Predicting Saltwater Intrusion Effects on Tidal Freshwater Forested Wetlands of the Lower Apalachicola River

by

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ABSTRACT

Tidal freshwater forested wetlands occupy low relief coastal areas and provide important and diverse ecological and socioeconomic services. Because they are exposed to upland runoff and tidal flooding, they are highly sensitive to fluctuations in seal level and freshwater input. Climate change, land use/cover change and increase in demand for freshwater put pressure on these systems. The dynamics and distribution of these communities are poorly understood. Very few studies explored the impacts of long term variations in salinity and freshwater input to these wetlands. The overall goal of this study was to describe the hydrologic conditions, salinity fluctuations, and forest cover changes along a tidal gradient at two distributary rivers of the Apalachicola River, namely St Mark’s and East River, in northwest Florida, USA. Along these tidal rivers, salinity was affected by Aplachicola tidal stage, Apalachicola River discharge, local precipitation and evapotranspiration.

Six monitoring sites were selected to observe water level and salinity along the East and St. Mark’s Rivers. The water level and salinity data collection began in December 2014 and continued until December 2015. To assess salinity changes under multiple forcing variables, artificial neural network (ANN) based models were developed for various sections of the rivers and the Apalachicola Bay. Because Apalachicola Bay salt water move inlands along the tidal rivers, bay salinity was one of the inputs to the models developed for the sites on the distributary rivers. Therefore, the Apalachicola Bay model was developed at the first step which was followed by six more models along the tidal rivers. All the models had very good skills in predicting salinity. Sensitivity analysis showed that salinity on tidal rivers was mainly driven by river discharge and tide level. Other variables such as wind speed and direction, water temperature, local precipitation, etc. had relatively very little effect. Salinity gradually decreased as the distance from the bay increased. Rivers essentially became freshwater after 15 river km.

The ANN models were later used to generate historical salinity levels spanning from 1985 to 2015 along these rivers and they were related to wetland species assemblages. Twenty two forest survey monitoring stations ($500m^2$) were established (each river has eleven stations) along a tidal gradient near the edge of the St. Marks River (n=11) and the East River (n=10). To calculate the importance value for each canopy species (>2.5 cm DBH), species basal area and density were used. The species were classified as tidal and non-tidal based on a previous study (Anderson and Lockaby 2013). The results showed that there was a clear and rapid transition from tidally dominant to a mix of tidal and non-tidal forest species. At both distributary rivers, the average daily salinity reached to 3 ppt when saltwater intrusion happened. The species distribution was compared with the saltwater intrusion averaged less than two days per year and the results showed that forested wetlands were composed of a mixed assemblage. When the intrusion exceeded two days per year, forest species transitioned to mixing of tidal species. Both rivers showed species composition shifts along the rivers; however, it was uncertain how much saltwater intrusion were affecting species composition.

In summary, results indicate that salinity in these systems are mainly driven by river discharge and tide level and the average salinity level changes gradually along the tidal gradient. Forest species distribution was also parallel to this salinity level changes. While tidal species are dominant at the high salinity level forest survey sites, non- tidal species are observed along the low salinity level forest survey sites. There is a strong relationship between species distribution and average salinity level and its frequency and duration. These species distribution will likely shift in the future because in response to sea level rise; salt water will reach further inlands and it will likely change the frequency and duration of salt water intrusion along these rivers.

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INTRODUCTION

Tidal freshwater forested wetlands are found between terrestrial and aquatic zones and are common near the outlets of the coastal rivers (Mitsch and Gosselink 2007). These wetlands are valuable resources due to the ecological services they provide. For instance, they are often part of the migration zones for birds during the winter time and a breeding area for both coastal fisheries and migratory birds (Costanza et al. 1998). Shallow water tidal habitats that include salt marshes to tidal freshwater wetlands also support the various prey species that are food for a large number of anadromous and marine fishes (Gunderson et al. 1990, Haley 1982, Simenstand et al. 1982). Anadromous species often occupy the tidal freshwater wetlands riparian zone because these areas provide suitable habitat (abundant insects communities, shade, refuge from predation) (Simenstand et al. 1982, Thorpe 1994) through their unique hydrologic characteristics.

In eastern United States, tidal freshwater forested wetlands can be found in the Atlantic coast and Gulf of Mexico coastline that extends from Maryrland to Texas. (Odum 1988; Mitsch and Gosselink 2007). Conservative estimates by Field et al. (1991) indicate that there are approximately 200,000 ha of tidal freshwater swamps along the coast of the Southeastern United States (Field at al. 1991). Tidal freshwater includes both forests and marshes and are considered vulnerable to increased saltwater intrusion due to changes in relative sea level and reduced freshwater flows (Doyle et al. 2007). The normal fluctuation of tides and river discharge has been shown to vary seasonally and annually and as a result it is difficult to estimate the upriver extent of tidal forested wetland boundaries (Doyle et al. 2007).

Where the rivers meet the ocean they are influenced by tides and other oceanic forces. These rivers are named ‘tidal rivers’ however depending upon the magnitude of river discharge, these lower sections of the river may retain freshwater conditions (Hoitink and Jay 2016). The amount that tidal water moves upriver and salinity levels that occur depend on the elevation of the river mouth and its geomorphology (Doyle et al.2007). Tidal waters can normally reach further inland on larger rivers (Hoitink and Jay 2016). Tidal fluctuations and tidal asymmetry also play important roles (Hoitink and Jay 2016) Tides are highly predictable and fluctuate because of the combined forces of the sun and moon and the rotation of the Earth. In a given day, there may be two pairs of low and high tides. If these high tides and/or low tides are nearly the same height, the pattern is considered semidiurnal (Hoitink and Jay 2016;Figure 1. 1). There can also be extra-high and -low tides. These extra-high tides (spring tides) appear at the time of the new and/or full moon, and the sun, moon, and Earth must be in alignment. One week after a spring tide, the gravitation pull of the sun counters the moon’s gravitational force and creates extra-low (neap) tides (Sumich 1996). Tidal bore is another important circumstance that can affect the salinity level in tidal rivers. They are vertical walls at the surface of water; during a flood, tide bores travel upriver “tens of kilometers” in shallow estuaries (Hoitink and Jay 2016). In this situation, the wave movement can cause a solarity wave before the tidal wave energy is gone (Hoitink and Jay 2016).

Salinity and flood regime along tidal wetlands have been impacted by sea level rise, and this change plays an important role on current and future vegetation communities (McKee and Mendelssohn 1989, Broome et al. 1995, Williams et al. 1999). However, there is no easy way of estimation for the direction and timing of these changes (Pereira et al. 2010, Bellard et al. 2012). Over the past few decades, studies on vegetation shifts along the coasts has increased. For example, based on a field survey in Southern New England, Field et al. (2016) detected in a tidal marsh zone that vegetation cover area decreased because of sea- level rise however this decrease was balanced by landward migration by the community. Their survey results showed that the tidal marsh vegetation shifts were observed over a large area. Low mortality and high growth rates were also found at the forest boundary due to shifts that did not extend into that zone. In another study, Stagg et al. (2016) observed the tidal freshwater forested wetlands at the Georgetown South Carolina and they found that oligohaline marsh capacity to recover from sea-level rise over a five years period. Tidal marshes were more resilient than tidal freshwater forested wetlands. They found that elevation loss was observed in the study area, and tidal fresh water forested wetlands were affected by subsurface process (root zone expansion and/or compaction).

Another parameter affecting the salinity level in the tidal freshwater forested wetlands is river discharge. Coupled with tidal stage, these factors create a unique salinity range in tidal freshwater forested wetlands. Because of this, tidal freshwater forested wetlands are highly sensitive to river basin management, which can affect the amount and frequency of freshwater delivery. Although waters in these wetlands are usually fresh, their proximity to oceanic waters means that occasionally they are exposed to mixed saline waters. However, hydrology of the system is changed by river management practices (Ward 1998). Under normal conditions there is a negative correlation between salinity and river flow, and if the river flow from the upstream is reduced, the salinity level may increase which can damage biological communities (Copeland, 1966). Tidal forested wetlands must receive an adequate flow of freshwater in order to keep surface water salinities at or below 0.5 parts per thousand (ppt) (Cowardin et al. 1979). Changing river flow regimes related to the operation of dams within a basin may reduce flow and cause greater salinities in these freshwater tidal zones. In the Mid-Atlantic States, nearly 90 hydroelectric dams were built along the coastal plain rivers (Schneider et al. 1989). These dams are an important reason for changing hydrologic conditions and changes in the amount and frequency of peak river discharge (Livingston 2008).

Tidal freshwater forested wetlands along the Apalachicola River are the focus of this study. The Apalachicola River (Figure 1. 2) is the biggest alluvial rivers in Florida (Anderson and

Lockaby 2012). The river is part of the Apalachicola-Chattahoochee-Flint (ACF) River basin (50 688km^2; Anderson and Lockaby 2012). The Apalachicola Bay is generally shallow and its depth is normally around 2 m (Freeman et al. 2012). Jim Woodruff Lock and Dam was built in 1952 at the confluence of the Chattahoochee and Flint Rivers to create the Seminole Lake reservoir. It empties into Apalachicola River which drains to the Gulf of Mexico. The operation of the Jim Woodruff Lock and Dam plays a critical role in providing freshwater to the Apalachicola River. River management here and elsewhere in the ACF basin is a critical component of water supply for Georgia, Alabama, and Florida, as this system provides drinking water for millions of people. Discharge, as moderated by JW dam, along with tide levels govern the extent of salt water intrusion along the Apalachicola River- Bay system. Tides in Apalachicola Bay are semi-diurnals and range around 1-m in height (micro-tidal) (Doyle et al. 2007).

Vegetation along the lower tidal sections of the Apalachicola River consists of a variety of marsh and forested wetland communities. In the tidal freshwater forests, several species are common. Anderson and Lockaby (2011) observed swamp tupelo (*N. biflora*), bald cypress (*T. distichum*), water tupelo (*N. aquatic*), cabbage palm (*S. palmetto*), Carolina ash (*F. carolina*), overcup oak (*Q. nigra*), and Ogeechee tupelo (*N. ogeechee*) along the Apalchicola River (Anderson and Lockaby 2011). These are similar to species seen in other freshwater tidal swamps. Along a similar zone on the Savannah River in Georgia, USA, Duberstein and Kitchens (2007) noted water tupelo (*N. aquatic*), swamp tupelo (*N. biflora*), water oak (*Q. nigra*), ash (*Fr. ssp*), sweetgum (*Liquidambar styraciflua*), red maple (*Acer rubrum*), bald cypress (*Taxodium distichum*), American hornbeam (*Carpinus caroliniana*). Some of these species are considered ecologically and economically significant. For instance, Ogeechee tupelo is economically valuable because Tupelo honey, one of the world's most renowned honeys, is produced from the Ogechee tupelo along the Apalachicola River and its tributaries Northwest Florida (Watson 2016).

The Apalachicola system is an extremely clean and productive body of water, a type of system that is uncommon in the United States (Livingston 2008). It contributes to the productivity of Apalachicola Bay which supports 90% of Florida's and 10% of nationwide oyster harvest, along with being highly important to shrimp harvests (Huang et al. 2001). The Apalachicola River-Bay system promotes high phytoplankton productivity (Boynton et al. 1982) because the amount of freshwater is relatively high compared to others (Myres and Inverson 1981). Phytoplankton productivity is a major contributor to estuarine food webs along the Gulf coast (Livingston 2008). If the salinity of the bay and the along the Apalachicola distributaries change in the near future due to sea level rise and/or increased/reduced freshwater input all these species might become stressed. Therefore, it is vital to study the potential changes in the salinity levels in this sensitive system in response to various environmental variables.

STUDY GOALS AND OBJECTIVES

The overarching goal of this study is to describe the hydrologic conditions, salinity fluctuations, and potential forest cover changes in tidal freshwater forested wetlands along the St. Mark's River and East River distributary rivers to the Apalachicola River near Apalachicola, Florida. The specific objectives to reach this goal are as follows:

1. Create an Artificial Neural Network (ANN) based salinity model using long term data series to predict historical (1985 to 2015) salinity in Apalachicola Bay.
2. Utilizing the bay model, develop separate models along the St. Mark's River and East River tidal freshwater wetlands. Predict historical salinity fluctuations along the tidal reach of these rivers.
3. Describe and compare tidal freshwater forests and distributions of the tree species along a tidal gradient.
4. Relate the characteristics of salinity regime and forest types along the river's tidal reach (upstream to downstream).

Artificial neural networks (ANN) Models

The relationship between salinity, freshwater inflow, tide level, wind speed and direction, temperature is a very complex one. Complex hydrodynamic models that are based on physical process are useful in predicting salinity. However, such models require availability of spatially variable input data, which is difficult to obtain. Artificial intelligence-based models such as artificial neural networks (ANNs) can be viable options for such complex, highly non-linear problems. Artificial Neural Network (ANN) models are data driven approaches that can help clarify complex relationships between inputs and outputs (Maier and Dandy, 1996; Neural Ware Inc., 1991; Hubick, 1992; Maren et al., 1990) without the need of a detailed understanding of its physical characteristics. The application of ANN in various hydrological predictions has been extensively evaluated and published in recent years (Ha and Stenstrom, 2003; Sahoo et al., 2006; Singh et al., 2009; Kalin et al., 2010; Palani et al., 2011; Gazzaz et al., 2012; Isik et al., 2012; Amiri et al. 2012; Rezaeianzadeh et al. 2013, 2014). Artificial neural networks are widely used in hydrology. This study aims to use ANN as a forecasting tool to determine the historical salinity regime in the bay first, then in the wetland plots along the tidal rivers.

The simple format of ANN architecture includes three layers: the input, the hidden layer, and the output (Figure 1. 3). Training of ANN (learning) includes three elements, namely weight, transfer function, and learning laws. While the relative importance of the inputs defines the weights between the neurons, the learning laws examine the adjustment of the weights during training according to an algorithm. In addition, the transfer function controls the generation of the outputs from a neuron (Caudil, 1987). Numerous forms of ANN model architecture are present. Flow and processing are two ways to categorize ANN. There are multiple connections between hidden layers, inputs, and outputs in a feed-forward network (Dawson and Wilby, 2001). One of the most popular forms of ANN architecture has been the multilayer perceptron, which does not include any connection between neuron outputs and inputs. In the multilayer perceptron network, the input data flows in one direction. In this study, the transfer function is tangent sigmoid, which is defined for any variable *S* as

$$f(s) = \frac{2}{1+\frac{1}{e^{2s}}} + 1 \quad (1)$$

Different training algorithms can be used in ANN models. This study utilizes the Levenberg-Marquardt training algorithm. The Levenberg-Marquardt algorithm can approach second-order training speed in the absence of a Hessian matrix (More, 1978). In order to get the most efficient solution, error will be assessed by the Mean Square Error (*MSE*) function, which is one of the commonly used error functions. A low *MSE* indicates an efficient model.

$$MSE = \frac{1}{n} \sum_{i=1}^{n} (S_i - O_i)^2 \quad (2)$$

where, S_i is the ANN output (simulated), O_i is the target (observation), and n is the number of data.

Long Term Salinity Prediction along the St. Mark's River and East River

Salinity level can change in time and location and such changes contribute to the spreading of estuarine ecosystem properties (Morey and Dukhovskoy 2011). In tidal freshwater forested wetlands, periodic saltwater intrusion occurs and increases in likelihood the closer rivers are to the coast. As a results, salinity exposure increases along the lower reaches of rivers to the point where tree growth rate is decreased (Chapin 1991) and eventually riparian lands are unsuitable for forests, transitioning to shrub or marsh conditions (Powell et al. 2016). Increasing salinity causes osmotic stress which is the important mechanism affecting the growth of tree species (Powell et al. 2016). Because of the proximity to the coast, hydrogen sulfide can be commonly produced in high quantities in wetland sediments and is a toxic chemical to plants which can further restrict plants (Hakney and Avery 2015). Therefore, it is important to clarify the long term hydrologic and salinity characteristics at this study site to clarify the effects of tidal fluctuations and variations in Apalachicola River discharge. The ANN models developed will be used to estimate future salinity levels in the bay and in selected sites along the distributary rivers.

A Sign of Tidal Gradient: Forest Species Distribution along the Tidal Rivers

Vegetation in tidal forested wetlands is primarily freshwater dependent. Brackish tidal marshes typically separate tidal freshwater forested wetlands from coastal waters; however, the hydroperiod of these ecosystems vary and can be strongly influenced by the local tidal range. Long term exposure to elevated salinities (particularly those >0.5 ppt) is considered detrimental to most tree species although some species are considered more sensitive to salinity than others (Anderson and Lockaby, 2011). At the downriver range of freshwater forested wetlands, saltwater intrusion is more common. Because of that, forest species that are in the freshwater tidal range are typically adapted to periodic saltwater intrusion. When wetlands transition in type from tidal to non-tidal, it is expecting that a significant shift in salinity occurs along with change in hydrology (Day et al., 2007). As a result, the composition of forest species has been shown to shift as riparian transition from tidal to non-tidal sections. Forest structure can also change along this transition zone and shows different conditions (Anderson et al., 2013). Some non-tidal forest composition moves to the upper tidal transition zone and these areas likely will have significantly less occurrence of saltwater intrusion.

In this study, Models estimating historical wetland salinity (1995–2015) along this gradient will be compared spatially to the results of forest survey data to examine for potential relationships between historical salinity patterns (frequency and average number of saltwater intrusion events; >5 ppt) and species occurrence across a tidal gradient. This chapter will explore the relationship between tidal influence and the distribution of forest species along a tidal gradient at St. Mark’s River and East River.

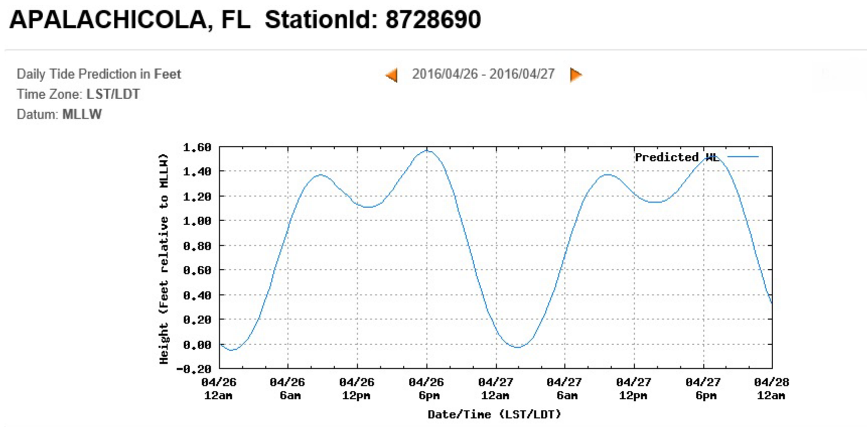

Figure 1. 1: Example tide level in the Apalachicola Bay; two low-level and two high-level tides are shown in a day (semidiurnal). Data source: http://tidesandcurrents.noaa.gov/noaatidepredictions/NOAATidesFacade.jsp?Stationid=8728690

Figure 1. 2: Apalachicola Bay and lower downstream of the Apalachicola River and its tributaries St. Mark's River and East River in Florida, USA.

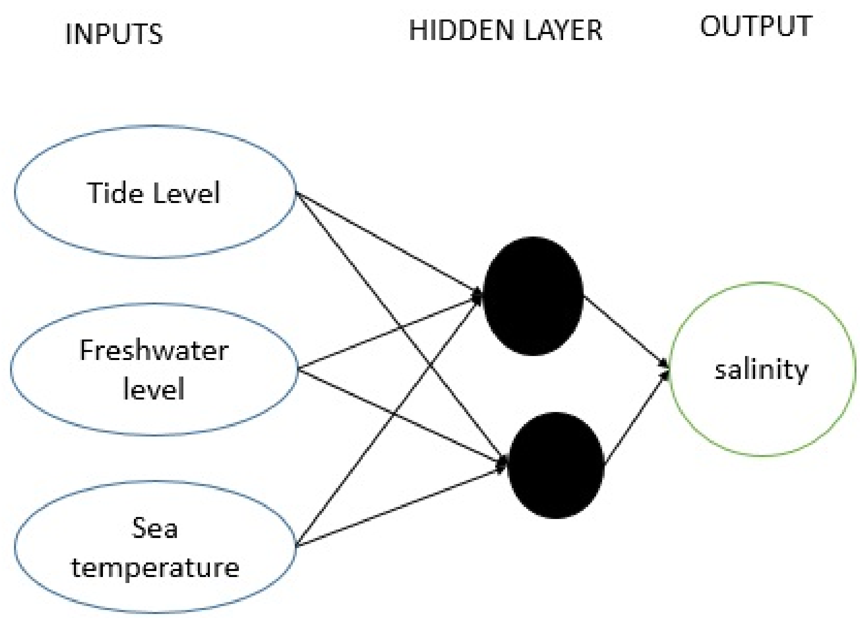

Figure 1. 3: An example of feed-forward artificial neural network (ANN) structure with three vectors as inputs, 1 hidden layer with 2 neurons and one output vector.

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Chapter 2. Predicting Salinity along a Tidal Freshwater Gradient of the Lower Apalachicola River System

ABSTRACT

Hydrology and salinity play an important role in the health and function of coastal ecological systems, such as wetlands. Along the coastal rivers, salinity determines the type of wetlands that occur, therefore it is important to understand how far saltwater can access inlands. This chapter presents a modeling approach for long term prediction of saltwater level change along the St. Marks and East River, distributary rivers for the Apalachicola River near Apalachicola, Florida. Along the lower reaches of these river systems are extensive freshwater tidal forested wetlands. Salinity in these tidal rivers are affected by Apalachicola Bay tidal stage, Apalachicola River discharge, local precipitation, and evaporation. Models were developed to understand the relationship between salinity and hydrographic variabilities within the Apalachicola Bay and in distributary systems along the St. Marks and East Rivers. To assess salinity changes under multiple forcing variables, an artificial neural network (ANN) model was developed for various sections of the rivers and Apalachicola Bay. A number of different input combinations were experimented as predictors to create the best model. Results showed that the discharge measured at a nearby USGS river gage (Sumatra Gauge) and bay tide level were the main drivers of the Apalachicola Bay salinity. When the model was provided with one-day lag of the bay salinity with all other inputs (discharge, tide level, sea temperature, evaporation, wind direction and wind speed), model accuracy increased significantly. However, discharge, tide level, and sea temperature were eventually selected as input data in the developed model in order to predict water salinity in the Apalachicola Bay, because other input data sets (wind speed, wind direction one-day lag salinity) were not available between 1985 and 2015. The bay salinity prediction model was used as an input for the small scale models that were set along

the St. Marks River and East River. To examine salinity along the St. Marks River and East River, six ANN models (small scale models) were created. To create these models, salinity and water level data were collected at six measurement sites along the two rivers. The small scale models were fed by different inputs including the Apalachicola River discharge, tide level of the Apalachicola bay, simulated salinity level from the large scale bay model, evapotranspiration and temperature. Results showed that tide levels in the Apalachicola Bay and Apalachicola River discharge were the main drivers of salinity in downstream sections of the St. Marks River and East River. When the annual average salinity decreased below 0.10 ppt, salinity level was mainly driven by the Apalachicola River discharge.

INTRODUCTION

Coastal zones and river-dominated estuaries are important and often very productive ecosystems. Due to climate change and anthropogenic pressure, these ecosystems may become impaired, resulting in the decline of important ecosystem functions and habitats. Engineering structures on rivers, changes in river flow, and sea level rise have a great impact on the salinity of estuaries. The Apalachicola River/Bay system is influenced by all of these factors. This estuary is located in northwest Florida where the Apalachicola-Chattahoochee-Flint (ACF) River system drain to Gulf of Mexico (Figure 2. 1). The ACF basin consists of 16 major tributaries and is important for agricultural, municipal, and industrial use (Havens et al. 2013). Water allocation among these diverse uses has been a contentious issue in the Southeast. River discharge in the Apalachicola River is largely controlled by the Jim Woodruff Dam which was constructed on the river in 1952 (Figure 2. 1).

The Apalachicola River drains into the Apalachicola Bay which is a highly productive ecosystem and an important fishery (Huang 2002). This system is home to 131 freshwater and estuarine fish species, including several significant Gulf of Mexico species (Gulf sturgeon and oyster), and supplies 90% of Florida oyster production (Whitfield and Beaumariage1977, Huang 2010, Villegas and Roberts 2012). River flow to the bay is an important consideration as because this flow influences the bay's salinity and temperature; increases in salinity and temperature can greatly increase oyster disease and parasite risks (Livingston et al. 2000). Additionally, waters with higher salinities generally support more oyster predators such as whelks, drills, and conch (Haven et al. 2013). Freshwater tributaries associated with Apalachicola River contribute nutrients to the bay and help maintain estuarine salinity (Huang 2002, 2010). The incoming freshwater, however, is influenced by various factors such as coastal runoff, river discharge, local precipitation, evaporation etc. (Morey and Dukhovskoy 2012). This results in freshwater salinity varying in space and time.

The Woodruff Dam (Figure 2. 1) plays a critical role in controlling the amount of freshwater flow to the Apalachicola Bay. Due to the direct correlation between bay salinity and river flow, reduced flows can lead to increasing salinity levels in the bay (Copeland 1966). During drought years, the reduced flows can be particularly problematic and can affect estuarine biological communities (Copeland 1966). A number of recent, severe low-flow periods impacted the river between 2006 and 2008. Such drought years are especially worrying as droughts normally coincide with the increase in water demand throughout the basin (Anderson and Lockaby 2012). ACF Basin management actions have recently faced legal issues because of Georgia's plan to divide the flow that feeds water reservoirs along the Atlanta region (Morey and Dukhovskoy 2012). Since 1950, considerable growth has occurred in and around the city of Atlanta (Huang, 2010) which has increased the demand for water and the need to increase water storage in the region.

In addition to being affected by river flow, the estuary hydrodynamics are also affected by other freshwater sources such as coastal run off and the connections between the bay and the Gulf of Mexico. Seawater generally comes to the bay from the east and exits from West

Pass, Indian Pass, and the manmade Sikes Cut (Figure 2. 2) (Sun and Koch 2001). The saltwater and freshwater are well mixed in the Apalachicola Bay (Sun and Koch 2001); however salinity levels in the bay usually varies spatially. The Apalachicola River mouth is located at northwest part of the bay (Figure 2. 2), so the north and west parts tend to be fresher than other sites (Sun and Koch 2001).

The freshwater tidal zone represents the interface of downstream river flow and upriver movement, and Apalachicola system shows a natural river-bay interaction. In this tidal freshwater zone, periodic saltwater intrusion is normal (Anderson and Lockaby 2012). In the Apalachicola system, vegetation changes are linked with variation and duration of saltwater intrusion (Letizman et al. 1982). Salt water intrusion is driven by winds and tides in the Apalachicola system (Livingston 1984).

To understand the general hydrodynamics in estuarine systems, it is important to examine long term datasets for a wide variety of physical parameters (Tsou and Matheson 2002). Predictive models can be used as a tool to clarify effects of the flow alterations, sea-level rise, and other climate variables. Numerous studies have been conducted to understand the long term salinity range changes in the Apalachicola Estuary (Sun and Koch 2001, Zhang 2003), but to the best of our knowledge no such study exists in the tributaries. In this study, an artificial neural network (ANN) model was used to predict long term salinity in the Apalachicola Bay and, using these data, determine the salinity regime of two distributary rivers near the mouth of the Apalachicola River: The East River and the St. Marks River (Figure 2. 2). Examining the salinity regime and the Apalachicola River management effects in tidal freshwater forested wetlands ecology is the main goal of this study.

METHODOLOGY

Study Site

Apalachicola Bay is located at the mouth of the Apalachicola River in the Florida panhandle region, USA. The barrier islands St. Vincent Island, St. George Island, and Dog Island border the bay (Figure 2. 2). As these islands constitute the boundary of the bay, it is a multiple-inlet bay. Between the islands, there are four inlets (Morey et al., 2012): West Pass, Indian Pass, East Pass, and Alligator Pass connecting the bay to the Gulf of Mexico.

The Apalachicola River watershed covers an area of 48,500 km^2. The river drains western Georgia, southeastern Alabama, and northern Florida (Livingston, 1991, Figure 2. 1). In volume, the Apalachicola River is the 21st largest river in the United States and the largest in Florida (Ward et al., 2005; Light et al., 2006). At the USGS Sumatra gauge (Figure 2. 2), which measures the discharge of the Apalachicola River, the average yearly (2002 to 2007) discharge of the river is approximately 400 m^3/s (http://waterdata.usgs.gov/nwis/uv?02359170). Salinity in the Apalachicola Bay varies between 0.6 and 27 part per thousand (ppt) (2002 to 2007; http://cdmo.baruch.sc.edu/get/export.cfm) mainly because of the temporal variations in freshwater input from the river. The Apalachicola Bay's average salinity level is generally lower than other bays because the water level is shallow and the freshwater input is higher than many other bays located along the Gulf of Mexico and Atlantic coast (Livingston 1984).

Field Data Collection

For this study, two groups of salinity models were developed. Small scale models consisted of six individual models developed along the St. Marks and East Rivers (Figure 2. 2 and Figure 2. *3*). The Apalachicola Bay model (large scale) was developed to test the utility of ANN models in salinity prediction as well as generate historical salinity data for the Apalachicola Bay which served as input to the small scale models.

To develop the small scale river models, water level and conductivity data were collected continuously at six points along the St. Marks and East Rivers. Three points were located along the approximate extent of freshwater tidal reach of each river (Figure 2. 3). Along each river, sites were selected along relatively straight sections of the river and where a small, drainage outlet connected riparian forested wetlands to the river. Within each drainage outlet near the river, a global positioning system unit (GPS) was used to determine the geographic coordinates of each sites (East River One (ER-1; 6.28 rkm – river kilometer), East River Three (ER-3; 7.39 rkm), East River Five (ER-5; 14.12 rkm) and St. Marks River One (SMR-1; 4.87 rkm), St. Marks River Three (SMR-3; 12.48 rkm), and St. Marks River Five (SMR-5; 16.69 rkm))

A Solinst Levelogger pressure transducer (Solinst Levelogger Model 3001) was installed and used to measure water levels (Figure 2. 4 A). To measure water conductivity, a conductivity data logger (HOBO U24-001) was installed. Each instrument was attached to a fence post and inserted into the drainage outlet so that the instruments were near the bottom of the outlet and most likely to stay inundated (Figure 2. 4 A). The fence post was stabilized at the deepest point of the outlet, and the conductivity logger and pressure transducer was attached to the bottom side of the fence post. The Levelogger measures absolute pressure using the Hastelloy pressure sensor. Air pressure data are needed to make a correction on the water level. Therefore, we also installed a single barometer on a tree to measure the air pressure at St. Marks River (Figure 2. 4 B). Each instrument was programmed to collect data on an hourly basis. Instruments were installed on 2/20/2015 and retrieved on 6/3/2016. Subsequently, the conductivity data sets were transformed to salinity level (part per thousand) by the Hobo conductivity logger software, and daily averages were calculated for each data set.

Other Data Sources

To predict the historical salinity at these six points, long term data sets of Apalachicola Bay tide level, Apalachicola River flow, and sea water temperature between 1985 and 2015 were accessed. The daily average discharge of the Apalachicola River between 1985 and 2015 were downloaded from USGS Sumatra gauge (902359170) (Figure 2. 2) (http://waterdata.usgs.gov/nwis/uv?02359170). There was a gap between 1986 and 1989 discharge data. However, stage-discharge (h-Q) data were available for this period from the USGS. We fit a second degree polynomial to that data and used it to fill the missing flow values.

$$Q = 121h^2 - 2971h + 7598 \quad (1)$$

Daily tide level data for 1985-2016 were downloaded from the National Oceanic and Atmospheric Administration (NOAA), Apalachicola Station (8728690) (Figure 2. 2) (https://tidesandcurrents.noaa.gov/noaatidepredictions/NOAATidesFacade.jsp?Stationid=8728690) (Figure 2). Daily evaporation in mm/day was calculated using the Hamon Method.

$$PET = 29.8 * \frac{D * es}{T + 273} \quad (2)$$

where, D is sunshine hours on a given day, e_s is saturated vapor pressure in kPa, and T is daily temperature (C^0). Daily sun shine hours were calculated based on Dry Bars altitude and longitude in the bay.

Daily salinity and water temperature data sets for the Apalachicola Bay for 2002-2016 were downloaded from Apalachicola National Estuarine Research Reserve (ANNER) Dry Bar station (Figure 2. 2) (http://cdmo.baruch.sc.edu/get/export.cfm). Daily wind speed and wind direction data sets were also downloaded from ANNER's meteorology observation station.

Although water temperature affects salinity, water temperature data were not available before 2002. Therefore, a model was created for estimating water temperature using air temperature as input. Air temperature were downloaded for the pre 2002 period from the NOAA Apalachicola weather station (http://www.ncdc.noaa.gov) (Figure 2. 2).

Salinity Prediction with ANN Models

Artificial Neural Network (ANN) is a data driven approach that can help clarify complex relationships between inputs and outputs (Maier and Dandy, 1996; Neural Ware Inc 1991; Hubick, 1992; Maren et al. 1990). This study included two different ANN models: large scale (the model of The Apalachicola Bay) and small scale (the models along the St. Marks and East Rivers). The bay's salinity predicted by the large scale model served as input for the small scale tidal river models. To assess the correlations between various inputs (discharge, tide level, sea temperature, evaporation, wind speed and cosine of wind direction) and salinity, rank-based scatterplots were created (Figure 2. 5). The Pearson and Spearman correlation analyses were performed using Statistical Analysis Software (SAS).These inputs were fed into ANN to predict the salinity in Apalachicola Bay (large scale model). Since the study sites in the tidal river systems are located along a salinity gradient, different combinations of inputs (Apalachicola River discharge, tide level, air temperature, evapotranspiration, precipitation, and simulated bay salinity from the large scale model) were tested to predict salinity along the tidal rivers (small scale models).

Performance measures such as *AIC*, *BIC* and mean square error (*MSE*) were used to select the optimal model. In selecting the best model the Akaike's information criterion (*AIC*) and Bayesian information criterion (*BIC*) were used, a standard approach in ANN modeling. *AIC* and *BIC* takes into account both descriptive accuracy of the models and model complexity (parsimony) (Wagenmarkers and Farrell 2004). There is no consensus in the literature one being superior over the other, thus both are commonly used. The equations given in Qi and Zhang (2001) were used in this study.

$$AIC = log\ (\sigma^2_{MLE}) + 2m/n \quad (3)$$

$$BIC = log\ (\sigma^2_{MLE}) + m\ log\ (n)\ /\ n \quad (4)$$

where, m is the number of data points, n is the number of parameters in the model and σ^2_{MLE} is the Mean Square Error (*MSE*) between the observed and simulated data. The model performance was evaluated by coefficient of determination (R^2), Nash-Sutcliffe efficiency (E_{NASH}), and bias ratio (R_{BIAS}) (Kalin et al. 2010).

In creating the ANN models, the number of hidden layers was set to 1 in order to avoid overtraining the results and to minimize error. The number of hidden neurons was varied from 2 to 20. The models were constructed in MATLAB version 8.4.0 (2014). Separate models were created for predicting salinity at each river station. To predict the salinity, 70% of the data series were selected randomly and assigned for training, and 30% of the data series were used for testing purposes (Rezaeian et al. 2007). Because it is important that random data include good distribution of low and high numbers, Levene’s Test was applied to examine the equality of variances between training and testing groups. The data sets were normalized between 0.05 and 0.95 using the normalization function in MATLAB. These steps were followed for both, the large scale and small scale models.

RESULTS AND DISCUSSION

Large Scale Apalachicola Bay Model

The observed mean daily salinity was 20.45 ppt in between 2002 to 2007. The minimum and maximum salinity were 0.11 ppt and 35.94 ppt, in the bay. There were 1812 days of data after the gaps were removed between 2002 and 2007. The tide level reached a maximum of 1.45 m and the lowest tide level was -0.27 m over this period of time. The average discharge measured at USGS Sumatra Gauge was around 600 m^3/s for this time period; maximum discharge was 4700 m^3/s and the minimum discharge was around 140 m^3/s.

The results of the Pearson and Spearman Correlation analyses (Table 2. 1) clarified that there was a strong relationship between bay salinity (*C*) and discharge (*Q*) (Pearson r = -0.60, p<0.001; Spearman r = -0.66, p<0.001). High correlations were also observed between the salinity on a given day, *C(t),* and the salinity of the previous day, *C(t-1)*. Tide level had a very weak but significant correlation with salinity (Spearman r = 0.051, p<0.001; Table 2. 1). Similarly, weak but significant correlations were observed with sea temperature (*T*) and evapotranspiration (*ET*). Having statistically significant correlations between the analyzed variables, we evaluated various input combinations for constructing the best ANN model.

Different combinations of inputs were evaluated for the large scale model (Table 2. 2). The input combination for the most optimal large scale model (lowest *AIC*) included tide level (*Y*), Apalachicola River discharge (*Q*), and sea temperature (*T*) (training dataset: *AIC* = 1.15, *RMSE* = 3.58, E_{Nash} = 0.53; testing dataset: *AIC* = 1.25, *RMSE* = 3.75, E_{Nash} = 0.49; Table 2). Since *C(t)* is highly correlated with *C(t-1)*, *C(t-1*) was also added to the input combinations to construct a new set of models. In addition, we added *Q(t-1)* and *Y(t-1)* due to their high correlations with salinity. The inclusion of *C(t-1)* significantly increased the strength of the model (training dataset *AIC* = 0.88; *RMSE* =2.57 E_{Nash} = 0.82; testing dataset: *AIC* = 0.96, *RMSE* = 2.54, E_{Nash} = 0.83;Table 2. 3). The model developed by including all three lag variables (i.e., *C(t-1)*, *Q(t-1)* and *Y(t-1)*) resulted in a stronger model than one with no lag variables, but was weaker than the one that had only *C(t-1)* as the lag (training dataset *AIC* = 0.92, *RMSE* = 2.62 E_{Nash} = 0.83; testing dataset: *AIC* = 1.03, *RMSE* = 2.62 E_{Nash} = 0.83;Table 2. 3). Although the models with lag variables were shown to be more accurate for predicting salinity in the bay, we did not utilize them because they put some limitations for their future applications and when/where there is no salinity data available. Hence, to predict the historical salinity in the bay, the model fed by discharge, tide level and sea temperature was chosen. This model proved to be more (most?) accurate for simulating the salinity levels at 15-22 ppt and incapable of predicting the variations in extreme low and high salinity levels (Figure 2. 6).

Small Scale Tidal River Models

The small scale models were created from 1.5 years of observed salinity data. There were six data observation sites across the East and St. Marks River (Figure 2. 3). The period and duration of salinity time series were not identical across all sites because the conductivity loggers malfunctioned at different times. As a result, data was omitted because of apparent temporary fouling on the conductivity logger. A description of small scale model results at each river site is provided below.

East River (ER)-1

At this site, there were 461 daily average measures of salinity. The mean daily salinity (*C*) was 0.39 ppt, minimum salinity was 0.01 ppt, and the maximum salinity was 14.9 ppt. The maximum and minimum discharge (*Q*) were 4757 m^3/s and 270 m^3/s, respectively, corresponding to these data. The highest tide level (*Y*) was 1.02 m and the salinity level reached a maximum 10.81 ppt When discharge increased salinity levels dropped and vice versa (Figure 2. 8 A). This behavior is something we expected and it confirmed the validity of data. Tide level had a statistically meaningful relationship with salinity (Spearman r = 0.30; p<0.001;Table 2. 4). Also, the simulated bay salinity had a moderate correlation with the site salinity (Spearman r = 0.39, p<0.001). There was a very weak correlation between salinity and precipitation (Spearman r = 0.03, p < 0.001).

Different input combinations were experimented to analyze potential interactions. Model performances were not significantly different from each other (Table 2. 5). Discharge and tide level input combination had the lowest *AIC* (training dataset *AIC* = -1.21, *RMSE* = 0.21 E_{Nash} = 0.94; testing dataset: *AIC* = -0.80, *RMSE* = 0.27, E_{Nash} = 0.93; Figure 2. 8 C and D). Therefore, this model was chosen for predictions.

ER-3

For this site, we calculated daily average salinity and water level for 401 days. In this time period, the mean salinity was 0.26 ppt while the minimum and maximum salinity were 0.01 ppt and 9.33 ppt. Maximum river discharge corresponding with these days was 4757 m^3/s and minimum was 279 m^3/s. Salinity peaks were observed on the same days as in ER-1 (Figure 2. 9 A and B) indicating the presence of high tides. While the salinity reached nearly 11.5 ppt at ER-1, it was nearly 9.2 ppt at ER-3 on the same day. The mean salinity was lower than the mean salinity at ER-1, as expected, given ER-3's greater distance inland.

There was a moderate correlation between salinity and tide level at this site (Spearman r = 0.32, p<00.1). Salinity, discharge, precipitation, and simulated bay salinity had a very weak correlation (Table 2. 6). Although, the discharge and tide level had a weak correlation with salinity, the model had the lowest training *AIC* and *BIC* (Table 2. 6) with these variables. Using tide level and river discharge input in combination, the model performed very well in simulating salinity at this sites (E_{Nash} (training)=0.91; E_{Nash} (testing) =0.55); Figure 2. 9).

ER-5

This site had less data available because the conductivity logger and transducer were stolen sometime between 12/10/2014and9/18/2015. Therefore, the number of observation days (229) was lowest at this site. This site is the most upstream site on East River. The mean salinity was 0.14 ppt at this site, and the minimum and maximum salinity was 0.10 ppt and 0.21 ppt respectively. The maximum river discharge at this site was 1203 m^3/s and the mean discharge was 533 m^3/s. The two highest tides on 10/4/2015 and 10/27/2015 had almost no impact at all on the salinity of this site (Figure 2. 10 A and B). The average salinity at this site was significantly lower than the average salinity of the other two sites.

Pearson and Spearman correlation results show there was a statistically significant relationship between salinity, Apalachicola River discharge and evapotranspiration (Table 2. 4). The ANN model was run with different input combinations, and the best model included tide level, river discharge, simulated bay salinity, and evapotranspiration (Table 2. 7). Figure 2. 9 shows the comparison of model generated salinity versus observed salinity (training dataset AIC = -3.84, $RMSE$ = 0.01 E_{Nash} = 0.76; testing dataset: AIC = -3.29, $RMSE$ = 0.01, E_{Nash} = 0.71).

St. Marks River (SMR)-1

Daily average salinity and water level was calculated for 294 days at this location. The mean salinity here was 0.65 ppt, the maximum was 12.59 ppt and the minimum salinity was 0.02ppt. The maximum tide level observed corresponding with this time period was 1.02 m. During this observation period, the maximum river discharge was 1203 m^3/s and the minimum discharge was 270.42 m^3/s (Figure 2. 11). The highest tide level was on 10/27/2015. Salinity increased with tide level (Figure 14-B). When river discharge increased, the salinity was clearly lowered (Figure 14-AB). The Spearman and Pearson correlation results (Table 2. 4) showed a strong relationship between salinity, tide (Spearman r = -0.56, p<0.001), discharge (Spearman r =0.63, p<0.001), and simulated bay salinity (Spearman r = 0.65, p<0.001). However, evapotranspiration and precipitation had weak correlations with salinity.

The best model performance was captured from the model using tide level and discharge as inputs (training dataset *AIC* = -3.32, *RMSE* = 0.34 E_{Nash} = 0.92; testing dataset: *AIC* = 0.18, *RMSE* = 0.28, E_{Nash} = 0.88; Figure 2. 11 C and D). Model predictions were consistently underestimated for salinities <0.8 ppt and overestimated for salinities in the range of 0.8 – 5 ppt. The model accurately predicted salinities >5 ppt.

SMR-3

This site is located near the middle of the St. Marks River study transect. A total of 294 days of daily average salinity and water level was calculated after deleting data gaps. On 10/2/2015, tide level was elevated to a maximum of 0.16 m and the salinity increased to a maximum of 7.95 ppt (Figure 2. 12 A and B). This coincided with the day of highest salinity observed at SMR-1. The minimum salinity was 0.02 ppt and the maximum salinity was 7.95 ppt, with mean salinity 0.19 ppt. The mean river discharge was 548 m^3/s corresponding to this time period. When the discharge increased, salinity was lowered (Figure 2. 12 A). Salinity levels were driven by higher than normal tide levels (Figure 2. 12 B). On10/27/2015, the daily average salinity level was nearly 8.4 ppt at this station and at same day daily average salinity was 12.5 ppt at the downstream site SMR-1.

At this site, strong correlations were observed between salinity, discharge (Spearman r = 0.511 p<0.001;Table 2. 4), and evapotranspiration (Spearman r = -0.604 p<0.001). To predict historical salinity, models were created with different input combinations (Table 2. 9). The training and testing *AIC* results were not very different from each other. The model including tide level and discharge had the lowest *AIC* and highest performance (training dataset *AIC* = -2.09 *RMSE* = 0.07 E_{Nash} = 0.98; testing dataset: *AIC* = -1.42, *RMSE* = 0.10, E_{Nash} = 0.77). Figure 2. 12 C and D illustrate high model performance for training and testing datasets. The difference between observed data and simulated data never exceeded 0.2 ppt and the residuals averaged around 0.05 ppt. The model accurately predicted a wide range of salinities; however, for very low salinities under 0.25 ppt the model predictions show slight underestimation or overestimation.

SMR-5

The site is the farthest from the bay on the St. Marks River transect. There were 273 days of daily average salinity and water level observations at this site. The mean salinity was 0.13 ppt. During this data collection period, the maximum tide level observed was 1.02 m, corresponding to a high salinity level of 0.50 ppt. The mean river discharge during this period was 538 m^3/s. This site did not experience elevated salinity (Figure 2. 13). There was a very small increase in salinity on 10/27/2015 during the highest observed tide level, but the observed salinity was still <0.1 ppt. There was a strong correlation between salinity, discharge (Spearman r = 0.764, p<0.001) and evapotranspiration (Spearman r = -0.57, p<0.001; Table 2. 4). Also, there was a moderate correlation between simulated bay salinity and SMR-5 salinity (r= -0.48, p<0.001).

At this site, the lowest *AIC* (training dataset *AIC* = -2.87 *RMSE* = 0.02E_{Nash} = 0.88; testing dataset: *AIC* = -2.76, *RMSE* = 0.02, E_{Nash} = 0.83;Table 2. 10) was captured by the model that included tide level, river discharge, simulated bay salinity, evapotranspiration, and precipitation. Model performance for training and testing datasets were very high (Figure 2. 13).

SUMMARY AND CONCLUSIONS

Freshwater forested wetlands in coastal regions are highly sensitive ecosystems. Salinity from the ocean and freshwater inflow from the terrestrial land and river floods need to be balanced so that salinity levels in the soil stays at 0.5 ppt which is unique for these ecosystems.

Unfortunately, sea level rise, changes in precipitation patterns and amounts due to climate change, and management of rivers systems and watersheds (e.g. damming, excess water withdrawal from rivers or aquifers, etc.) are threating to change this balance. Examining the salinity regime and the Apalachicola River management effects in tidal freshwater forested wetlands ecology was the main goal of this study. It is not only important to understand the existing salinity regime and fluctuations in this area, but also assess consequences of future potential changes in sea level and freshwater inflow. In that sense models are indispensable tools to answer such questions.

In this chapter, artificial neural networks (ANN) based models were developed to explore the potential impacts of sea level rise and changes in Apalachicola River discharge on the salinity of Apalachicola Bay and along the St. Marks River and East River tributaries. Although it is a black box type model, ANN's are widely used in highly complex, nonlinear systems. Result showed that $Y(t)$, $Q(t)$, $Q(t\text{-}1)$, $T(t)$, and $C(t\text{-}1)$ were significantly correlated to $C(t)$. Having high correlations with the day before values clearly indicate the memory effect, which is a common phenomenon in natural systems. On the other hand, inclusion of $Q(t\text{-}1)$ and $C(t\text{-}1)$ in a model makes forecasting over a long period very difficult. Therefore, they were excluded from the forcing variables in predictive models.

ANN type of models are known to require extensive amounts of data. To develop a good, reliable ANN model one needs a long data. The longer the data the better the model will be. In this study certain variables had very long data records (tide level and river discharge), but not all of them (wind speed and wind direction). However, we were able to predict them to extend the record length. For instance, sea temperature data was not available for the whole period, but we were able to generate missing periods using air temperature, which is widely available.

The ANN model developed for predicting bay salinity was able to simulate salinity levels with good accuracy, both during training and testing periods. Since the Apalachicola Bay salinity drives salinities along the St. Marks River and East River, the salinity predictions from the bay can be used as input in ANN models at the individual sites (along the St. Marks River and East River) to increase model predictive skills. The small scale models along the rivers, especially the ones closer to the bay (ER-1, SMR-1), had very good performances during training and testing periods.

Model sensitivity results show that salinity levels at sites closer to the bay are driven by the tide level and the Apalachicola river discharge. At the middle sites ER-3 and SMR-3, the average salinity was more impacted by the tide level. Therefore, more frequent high tides will increase the average salinity at these sites. The SMR-3 site had much lower salinity levels than ER-3 because St. Marks River intersection zone is larger than East River connection to the Apalachicola River. Therefore, the best model results were obtained with tide level and Apalachicola discharge input combinations. At the most upstream sites ER-5 and SMR-5, model results showed that tide level, discharge, precipitation, and evaporation all contribute to affect the salinity. At the SMR-5 station the salinity was mostly controlled by the Apalachicola River discharge likely because of a large, close-by intersection between the Apalachicola River and St. Marks River that readily delivers freshwater from the Apalachicola River.

There was a gradual reduction in salinity moving from the downstream to upstream at the East River. However, the St. Marks River had a different behavior. Salinity levels were generally lower along this river because of a bigger more immediate connection with the Apalachicola River. The East River's connection is smaller than St. Marks River and this leads to elevated levels of salinity. At the ER-5, the salinity level was higher than STM-5 because there is a more restricted connection with the Apalachicola River. Because of this, the amount of freshwater delivered from the Apalachicola River to the East River is probably lower than STM-

5. There is another smaller connection between St. Marks River and East River further down river, however these results suggest that the East River maintains higher salinity than the St. Marks River. Therefore, it is difficult to say if ER- 5 salinity level is being controlled by the Apalachicola River discharge as much as SMR-5. ER-5 could be the transition zone from tidal to non- tidal while SMR-5 behaves like a freshwater site. Dominant forest canopy species in the riparian swamps would suggest this is the case (see Chapter 3).

The models developed in this chapter are used in the next chapter to answer some what-if type scenario questions. In the next chapter combinations of sea level rise and increase/decrease in Apalachicola River discharge scenarios are developed. The implications of those potential changes on the salinity levels in the bay and at those individual sites are assessed with the ANN models developed in this chapter. Potential changes in the salinities are correlated to vegetation characteristics and qualitative projections are made based on those individual scenarios.

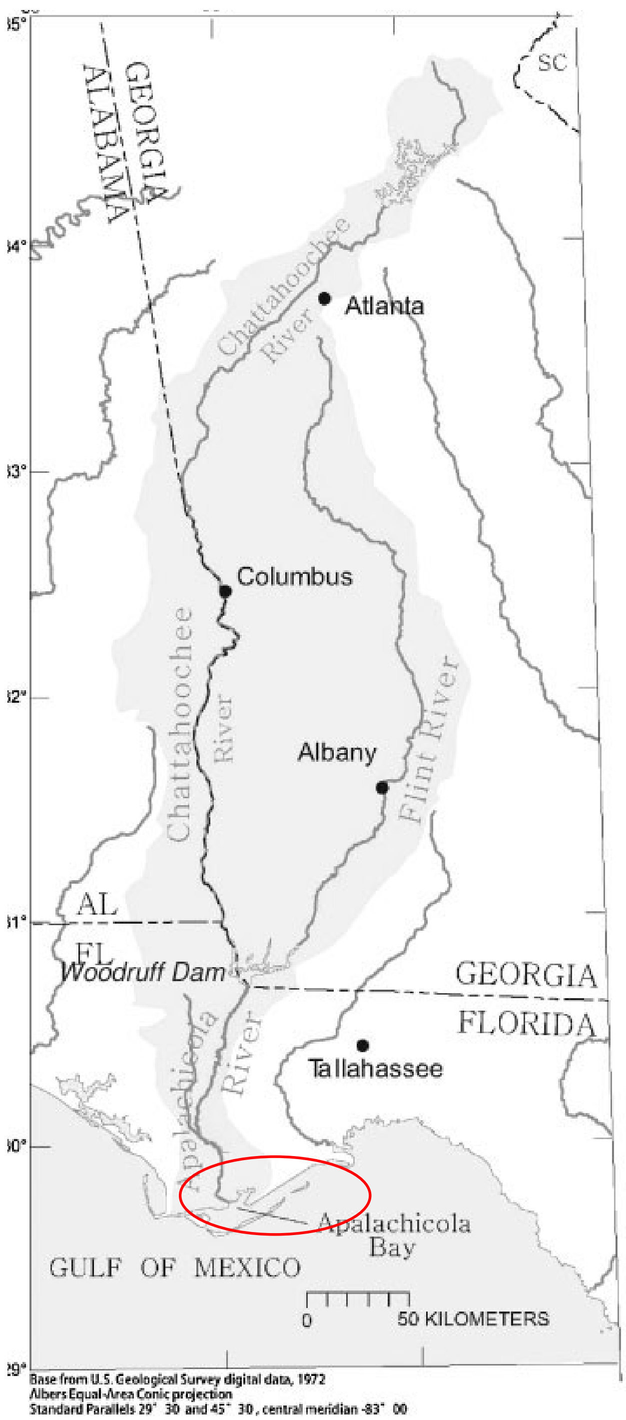

Figure 2. 1 : Location map of Apalachicola Chattahoochee and Flint (ACF) River Basin. Circled area indicates study focal area.

Figure 2. 2: Map of Apalachicola Bay including the Apalachicola River, contributing distributary rivers, and barrier island passes, which are entrance and exist of saline water, around the bay

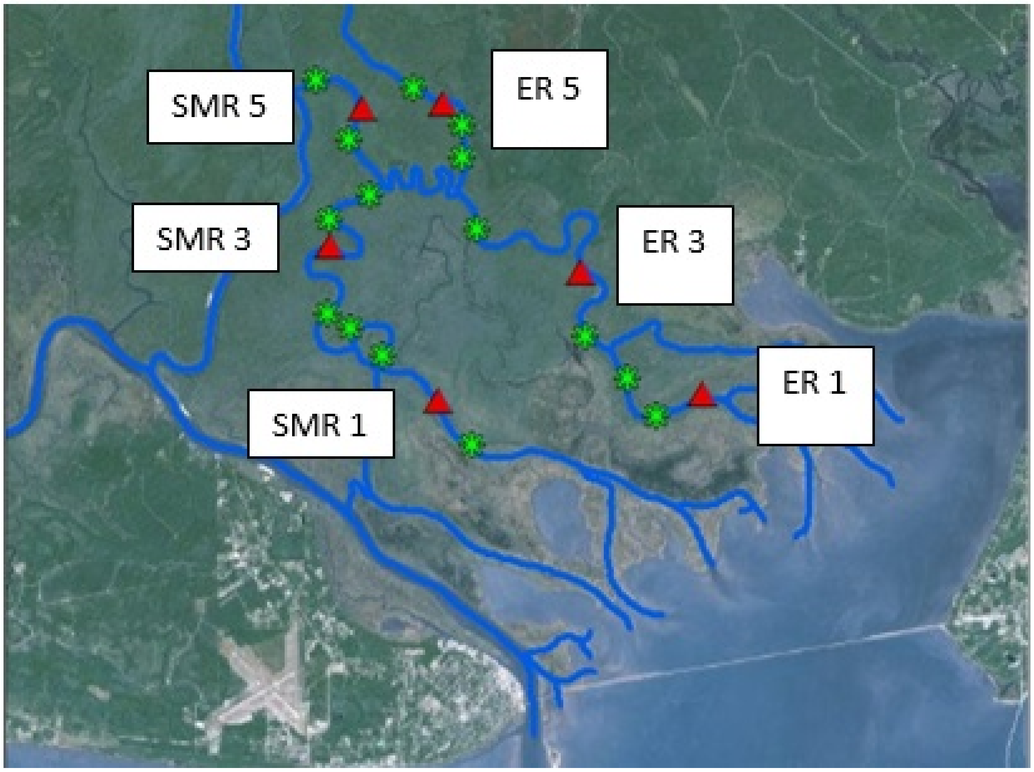

Figure 2. 3: Freshwater tidal river sites along the East River and St. Marks River. Red triangles show the small scale models salinity observation stations and green stars are sign for forest monitoring stations on the St Mark River (SMR) and East River (ER).

Figure 2. 4: Photographs of (A) conductivity logger and pressure transducer located at site SMR-3 along the St. Marks River, (B) Solinist barometer, and (C) a drainage inlet creek along the East River.

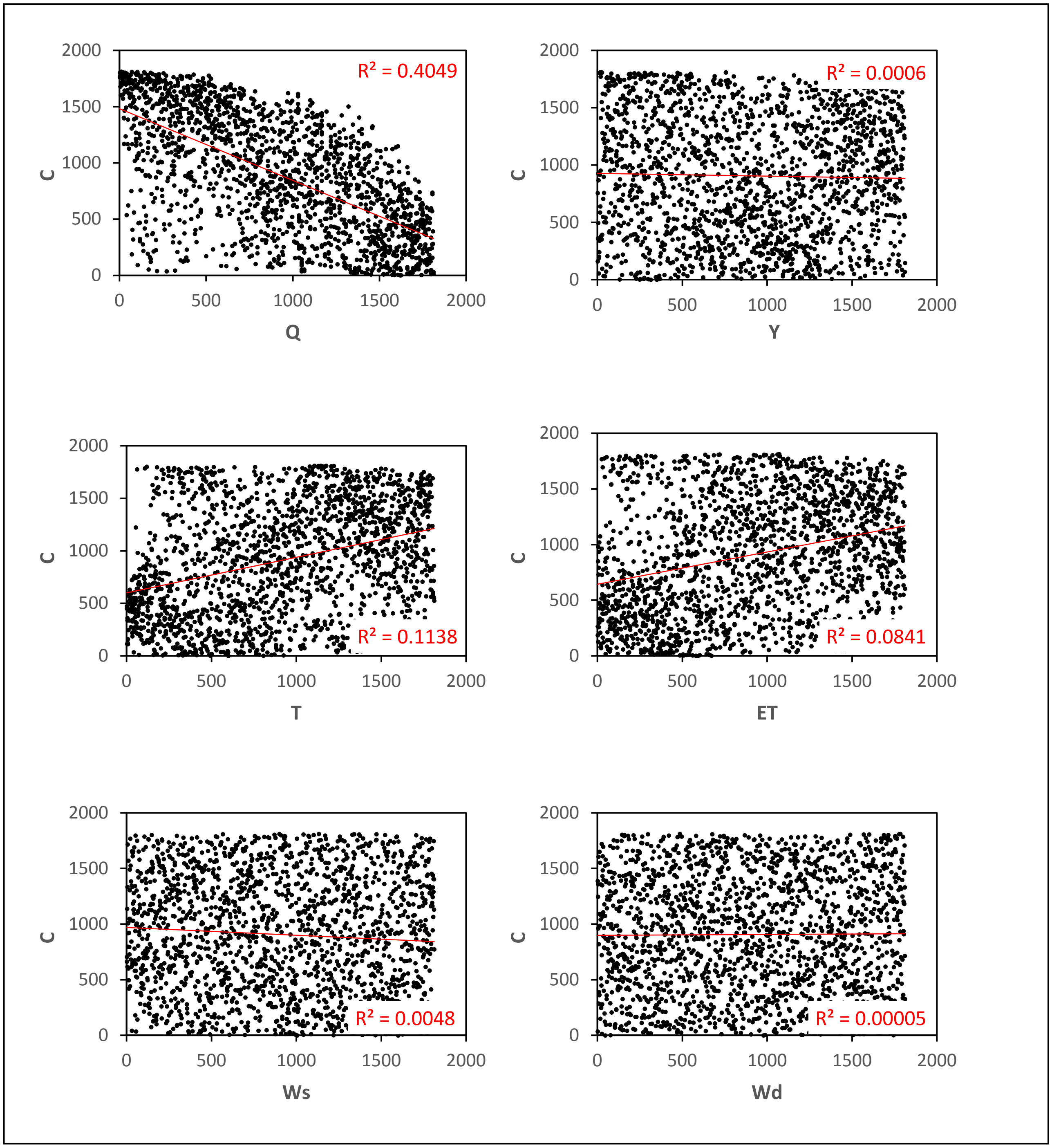

Figure 2. 5: Scatter plots showing rank of Apalachicola Bay salinity (C) verses rank of river discharge (Q), tide level (Y), water temperature at the Bay (T), evapotranspiration (ET), wind speed (Ws), and cosine of wind direction (Wd).

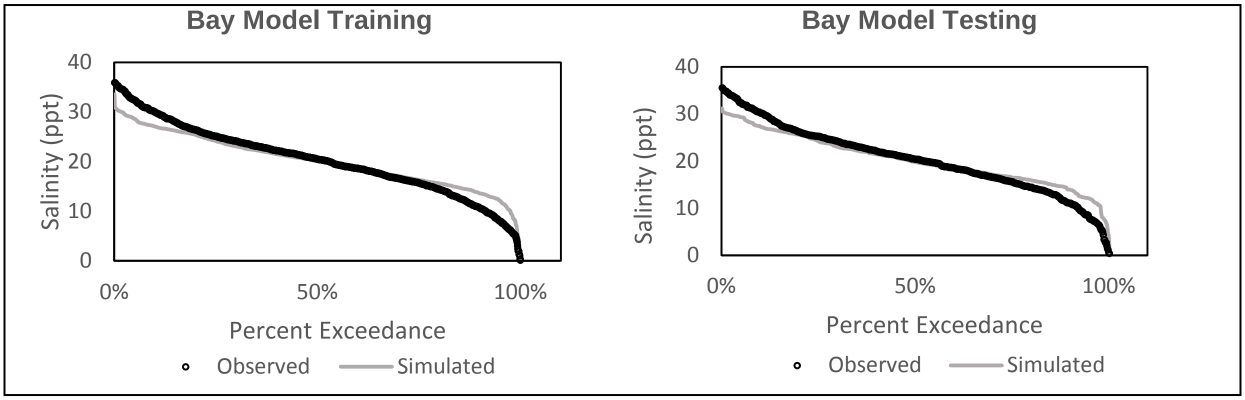

Figure 2. 6: ANN model performance (training and testing) for salinity at the Apalachicola Bay

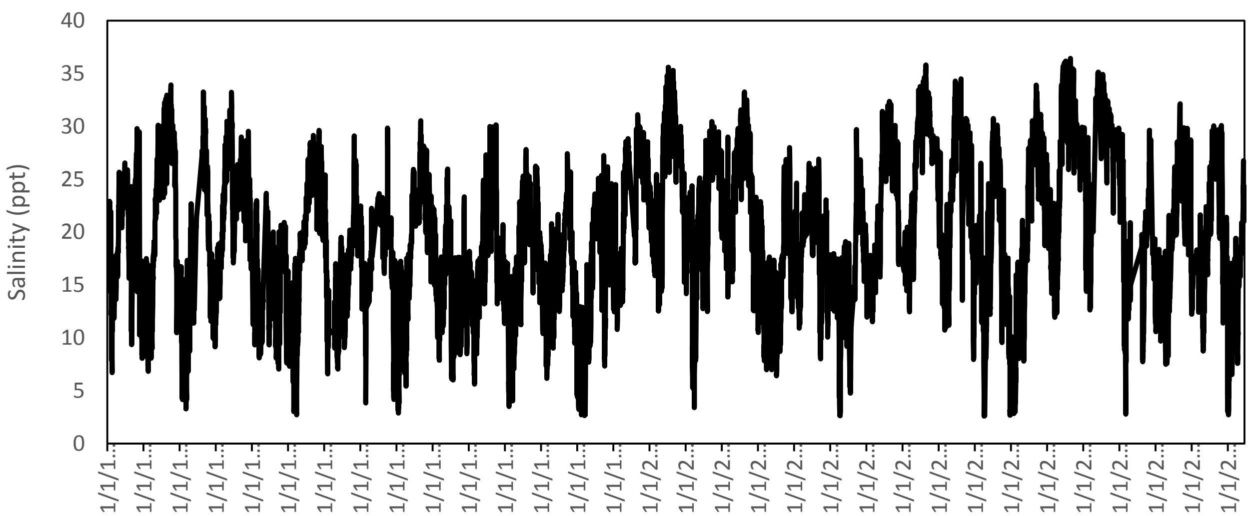

Figure 2. 7: Estimated salinity time series in the Apalachicola Bay with the ANN model.

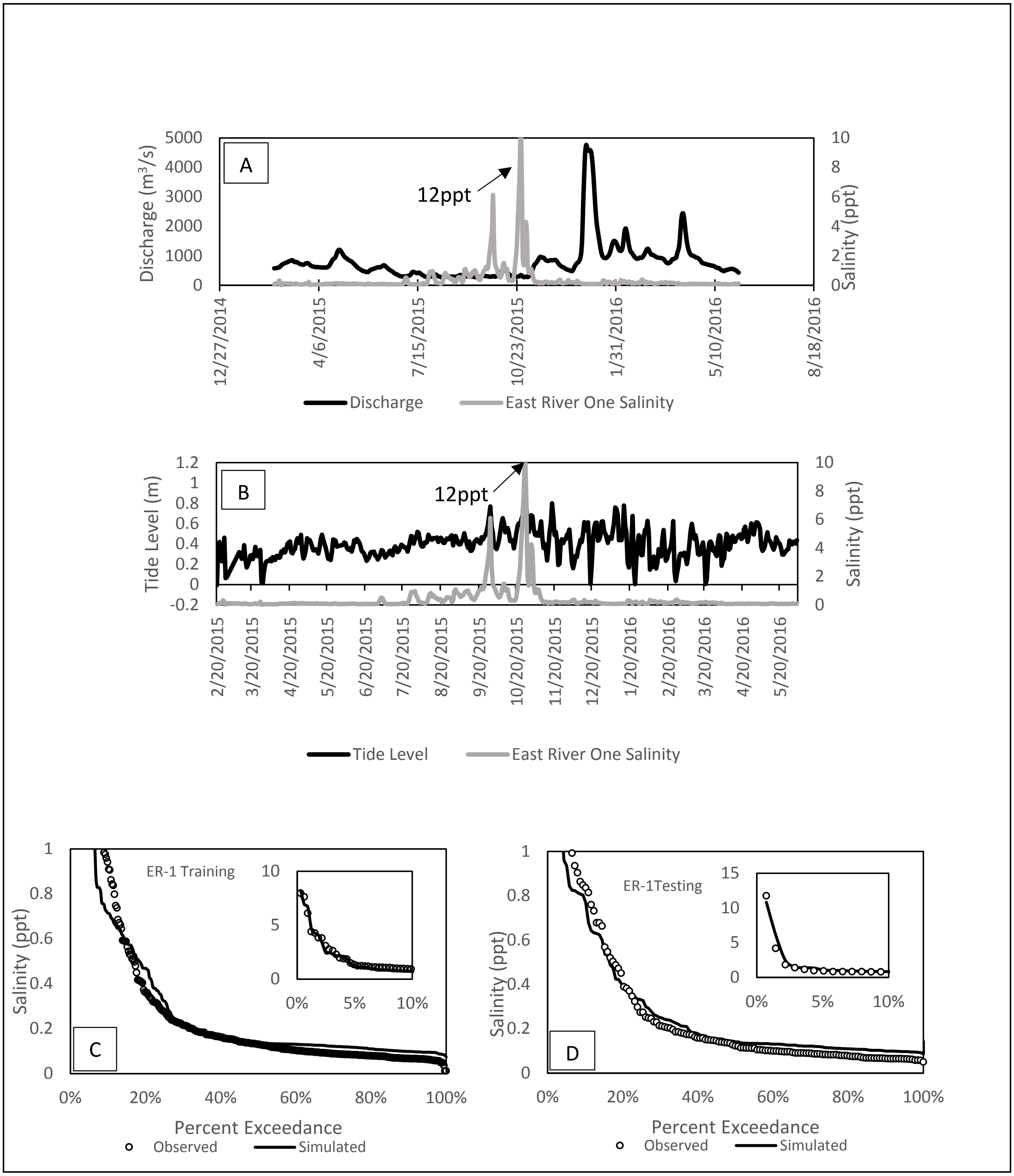

Figure 2. 8: (A) ER-1 salinity and discharge graph; (B) ER-1 salinity and tide level graph; (C and D) ANN model performance (training and testing) at ER-1. The small graphs show the full performance of ANN model (training and testing) predictions.

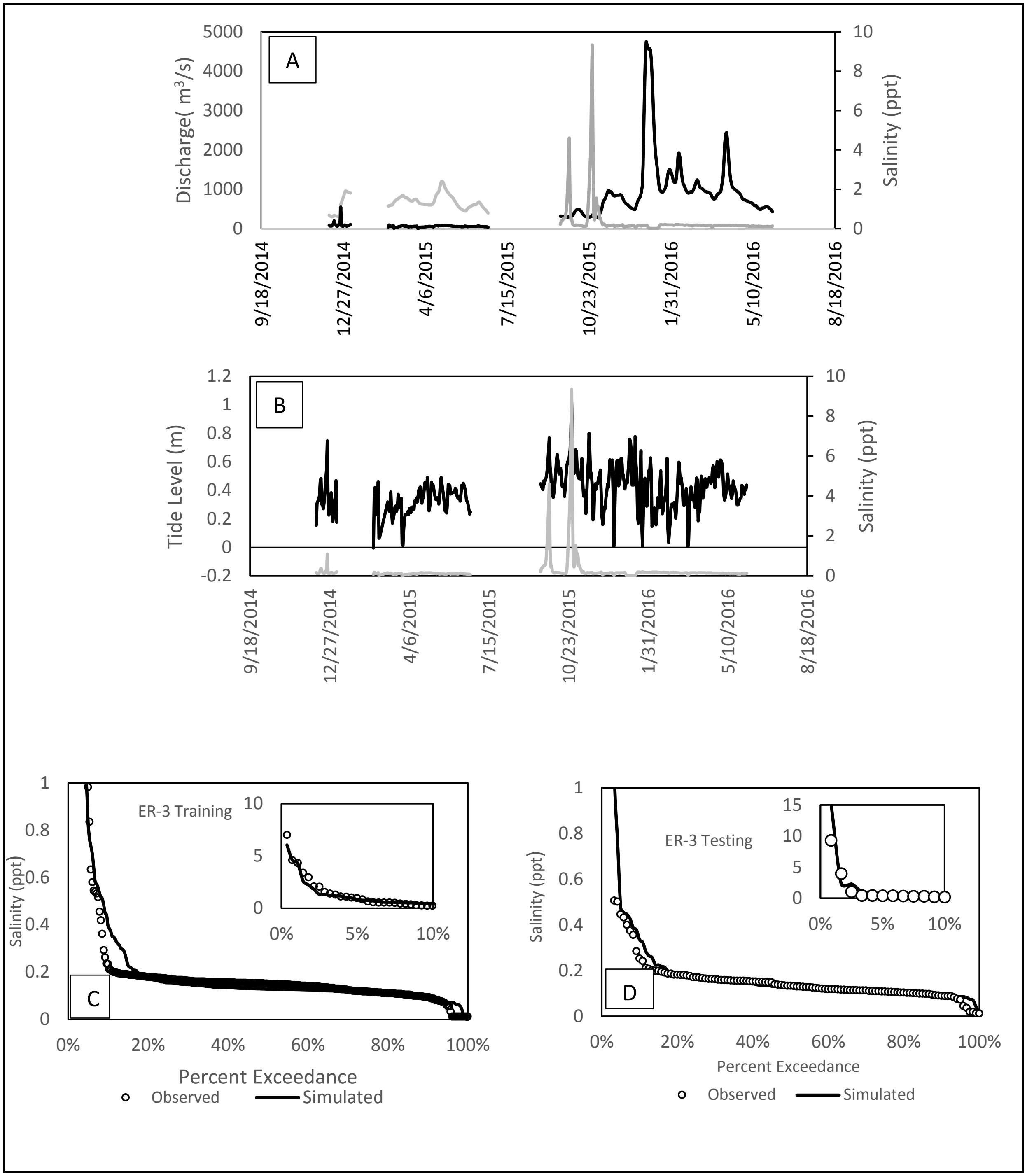

Figure 2. 9: (A) ER-3 salinity and discharge graph; (B) ER-3 salinity and tide level graph; (C and D) ANN model performance (training and testing) at ER-3. The small graphs show the full performance of ANN model (training and testing) predictions.

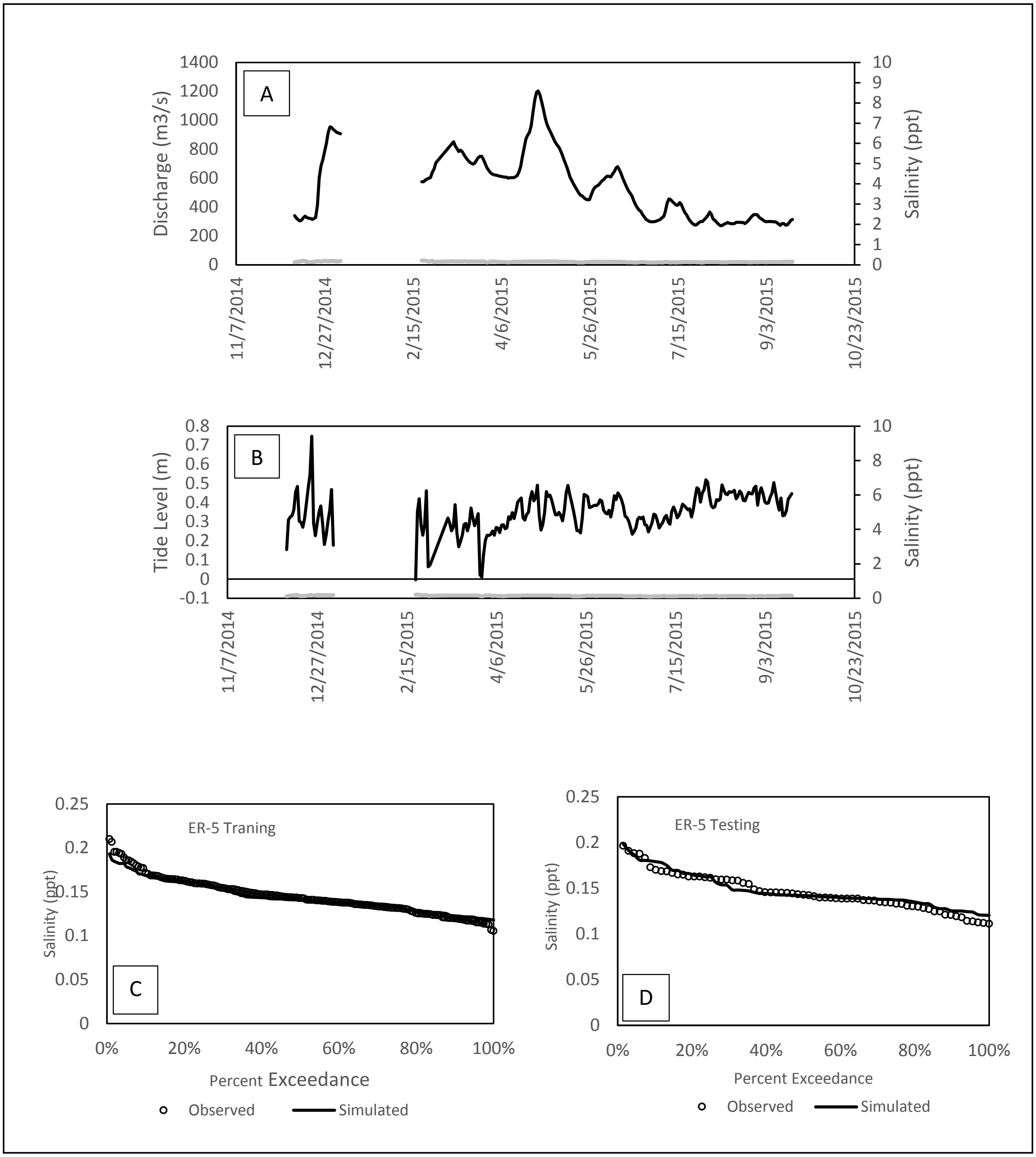

Figure 2. 10: (A) ER-5 salinity and discharge graph; (B) ER-5 salinity and tide level graph; (C and D) ANN model performance (training and testing) at ER-5. The small graphs show the full performance of ANN model (training and testing) predictions.

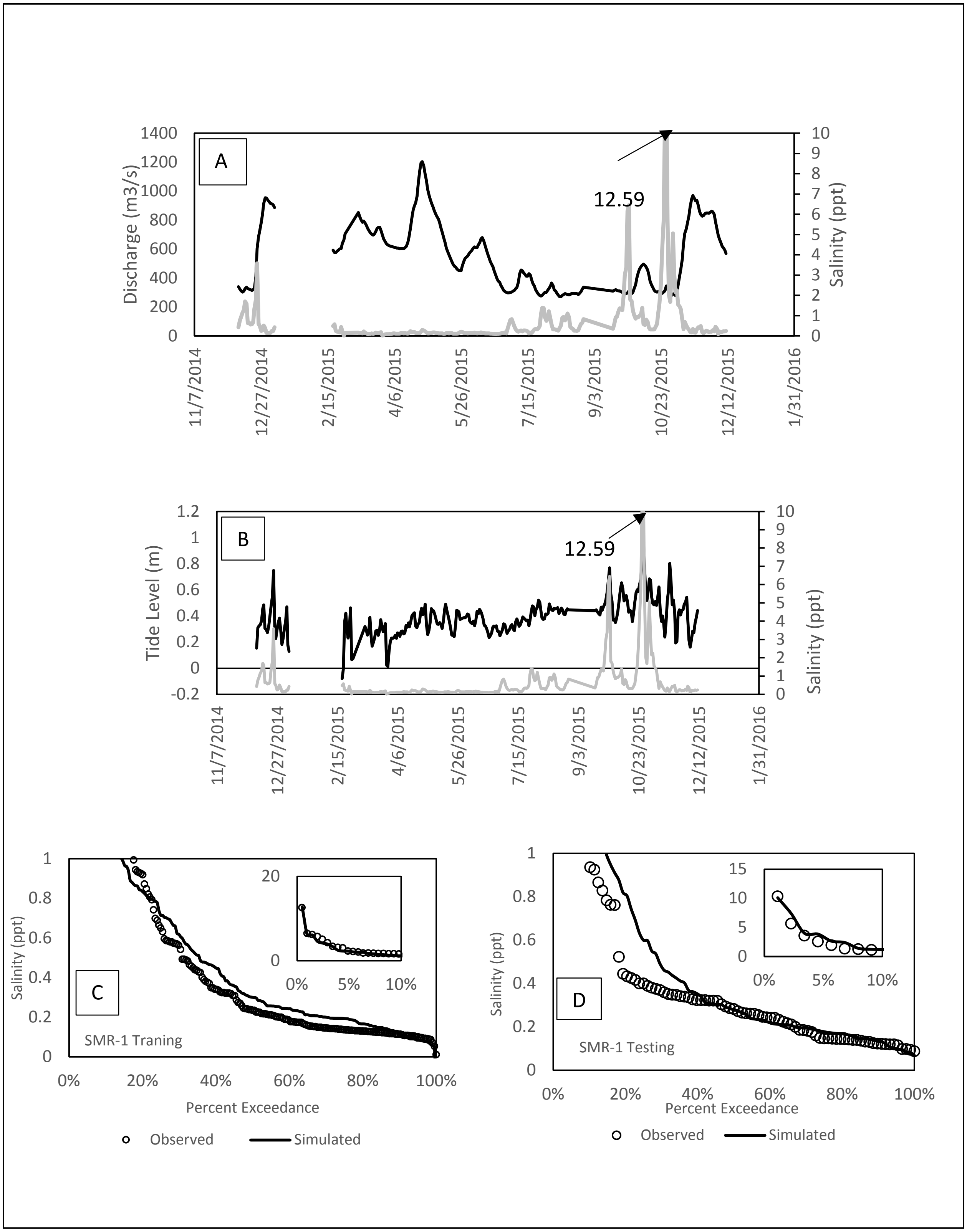

Figure 2. 11: (A) SMR-1 salinity and discharge graph; (B) SMR-1 salinity and tide level graph; (C and D) ANN model performance (training and testing) at SMR-1. The small graphs show the full performance of ANN model (training and testing) predictions.

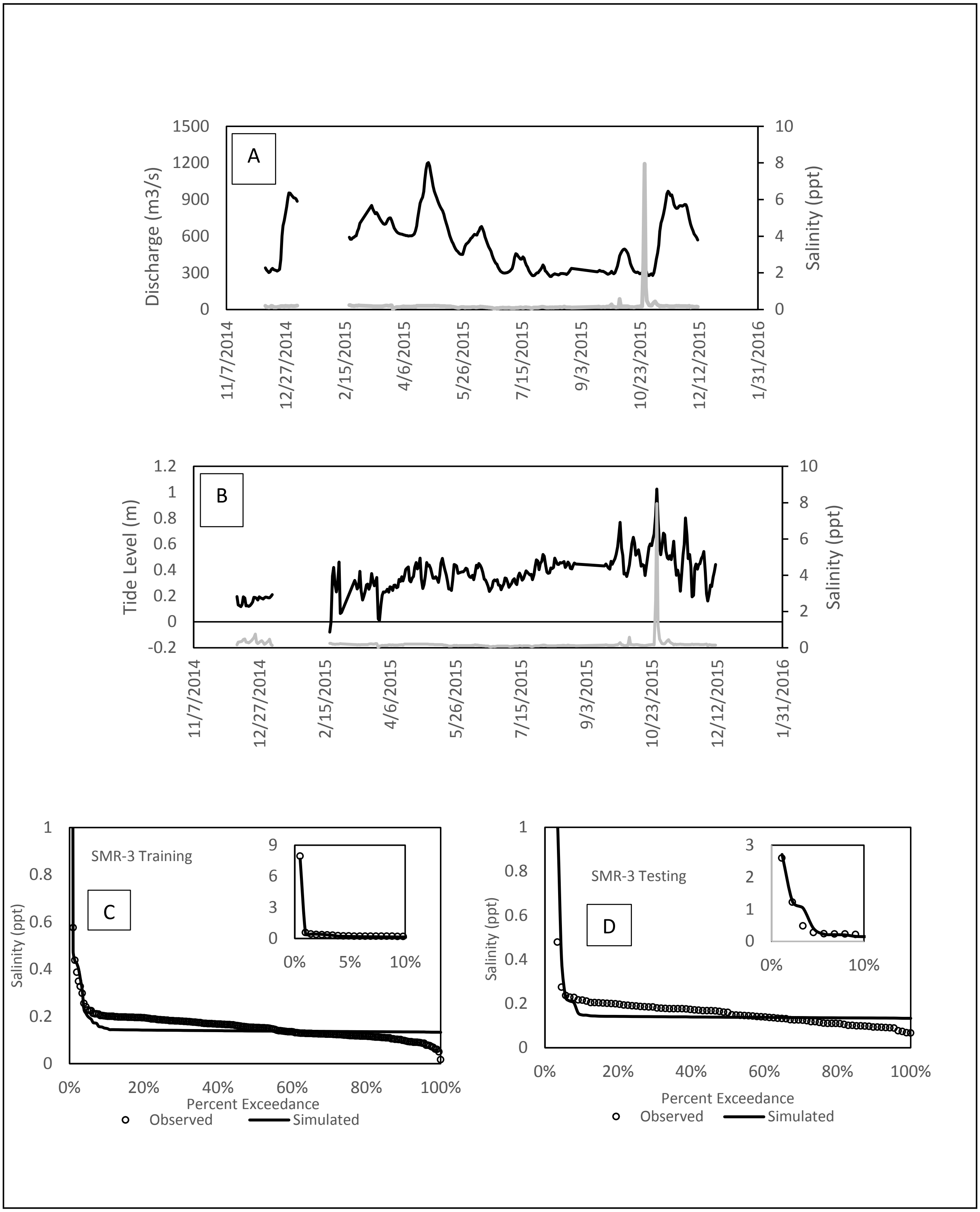

Figure 2. 12: SMR-3 salinity and discharge graph; (B) SMR-3 salinity and tide level graph; (C and D) ANN model performance (training and testing) at SMR-3. The small graphs show the full performance of ANN model (training and testing) predictions.

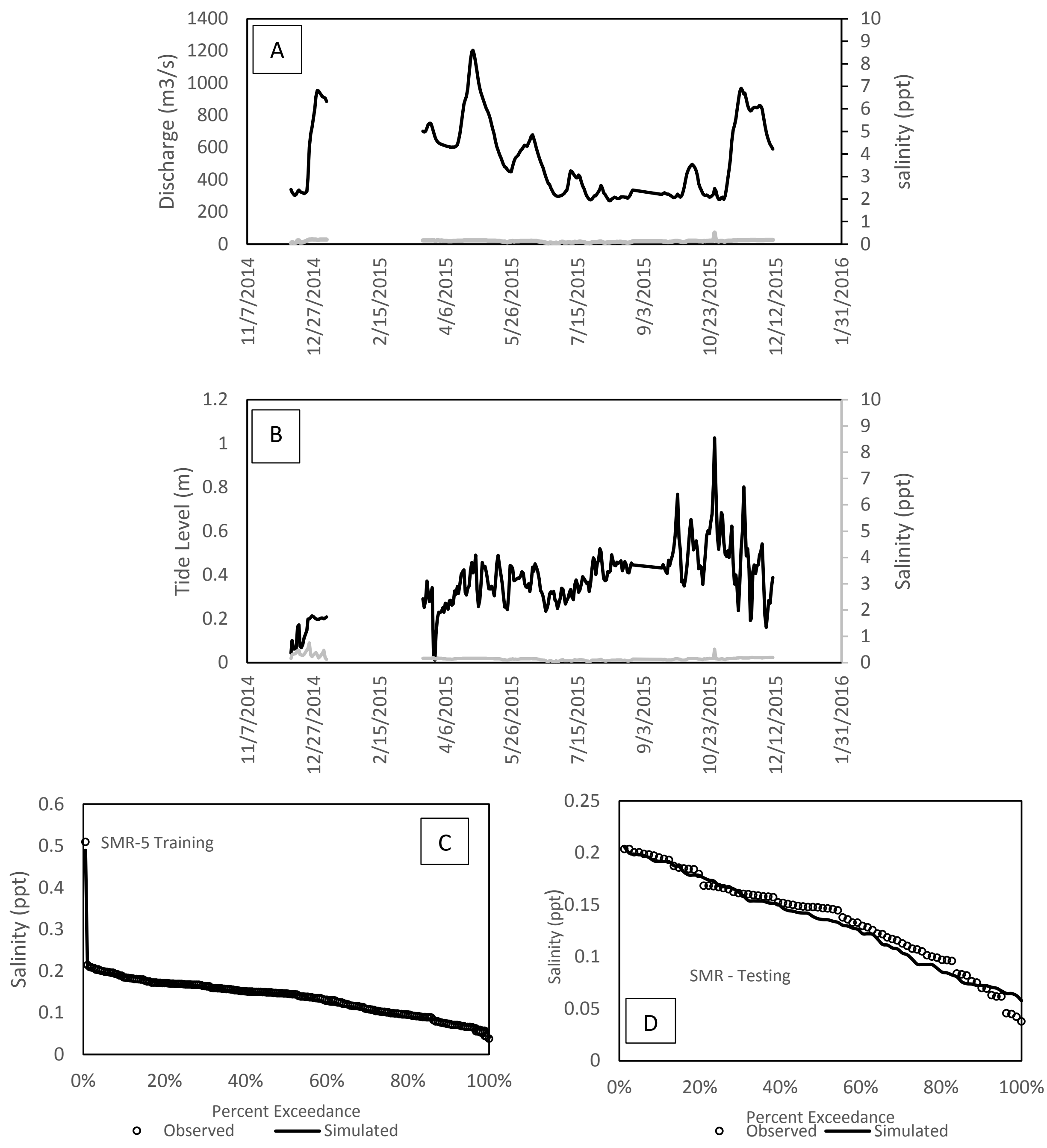

Figure 2. 13: SMR-5 salinity and discharge graph; (B) SMR-5 salinity and tide level graph; (C and D) ANN model performance (training and testing) at SMR-5. The small graphs show the full performance of ANN model (training and testing) predictions.

Table 2. 1: Pairwise Pearson and Spearman correlations between the Apalachicola Bay salinity (C) and river discharge (Q), tide level (Y), sea temperature (T), evapotranspiration (ET), wind speed (WS) and cosine of wind direction (WD). All correlations were sign

	Q	***Y***	***T***	***ET***	***WS***	***(WD_{cos})***
Pearson	-0.60	-0.04	0.29	0.18	-0.01	0.09
Spearman	-0.66	0.05	0.33	0.22	0.02	0.12

Table 2. 2: ANN performance under different input combinations at Apalachicola Bay.

		Training				**Testing**			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	6	0.48	3.63	1.16	1.18	0.45	3.68	1.22	1.25
Q, Y, T	6	0.53	3.58	1.15	1.18	0.49	3.75	1.25	1.29
Q, Y, T, ET	6	0.55	3.60	1.17	1.20	0.48	3.88	1.31	1.36

Table 2. 3: Model results at Apalachicola Bay which was run with one-day lag of inputs

		Training				**Testing**			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y, T, ET, Q(t-1)	6	0.52	3.45	1.14	1.18	0.48	3.59	1.26	1.32
Q, Y, T, ET, Y(t-1)	6	0.52	3.55	1.16	1.20	0.49	3.69	1.28	1.34
Q, Y, T, ET, C(t-1)	6	0.82	2.57	0.88	0.92	0.83	2.54	0.96	1.02
Q, Y, T, ET, Q(t-1), Y(t-1)	6	0.56	3.58	1.18	1.22	0.52	3.78	1.32	1.39
Q, Y, T, ET, Q(t-1), Y(t-1), C(t-1)	6	0.83	2.62	0.92	0.96	0.83	2.62	1.03	1.10

Table 2. 4: Pearson and Spearman Correlations between salinity (C) and discharge (Q), tide level (Y), evaporation (ET), precipitation (P), and simulated bay salinity (C_{Bay}) at the Saints Mark and East Rivers.

		Q	***Y***	***ET***	***P***	***C_{Bay}***
East River (ER-1)	Pearson	- 0.19	0.45	-0.12	0.06	0.42
	Spearman	-0.32	0.30	-0.37	0.03	0.39
East River (ER-3)	Pearson	- 0.15	0.42	-0.07	0.10	0.17
	Spearman	0.05	0.32	-0.34	0.11	-0.19
East River (ER-5)	Pearson	0.59	-0.03	-0.57	0.01	-0.50
	Spearman	0.63	-0.03	-0.60	-0.06	-0.26
St. Marks River (SMR-1)	Pearson	-0.33	0.63	-0.20	0.11	0.47
	Spearman	-0.56	0.63	-0.34	0.14	0.65
St. Marks River (SMR-3)	Pearson	-0.03	0.36	-0.11	0.07	0.11
	Spearman	0.51	0.18	-0.60	0.01	-0.45
St. Marks River (SMR-5)	Pearson	0.62	0.23	-0.47	0.06	-0.48
	Spearman	0.76	0.06	-0.57	0.06	-0.46

Table 2. 5: ANN performance under different input combinations at ER-1

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	5	0.94	0.21	-1.21	-1.18	0.93	0.27	-0.80	-0.78
Q, Y, C_{Bay}	5	0.93	0.23	-1.10	-1.06	0.92	0.29	-0.67	-0.64
Q, Y, C_{Bay}, ET	5	0.93	0.20	-1.19	-1.14	0.92	0.29	-0.60	-0.57
Q, Y, C_{Bay}, ET, P	5	0.93	0.20	-1.15	-1.09	0.62	0.64	0.14	0.18

Table 2. 6: ANN performance under different input combinations at ER-3

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	6	0.91	0.16	-1.38	-1.34	0.55	0.38	-0.41	-0.39
Q, Y, C_{Bay}	6	0.85	0.22	-1.07	-1.02	0.90	0.25	-0.66	-0.64
Q, Y, C_{Bay}, ET	6	0.92	0.15	-1.33	-1.27	0.65	0.36	-0.26	-0.23
Q, Y, C_{Bay}, ET, P	6	0.74	0.15	-1.32	-1.26	0.77	0.18	-0.75	-0.72

Table 2. 7: ANN performance under different input combinations at ER-5

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	7	0.52	0.01	-3.72	-3.68	0.54	0.01	-3.54	-3.53
Q, Y, C_{Bay}	7	0.63	0.01	-3.69	-3.63	0.60	0.01	-3.25	-3.22
Q, Y, C_{Bay}, ET	7	0.76	0.01	-3.84	-3.78	0.71	0.01	-3.29	-3.26
Q, Y, C_{Bay}, ET, P	7	0.73	0.01	-3.86	-3.78	0.63	0.01	-3.21	-3.17

Table 2. 8: ANN performance under different input combinations at SMR-1

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	6	0.92	0.34	-0.32	-0.29	0.88	0.28	0.182	0.14
Q, Y, C_{Bay}	6	0.84	0.51	0.09	0.20	0.92	0.37	0.75	0.70
Q, Y, C_{Bay}, ET	6	0.93	0.33	-0.14	-0.01	0.80	0.33	0.97	0.92
Q, Y, C_{Bay}, ET, P	6	0.87	0.37	-0.10	0.25	0.77	0.46	1.56	1.50

Table 2. 9: ANN performance under different input combinations at SMR-3

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	3	0.98	0.07	-2.09	-2.05	0.77	0.10	-1.42	-1.45
Q, Y, C_{Bay}	3	0.85	0.07	-2.13	-2.10	0.66	0.09	-1.54	-1.56
Q, Y, C_{Bay}, ET	3	0.99	0.05	-2.29	-2.26	0.65	0.04	-2.20	-2.22
Q, Y, C_{Bay}, ET, P	3	0.99	0.05	-2.42	-2.39	0.56	0.04	-2.36	-2.38

Table 2. 10: ANN performance under different input combinations at SMR-5

		Training				Testing			
Inputs	**# of Neurons**	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***	***E_{Nash}***	***RMSE (ppt)***	***AIC***	***BIC***
Q, Y	10	0.88	0.02	-2.87	-1.88	0.83	0.02	-2.76	-1.96
Q, Y, C_{Bay}	10	0.94	0.01	-3.06	-1.82	0.82	0.02	-2.98	-1.90
Q, Y, $C_{Bay,}$ ET	10	0.82	0.02	-2.67	-1.52	0.64	0.02	-2.57	-1.60
Q, Y, $C_{Bay,}$ ET, P	10	0.93	0.01	-3.05	-1.81	0.85	0.02	-2.94	-1.89

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Chapter 3. Examining Salinity Patterns and Freshwater Forested Wetlands Along a Tidal Gradient

ABSTRACT

Along coastal rivers, tidal freshwater forested wetlands can occur where proximity to tidal waters occur. Moving upriver from the bay, it is usually apparent where these forested wetlands start as they normally transition from brackish or freshwater marshes. However, it is hard to identify the boundary of the tidal freshwater forested wetlands because tidal fluctuations can be subtle and it is unclear if forest species and edaphic conditions shift along this transition. This chapter details a study illustrating the change in forest community composition and corresponding salinity regime along the St. Marks River and East River, both distributary rivers of the Apalachicola River, in northwest, Florida. To document changes in forest communities, survey plots (500 m^2) were established along a freshwater tidal gradient near the river-edges of the St. Marks River (n=11) and the East River (n=10). Species basal area and density were used to calculate importance values for each canopy species (>2.5 cm DBH) detected. Based on a previous study (Anderson and Lockaby 2013) that identified species indicative of tidal and non-tidal forests, aggregate importance values were calculated to compare the changing contribution of tidal and non-tidal species across the gradient. Within the study reach, both rivers showed a transition from tidally dominant to a mix of tidal and non-tidal forest species. Using daily salinity models developed for these wetlands, we evaluated 30-year modeled and interpolated measures of salinity across the gradient to characterize salinity patterns in this zone, particularly focusing on transition areas between tidal and mixed forest. For both rivers, where saltwater intrusions (daily average >3 ppt) averaged less than two days per year, forested wetlands were composed of a mixed assemblage. Where average saltwater intrusion exceeded two days per year, forest species transitioned to an assemblage of entirely tidal species. Although it is uncertain how much salinity or other aspects of tidal influence are affecting species

composition, both rivers showed shifts that corresponded to salinity albeit at different proximities from the bay. These similarities may assist with modelling efforts for predicting forest shifts that may occur with possible changes to sea level or river discharge.

INTRODUCTION

Tidal freshwater forested wetlands are important components of coastal forests. These wetlands are located along tidally influenced rivers and are normally bordered by upstream non- tidal wetland forests and downstream tidal marshes (Barendregt and Swarth 2013). Tidal pulses are a controlling factor on tidal freshwater wetlands, however, these ecosystems are generally characterized by low salinities year round (<0.5ppt; Doyle et al. 2007) due to the influence of river flows. Periodic spikes in water salinity can occur however whenever extremely low river flows or high tidal surges occur (Doyle et al. 2007). Salinity changes are expected to accompany sea level rise (Craft et al. 2007) because of the increasing reach of more saline waters up rivers. There are some questions regarding the long-term response of these forests to the influence of sea level rise (Anderson et al. 2013). Hydrology is known to be the most important factor influencing the structure and composition of wetlands (Mitsch et al. 2007). When wetlands transition from tidal to non-tidal, a significant hydrologic shift occurs (Day et al. 2007). The influence of tides gradually decreases further upriver (Baldwin 2007) and there has been some documentation about the differences between tidal influences and forest communities. Forest structure has also been shown to change along this transition zone with tidal freshwater swamps often displaying smaller tree stems at higher densities (Anderson et al. 2013). However, potential changes in these ecosystems are currently poorly understood. Quantifying saltwater intrusion and changes in river discharge impacts is challenging but necessary task to facilitate a better understanding of possible changes to the ecosystem with potentially important biological, social, and economic consequences.

Saltwater intrusion and flooding are two important conditions that directly affect the tidal freshwater forested wetland occurrence and species composition (Krauss et al. 2009).Tidal freshwater wetlands typically get adequate freshwater flows that keep surface water salinities < 0.5 parts per thousand (ppt) (Corwadin et al. 1979). When the salinity level regularly approaches 2 ppt, freshwater forested wetlands are inclined to shift towards more oligohaline or brackish marsh systems (Hackney et al. 2007). In areas where salinities increase, conversion of tidal swamps to brackish marsh may take some time because trees that are located in tidal freshwater forested wetlands are adapted to the effect of periodic and brief saltwater intrusion (Yanosky et al.1995, Williams et al. 2003). Plant species can become adapted to habitat conditions because previous generations have already survived under these conditions (Sussu and Turesson 1922).

At the downstream range of tidal freshwater forested wetlands, saltwater intrusion is more frequent and seems to elicit shifts in forest composition and structure. For instance, tidal freshwater forests tend to have greater tree densities than other floodplain forests in the Southern United States (Brison et al. 1985; Krasus et al. 2009). The tidal influence on riparian wetlands is considered in part a product of the sediment surface elevation which influences the amplitude, duration and frequency of tides (Morris et al. 2002). As the riparian landscape becomes more elevated upstream, tidal connection declines and becomes less important. However, if it is a flatter area with a low elevation, saltwater intrusion can reach further inland. For example, the tides of coastal Louisiana are considered micro-tidal (<1m in range), however, this area has a very gradual elevation that is fairly flat and therefore it has a wide range of saltmarsh and freshwater tidal wetlands (Krauss and Duberstein 2010). Within a tidal freshwater forested wetland, the frequency and depth of flooding also varies depending upon microtopography (Reinhardt, 1992). Hummocks and hollows are microtopographic features common in tidal freshwater forested wetlands. Hummocks are slightly elevated (average +15 cm) surfaces that rise above the bottom elevation while hollows are low areas which are commonly flooded during tidal flooding (Lampela et al. 2016). Fallen trees have been noted to be important for the formation of hummocks in tidal swamps while the hollows are determined by the remaining balance of land not in hummocks (Stribling 2007). These features create heterogeneous environments that support various tree species (Smith 1997). Because hummocks tend to be less inundated, they often support species more adapted to just periodic flooding compared to species that frequent hollows where they are flooded longer and have slower decay rates and nutrient cycling (Johnson and Damman 1991).

Some freshwater wetland tree species are more tolerant of flooding and saltwater intrusion. However even these tolerant species have a limit to salinity exposure (Krauss et al. 2007). These species may survive longer under saltwater stress by excluding excess ions in order to maintain an optimal turgor pressure (Pezeshki 1990). In some species, Cl^- and Na^+ are transported to the leaves by glycophytes in conjunction with compartmentation mechanism that brings salts in to the vacuoles to minimize cytoplasm damage (Pezeshki 1990). Physiological studies have found that new seeds of bald cypress (*Taxdium distichum*) which is tolerant to some salt exposure, can germinate and survive after flooding conditions (Allen et al.1996). However, these seeds can die when located in salt water (>10ppt) for two weeks (Allen 1997). Adult bald cypress can tolerate salinities up to 8 ppt for short durations (Corner et al. 2005) and are resilient to submergence (Anderson and Pezeshki 2000). In the southeast United States, bald cypress (*T. distichum*) and swamp tupelo (*Nyssa biflora*) are common tree species in many tidal freshwater rivers (Krauss and Duberstein 2010). Although generally tolerant to flooding, their growth is slowed by flooding (Shanklin and Kozlowski 1985, Flynn 1986, Effler and Goyer 2006). In addition to bald cypress, earlier forest surveys along the tidal sections of the Apalachicola River showed that some other common tree species that were indicative of tidal sections of the river included red maple (*Acer rubrum*), pumpkin ash (*Fraxinus* *profunda*), sweet bay (*Magnolia virginiana*), and cabbage palm (*Sabal palmetto*). Non-tidal communities were typically represented by Carolina ash (*Fraxinus caroliniana*), water tupleo (*Nyssa aquatica*), and Ogechee tupleo (*Nyssa ogeche*) (Anderson and Lockaby, 2011). Examples of freshwater forest communities associated with tidal conditions have been detected elsewhere as well. The transition along the tidal swamps of the Suwanne River in central Florida showed that the downriver forests had only five different species, and the most important one was *Taxodium distictum* in terms of basal area (Light et al. 2002). Furthermore, the upper tidal river forest differed from the riverine forest, and *S. palmetto* was the most important species in basal area. *Fraxinus profunda* represented an important species in the upper tidal forest, and the most important species was *Nyssa aquatica* in terms of basal area (Light et al. 2002).

Although comparisons have been made between tidal and non-tidal forested wetland communities, less is known about where and the nature of how these forests transition and what conditions related to tidal influence may elicit these shifts. In this study we report thirty years of salinity fluctuations along two tidal rivers: the St. Marks River and East River, both distributary rivers from the Apalachicola River to Apalachicola Bay in northwest Florida, USA. We used forest survey data along a tidal gradient along with 30 years of modeled/interpolated salinity data (Chap. 2) to inspect for possible relationships between salinity and tree species composition. This study was designed to help understand tidal influence along these rivers. Salinity as a measure of tidal influence was expected to play an important role on the community composition and distribution of these tidal forested wetlands.

METHODOLOGY

Study Site

Tidal forests were monitored along the St Marks River and East River starting at the forest-marsh boundary (Figure 3. 1). Eleven forest survey plots were distributed along 18 river-km (rkm) on the St. Marks River (Figure 3. 2). The East River had ten forest survey plots extending 15 rkm (Figure 3. 2). Forest survey plots were located between and at hydrology sampling sites (Figure 3. 2). Reported soil types of the wetlands along these rivers include Chawan, Brickyard, and Kenner soils, which are frequently flooded (Sasser et al. 1994) and are characterized by poor drainage, very high runoff, water depths of 0-15 cm, and non-saline to very slight saline (0.0 to 2.0 ppt) conditions. Along these two rivers, hydrology at the tidal freshwater forested wetlands is heavily influenced by seasonal discharge of the Apalachicola River and tidal pulses.

Forest Data Collection

Sampling was conducted on 8-9 October 2015 at the St. Marks River and on 16-17 November 2015 at the East River. Each of the 21 samplings site consisted of a 500-m^2 circular plot that was located 60 m from the river's edge. Site locations along each river were spaced at relatively equal distances between sampling points and located within straight sections of the river. Within each plot all tree species with diameters >2.5 cm at breast height (DBH, 1.3-m high) were identified and their diameters measured. Because three species of ash (*Fraxinus* spp.) are known to occupy the lower Apalachicola wetlands (Anderson *et al*. 2013) and reliable identification is only achieved through inspection of samaras (which were not always available), all *Fraxinus* were identified only to genus.

Forest Data Analyses

Using forest survey data collected at each site, species density and basal area were calculated per plot and used to calculate relative density and dominance per species using the following equations:

$$\text{Relative density } = \frac{\text{number of individuals of the species}}{\text{number of individuals of all species}} \quad (1)$$

$$\text{Relative dominance } = \frac{\text{total basal area of the species}}{\text{total basal area of all species}} \quad (2)$$

The importance value (IV) of each species per plot was calculated the sum of these values:

$$\text{IV} = \text{ relative density} + \text{ relative dominance} \quad (3)$$

For each river, we examined the relative contribution of tidal and non-tidal tree species relative to the distance to river mouth to examine potential trends and thresholds linked to tidal influence. We used indicator species identified for tidal and non-tidal freshwater forested wetlands along the Apalachicola River (Anderson and Lockaby 2011). The contributions of indicative tidal species (*N.biflora, T.distichum, M. virginiana,* and *S. palmetto*) and non-tidal species (*N. aquatica, N. ogechee, Q. lyrata, and Q. nigra*) were calculated as cumulative importance values (IV_{Tidal} and $IV_{Non\text{-}tidal}$) using the following equations:

$$IV_{Tidal} = IV_{N.\ biflora} + IV_{T.\ distichum} + IV_{M.\ virginiana} + IV_{S.\ palmeto} \quad (4)$$

$$IV_{Non\text{-}tidal} = IV_{N.\ aquatica} + IV_{N.\ ogechee} + IV_{Q.\ lyrata} + IV_{Q.\ nigra} \quad (5)$$

Therefore, IV scores could range between 0 (absent) and 2 (entire composition).

Relation between Tidal Freshwater Forested Wetland and Salinity

Models estimating historical wetland salinity (1985–2015) along this tidal gradient were used along with the results of the forest survey data to examine for potential correspondence between forest composition and historical salinity patterns. For the purposes of this study, we considered saltwater intrusion to be the single (hourly) occurrence of salinity >5 ppt. Although other conditions related to tidal influence may affect species composition (hydro period, sulfide toxicity), salinity exposure is an important indicator of tidal influence and likely a primary factor driving species composition (Board 2012). As indicated in Chap. 2, salinity data was collected and models developed for three stations across the tidal gradient of each river. These models were created by different input combinations of tide level, river discharge, evapotranspiration, and temperature. . Forest survey plots were also located at these six stations and used to make 30-yr salinity predictions. Using these 30-yr salinity data sets, a comparable data set was developed for all forest survey plots through linear interpolation Interpolated daily average salinity between forest plots was calculated for each plot based on its relative distance to the river mouth and modeled salinity monitoring stations.

The small-scale wetland salinity models and interpolated results for each plot were summarized by daily average salinity in addition to minimum, maximum, median, and 25th and 75th quartile measures. Daily average salinity level >3 ppt was used as a conservative threshold of saltwater intrusion to account for hourly fluctuations in salinity that exceeded 5 ppt at least part of the day. This was confirmed by using field data, where on all days where an hourly measure exceeded 5 ppt, the corresponding daily average was >3 ppt. Using these modeled and interpolated data, the average number of days and events (continuous days) per year where salinity was >3 ppt was calculated for each forest plot. These measures were used to 1) characterize the frequency of saltwater intrusion across the tidal freshwater forested zone, 2) compare salinity regimes between the two rivers and 3) compare/identify possible salinity thresholds where important forest species shifts occur.

RESULTS AND DISCUSSION

Forest Species

A total of 22 species of canopy trees were identified across all river plots (Table 3. 1, Table 3. 2). Species indicative of tidal conditions dominated over much of the swamps along the tidal gradient. Based on calculated IVs averaged across all plots, the three most common species along the East River were swamp tupelo (IV= 0.28), ash (IV= 0.27) and bald cypress (IV = 0.21).The most common forest species along the St. Marks River were bald cypress (IV = 0.30), water tupelo (IV = 0.29), and ash (IV = 0.23). Species composition was fairly consistent with results previously reported in Anderson and Lockaby (2011) along the Apalachicola River. They detected 4 communities based on indicator species analysis including two tidal communities (swamp tupelo – bald cypress and cabbage palm – sweet bay) and two non-tidal communities (Ogeechee tupelo – overcup oak and water tupelo –Carolina ash) (OT – OO andWT-CA).

Coastal plants occupy different habitats that often depends on their tolerance to various levels of salinity (Barbour and Davis 1970, Bertness et al. 1992). Although some plants can survive in elevated salinities, they also grow best in freshwater (Crain et al. 2004). In this study, the salinity level distribution differed between St. Marks and East River, and tree species composition showed parallel changes. In general, tidal species had higher importance values at downstream forest survey sites where the average salinity was higher than forest survey plots further inland. At the East River, bald cypress (*T. distichum*) and swamp tupelo (*N. biflora*) species were found at all of the forest survey study sites to plot 11 (Table 3. 1). Between forest survey plot 2 and 5, these tidal species were dominant (Table 3. 1). However, bald cypress was well distributed along the study reach of both rivers. There was only one site where bald cypress was not detected (East River forest survey plot 6). Swamp tupelo (*N. biflora*) had the highest importance value at the forest survey plot 2 (IV = 1.11) and bald cypress had the second highest importance value (IV = 0.69). Bald cypress (*T. distichum*) is tolerant of low salinity although its growth rate decreases when the salinity levels exceeds 1 ppt (Powell et al. 2016). Swamp tupelo is considered more sensitive to saltwater, flooding and temperature conditions than bald cypress (Krausset al. 2009). Swamp tupelo has been shown to occupy continuously saturated soils that are not deeply flooded and therefore may be well suited for intertidal sections frequently flooded by tides (Putnam 1951, Klawitter 1962, Harlow and Harrar 1969, Outcoalt 1990). Another study found that swamp tupelo seeds could not germinate (because seeds death) at 2 ppt salinity after only 70 days of exposure (McCarron et al. 1998). If the tide level duration and frequency increase, swamp tupelo may not regenerate at these locations, which could change the community composition.

Cabbage palm was the other tidal species that had the high level of importance value (highest; IV = 0.53, plot 5). Cabbage palm is also a somewhat salt tolerant species (Brown 1973) with seeds that are highly tolerant to flooding and saline conditions (Perry and Williams 1996). This may allow cabbage palm to occupy sites along lower coastal rivers where saltwater intrusion is more common (Corner and Askew 1993). Due to its higher tolerance to salt water intrusion and large distribution along tidal rivers, if the salinity levels increase in this tidal river, cabbage palm may increase in frequency. Cabbage palm was observed in all but three plots along the East River.

River Salinity Regime at the Tidal Freshwater Forested Zone

Using the modeled and interpolated salinity data, the average salinity level along the St. Mark's River and East River were examined. The first forest plot (2) on the East River was 2.98 rkm from the bay (Table 3. 3). As expected, because it is the nearest site to the bay, the 30-year average salinity was higher (0.33 ppt;Table 3. 3) than sites further up the East River. Similar differences were detected along the St. Mark's River (Table 3. 4). Even though, the East River forest survey plot 2 was located close to the bay (2.98 rkm; Table 3), the St. Mark's River forest plot 2 (6.29 rkm; Table 3. 4) had a higher salinity than the East River. Elevation differences between these two rivers might explain some this difference. Because of a more gradual slope, tidal waters may be allowed to protrude further up the St. Mark's River so the average salinity was higher. However, the St. Mark's River was sharply fresher at the forest plot 6 and above this site (Table 4). The change in average salinity level along the East River was more gradual. After forest plot 6, upstream average salinity level was <0.2 ppt. Table 3 and Table 4 shows the ANN model and interpolated prediction results for 30 years. It should be noted that ANN models can make over estimates (Maier and Dandy 2000) which would explain how the maximum daily average salinity level results at plots 3-5 are higher than plots 1 and 2 at the East River (Table 3. 3). Measures of the mean, median 25th quartile, and 75th quartile eliminate extreme measures and showed the expected progression of salinities decreasing with increased distance of the plot from the bay (Table 3. 3, Table 3. 4).

Along the East River forest plots, there was a section (plot 7;Table 3. 5) where all tidal and non- tidal species were present; however, water tupelo (IV = 0.63) was the defining species of the community. The average salinity at this plot was 0.18 ppt while the previous forest survey plot average salinity was 0.28 ppt, and the maximum salinity range significantly lower than previous plot (plot 6 = 1.96 ppt, plot 7 = 0.66). Due to that, the community composition was represented by a non- tidal species. Results from this plot were well outside the typical trends for this reach suggesting that there something atypical was driving community composition (perhaps an alternative freshwater source) and this plot was omitted as an outlier data point. This was supported by the amount of swamp tupelo at this study plot and others around it. Salt water tolerances of these two species are very different, for example, Krauss et al. (2009) found that, water tupelo was limited to salinity level <1ppt while swamp tupelo could survive at up to2.1 ppt, (Krauss et al. 2009). Penfound and Hathaway (1938) also found that water tupelo was only found in freshwater (0 ppt) in their study site in Louisiana (Penfound and Hatway, 1938) although water tupelo survived under high level of salinity pulses (up to 21 ppt) resulting from hurricanes (Conner and Inabinette 2003).

In the East River, tidal species importance value (IV_{Tidal}) was significantly higher than non- tidal species ($IV_{Non\text{-}tidal}$) until plot 7 (11.4 rkm far from the bay; Figure 3). Only Ogeechee tupelo and water tupelo were found (Ogeechee tupelo: plots 1, 2, and 4; water tupelo: plot 5) (Table 3. 5). After this forest survey plot, $IV_{Non\text{-}tidal}$ increased, and at plot 9, Ogeechee tupelo had its highest importance value (IV = 0.39). Forest survey plot 11 was strongly different than others. At this plot, the sum of the tidal species importance value was the lowest all along the study sites and bald cypress was the only tidal species (IV = 0.06). Water oak was the representative species *(*IV = 0.48) at this study site.

Salinity level changes changed sharply at the St. Marks River. High average daily salinity level was predicted at plot 1 (0.83 ppt) through 5 (0.33 ppt). At the forest plot 6, average daily salinity levels were 0.16 ppt (Table 3. 4). Tidal species (cabbage palm, bald cypress, sweet bay, and swamp tupelo) and one of the non- tidal species (Ogeechee tupelo) were observed at the St. Marks river forest survey plot 1. Swamp tupelo (IV = 0.42) was the most representative species in this community. In plot 2, there were only cabbage palm and bald cypress, and their importance values were significantly different from the each other. Cabbage palm importance value was 0.63 while the bald cypress had 0.10. Along the St. Marks River, almost all species were observed except water oak and sweet bay at plot 6, which was 12.4 rkm far from the bay (Table 3. 4). Non- tidal species were generally representative species after forest plot 6.

When daily tidal pulses occur they can occur from one to several days. For example, along the East River, salt water intrusion events lasted at most for two days based on 30-year model/interpolation results (Figure 3. 7) while it occurred for up to six days for the St. Mark' River (Figure 3. 8), in average thirty years (1985-2015) predicted salinity. If we look at the frequency of average daily salinity level ≥3 ppt over thirty years, it was almost two days in the

East River (Figure 3. 7), and was nearly fourteen days (Figure 3. 8) in the St. Marks River. Saltwater intrusion affects the species distribution along the tidal river and is important for mature tree survival and recruitment because it has an important effect on germination of a tree seeds. When salt water intrusion (average daily salinity ≥3ppt) was less than one day per year, the tidal species importance value and the non-tidal species importance values were more comparable. Tidal species importance values (IV_{Tidal}) were higher than $IV_{Non\text{-}tidal}$ in the East River. If saltwater intrusion occurred on average more than one day per year, there was a significant difference between IV_{Tidal} and $IV_{Non\text{-}tidal}$ and non-tidal species generally disapperaed. The St. Marks River did not have the same characteristics. Along the St. Marks River, if the saltwater intrusion occurred on average less than one day per year, the IV_{Tidal} and $IV_{Non\text{-}tidal}$ were slightly different, but non- tidal species IV was higher than tidal species. Even though saltwater intrusion occurred for up to 3 days the $IV_{Non\text{-}tidal}$ was higher than IV_{Tidal}, but again not much more different from each other's. In the St. Marks River, the significant shift was obtained when the annual average occurrence of saltwater intrusion exceeded five days.

CONCLUSION

This study demonstrated that community composition changed along the St. Marks River and East River in which was related to proximity to Apalachicola Bay and corresponding tidal influences including saltwater intrusion frequency, duration, and average salinity among likely factors driving the patterns. Saltwater intrusion functions as a stressor throughout the forest survey sites and its affects are varied along the rivers because of differences in saltwater intrusion reach further inland that is likely related to elevation of the river, river discharge and tidal stage. In likely response to the detrimental effects of saltwater intrusion throughout the tidal rivers, non-tidal species' (*N. aquatica, N. ogeche, and Q. lyrata.)* importance values were reduced downstream and significantly higher at the upstream plots. Conversely, tidal species importance value (*T. distictum, S. palmetto, M.virginiana, N. biflora*) were primarily highest in the lower downstream area. At both rivers there was a critical point where tidal and non- tidal species importance values became similar to each other. At the St. Marks River that point was observed at plot 6 (12.48 rkm) while at the East River the same characteristic was at plot 7 (11.42 rkm). The community composition at these sites became mixed with tidal and non-tidal species. Because non- tidal species importance values were higher at these forest survey sites, it seems tide level could not reach these sites as much as lower sites and these sites seemed to be a transition zone between tidal and non-tidal. However, tidal species importance values decreased and non- tidal species importance values increased at ≈15 rkm from the bay for both rivers (Figure 3. 9). It appears that the tidal influences that affect species composition begin to diminish from these points.

The duration and frequency of saltwater intrusion events also varied along the tidal reach and likely influenced tidal and non-tidal species importance values. Because tide level duration and frequency varied between the rivers, tidal and non-tidal species importance values also differed between two rivers. This study predicted the historical annual average tide intrusion days, and we obtained that importance value of the tidal and non-tidal species varied depends on how many days tide stayed at the forest survey zone. Based on these results, if saltwater intrusion was less than one day, the tidal species average importance value was slightly higher than non- tidal species; however, there were a significant drop in non-tidal species importance value when the annual occurrence of saltwater intrusion occurred (on average) more than one day in the East River. It was different at the St. Marks River. If saltwater intrusion occurred on average less than one day per year, non- tidal species importance values became slightly higher than the tidal species. Furthermore, when saltwater intrusion was >3 days per year, non-tidal species importance values diminished significantly.

In conclusion, while the tidal species gradually declined in importance value along the East River and St. Marks River, there was a clear break point where tidal communities' species became mixed communities (e.g., forest plot 6 along the St. Marks River). Although it is uncertain which factors or combination of tem associated with tidal influence affect species, salinity level is likely one of the primary drivers of community composition along these tidal rivers. Community composition and distribution showed different characteristics and possible responses along these two tidal rivers which likely reflects differences flow patterns and river geomorphology. Using metrics such as long term average salinity and the duration or frequency of saltwater intrusion may be important measure that gage tidal influences that affects forest community composition along rivers and may be used to improve predictions of species shifts as a result of future environmental and management changes.

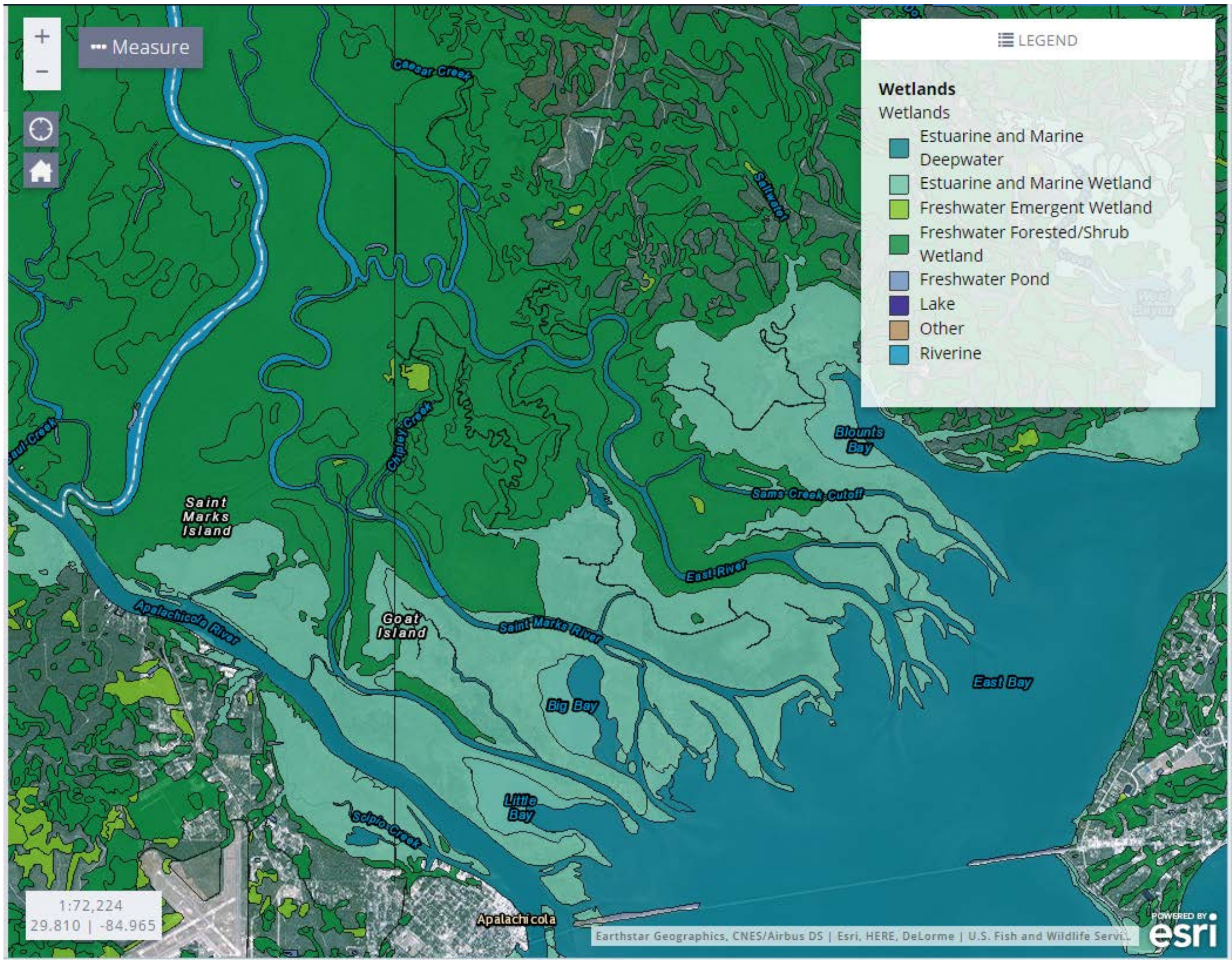

Figure 3. 1: Wetland types along the lower Apalachicola River, Florida, USA as mapped from the National Wetland Inventory (wetland mapper: https://www.fws.gov/wetlands/data/mapper. 2016.

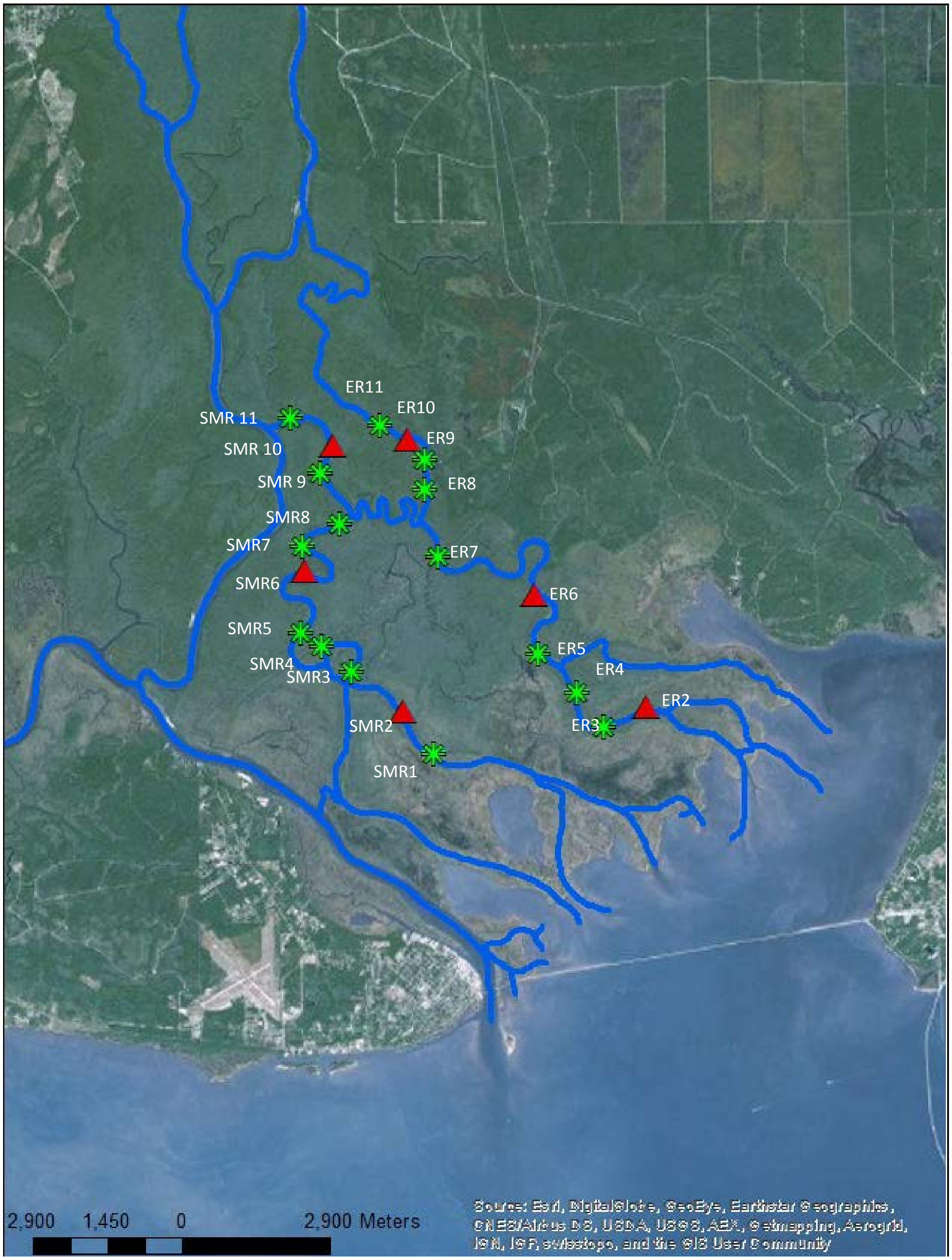

Figure 3. 2: Location of forest survey plots along the St Marks River (SMR) and East River (ER). Red triangles denote survey sites located at salinity/hydrology monitoring stations.

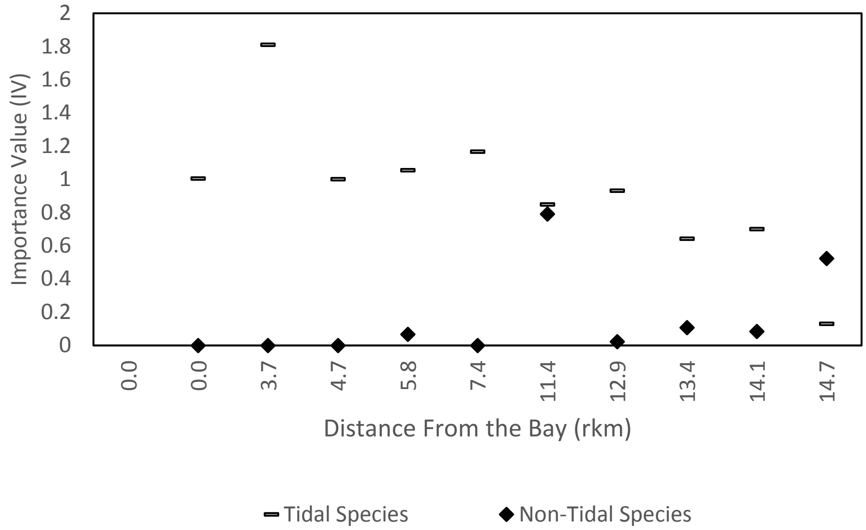

Figure 3. 3: Aggregate importance values for designated tidal species (IVTidal; swamp tupelo, bald cypress, cabbage palm and sweet bay and non-tidal species (IVNontidal; water tupelo, water oak, Ogeechee tupelo and overcup oak) relative to distance to river mouth at the East River

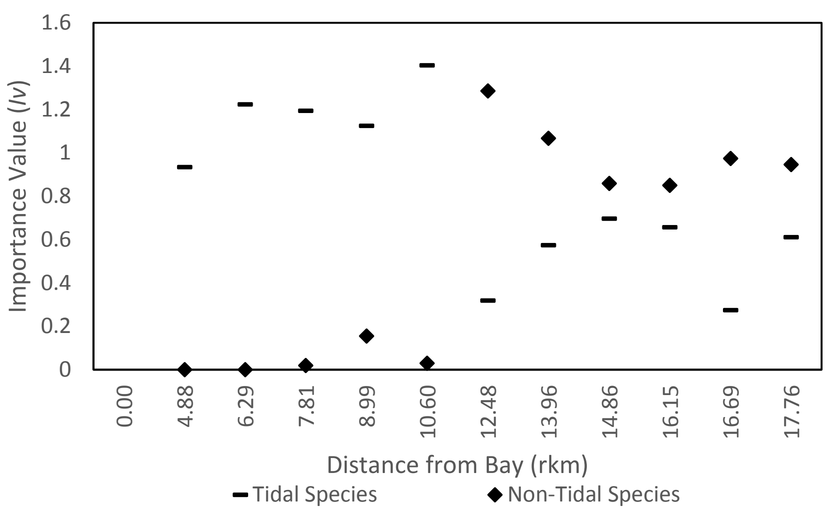

Figure 3. 4: Aggregate importance values for designated tidal species (IVTidal: swamp tupelo, bald cypress, cabbage palm and sweet bay and non- tidal species (IVNontidal: water tupelo, water oak, Ogeechee tupelo and overcup oak) relative to distance to river mouth at the St. Marks River

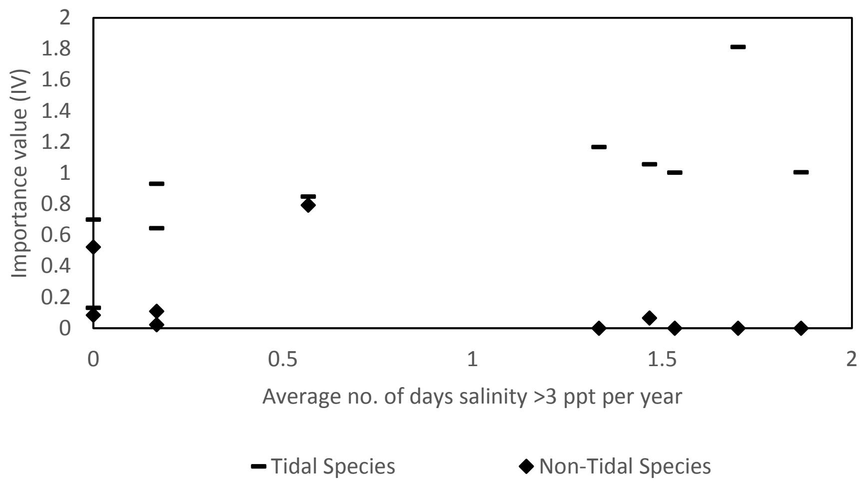

Figure 3. 5: Aggregate tidal and non-tidal species importance values (IVTidal and IVNontidal) distribution vs. average number of days of saltwater intrusion (>3 ppt) per year using 30-year modeled/interpolated salinity data for forest plots along the East River.

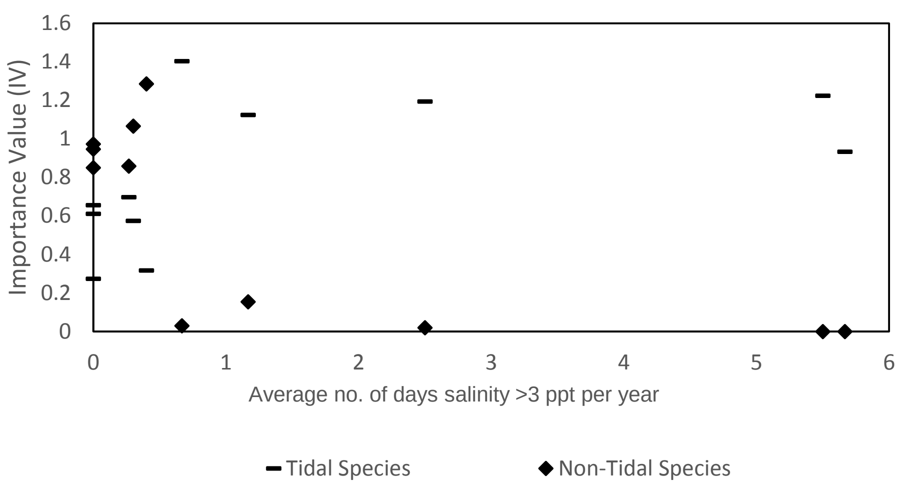

Figure 3. 6: Aggregate tidal and non-tidal species importance values (IVTidal and IVNontidal) distribution vs. number of days of saltwater intrusion (>3 ppt) per year using 30-year modeled/interpolated salinity data for forest plots along the St. Marks River.

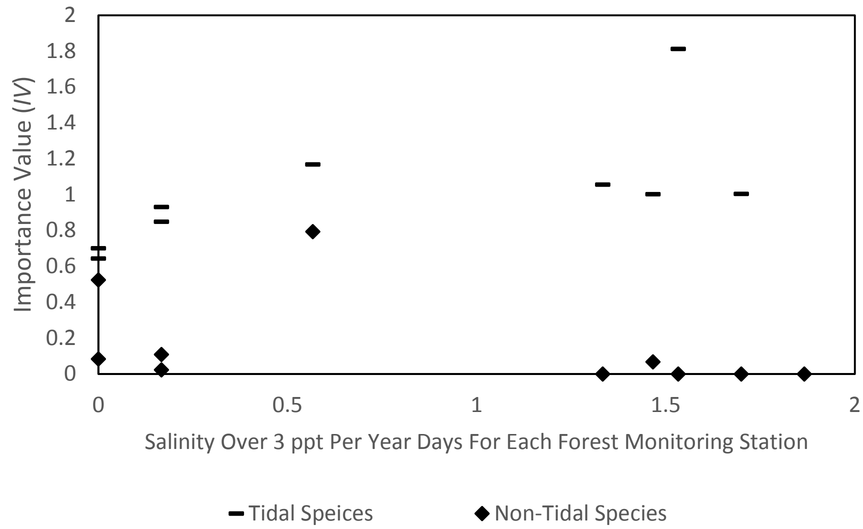

Figure 3. 7: Aggregate tidal and non-tidal species importance values (IVTidal and IVNontidal) distribution vs. on number of average events of saltwater intrusion (>3 ppt) per year using 30-year modeled/interpolated salinity data for forest plots along the East River

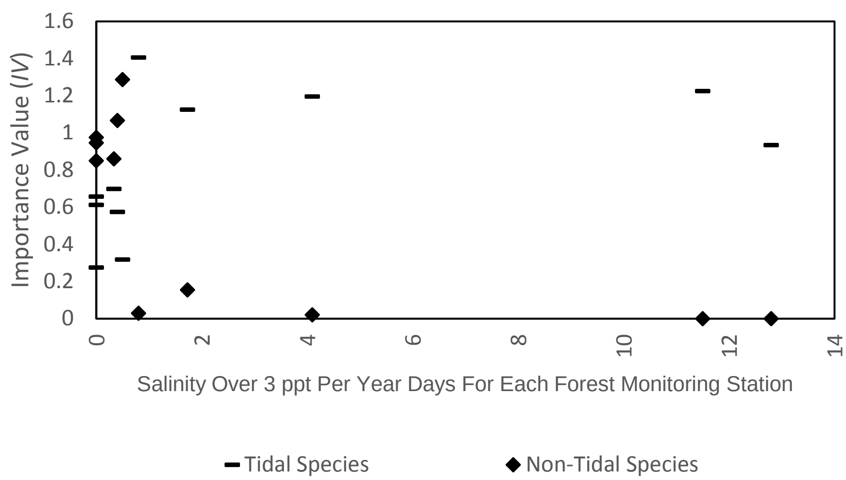

Figure 3. 8: Aggregate tidal and non-tidal species importance values (IVTidal and IVNon-tidal) distribution vs. number of average events of saltwater intrusion (>3 ppt) per year using 30-year modeled/interpolated salinity data for forest plots along the St. Marks River

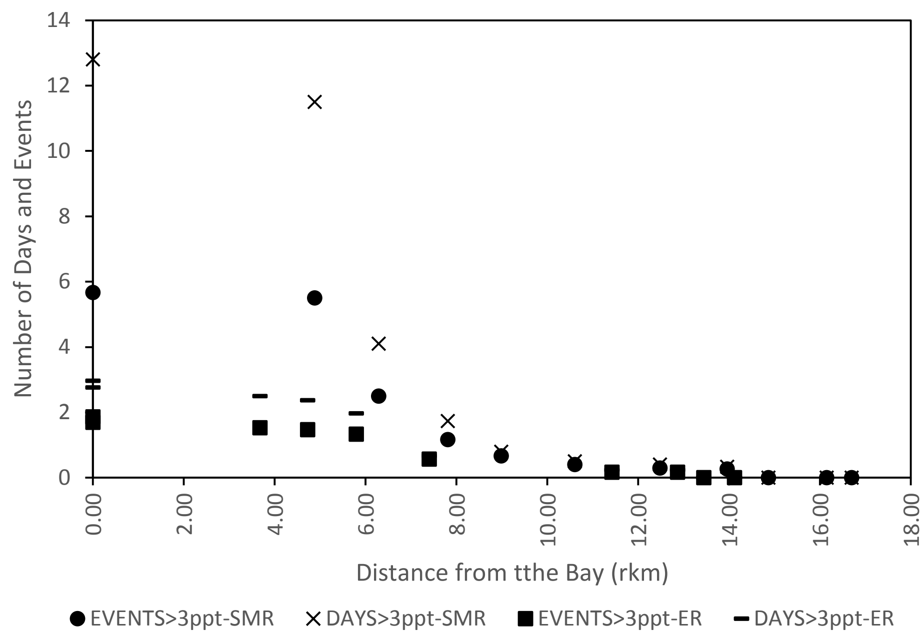

Figure 3. 9: Distribution of average number of days and events per year that salinity ≥3ppt along the tidal rivers (ER; East River, SMR; St. Marks River) vs. distance from bay.

Table 3. 1: Occurrence of tree species in forest survey plots along the East River, Apalachicola National Estuarine Research Reserve, Florida during 2015. X = present.

		Forest Survey Plots									
Species	**Common Name**	2	3	4	5	6	7	8	9	10	11
Fraxinus spp.	Ash	X	X	X	X	X	X	X	X	X	X
Taxodium distichum	Bald cypress	X	X	X	X	X		X	X	X	X
Cephalantus occidentalis	Button bush	X	X							X	
Nysee ogeechee	Ogeechee tupelo	X	X		X		X	X	X	X	
Acer rubrum	Red maple	X	X	X	X	X	X	X	X	X	X
Nyssa biflora	Swamp tupelo	X	X	X	X	X	X	X	X	X	
Magnolia virginiana	Sweet bay	X	X	X	X	X	X	X	X	X	X
Myrica cerifera	Wax myrtle		X	X	X	X					
Ulmus americana	American elm			X	X	X		X	X	X	X
Sabal palmetto	Cabbage palm			X	X	X	X	X	X	X	
Ilex cassine	Dahoon holly			X		X					
Persea borbonia	Swamp bay			X	X	X	X				
Nyssa aquatica	Water tupelo				X						X
Quercus lyrata	Over cup oak						X			X	
Planera aquatica	Planer elm						X				
Quercus nigra	Water oak						X	X	X	X	X
Salix nigra	Black willow							X			
Liquidambar straciflua	Sweet gum									X	X
Carpinus caroliniaana	Horn Beam									X	X
Carya aquatica	Water hickory									X	X
Populus heterophylla	Cotton wood										X

Table 3. 2: Occurrence of tree species in forest survey plots along the St. Marks River, Apalachicola National Estuarine Research Reserve, Florida during 2015. X = present.

		Forest Survey Plots										
Species	**Common Name**	1	2	3	4	5	6	7	8	9	10	11
Fraxinus spp.	Ash	X	X	X		X	X	X	X	X	X	X
Taxodium distichum	Bald cypress	X	X	X	X	X	X	X	X	X	X	X
Cephalantus occidentalis	Button bush			X								
Nysee ogeechee	Ogeechee tupelo	X					X	X	X	X		X
Acer rubrum	Red maple	X	X	X	X	X	X	X	X	X		X
Nyssa biflora	Swamp tupelo	X	X	X	X	X	X	X	X	X		
Magnolia virginiana	Sweet bay	X		X	X	X	X					
Myrica cerifera	Wax myrtle	X	X	X	X							
Ulmus americana	American elm	X	X	X	X	X	X	X	X	X		X
Sabal palmetto	Cabbage palm	X	X	X	X	X	X				X	
Ilex cassine	Dahoon holly					X						
Persea borbonia	Swamp bay		X	X								
Nyssa aquatica	Water tupelo				X		X	X	X		X	X
Quercus lyrata	Over cup oak			X		X	X				X	
Planera aquatica	Planer elm			X	X				X	X	X	X
Quercus nigra	Water oak							X	X	X	X	
Salix nigra	Black willow	X			X							
Liquidambar straciflua	Sweet gum										X	
Carpinus caroliniaana	Horn Beam						X	X	X	X		X
Carya aquatica	Water hickory											X
Populus sp.	Cotton wood											
Juniperus virginiana	Red cedar		X									
Quercus laurifolia	Laurel oak											

Table 3. 3: Model and interpolation results of 30-year daily average salinity data. Mean, minimum, maximum, median, and 25th quartile and 75th quartile of salinity, and and frequency of days and events of saltwater intrusion (average daily salinity >3ppt days/year

Sampling Station	**Distance from rivermouth (rkm)**	**Mean daily salinity (ppt)**	**Min. salinity (ppt)**	**25th quart. salinity (ppt)**	**Median salinity (ppt)**	**75th quart. salinity (ppt)**	**Max. salinity (ppt)**	**No. of saltwater intr. (days/yr)**	**No. of saltwater intr. (events/yr)**
2	2.77	0.33	0.07	0.13	0.14	0.29	12.64	2.96	1.86
3	3.67	0.32	0.06	0.13	0.14	0.27	16.31	2.76	1.70
4	4.72	0.31	0.06	0.13	0.15	0.26	20.53	2.50	1.53
5	5.79	0.30	0.05	0.13	0.14	0.24	24.88	2.36	1.46
6	7.36	0.28	0.01	0.12	0.14	0.16	31.24	1.96	1.33
7	11.42	0.18	0.07	0.12	0.13	0.14	12.72	0.66	0.56
8	12.86	0.14	0.09	0.11	0.12	0.13	6.02	0.23	0.16
9	13.44	0.13	0.10	0.11	0.11	0.11	3.32	0.23	0.16
10	14.12	0.11	0.10	0.11	0.11	0.11	0.24	0.00	0.00
11	14.72	0.11	0.06	0.11	0.11	0.11	0.24	0.00	0.00

Table 3. 4: Model and interpolation results of 30-year daily average salinity data. Mean, minimum, maximum, median, and 25th quartile and 75th quartile of salinity, and and frequency of days and events of saltwater intrusion (average daily salinity >3ppt days/year

Sampling Station	Distance from river mouth (rkm)	Mean daily salinity (ppt)	Min. Salinity (ppt)	25^{th} quart. (ppt)	Median (ppt)	75^{th} quart. (ppt)	Max. Salinity (ppt)	No. of saltwater intr. (days/yr)	No. of saltwater intr. (events/yr)
1	4.87	0.83	0.20	0.23	0.42	1.19	16.35	12.8	5.66
2	6.28	0.75	0.19	0.21	0.37	1.00	15.52	11.5	5.50
3	7.80	0.60	0.17	0.19	0.31	0.79	14.42	4.1	2.50
4	8.98	0.50	0.16	0.18	0.27	0.79	13.57	1.73	1.16
5	10.60	0.33	0.15	0.16	0.21	0.62	12.65	0.80	0.66
6	12.48	0.16	0.13	0.14	0.14	0.40	11.90	0.50	0.40
7	13.95	0.15	0.10	0.14	0.12	0.15	8.87	0.40	0.30
8	14.86	0.14	0.10	0.14	0.12	0.15	6.18	0.33	0.26
9	16.14	0.13	0.09	0.15	0.11	0.15	2.3	0.00	0.00
10	16.69	0.13	0.06	0.15	0.09	0.16	0.92	0.00	0.00
11	17.76	0.12	0.03	0.14	0.07	0.17	0.84	0.00	0.00

Table 3. 5: Species importance values of tidal and non-tidal indicator tree species at forest sampling plots along the East River.

		Tidal Species				***Non-Tidal Species***			
Plot number	**Distance from the rivermouth (rkm)**	***Taxodium distichum***	***Nyssa biflora***	***Magnolia virginiana***	***Sabal palmeto***	***Nyssa aquatica***	***Nyssa ogeechee***	***Querqus nigra***	***Querqus lyrata***
2	2.7	0.70	0.27	0.00	0.00	0.00	0.01	0.0	0.0
3	3.6	0.41	1.10	0.20	0.00	0.00	0.02	0.0	0.00
4	4.7	0.40	0.09	0.10	0.35	0.00	0.00	0.0	0.0
5	5.7	0.13	0.36	0.00	0.53	0.06	0.03	0.0	0.0
6	7.3	0.17	0.25	0.20	0.47	0.00	0.00	0.0	0.0
7	11.4	0.00	0.43	0.30	0.08	0.63	0.10	0.15	0.02
8	12.8	0.26	0.40	0.20	0.06	0.0	0.10	0.02	0.0
9	13.4	0.12	0.08	0.00	0.41	0.0	0.40	0.10	0.0
10	14.1	0.13	0.08	0.20	0.24	0.0	0.02	0.08	0.11
11	14.7	0.06	0.00	0.00	0.00	0.05	0.00	0.47	0.0

Table 1. Species importance values of tidal and non-tidal indicator tree species and at forest sampling plots along the St. Marks River.

		Tidal Species				***Non- Tidal Species***			
Plot number	**Distance from the rivermouth (rkm)**	***Taxodium distichum***	***Nyssa biflora***	***Magnolia virginiana***	***Sabal palmeto***	***Nyssa aquatica***	***Nyssa ogeechee***	***Querqus nigra***	***Querqus lyrata***
1	4.8	0.12	0.42	0.20	0.20	0.00	0.05	0.00	0.00
2	6.2	0.09	0.00	0.00	0.62	0.00	0.00	0.00	0.00
3	7.8	0.38	0.64	0.20	0.12	0.00	0.00	0.00	0.02
4	8.9	0.17	0.30	0.40	0.45	0.15	0.00	0.00	0.00
5	10.6	0.81	0.30	0.11	0.08	0.00	0.00	0.00	0.03
6	12.4	0.08	0.13	0.00	0.03	1.04	0.13	0.00	0.10
7	13.9	0.23	0.14	0.00	0.00	0.31	0.69	0.05	0.0
8	14.8	0.39	0.08	0.00	0.00	0.49	0.29	0.07	0.0
9	16.1	0.20	0.35	0.00	0.00	0.00	0.69	0.15	0.0
10	16.6	0.21	0.00	0.00	0.06	0.62	0.0	0.06	0.28
11	17.7	0.56	0.00	0.00	0.00	0.55	0.40	0.00	0.00

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