree, Yanxi, Birdy, and PalmGlade; see Figure 4-1) were directly observed. Tides intruded inland along canals when tidal stage at Fort Pulaski exceeded approximately 5.0 ft referenced to NAVD 1988. This tidal stage occurred approximately 24 days per year (Table 4-2), and tended to coincide with the biweekly spring tides. Tidal influx along canals at Wormsloe occurred 1 to 2 hours after the time of high tide stage at Fort Pulaski. There was some variation among sites, since geographical location of canals relative to tidal inlet varied.

Observation and measurements indicated that all tidal water that flowed along the canals and into the depressions did not flow back out. In fact, tide water continued to flow downslope along the canals into the depression. Water accumulating in the depressions eventually soaked into the ground. Both surface water and surface soils tended to be saline. Salinity varied thru time (Table 4-3).

Table 4-2. Number of days per year tides at Fort Pulaski exceeded defined stage (ft, NAVD). Stage of 4.75 to 5.0 ft correlated with tidal flow along canals at Wormsloe.

Year	4.5 ft	4.75 ft	5 ft
2010	51	20	7
2011	48	17	3
2012	53	26	17
2013	88	48	24
2014	86	38	24

Table 4-3. Water depth and salinity in depressions.

	Water Depth (cm)					Salinity (ppt)				
Site	Yanxi	Palm	Benttree	Birdy	VP3	Yanxi	Palm	Benttree	Birdy	VP3
6/4/2012	13	17 cm	Soil	Soil	Soil	24	30	---		
6/5/2012	rain									
6/6/2012	23 to 42	3 to 8	4 to 11	10 to 34	Soil	18	22	15	13	
7/10/2012	Soil	Soil	Soil	Soil	Soil	4	8	4	4	3
10/16/2012	Soil	5 to 10	Soil	Soil	Soil	0	25	0	3	0
11/14/2012	Tide IN	Tide IN	Tide IN	Tide IN	Soil	30	30	30	30	0
3/5/2013	15 to 17	2 to 4	12 to 17	27 to 30	4 to 5	4	13	5	3	0
3/22/2013	16 to 18	1 to 3	5 to 7	9 to 10	14 to 18	10	7	8	3	0
rain overnight										
3/23/2013	24 to 30	8 to 9	14 to 17			3	3	3		
4/3/2013	23 to 40	7 to 10		10 to 13	9 to 12	5	7		4	0
4/20/2013	24 to 30	8 to 11	19 to 21	20 to 23	14 to 16	3	5	3	3	0
6/25/2013	11 to 12	6 to 7	Soil	3 to 4		21	20	4	21	
8/18/2013	19 to 22			18 to 21	15 to 17	4			3	0
8/20/2013	Tide IN			Tide IN						
11/3/2013	Soil						0			
11/5/2013				Tide IN						
2/26/2014	Tide IN			Tide IN						

DISCUSSION

The detectability of microtopographic features was related not only to their own physical dimensions, but also to their context. Still sites were not detectable solely from the DEM, whereas all deep canals were. Both had similar differences below the ground elevation of the surrounding area (1 to 1.5 m). Still sites meet the criterion for detection based upon vertical difference, but their small horizontal dimensions were smoothed out in DEM processing or were represented by 2 or so pixels. Those features that were linear, with greatest difference from the surrounding area were most detectable. Microtopographic features differing from the surrounding ground surface by 15 to 75 cm were evident on the DEM with a reported 95% confidence interval of RMSEz of 24 cm.

Automated methods to extract features by ground elevation alone were not successful since the ground elevations of features significantly overlapped.

While linear features could be detected and delineated, extraction of elevation values of point values suggested error in approximately 45% of points in canals. Histograms of differences between points in canals and points outside of canals frequently indicated that canals were at the same elevation or higher in elevation. Points in canals with either no difference or higher in elevation than surrounding land surface are errors in the microtopographic representation of the DEM.

The context and structure of features can provide clues to their original purpose. Historically, in this region, property boundaries were sometimes delineated with shallow ditches. Since all shallow canals at Wormsloe connected to deep canal, tide or depression, their purpose was more likely drainage. The fact that the study area has been passed down thru direct ancestors since 1735 suggests that there was no need to delineate property boundary. The context and microtopography of large canals provides fewer clues to original purposes. Was the purpose to drain interior freshwater, to lower water table of surrounding areas to enhance suitability for farming, or to provide a catchment for freshwater to be used during periods without rain. Canals do not have a steady slope from inland depression to tidal marsh. Consequently, gravitational drainage of depression downslope to marsh is unlikely. The elevation "hump" detected in four of the deep canals may be derived from more recent siltation. Regardless of what original slope might have been, the slope of canals today means that enforcing hydrology as a flat surface may mis-represent inputs for modeling of hydrological processes. Sluice gates were probably in place to control water flow in the past. But, present-day lack of sluice gates means that tidal influx occurs, and the elevation "hump" means that water drains downslope into depression rather than flowing out as the tide ebbs.

Landscape changes have been correlated with decrease in seasonal ponds (Nuno, Tibor and Jan 2011), and drying of seasonal ponds has been noted for coastal Georgia islands (Vance et al. 2010). The bottom of the still sites were at similar ground elevation as seasonal ponds. Surficial freshwater (e.g. ponding from rainfall) was not observed in any still site during 2010 – 2014, although surficial water in these still sites was observed in 2008 (M. Madden, personal communication). The lack of surficial freshwater in still sites for 4+ consecutive years suggests that the freshwater lens has been depleted. Soil cores to 1.5 m (2 in still sites and 4 in depressions) either reached brackish water or remained dry, rather than hitting freshwater. The freshwater lens is usually within 2 m of the surface in this region (Peck and Payne 2003).

During rainy periods, tidal salt water mixed with fresh rainwater in the depressions, resulting in brackish water (3 ppt to 25 ppt). During dry periods, the depression simply filled with tide water. The soil itself retained salt which leached out when it rained, making rainwater salty (2 to 5 ppt). Indications are that sufficient rainwater will eventually flush the salts downward. Standard estimates are that 15 cm of freshwater are required to flush 50% of salts out of the top layer of soil. However, the frequency of tidal influx re-starts the cycle of salt deposition on soil surface that needs to be flushed by rainwater.

CONCLUSIONS

While microtopographic features may be detectable on the DEM, extraction of point elevation values showed errors at approximately 45% of the points. Microtopographic features of interest in other areas may differ from those in this study of coastal Georgia. However, understanding scale of features that can be predicted from the lidar data set will help in planning future studies. Understanding which features may lack detection will allow researchers to target field verification.

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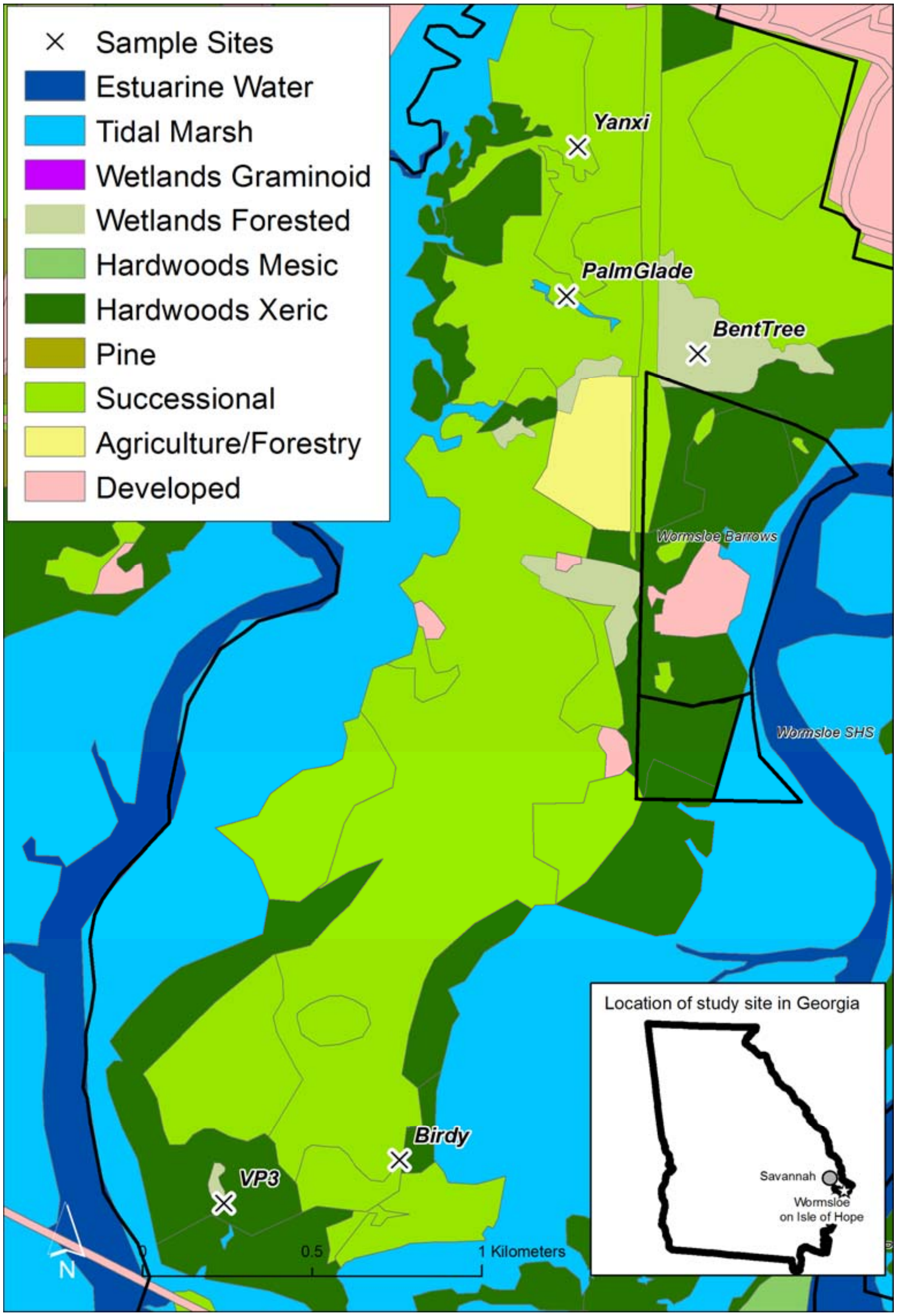

Figure 4-1. Location of the study area and of specific study sites at Wormsloe on the Isle of Hope, Chatham County Georgia, approximately 8 km southeast of Savannah.

Figure 4-2. Photographs of seasonal ponds and illegal liquor distillation sites. Top left: Surface water in depression Birdy. Water was brackish. Note distinct ecotone between vegetation communities, which was typical of natural depressions in the study area. Top right: Same depression as at right, but dry. Bottom left: Still well site (~1.75 m below ground surface) that was dug to provide access to freshwater for illegal liquor distillation. The burrows are active denning sites, not part of the historical still well. Bottom right: Old barrels also typical of still sites.

Figure 4-3. Photographs of canals and roads typical of the study site. Top left: A deep, wide canal, with sharp, clear sides. Canal was approximately 2 m wide by 1.25 to 1.5 m deep. Top right: A shallow, narrow canal, with indistinct sides. Canal was approximately 1 m wide by 50 to 75 cm deep. Water in canal was from tidal influx, approximately 250 m from salt marsh along adjoining large canal (see above). Bottom left: Example of unpaved dirt road. Roads were 2–3 m wide; ground surface relative to surrounding forest was difficult to detect in the field.

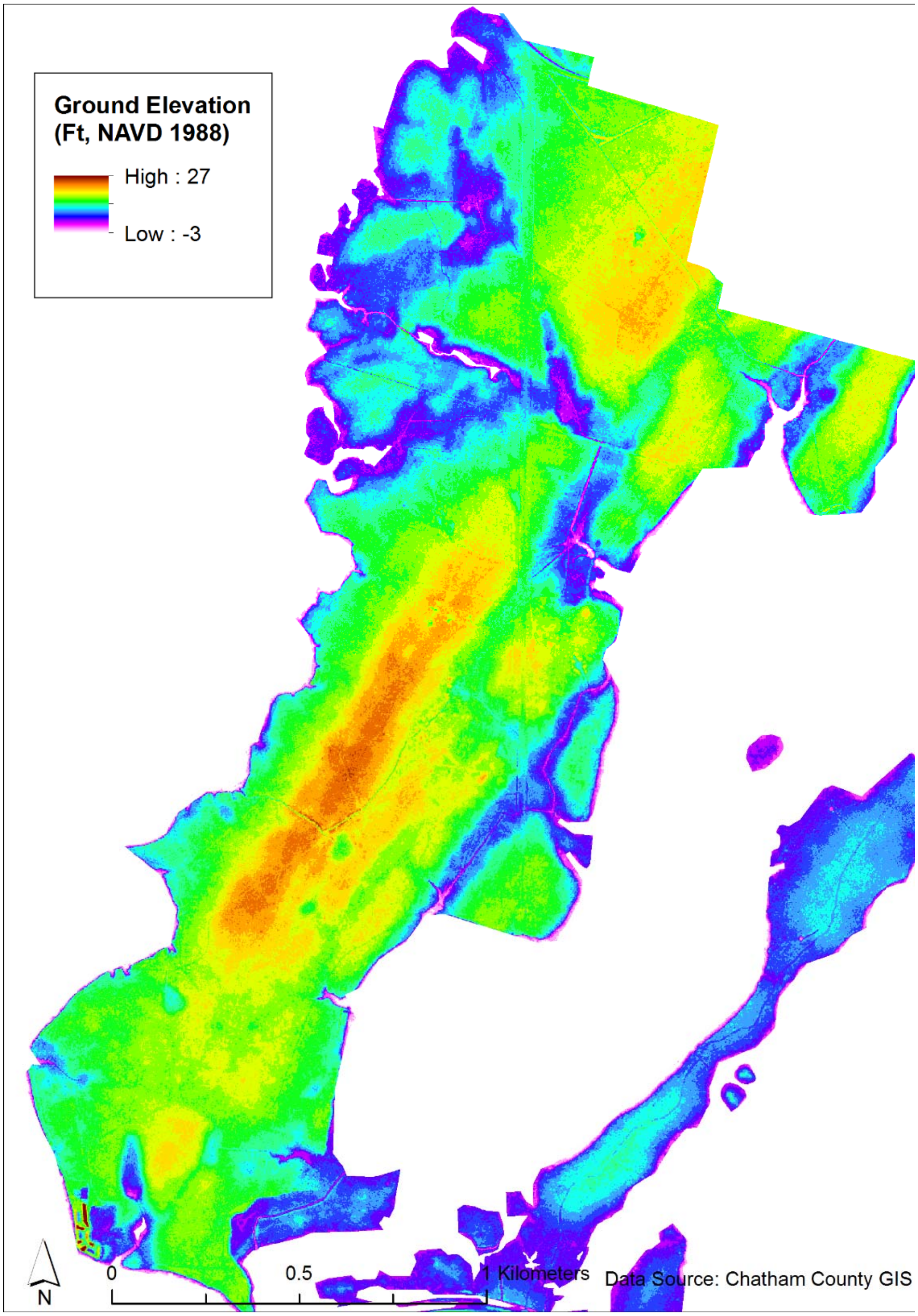

Figure 4-4. Digital elevation model (DEM) of Wormsloe, on the Isle of Hope. DEM generated from lidar point cloud data and has 1 m pixel size.

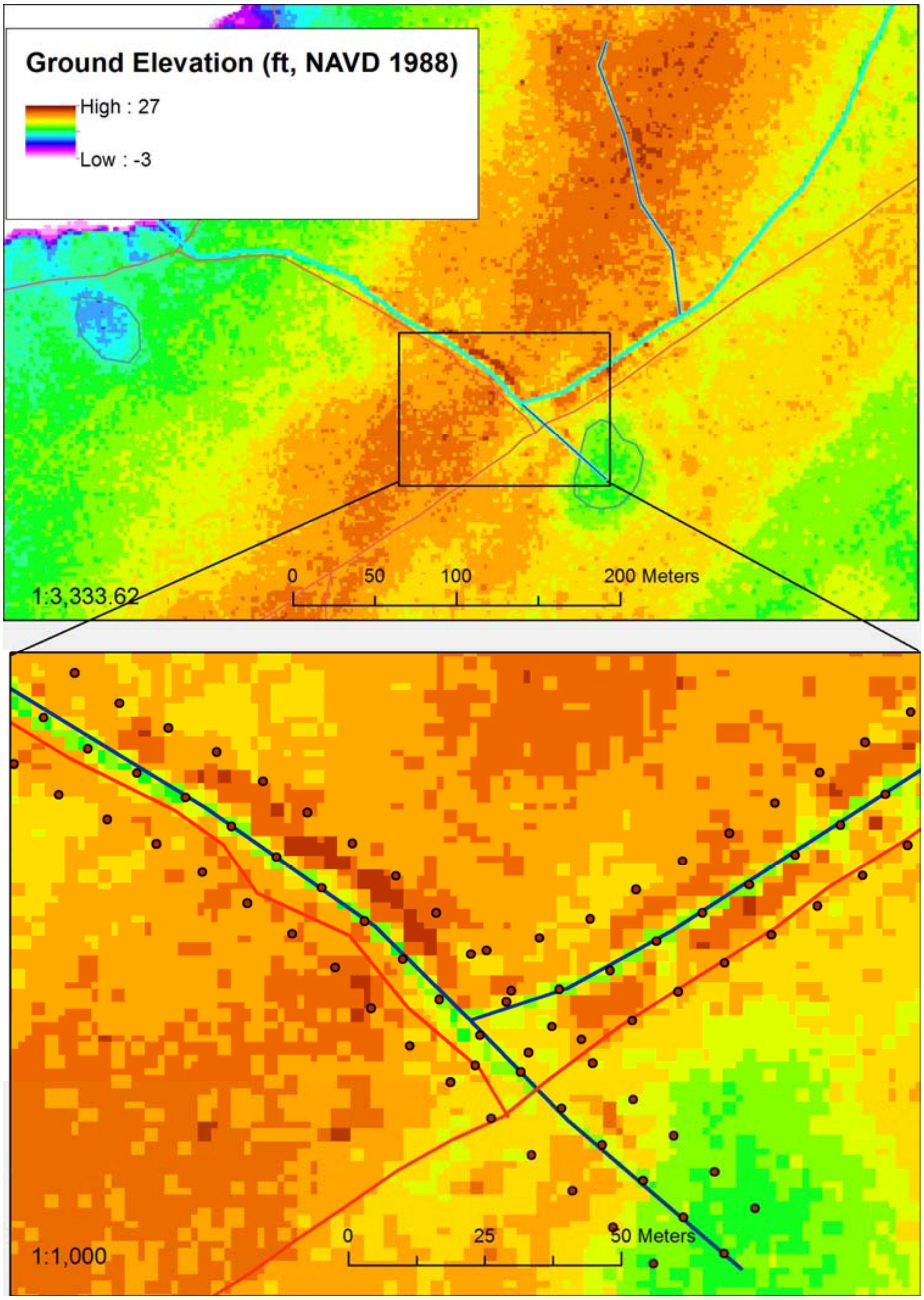

Figure 4-5. Example of point placement to calculate elevation of feature and of the surrounding area.

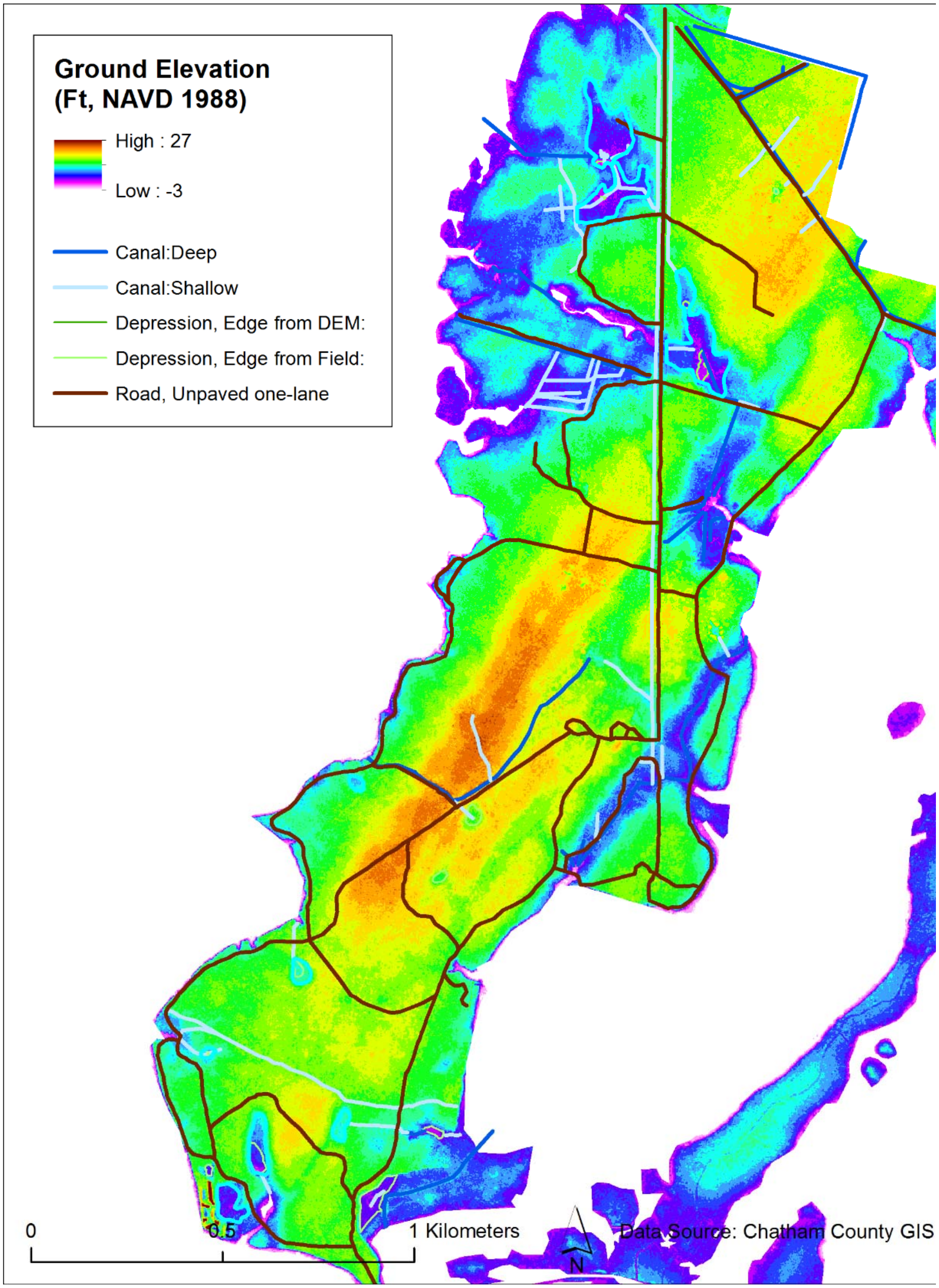

Figure 4-6. Microtopographic features at Wormsloe.

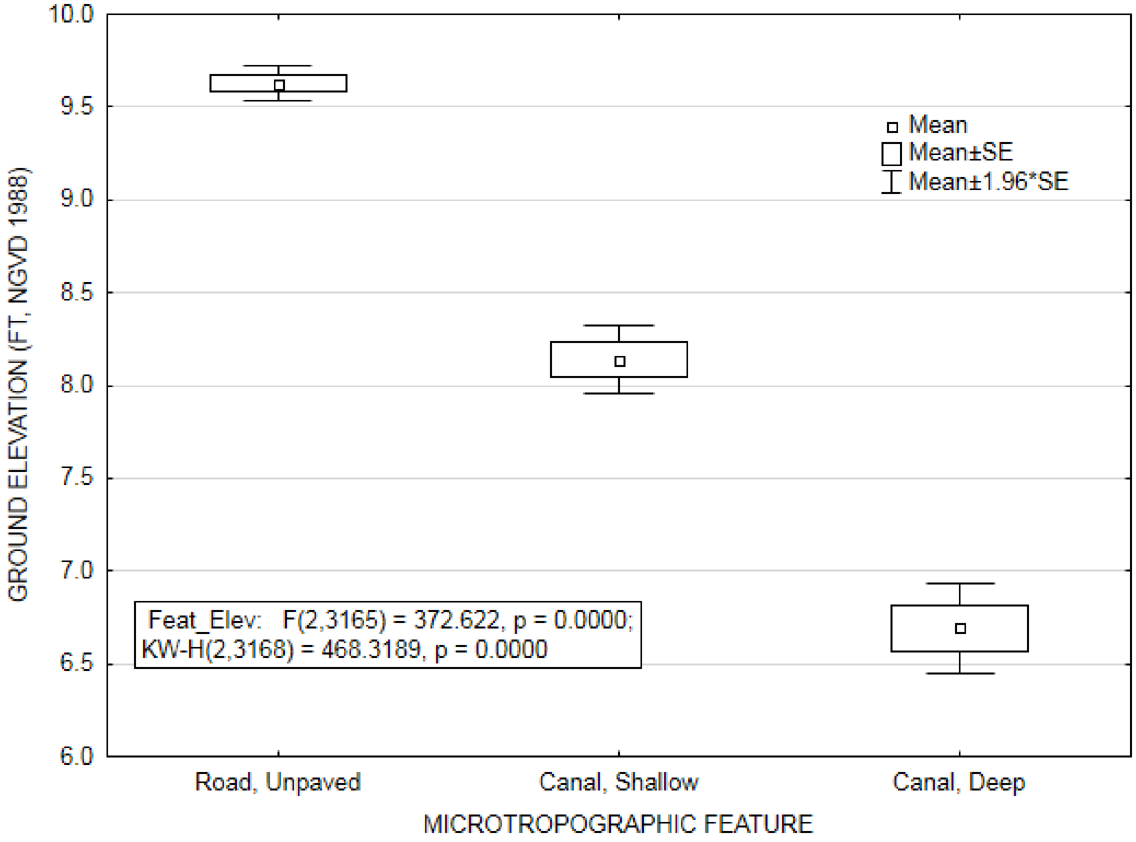

Figure 4-7. Box and whiskers plots of average elevation by linear feature type.

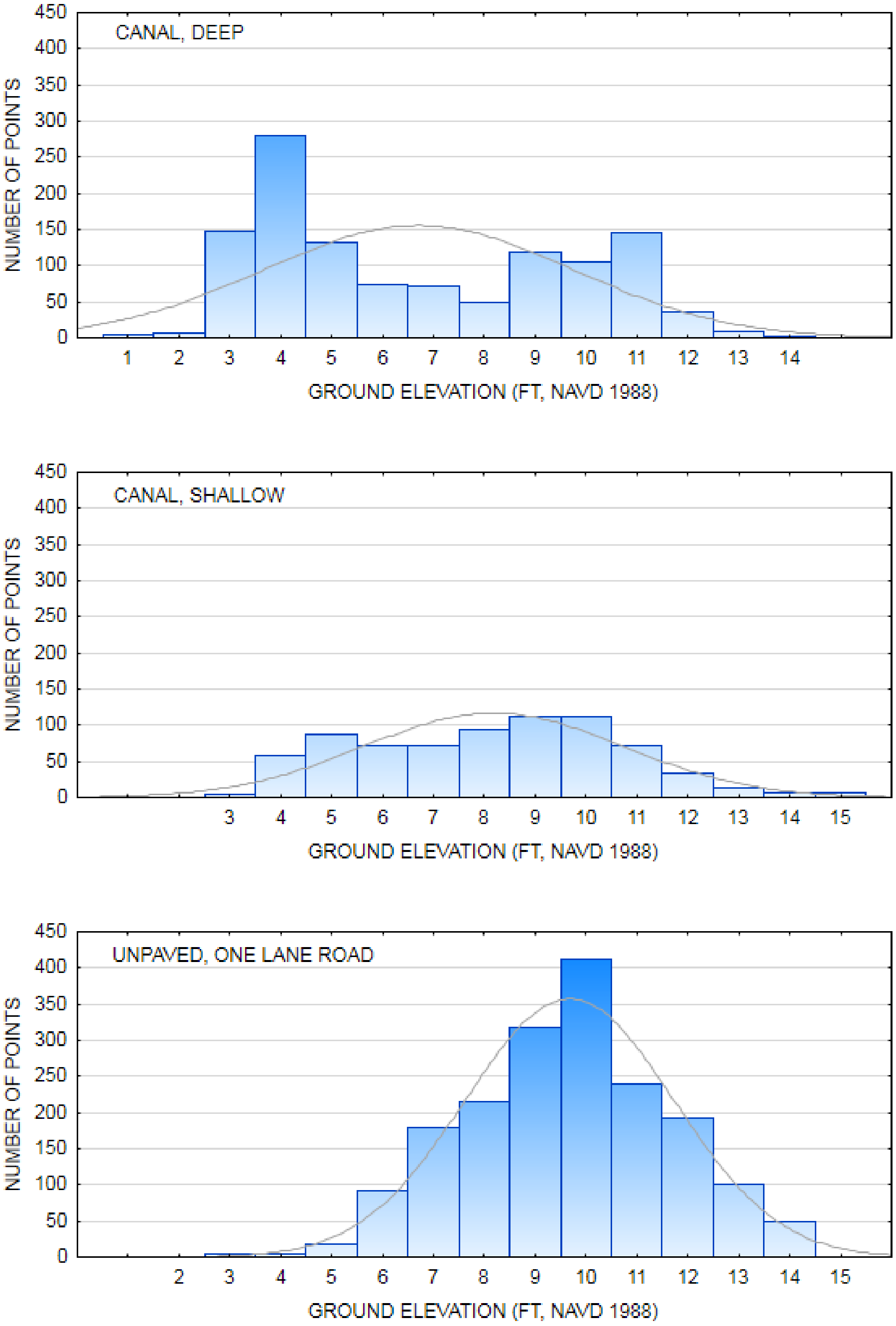

Figure 4-8. Histograms of ground elevation by linear feature type (deep canal, shallow canal, or road).

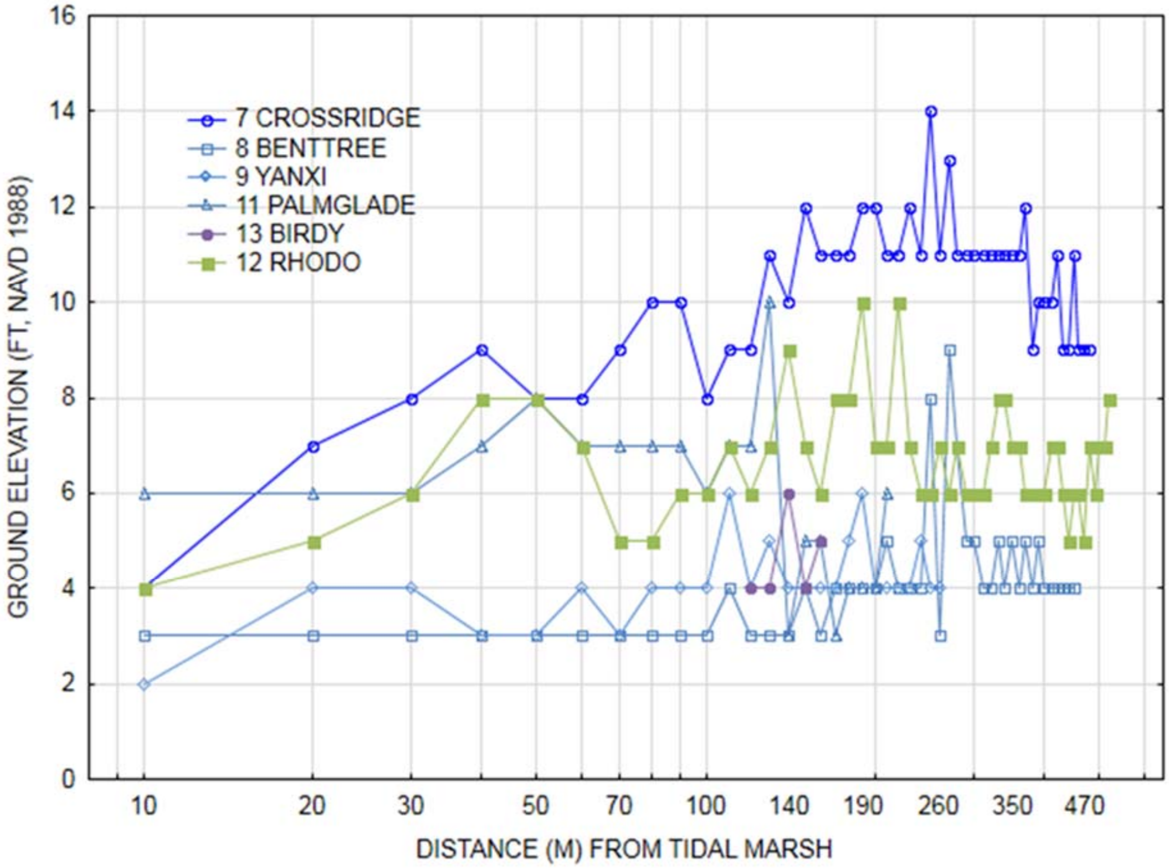

Figure 4-9. Line plots of ground elevation every 10 m along 6 deep canals.

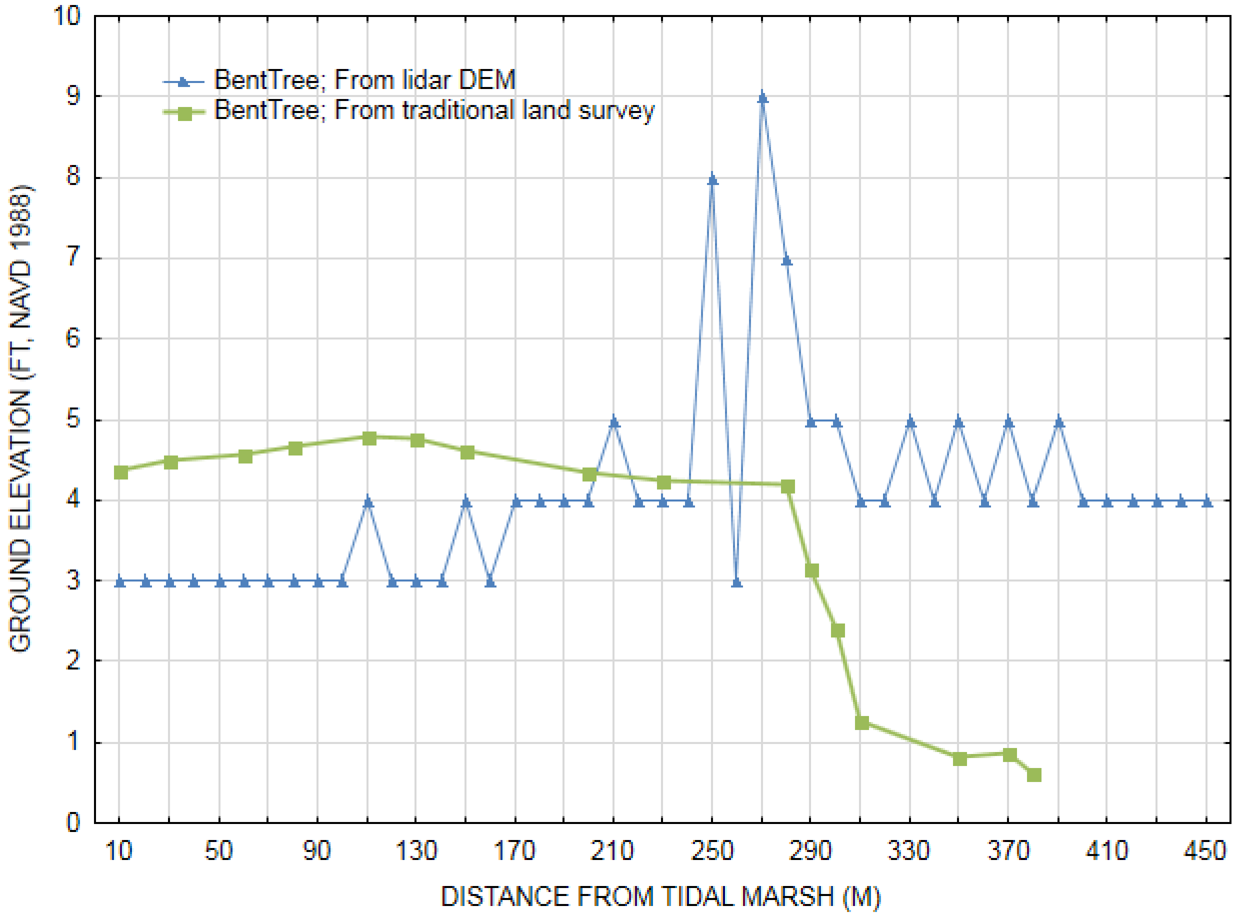

Figure 4-10. Results of field verification of ground elevation as determined by traditional ground level survey compared to ground elevation as determined by lidar DEM. Elevation along four canals were verified in the field; all show discontinuity between the two elevation sources.

B

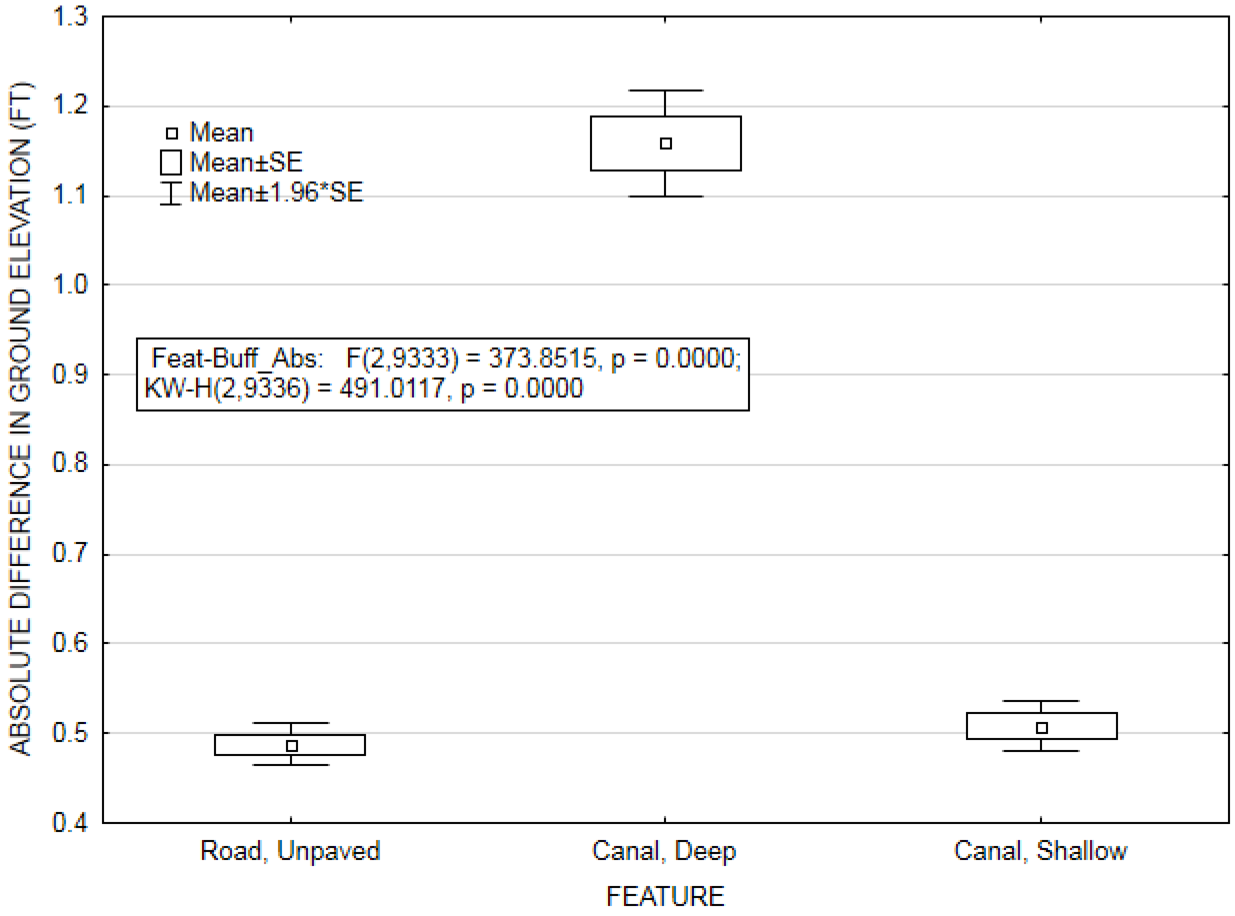

Figure 4-11. Box and whiskers plots of difference in elevation between feature and adjacent ground surface.

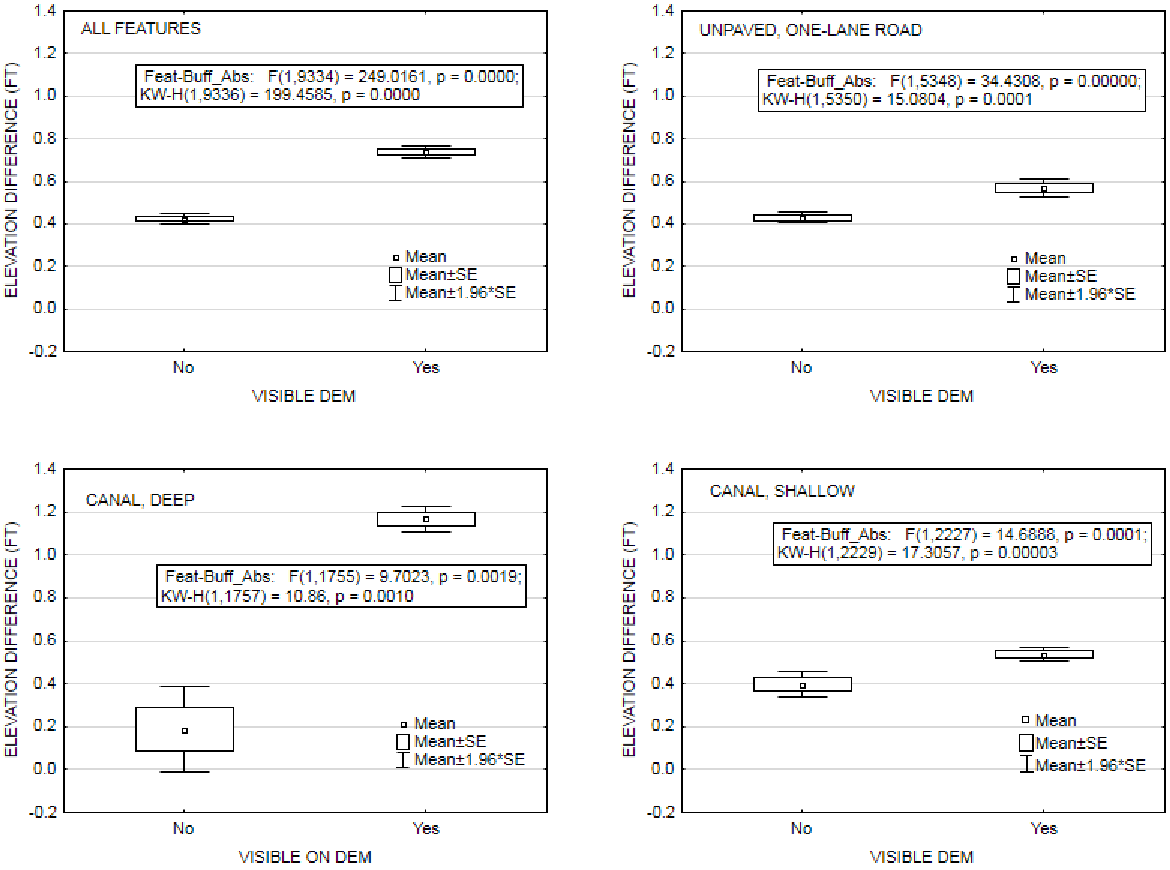

Figure 4-12. Box and whiskers plots of average elevation by linear feature type (all features, combined, unpaved one-lane road, deep canal, or shallow canal) categorized by whether segment was visible at 1:500 scale on DEM.

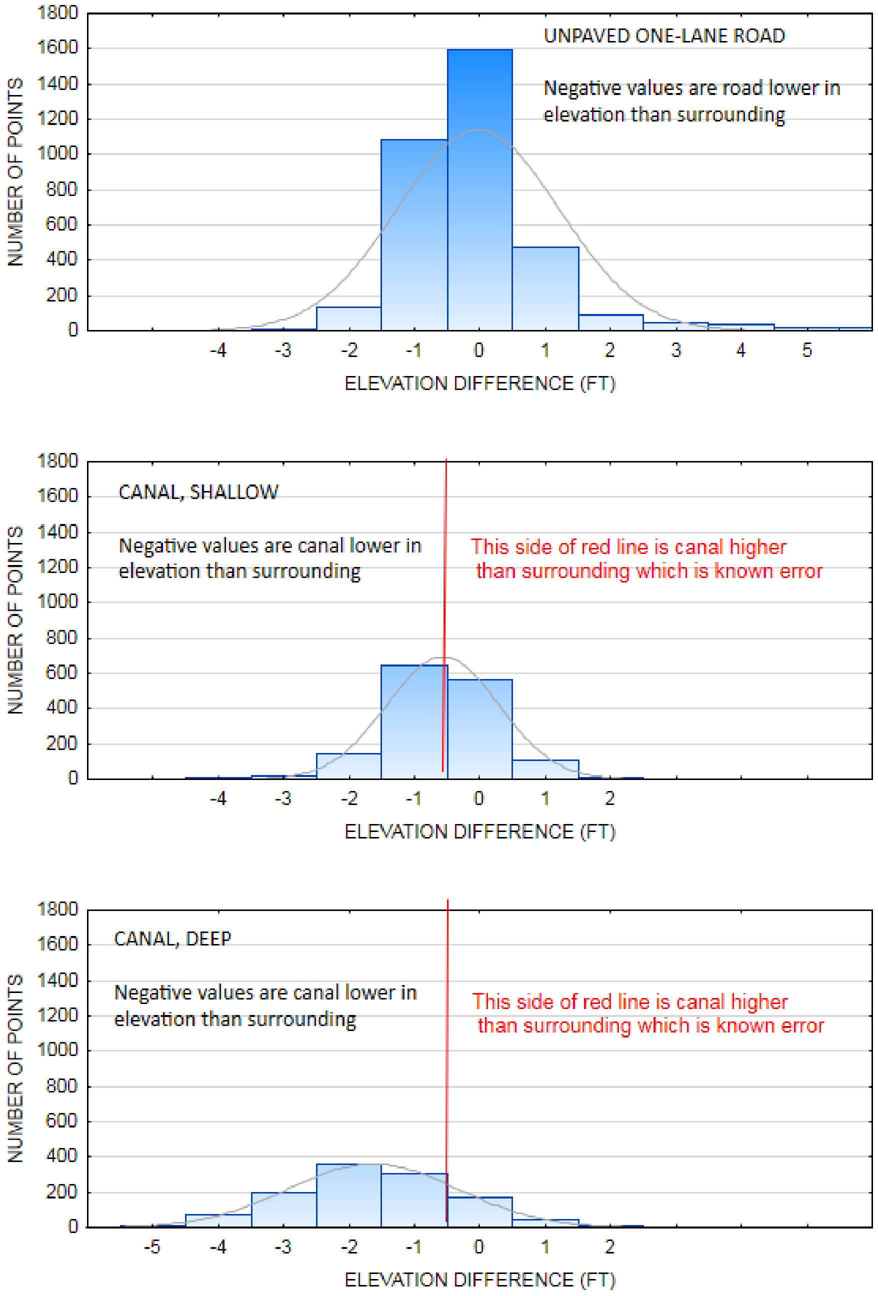

Figure 4-13. Histogram of differences between ground elevations of points within feature versus points placed 10 m on either side, categorized by feature type (deep canal, shallow canal, road).

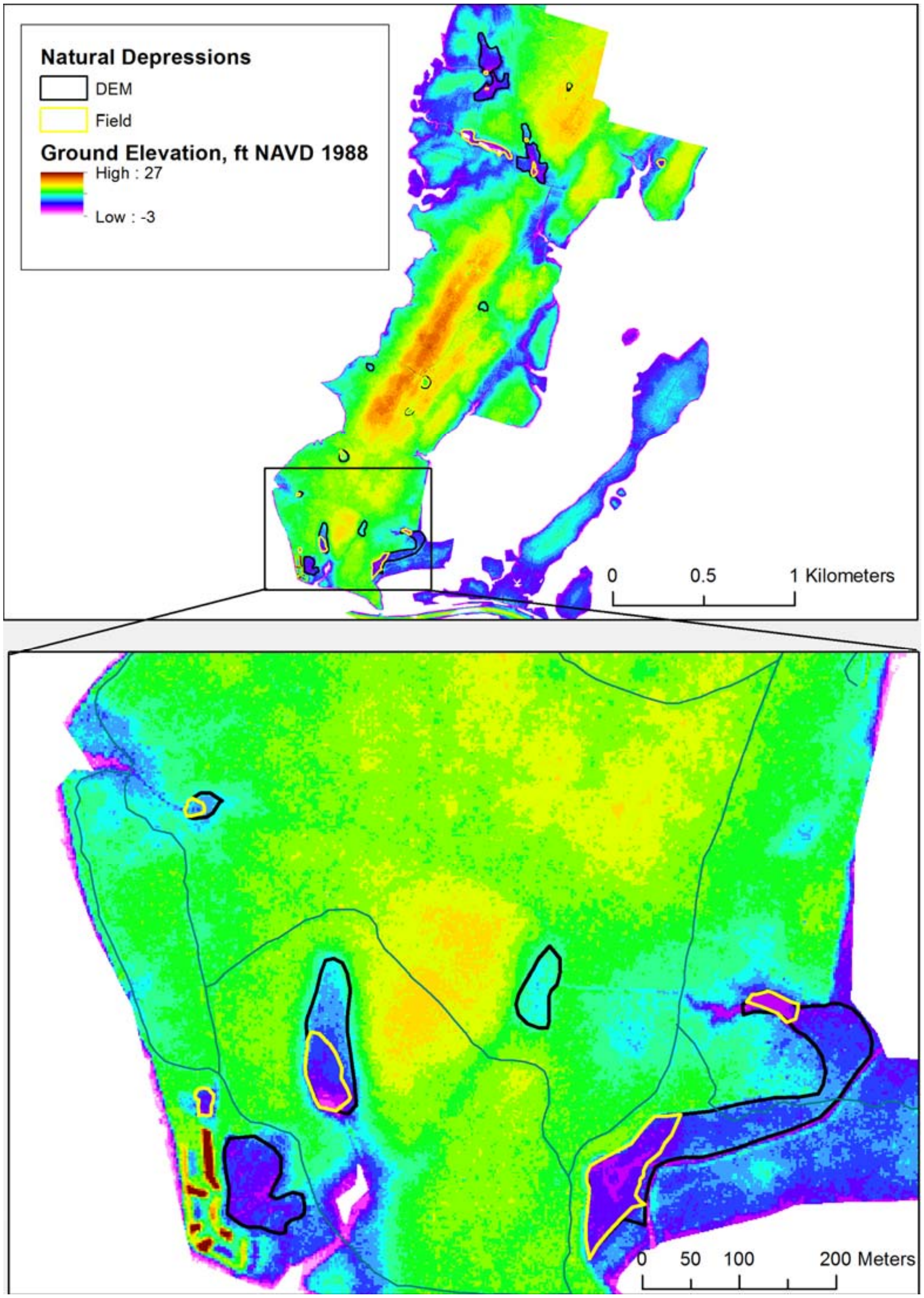

Figure 4-14. Extent of natural depressions as determined by the DEM and from field indicators of hydrology.

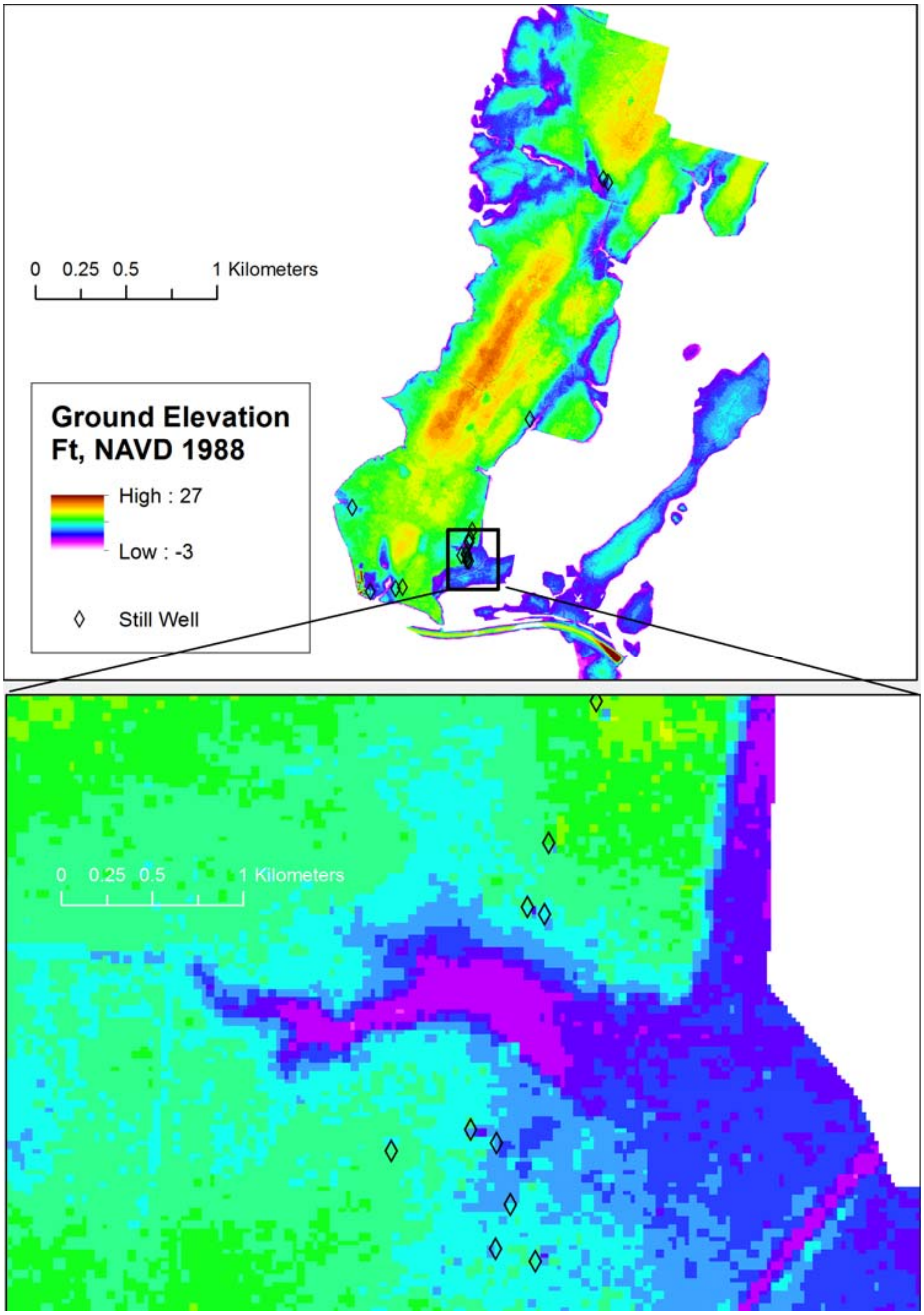

Figure 4-15. Location of still sites at Wormsloe, Isle of Hope.

CHAPTER 5 CONCLUSIONS

This study met its objectives to inventory herpetofaunal species on Wormsloe State Historic Site (Wormsloe) on Isle of Hope, a previously unstudied island (Chapter 2), to relate the amphibian distribution across 12 islands in the Georgia Seas Islands to species life-history traits and landscape (Chapter 3), and to characterize features influencing the freshwater resources which support amphibian breeding at Wormsloe using lidar-derived DEM (Chapter 4).

Chapter 2, Herpetofaunal Inventory of Wormsloe State Historic Site, documented 21 herpetofaunal species, compared to 74 for the mainland. Lower species diversity on an island relative to the mainland is not unexpected, given that larger areas tend to support greater species richness. Aside from species-area relationships, some species present on the mainland or other islands are likely absent from Wormsloe since it lacks certain micro-habitats (e.g. wetlands with hydroperiod more than 6 months, streams). Analyses indicated that the three seasons of trapping sampled approximately 80% of predicted maximum species richness for Wormsloe; only 3 to 4 additional reptile species and 1 amphibian species are likely to be added with further trapping.

While Wormsloe continues to support herpetofauna, abundances are skewed to a few, generalist species of typical of mesic forests with short hydroperiod wetlands. Unexpectedly, most species had low abundance with only 10 or more individuals captured for 6 of the 21 species. Amphibians were an order of magnitude more abundant than reptiles. Amphibians can have explosive population changes, so it is difficult to determine if skewed abundances reflect the below average rainfall during part of the trapping period or sustained longer term trend of declining populations. The changing landscape context of Wormsloe may also influence its' ability to continue to support herpetofaunal; from 1974 to 2008, the area in a 2.5 km buffer around Wormsloe shifted from 9% developed to 24%. The development now surrounding Wormsloe fragments the natural landscape linkages that existed and may be affecting local hydrology thru altered groundwater recharge.

Chapter 3, Amphibian Diversity in the Georgia Sea Islands and Implications for Long-term Conservation, placed the amphibian diversity at Wormsloe into a regional context of 12 other conservation areas. Reptiles were not included since abundances and diversity were low at Wormsloe and recent species distribution data are lacking for many other islands. Amphibians are considered indicators of overall ecosystems health since they require both wetland and upland habitats. As such, regular monitoring of amphibian populations is being implemented on all lands under federal jurisdiction (Byrne 2007, USGS 2012) and most recent field data comes from these surveys.

As a collective, the Georgia Sea Islands support 88% (23 of 26) of the amphibian diversity as the mainland. Individual islands, however, support from 19% to 69% of the species present on the mainland. Larger islands supported a greater number of species. Previous studies suggested that island geological age was also a factor, with older islands supporting greater species richness. However, since the older Pleistocene-core islands (Ossabaw, St Catherine's, Sapelo, St Simon's, Jekyll, Cumberland) are all larger than the Holocene islands, it is not possible to separate the effects of island age from island size. There was a stronger correlation between species diversity and area for larger islands. Smaller islands had weaker correlations; this may be an example of the small island effect related to vicariance in species colonization and persistence or to differences in types of habitats.

Micro-habitat features critical for breeding, e.g. seasonal ponds, are still under-represented even by the finer scale Georgia Department of Natural Resources (GA DNR) vegetation maps used in this study. Depressions at Wormsloe frequently occurred within forested vegetation communities indicative of uplands. Also, GA DNR map did not include any wetland habitat on one island for which eight seasonal pond-breeding amphibian species were recently confirmed (Byrne et al. 2011). Consequently, the area of a critical micro-habitat may be as influential as total land area. While lidar derived DEMs facilitate detection of areas likely to support seasonal freshwater, tying structure to hydrological function was difficult (see below).

Species occurring on more islands shared life-history traits of requiring less than 60 continuous days of freshwater to breed, having greater total reproductive output (greater total number of eggs and greater longevity), and with adults showing some tolerance to brackish. Some of the species absent from an island lacked these traits. Other species absent did not; their absence may be related to land-use legacies from farming. For example, because farming cleared old growth forests and disturbed soils, salamanders relying upon upland forested habitats may have been extirpated. While the forests regrew once farming ceased, the successional forests provided altered habitat conditions and either the development in the surrounding landscape or distance across saltwater marshes may have hindered recolonization. Since life history traits of species colonizing Holocene islands and persisting on Pleistocene island were correlated, retaining species on any conservation area should be a management priority.

It is unclear if shift in body size on islands compared to mainland population applies to amphibians in the Georgia Sea Islands (e.g. Bergmann's Rule for insular populations becoming either larger if released from predation or smaller if resource limited). Body size of a common toad at Wormsloe was significantly less than on the mainland, but this may not be true for all islands in the region. Body size has been shown to influence extirpation and colonization in transforming landscapes (Allen, Forys and Holling 2010). Until data on body size for a number of the Georgia Sea Islands are available, Bergman's Rule for insular populations and hypotheses relating species extirpations and colonizations to body size discontinuities cannot be tested in this region.

Chapter 4, Lidar Detection of Microtopographic Features in a Forested Landscape with Low Relief and Extensive Land-use History, explored detection of microtopographic features of importance to seasonal freshwater at Wormsloe State Historic Site. These features included the land-use legacies features of canals and deep wells dug out for ill-legal liquor distillation stills. While microtopographic features could be visually interpreted from lidar derived DEM, the overlap in ground elevations that the features occurred at limited automated extraction of features.

Tying ground surface structure to hydrological processes is challenging. Tidal influx inland was estimated to occur approximately 24 to 50 days per years along the canals connecting many of the four of the largest seasonal freshwater ponds to tide. The impact persists, since the tidal salt water percolates into the ground. Rain water then simply leaches the salts out of the ground. There are few hydrologic gages of the freshwater lens or surficial water levels in the study area (18 spread over 180 km, but clustered around Brunswick and

Savannah) (Peck, Painter and Leeth 2009). Anecdotal evidence for decrease of both the freshwater lens and in the duration of seasonal ponds exist. There were numerous small open pit wells associated with bootlegging still sites located throughout the Isle of Hope that were operational in the 1920s to 1930s era. These wells would have held freshwater, yet surface water was not observed in these sites during this study. Six cores at Wormsloe found lack of freshwater lens within 1.5 m of the surface (data collected in December 2013). A freshwater lens may exist deeper below the surface. But the soil between the surface and 1.5 m was saline, suggesting that salts are either leaching down from tidal influx or leaching up from saltwater wedge. Other evidence for decreased hydrology is presence of trees that require longer hydroperiod than observed. One particular site at Wormsloe has several adult *Taxodium distichum* (bald cypress), but no saplings or seedlings. Indeed, the site did not support freshwater hydroperiod longer than ~45 days during 2010 to 2014. The land-use history and changing land-use context of Wormsloe appears to negatively affect seasonal freshwater resources.

MANAGEMENT RECOMMENDATIONS

Management recommendations are made specifically for Wormsloe. However, they are likely of import to other conservation areas in the Georgia Sea Islands. Of concern is how the changing landscape context of Wormsloe may affect the ability of the remaining 21 herpetofaunal species to persist. Recolonization or “rescue” from adjacent areas is unlikely for islands surrounded by tidal marsh, and further fragmented and isolated by development on islands. Freshwater resources would have been limited historically in both spatial (natural depressions) and temporal (seasonality of rainfall and inter-annual variation in rainfall) extents. At Wormsloe, these originally constraints appear relatively unchanged; yet the processes that the same landscape support have been altered by canals, and perhaps surrounding development. Consequently, seasonal freshwater occurs over smaller spatial extent, as evidenced by lack of rainwater ponding, and over more limited temporal extent, as evidenced again by lack of rainwater ponding, but also with tidal influx along canals. Aside from tidal influx, the most obvious other influences at Wormsloe are an increase in surrounding development that alters groundwater recharge, and depletion of the freshwater lens necessary for surficial ponding of rainfall.

Management recommendations fall into two broad categories: monitoring of existing populations and protecting or enhancing freshwater resources to support amphibian breeding. While both categories are important along, they are inter-related and would function most effectively with action taken in both categories. Monitoring of amphibian communities would determine if measures to enhance freshwater resources are actually effective. Lack of breeding by common amphibians species in natural ponds over three years indicate that management actions should be taken soon.

1. Monitoring of existing populations
    a. Annual monitoring of amphibian breeding would catalog species persistence, and facilitate correlation between population dynamics and changing environs (Ray et al. 2014). Recent surveys documented the first occurrence of a non-native species (Byrne et al. 2011). Narrower time-windows of either extirpation or colonization would also be possible.

    Monitoring could be done either with remote vocal recorders or visual surveys for egg masses and tadpoles.
    b. Body size may predict persistence in changing landscapes, yet metrics on body size of for most species is lacking. Trapping during rainy periods when night time temperatures are above 7° C would be most likely to capture high number of adults during annual breeding migration.

2. Enhancing freshwater resources to support amphibian breeding
    a. Eliminating tidal influxes inland along anthropogenic canals with earthen plugs or sluice gates.
    b. Install gages to monitor depth and duration of seasonal freshwater ponds.
    c. Additional soil cores to determine if freshwater lens still exists. If so, install gage to monitor fluctuations in the freshwater lens.
    d. To facilitate ponding of rainwater in historical freshwater ponds, rapid infiltration of rainwater could be slowed by adding a layer of bentonite clay sealant, especially if the freshwater lens is depleted.

FINAL REMARKS

Sustaining existing populations of common species should be a priority for any conservation area. The fact that these conservation areas are islands should only heighten the importance of sustaining existing populations.

Resources management of regional amphibian populations is challenged by jurisdictional boundaries, as well as hydrological boundaries. The conservation areas included in this study occur in five of the six coastal Georgia counties, with ownership by private, state, and federal agencies. The conservation areas also crossed watershed boundaries; in fact, watershed boundaries bisected 9 of the 12 islands. The upstream portion of these watershed boundaries may intersect different counties. The complex jurisdictional and upstream watershed boundaries may complicate a conservation unit's efforts to protect its freshwater resources, inherited in part from its' upstream watershed.

The region developed under fluctuating sea levels 10,000 to 15,000 years before present, during the Pleistocene-Holocene transition (Hoyt 1967). Sea levels at a local tidal station, Fort Pulaski, have risen approximately 15 cm over the past 50 years. Historic maps of Wormsloe from 1897 show that some of the canals existed, even. Based upon naturally changing sea levels, canals connecting inland depressions to tidal salt marshes likely function differently than when initially built (late 1800s to early 1900s). Predictions of habitat changes based upon future sea level rise, approximately 30 cm per 100 years, suggest tide water will intrude inland more frequently and that the tidal marshes between the islands are likely to convert to more open water (Craft et al. 2009), further isolating the islands from an amphibian's perspective. The outer islands are the least likely to be completely inundated by future sea level rises since they tend to be higher in elevation and larger in area. Establishing baseline amphibian population data now will facilitate future correlations to changes in island area, connectivity, and in the timing or amount of precipitation. Any species extirpated from an island is unlikely to re-establish from natural dispersal as the amount of open water between islands increases, resulting in further isolation of populations.

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