

**TIME AND TIDE: UNDERSTANDING THE WATER DYNAMICS IN A TIDAL  
FRESHWATER FORESTED WETLAND**

**A thesis submitted in partial fulfillment of the requirements for the degree**

**MASTER OF SCIENCE**

**in**

**ENVIRONMENTAL STUDIES**

**by**

**BROOKE JAMES CZWARTACKI  
DECEMBER 2013**

**at**

**THE GRADUATE SCHOOL OF THE COLLEGE OF CHARLESTON**

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## **ABSTRACT**

### **TIME AND TIDE: UNDERSTANDING THE WATER DYNAMICS IN A TIDAL FRESHWATER FORESTED WETLAND**

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The Southeastern Atlantic Lower Coastal Plain is characterized by low topographic slope and drainage systems that interface with mesotidal estuaries, resulting in long reaches of freshwater tidal streams and adjoining tidal freshwater forested wetlands. Hydrodynamics are poorly understood in these spatially heterogeneous systems, characterized by interactions of seasonal river discharge, geomorphology, local climate patterns, and tide stage. A better understanding of the hydrology and associated linkages to ecosystem functions and services is needed to anticipate their response to sea level rise. Huger Creek, a fourth order tidal freshwater stream flowing into the East Branch of the Cooper River and eventually discharging into the Charleston Harbor estuary in South Carolina was studied to determine the effects of a decreasing tidal gradient on water table dynamics, soil moisture regime, vegetation, and decomposition rates within the riparian zone. Turkey Creek, a non-tidal tributary of Huger Creek was used for comparison. The local water table gradient was upstream along a 600 m stream reach, ranging from near the surface to subsurface depths of 70 cm, with semi-diurnal tidal pulses extending up to 50 m from the bank. Near the non-tidal convergence zone, an abrupt shift to fluvial dominated hydrodynamics was evident with tidal oscillations greatly reduced, and water table position mediated by topographic position, evapotranspiration, and precipitation. This suggests the tidal creek functions as a freshwater reservoir made available to the wetland through the daily tide cycle. Despite a contrasting hydrologic regime, soil oxidation depth in the rooting zone was similar among sites, therefore, riparian vegetation communities and below ground decomposition showed no significant relationship to tidal versus non-tidal designations of the floodplain. These results demonstrate the need to assess and delineate non-tidal and tidally influenced bottomland hardwood wetlands separately when considering their hydrology and function in the landscape. This knowledge is fundamental to understanding how sea level rise will affect habitat quality, nutrient exchange, and sediment transport to the estuary.

## ACKNOWLEDGEMENTS

First, I would like to thank my committee members for sharing their time and knowledge with me over the last three years of my graduate career. To my thesis advisor Carl Trettin, I truly appreciate the time, patience, and mentorship he offered to help take my broad love of wetlands and turn it into a challenging, yet rewarding research project. I am grateful for the time he took out of his workdays to communicate his expertise about floodplain soils and processes, steering me back on course when I got lost, teaching me to become a better writer, and showing me how to be surgical with a sharp shooter shovel. To Tim Callahan, for his challenging classes that introduced me to the wonderful world of hydrogeology, and that applied mathematics can be fun. Tim's sense of humor and encouraging words helped me through the rough patches of coursework and research. Many thanks are due to Tom Williams for sharing his regional hydrology and GIS knowledge, and to Bo Song for her help analyzing the vegetative data.

The past three years have been an amazing college learning experience and a period of personal growth. I got engaged, married, converted to Catholicism, served on a student board, learned to disseminate scientific research, and made new friends that I hope to keep forever. I often reflect on my undergraduate experiences that fostered my desire to study the natural world, and I owe thanks to Professor Woody Bousquet at Shenandoah University - it was in his Ecology class, that this tree-hugging hippie realized she would never like it in the medical field.

This project would not have possible without funding, and all the people who spent long, hard, and hot days in field installing wells, digging holes, paddling or wading into the murkiest depths of Huger Creek, and swatting mosquitoes. I had many helpers in the field, but I am especially appreciative of Julie Arnold, Artheera Bayles, Frank Powers, and Michael Griffin. Without Julie's ingenious engineering of stilling wells and data logger installations, helping me with anything that took batteries, and decoding the mysteries of Microsoft Excel, this project would have taken much longer. I would also like to thank the Graduate Program in Environmental Studies and the Graduate School of College of Charleston for the two years of funding I received.

Lastly, I would like to thank my family. First, to my Mom and Dad who have been supportive and patient throughout all the years I have spent in school. Finally, to my loving husband, Stevie, who witnessed all the ups and downs of graduate school. He reminds me often that in the short time we have spent in Charleston, we have met and exceeded nearly every goal we set for ourselves. Throughout this process, Stevie offered encouraging words, instilled confidence in me, and provided unwavering love and support, for which I am truly grateful.

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## CHAPTER 1. INTRODUCTION AND LITERATURE REVIEW

### Introduction

Tidal freshwater swamps of the Southeastern United States occur in floodplains situated near the coastal zone, along freshwater rivers that are subject to tides. It is estimated that tidal freshwater forests and tidal freshwater marsh combined, occupy over 200,000 hectares of the Southeastern Atlantic coast from Maryland to Texas, with the largest proportion in South Carolina (Field et al. 1991). Tidal freshwater forests exist in a relatively narrow range on river floodplains within the freshwater (salinity < 0.5 ppt [parts per thousand]) intertidal zone between upland riparian forests (or bottomland hardwood forests) and freshwater marsh communities. The size of a tidal freshwater forest and associated vegetative community is defined by the size of the river, predominant type of coastal energy, mean tide range, and elevation (Doyle et al. 2007; Anderson and Lockaby 2011a; Day et al. 2007).

The hydrologic regime in tidal freshwater forested wetlands is influenced by both tidal and fluvial processes, as they share characteristics of both riverine and estuarine systems (Doyle et al. 2007). Freshwater outflows combine with tidal forcing to create systems that are both spatially and temporally heterogeneous, characterized by seasonal river discharge, geomorphology, local climate patterns, and tide stage (Day et al. 2007). The presence of tide creates a bi-directional hydrologic gradient, which can influence both upstream and downstream water quality and sedimentation patterns (Brinson 1993a;

Krauss et al. 2009). Similar to bottomland hardwood forests, tidal freshwater forests are considered valuable habitat areas and hotspots for biodiversity, nutrient exchange, and biogeomorphic feedbacks (Conner et al. 2007a). The presence of a freshwater tide differentiates these systems from similar non-tidal riparian or bottomland hardwood forests, but also highlights their vulnerability.

Tidal freshwater forests are sensitive to small changes in climate; and their low topographic position makes them particularly susceptible to salinity intrusion and increased flooding from sea level rise, coastal subsidence, and storm surges from tropical storms (Krauss et al. 2009; Doyle et al. 2010). Tidal freshwater systems undergo periodic saline exposure during low river flow or from storm surges (Krauss et al. 2009; Anderson and Lockaby 2011a), but chronic exposure to even low concentrations of saline water (2 ppt) has shown to cause dramatic shifts in vegetative communities, chemical processes, and the delivery of ecosystem services (Hackney et al. 2007; Neubauer, 2011, Noe et al. 2012). The landward retreat and mortality of tidal freshwater forest communities by salinity intrusion from sea level rise has been well-documented (Brinson et al. 1985; Hackney et al. 2007; Doyle et al. 2010; Williams et al. 2012). In addition to saline intrusion, recent research suggests that the rate of sediment accretion within tidal freshwater forests ( $1.3 - 2.2 \text{ mm yr}^{-1}$ ) is not keeping pace with the average rate ( $3 \text{ mm yr}^{-1}$ ) of sea level rise (Craft 2012). Along the South Carolina coastline, sea level is rising between  $3.2$  (Charleston) –  $4.2$  (Myrtle Beach)  $\text{mm yr}^{-1}$  (NOAA 2013). As a result, South Carolina salt marsh tidal creeks are experiencing a headward erosion rate of  $1.9 \text{ mm yr}^{-1}$  (Hughes et al. 2009). Tidal freshwater forests are disappearing or retreating

landward, and brackish marsh platforms are expanding (Craft et al. 2009; Williams et al. 2012).

In addition to the threats associated with sea level rise and salinity intrusion, there are uncertainties related to the actual extent of tidal freshwater forests. The presence of salt tolerant vegetation assists in defining the boundaries between the saltwater, brackish, and freshwater vegetative communities, but locating the forested edge of the tidal zone is difficult due to the uninterrupted forest cover within the tidal/non-tidal convergence zone (Day et al. 2007). This uncertainty arises from multiple sources including a seasonally dynamic upper boundary, the lack of a well-defined classification system, and inconsistent terminology. The current estimates of land area occupied by tidal freshwater forest cover are based on coastal county surveys conducted by NOAA (National Oceanic and Atmospheric Administration), who delineated tidal freshwater forest and marsh based on the 1991 National Wetland Inventory (Field et al. 1991; Doyle et al. 2007). Thus, the estimate is likely conservative due to reliance solely on vegetative data in a system that is hydrologically complex (Doyle et al. 2007).

Tidal freshwater ecosystems represent an important mixing zone between the downstream aquatic and upstream terrestrial environment. A better understanding of the water dynamics at the upland terrestrial boundary is needed; tidal freshwater systems may be the most vulnerable ecosystem in the coastal zone. Prior to 1985, literature related specifically to freshwater forested ecosystems was scarce because this forest type was commonly grouped into existing classifications of bottomland hardwoods or riparian forests that experienced tidal influence (Wharton et al. 1982; Odum et al. 1984; Doumlele et al. 1985). In recent years, there has been an increase in research on tidal freshwater

systems due to encroaching saline water from changing climate patterns and sea level rise. Studies along the marsh/forest boundary have demonstrated that even small changes in hydrology or saline concentrations can result in large ecosystem shifts (Hackney et al. 2007; Doyle et al. 2010). Still missing from the literature, are studies focused on water dynamics present at the tidal/non-tidal forested boundary. In these upper reaches of the tidal zone, increased flooding patterns may significantly affect the landscape because trees species may not be well adapted to prolonged flooding. Finally, there are no reliable estimates on how much of this resource currently exists, or what will happen, when it is gone. Collecting and interpreting these types of data are necessary for researchers and managers to anticipate how they will respond to a changing climate and sea level rise.

## **Literature Review**

### ***The Coastal Environment***

The existence of tidal freshwater wetlands (tidal freshwater forest or swamp, and freshwater marsh) is related to tide range, coastal geomorphology, topographic slope, and the availability of fresh water to the coast (Doyle et al. 2007). Thus, to understand how these systems are distributed, it is necessary to begin at the outer coast and work inland. Coastlines are classified by the primary source of energy that drives coastal processes, which is defined by the mean tidal range and mean wave height (Hayes and Michel 2008). Tide range is simply the difference between the highest tide and the lowest tide, and coastal energy relates to the ratio of wave height to tide range. The largest proportion of tidal freshwater ecosystems (forest and marsh) are found along large rivers

that discharge to mixed-energy (tide and wave), mesotidal coastlines (Doyle et al. 2007). Tide ranges along the Southeastern Atlantic coast range from upper microtidal (tide range 1 – 2 m) along the coasts of Maryland, Virginia, and North Carolina, to mesotidal (tide range 2 – 3 m) along South Carolina and Georgia (Doyle et al. 2007). The Gulf of Mexico is lower microtidal (tide range 0 – 2 m) and wave dominated. Hence, the majority of the tidal freshwater forest occur in South Carolina (40,000 ha), Georgia (25,000 ha), and Virginia (28,000 ha) due to contributing factors of low topographic slope, a mixed energy coastline, and large tidal range (Figure 1).

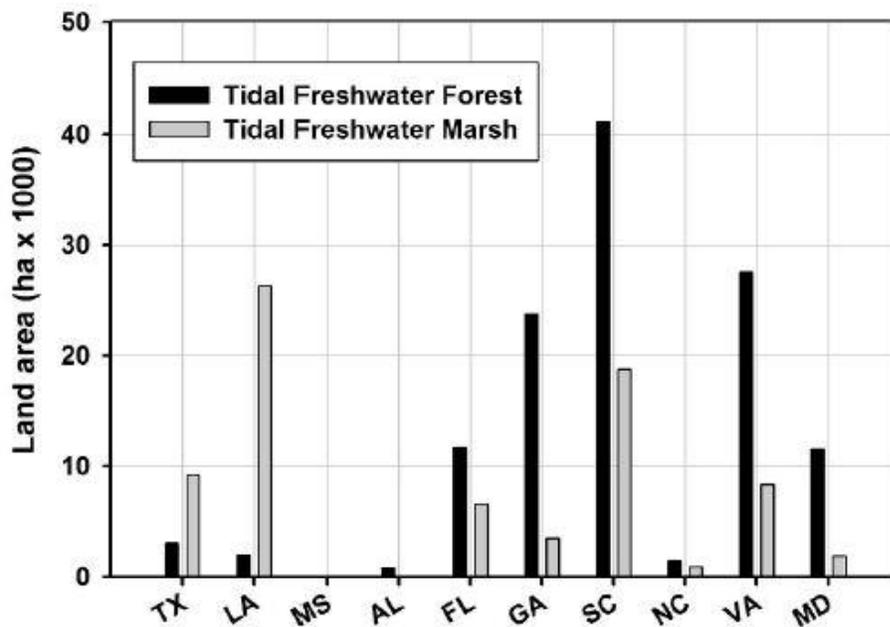


Figure 1. Land area estimated by Field et al. (1991) of tidal freshwater-forested wetlands (forest and swamp) and marsh by state in the Southeastern United States, based on 1991 National Wetlands Inventory (From: Doyle et al. 2007).

While Virginia has an upper microtidal tidal range, its connection to the Chesapeake Bay estuary explains the large area that is subject to tides. North Carolina and Florida are primarily wave-dominated coasts with an upper microtidal tidal regime, and relates to the smaller land area of tidal freshwater forests. Finally, along Texas and Louisiana there is a combined area of 7,000 ha of tidal freshwater swamp and 35,000 ha of tidal freshwater marsh. These coasts are wave dominated and tidal inlet poor characterized by long barrier islands backed with expansive marsh fringed lagoons (Hayes and Michel 2008).

### ***Tides and River Discharge***

In areas where tidal freshwater forested systems are prominent, the low topographic gradient, coupled with the large tidal range allows tides to propagate long distances upstream. Generally, tide range decreases with increasing distance from the coast. However, in funnel shaped estuaries, tide range decreases with increasing river depth, but then increases as the channel morphology narrows upstream (Mitsch and Gosselink 2007). Tidal energy is not a static factor and shows variation at differing temporal scales. Throughout the monthly lunar cycle, tide range (the difference between high and low tide level) and amplitude (height of the tide) will increase or decrease depending on the lunar phase (Hicks 2006). New and full moons produce larger amplitude spring tides and increased tide range. Conversely, first quarter and last quarter moons produce smaller neap tides with the least tidal amplitude and decreased tide range. The lunar orbit is elliptical, which causes the distance between the earth and moon to vary. Perigee is when the moon is closest to the earth, and during apogee, the moon is at its furthest distance from the earth (Hicks 2006). When a full or new moon coincides

with the lunar perigee, a perigean spring tide occurs. The resulting tide is very large with small differences between high and low tides because the gravitational pull on the earth is at its greatest.

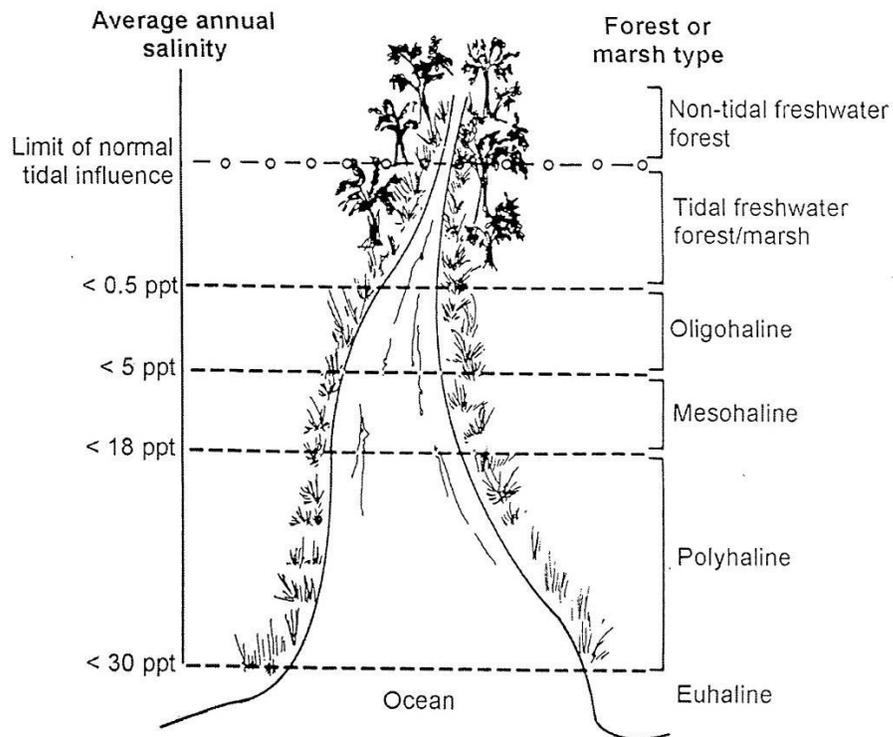
On a yearly basis, tidal range and extent is dependent upon several factors, including seasonal sea level, prevailing winds, and river discharge. Generally, sea level is at the lowest during the winter months and highest during the summer months (Doyle et al. 2007). Sea level fluctuations are due to oceanic thermal expansion, the proximity of planetary bodies, and seasonal changes in atmospheric pressure (Hicks 2006; Anderson and Lockaby 2011b). Prevailing winds also affect tide range by forcing wind driven tides. In the southeastern United States, prevailing winds in the spring and summer come from the southwest, and turn northeast during the autumn and winter months (SC DNR 2013). Southwest winds blow water up the estuary during the summer months resulting in greater tidal range, and northeast winds push water down towards the estuary during the winter months, resulting in both decreased tidal range and mean river stage.

River discharge variability arises from seasonal climate patterns and anthropogenic sources. Many rivers that discharge to the coast have been altered or impounded for municipal purposes with flows regulated by dam or tide gate releases. Generally, flows are highest during the winter and spring months coinciding with a period of higher rainfall and low evapotranspiration demands (Doyle et al. 2007; Anderson and Lockaby 2011b). During this time, the inland extent of tide may be dampened due to the interaction of low mean sea level and high river discharge. During the summer and fall, river flow is lower due to sporadic rainfall patterns and high

evapotranspiration rates. Consequently, the inland extent of tides may increase during the summer and fall due to higher mean sea level and low river discharge.

### ***Salinity and Flooding Gradients***

The frequency and depth of flooding is the primary controlling factor for vegetation patterns in wetlands (Rheinhardt and Hershner 1992; Mitsch and Gosselink 2007). However, along tidal gradients, salinity concentrations govern vegetation zonation and can override the effects of flooding (Odum 1988). Figure 2 is a conceptual diagram depicting the longitudinal salinity gradient present in the tidal river continuum.



**Figure 2. Conceptual diagram (not to scale) of salinity gradient and resulting vegetation communities found along a tidal river ecosystem (From: Odum et al. 1984).**

Salt marsh communities occupy the outer coastal and the adjacent estuary where salinity concentrations average 5 to more than 30 ppt. Vegetation zonation is distinct because few species are adapted to thrive in the harsh saline environment (Odum 1988). Smooth cordgrass (*Spartina alterniflora*) forms monotypic communities in the frequently flooded, low marsh zone, with dense colonies of black needlerush (*Juncus roemarianus*) in the less frequently flooded, high marsh elevations (Odum 1988; Wiegert and Freeman 1990). As salinity decreases upstream, species diversity increases and zonation becomes less distinct. In the oligohaline (brackish) marsh (0.5 – 5 ppt), colonies of smooth cordgrass and black needlerush are still common, but big cordgrass (*Spartina cynosuroides*) is more prominent, and is often mixed with bulrush (*Scirpus americana*) and pickerelweed (*Pontederia cordata*) (Wiegert and Freeman 1990). The most significant shift in vegetative community composition occurs in the zone where freshwater river outflow lowers average salinity concentrations to below 0.5 ppt. In South Carolina and Georgia the low marsh portion in the freshwater zone commonly includes spatterdock (*Nuphar luteum*), giant cutgrass (*Zizaniopsis miliacea*), and wild rice (*Zizania aquatica*). At higher elevations, smartweed (*Polygonum spp.*), cattail (*Typha spp.*), and rose mallow (*Hibiscus moscheutos*) become more common (Odum et al. 1984). The adjoining tidal forest, located at the landward edge of the marsh community represents an ecotone, where several emergent herbs and shrubs can occur in both habitats (high marsh and tidal forest/swamp). These include arrow arum (*Peltandra virginica*), jewelweed (*Impatiens capensis*), lizard's tail (*Saururus cernuus*), and wax myrtle (*Morella cerifera*) (Simpson et al. 1983; Odum et al. 1984; Sharitz and Pennings 2006). The flooding duration and frequency of tidal forests and swamps is similar to that of adjacent marshes, enabling

marsh species to thrive from year round soil saturation (Hackney et al. 2007), but not so intense that it excludes tree species. In this zone, the relative forest floor elevation, soil texture and type, and floodplain microtopography become important factors in determining forest structure and species distribution.

### ***Wetland Classification & Terminology***

Tidal freshwater forests exist in a narrow margin between the head of tide and freshwater marsh, and are distinguished from non-tidal forests by the presence of tidal hydrology for at least some part during the year (Day et al. 2007). A river is defined non-tidal where the mean tide range is less than 6 cm (Hicks 2006). However, due to seasonal and yearly variation of local climate, river discharge and sea level, the geographic location of this upper boundary can be dynamic. Additionally, tidal freshwater forests support a broad range of vegetative communities ranging from baldcypress and water tupelo stands in floodplains that are regularly flooded and maintain nearly constant soil saturation, to oak and gum forests that are seasonally flooded and closely resemble non-tidal bottomland hardwood communities (Light et al. 2002; Kroes et al. 2007). These descriptions make tidal freshwater forested wetlands difficult to categorize, hence terminology used is often confusing and inconsistent.

Currently, there is not a uniform classification system used to describe tidal freshwater forested wetlands (Table 1). This ambiguity in classification creates confusion about their role in the coastal zone. The most commonly used classification system is the U.S. Fish and Wildlife Service's Wetland and Deepwater Habitat

**Table 1. Wetland classification systems and related tidal freshwater forest/swamp terminology**

<b>Classification</b>	<b>Description</b>	<b>Water Regime Modifier</b>	<b>Reference</b>
Riverine (Stream Channel)	All wetlands and habitats contained within a channel (islands) with salinities < 0.5 ppt	Permanently flooded tidal, regularly flooded tidal, seasonally flooded tidal	Classification of Wetlands and Deepwater Habitats of the United States (Cowardin et al. 1979)
Palustrine Forested (Wetland)	All wetlands dominated by trees, shrubs and salinities less than 0.5 ppt.		
Upland (Zone E)	Upland zone of tidal creeks characterized by red maple, water willow, arrow-wood		The Ecology of Tidal Freshwater Marshes of the United States East Coast (Odum et al. 1984)
Fringe Wetland	Tidal swamp or marsh subject to astronomic tides, sea level controlled		A Hydrogeomorphic Classification for Wetlands (Brinson 1993b)
Tidal Freshwater Swamp	Sea-level controlled coastal wetland, no salinity		Brinson (1989)

Classification System, developed by the National Wetland Inventory. It used broad vegetation groups based on geology, but uses hydrologic descriptors and modifiers to distinguish between specific wetland types and classes (Cowardin et al. 1979; Mitsch and Gosselink 2007).

Tidal freshwater forests are categorized as either riverine (in-channel) or palustrine forested, which are denoted tidal by a water regime modifier. These wetland types are defined separate from the adjacent estuary system by a salinity threshold of less than 0.5 ppt and proximity to the open ocean. Estuarine wetlands are mostly open or only partially closed off to the ocean, where salinities can range from polyhaline (18 to 30 ppt) to mesohaline (5.0 to 18 ppt) further up the estuary (Cowardin et al. 1979). The riverine classification relates primarily to the area between the banks of a stream channel, and

describes both in-channel and forested cover. Palustrine forested wetlands include all wetlands dominated by trees. As riverine and palustrine classifications also apply to non-tidal wetlands, a water regime modifier must be used to denote that the wetland is subject to a freshwater tide. In a salt water tidal systems, three main types of flooding regimes exist subtidal, regularly flooded, and irregularly flooded, so to avoid confusion with saline environments, the non-tidal water regime modifiers are used, but the word “tidal” is added to differentiate them from a palustrine or riverine non-tidal system. Modifiers indicate wetland surface tidal inundation patterns, and include permanently flooded-tidal (the land surface is exposed less than once daily), regularly flooded-tidal (land surface is exposed at least one time daily), and seasonally flooded-tidal (land surface is flooded less than daily) (Cowardin et al. 1979).

Non-tidal palustrine wetlands are ubiquitous in the Southeastern Atlantic Coastal Plain, and are characterized by their horizontal, unidirectional surface flow with precipitation and overland flow as the primary sources of water (Brinson 1993a). These wetlands are typically associated with cypress-tupelo stands with wet regimes (deep flooding), and with the bottomland hardwood forest type in drier regimes. Tidal freshwater forests have similar vegetation communities, but the hydrologic regime is altered by tidal oscillations creating bi-directional flow and year round near saturated to saturated soils (Brinson 1993a). Fluctuating water table position from tidal flooding causes alternating periods of wet and dry corresponding with high and low tide cycling. The alternating pattern of soil saturation and desaturation is thought to increase decomposition rates and primary productivity (Brinson et al. 1981). With respect to hydrodynamics, the functionality of tidal freshwater forests is more similar to salt

marshes and estuarine wetlands than bottomland hardwood forests (Conner et al. 2007a). Therefore, if vegetative cover is the primary determinant for classification scheme (Cowardin et al. 1979), then tidal portions of palustrine wetlands may be inaccurately assessed due to the tidal /non-tidal forest continuum.

### ***Tidal Freshwater Forests as Riparian Systems***

Riparian wetlands exist at the interface of aquatic and terrestrial ecosystems, which are distinguished functionally by gradients of biophysical conditions, ecological functions, and biological communities (National Research Council [NRC] 2002). Their landscape position provides a hydrologic connection between water bodies and uplands by existing adjacent to rivers, streams, and lakes. In non-tidal riparian wetlands, the main sources of water are precipitation, groundwater discharge, overland flow, interflow, and surface runoff from the adjacent water body (NRC 2002). The gradient is largely unidirectional as water flows downhill, and water input from precipitation moves from the upland to the stream corridor via hillslope runoff, a collective mechanism, including overland flow and shallow surface flow. This process scours the substrate and mobilizes sediments that are deposited on the floodplain, or conveyed to the water body and transported downstream (Wharton et al. 1982).

Hydrologic processes combine with local geology, channel morphology, and bankside vegetation to create spatially variable sedimentation patterns with respect to particle size and structure. As water rushes across a substrate, sediment particles become suspended in the water column and transported to downstream locations. Sediments drop out of the water column according to size class, coarse fragments and sand particles are

first, followed by smaller silt particles, and finally, fine textured clays (Wharton et al. 1982). Creek bank levees are comprised of coarse-grained sandy sediments because they are the initial contact point between water and the floodplain. Interior floodplain sediments are stratified and comprised of finer grained sand, silts, and clays. The horizontal stratification of variable soil texture interacts with hydrology to drive floodplain biogeochemical processes (Wharton et al. 1982).

Hydrology and resulting hydrologic regime is the overarching factor that determines how a wetland functions in the landscape, and can be classified as hydroperiod; the frequency, duration and depth of flooding (Mitsch and Gosselink 2007). In riparian wetlands, river flood pulsing can be unpredictable and vary due to climate, seasonal river discharge or flooding events from storms (Junk, 1989; Tockner et al. 2010). However, under normal climate factors, riparian wetlands are usually flooded for some part of the year. Seasonal flooding usually occurs during the winter months when river discharge is high, trees are dormant, and evapotranspiration is low. At the start of the growing season, solar radiation drives transpiration and evaporative process causing ponded water to evaporate or recedes to below the ground surface. Additionally, groundwater flux to or from the water body may affect the water table position. This seasonal variation in hydroperiod is common in riparian wetlands adjacent to non-tidal water bodies. However, this process is altered when the riparian zone is connected to tidal water body. In a tidal riparian zone, the daily lunar or meteorologically driven tide is the primary mechanism driving the hydrologic regime of the wetland (Rheinhardt and Hershner 1992). The tidally driven hydroperiod, with regular intervals of wetting and

drying, creates an environment characterized by complex heterogeneous feedbacks unique to tidal forested ecosystems.

### ***Hydrologic Regime of Tidal Freshwater Forested Wetlands***

Studies in tidal freshwater forests and swamps have shown that wetland water table elevations are closely related to their associated tidal water bodies causing wetland soils to remain nearly always saturated, even during periods of low river flow; which is a key difference between non-tidal bottomland hardwood forests and tidal freshwater forests (Hackney et al. 2007). Hydroperiod is predominately affected by tides in tidal wetlands, but also from seasonal river flooding, groundwater discharge, and rainfall. Multiple inputs of water combined with microtopography, local climate (rainfall), elevation, geomorphology, source and type of soil parent material create a large degree of soil complexity (Anderson and Lockaby 2007; Day et al. 2007). Heterogeneity in floodplain soils causes physiochemical and biological processes such as decomposition and nutrient cycling to occur at different rates even within the same system. Differences in soil texture and floodplain microtopography create hydrologic microsites that determine the distribution of vegetative communities. These combined factors result in highly complex systems that are often difficult to generalize.

Flooding regimes present in tidal freshwater forested wetlands are broad and range from daily to seasonal tidal flooding, depending upon landscape position. It is important to differentiate here between a tidal freshwater forest and a tidal freshwater swamp. Swamps (both tidal and non-tidal) usually experience longer periods of flooding compared to forests. Forests may only be seasonally flooded, and possess an overall drier

hydrologic regime (Wharton et al. 1982). Additionally, tidal swamps are typically at a lower elevation in relation to mean sea level, and experience a greater degree of tidal flooding than tidal forests found at a higher relative elevation. Wharton et al. (1982) defined the upper limit of the tidal system as the point where natural levees prevent overbank flooding, but did not consider that in the uppermost regions tidal forcing is present in the subsurface, as evidenced by the rise and fall of the water table. This pattern has been observed in several studies (Rheinhardt 1992; Rheinhardt and Hershner 1992; Kroes et al. 2007; Duberstein and Conner 2009) where the water regime is described as seasonally tidal flooded. The presence of water table tidal forcing can inhibit the wetland to drain between tide cycles. This highlights how even under differing tidal regimes, forests and swamps share a commonality with regard to prolonged soil saturation. Hackney et al. (2007) studied several tidal freshwater swamps along the Lower Cape Fear River, and found that during periods when the tide did not flood the wetland surface, water levels declined below the soil surface. However, soils remained nearly saturated at all times because high water from tides occurred twice daily. The presence of hummock and hollow microtopography also affects soil saturation. Hummocks are elevated mounds of material (averaging +15 cm above the forest floor) comprised of mostly aerobic coarse material, and are large enough to support a few trees and shrubs (Duberstein and Conner 2009). Hollows are bowl shaped depressions at or just below the average wetland surface topography, characterized by long periods of saturation that restrict plant growth (Duberstein and Conner 2009; Courtwright and Findlay 2011). Hollows are thought to increase flood duration and soil moisture through depression storage and affect the frequency and depth of flooding (Courtwright and

Findlay 2011). Duberstein and Conner (2009) identified semi-diurnal tide signatures in groundwater hydrographs at backswamp sites (slough channels) and found persistently saturated soil conditions despite drought conditions. Another study conducted in tidal swamps along the Pamunkey River in Virginia found that the water table rose vertically in hollows corresponding with the high tide. It was noted that low tide cycles did not flood hollows, but also did not allow desaturation (Rheinhardt and Hershner 1992); suggesting that the low permeability soils did not allow for significant drainage during the low tide period.

While the general definition of the tidal freshwater forest hydroperiod seems to indicate prolonged soil saturation, not all areas of the floodplain remain permanently saturated due to elevation, landscape position, differences in soil texture, or distance from the water body. The localized soil moisture regime of tidal freshwater forests is just as dynamic as the flooding regime. Depth to oxidation varies spatially and temporally coinciding with seasonal sea level, water demand by plants during the growing season versus the dormant season, local climate, and topography (Day et al. 2007; Duberstein and Conner 2009; Anderson and Lockaby 2011a). Soils cycle between oxidized (unsaturated) and reduced (saturated) states that correspond to the position of the water table. In the oxidative state, a fraction of pore spaces in the soil matrix is filled with air, allowing oxygen to diffuse through the soil (Mitsch and Gosselink 2007). In a reduced state, the pore spaces are filled with water and dissolved oxygen concentrations decrease, consequently inhibiting oxygen uptake by plants. Soil oxidative state drives subsurface physiochemical and biological reactions, such as decomposition, and can define riparian zone vegetation communities (Courtwright and Findlay 2011).

### ***Organic Matter Decomposition***

Tidal freshwater ecosystems are known to have fast decomposition and nutrient exchange rates, where alternating cycles of wet and dry create optimal conditions for decomposition processes (Brinson et al. 1981; Ozalp et al. 2007). Tidal freshwater forested wetlands are thought to have the highest concentrations of soil organic matter due to persistent soil saturation (Wharton et al. 1982; Mitsch and Gosselink 2007). The average decomposition rate of organic matter in tidal freshwater forest is fast (decay rate ( $k$ ) = 1.8) compared to  $k = 1.1$  in riverine forests (Anderson and Lockaby 2007). During the decomposition process, a series of changes occurs including decaying, leaching, and the immobilization of nutrients (Ozalp et al. 2007). In forested systems, this process is driven by moisture, temperature, substrate quality, and time (Baker et al. 2001; Trettin and Jurgensen 2003). Studies on organic matter decomposition have used a variety of decomposition substrates including leaf litter, coarse woody debris, wood disks or stakes, and popsicle sticks both buried and on the soil surface (Baker et al. 2001; Ozalp et al. 2007; Romero et al. 2005; Courtwright and Findlay 2011). Comparisons of decomposition rates across studies are limited to the method (above or belowground) and substrate type, due to the differences in decomposer communities and organic matter quality. Leaf litter can take over a year to completely decompose (Ozalp et al. 2007); and woody substrates, such as popsicle sticks can take considerably longer due to wood resilience that prevents rapid fragmentation (Baker et al. 2001). However, it appears the rate of organic matter decomposition is most efficient in locations that alternate through aerobic and anaerobic conditions (Brinson et al. 1981). During aerobic conditions a variety of decomposers (bacteria, fungi, and fauna) are actively involved in the

decomposition process. During anaerobic conditions, the process is limited to work done by anaerobic bacteria (Trettin and Jurgensen 2003). Generally, prolonged periods of saturation or excessive dryness will retard the decomposition process by limiting the microbial community.

Findings from a study in a tidal freshwater swamp of the Hudson River, showed that decomposition rates were slower in continually flooded hollows compared to hummocks that alternated between wet and dry (Courtwright and Findlay 2011). Decomposition rates were strongly affected by moisture, temperature, and anaerobic versus aerobic respiration (Courtwright and Finlay 2011). In a yearlong decomposition study within a tidal freshwater swamp of the Pee Dee River, water tupelo leaf litter decomposition was most efficient at sites that were flushed daily by tide compared to those that were continually saturated (Ozalp et al. 2007). In their study, Ozalp et al. (2007) found only 12% (decay constant ( $k$ ) = 2.04) of the original mass remained after 196 days at the site that was flushed daily, compared to 20% after day 273 ( $k$  = 1.59) at the saturated site. In the Coosawhatchie River basin, the decomposition rates of popsicle sticks were evaluated over an 80-week period in two non-tidal forested wetlands with differing flooding regimes (Baker et al. 2001). The wet site was a frequently inundated sweetgum-swamp tupelo community (52% of 100 weeks), and the dry site was a less frequently inundated laurel oak stand (21% of 100 weeks). Results of the study indicated that 38.7 % ( $k$  = 0.564 yr<sup>-1</sup>) of mass remained at the dry site versus 9.4% ( $k$  = 0.906 yr<sup>-1</sup>) of original mass remaining at the wet site. Although the wet site was flooded considerably longer than the dry site, it was noted that flooding was not constant and several cycles of wetting and drying likely increased the rate of decomposition.

### ***Vegetative Communities***

The vegetative cover and composition of tidal freshwater forest is mainly the result of the long-term hydrological conditions present, but is also influenced by inter-annual flooding from seasonal sea level, river flow patterns, and rare episodic flooding events from storms (Rheinhardt and Hershner 1992; Day et al. 2007). The canopy composition is influenced by the relationship between the forest floor elevation and mean water level, which relates to flooding patterns (Day et al. 2007). Baldcypress – tupelo communities dominate where the swamp floor is at the same elevation as the mean high water level and experience prolonged flooding. Bottomland hardwood forest type dominates at higher relative elevations and experience short flooding duration (Wharton et al. 1982; Day et al. 2007). Wetland trees are adapted for life in hydric soils, but each possesses a threshold of waterlogging tolerance, which is the ability to tolerate flooding during the growing period before mortality will occur (Hook 1984; Theriot 1993). Hummock and hollow microtopography cause spatially complex inundation patterns and play a key role in vegetative structure (Anderson and Lockaby 2007; Duberstein and Conner 2009; Courtwright and Findlay 2011). Hummocks have shown to be important areas for plant respiration during anaerobic conditions (Duberstein and Conner 2009).

Forest type is also a function of topographic relief and elevation. In low lying Louisiana, most of the tidal swamps are pure stands of baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*), compared to higher elevation swamps and forests in South Carolina, whose canopies are mixed and comprised of baldcypress, water tupelo, swamp tupelo (*Nyssa sylvatica*), red maple (*Acer rubrum*), and Carolina ash (*Fraxinus caroliniana*) (Conner et al. 2007a).

**Table 2. Vegetation communities and tidal flooding regimes observed in previous studies of tidal freshwater forested wetlands.**

Study Site	Dominant Forest Type	Sub-canopy/ Understory	Tidal Flooding Regime	Reference
Pamunkey River, VA	Ash – blackgum (baldcypress)	Northern spicebush, common winterberry, highbush blueberry	Regular tidal flooding	Rheinhardt 1992; Rheinhardt and Hershner 1992; Rheinhardt 2007
	Maple – sweetgum (ash)	American hornbeam	Irregular tidal flooding, (water table rise)	
Pamunkey River, VA	Ash, blackgum, carpinus, red maple	Alder, smilax, Asian spiderwort, sedges	Regular flooding (high tides), Zone II	Doumlele et al. 1985
Hudson River, NY	Ash – maple	Alder, dogwood, honeysuckle, spice bush	Regular flooding (hhw flooded swamp)	Courtwright and Findlay 2011
Savannah River, GA	Water tupelo, swamp tupelo, water oak, sweetgum, red maple, baldcypress, carpinus	Alder, dahoon holly, wax myrtle, swamp doghobble, swamp bay	<u>Streamside</u> : irregular flooding <u>Backswamp</u> : (water table rise)	Duberstein and Kitchens 2007; Duberstein and Conner 2009
Grand Bay and Escatawpa River, LA & Pascagoula River, MS	Water tupelo, swamp tupelo, loblolly, Atlantic white cedar	Swamp titi, red maple, water tupelo	Flooded most of the growing season, irregular tidal regime- isolated by high levees	Keeland and McCoy 2007
Pocomoke River, MD	<u>Tidal site</u> : Baldcypress, red maple, ash, water tupelo, sweetgum	Not surveyed	<u>Tidal</u> : Short periods of inundation from high tide	Kroes et al. 2007
	<u>Fluvial site</u> : Carpinus, red maple, swamp chestnut oak, willow oak, ash, overcup oak, baldcypress	Not surveyed	<u>Fluvial</u> : Groundwater tidal signal 0.1 -0.2 m	
Suwannee River, FL	Baldcypress, pumpkin ash, swamp and water tupelo, wax Myrtle	Pumpkin ash, Carolina ash, Carpinus, wax myrtle	<u>Lower tidal</u> : Regular flooding <u>Upper tidal</u> : isolated by levees	Light et al. 2002; 2007
Waccamaw River, SC	Baldcypress – water tupelo, ash, red maple	Alder, knotweed sp.	Regular tidal flooding	Conner et al. 2007a; Cormier et al. 2012

Due to intense flooding patterns and prolonged soil saturation, the general structure of the tidal freshwater forest is scrubby, allowing light to penetrate the swamp floor (Baldwin 2007). Gaps of light allow a well-developed shrub and ground strata to thrive. Thus, species richness and diversity can rival both freshwater marshes and bottomland hardwood forests because the composition is frequently a mixture of both habitats (Baldwin 2007).

### **Need for Study**

In the past, research focusing specifically on tidal freshwater forested wetlands was infrequent (Brinson et al. 1981; Wharton et al. 1982; Simpson et al. 1983; Doumlele et al. 1985; Odum 1988; Field et al. 1991; Rheinhardt 1992; Rheinhardt and Hershner 1992), but has gained popularity in the past decade especially with the release of the book, *Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States* (Conner et al. 2007b). Locations of tidal freshwater forests and swamps associated with large river systems have been documented (See Table 2) and biogeochemical changes from sea level rise and climate change have been observed. However, a majority the literature focuses on salinity intrusion, vegetation patterns, and water dynamics along the marsh/forest boundary (Hackney et al. 2007; Krauss et al. 2009; Anderson and Lockaby 2011a; Cormier et al. 2012). Still lacking are detailed studies concerning hydrodynamics, biogeomorphic feedbacks, and seasonal patterns of surface water - groundwater interaction along the tidal/non-tidal transition zone. This research will interpret fine-resolution, hydrologic data and associated biologic feedbacks for a headwater system not currently experiencing saline intrusion. As sea level rise pushes

the tidal prism inland, non-tidal bottomland forests will become affected by freshwater tides, and it is unknown how this will alter the flux of nutrients and organic matter entering streams. Understanding those functional linkages is essential to considering the effects of sea level rise on estuarine health.

### **Research Objectives**

The overall goal of this study was to characterize the physical and biological aspects of a tidal freshwater forested wetland within the Huger Creek watershed. The research objective was to determine whether a freshwater tidal stream influences the hydroperiod and biological processes within the tidal freshwater riparian zone by quantifying differences in water table dynamics, soil moisture regime, vegetative communities, and decomposition rates between a tidally influenced and non-tidal wetland within the same drainage system.

Primary hypothesis: The soil moisture regime as determined from water table position tracked both spatially and temporally in a tidally – influenced freshwater forested wetland is consistently and measurably more wet and less variable than a non-tidal forested wetland assuming similar topographic positions.

Primary null hypothesis: The soil moisture regime of a tidally – influenced freshwater forested wetland is not wetter, nor less variable than a non-tidal forested wetland assuming similar topographic positions.

Corollary hypothesis 1: Vegetative communities will have discrete differences along the decreasing (tidal) wetness gradient (i.e., proximity to the tidally – influenced creek), and between the tidal and non-tidal riparian zones.

Corollary hypothesis 2: Organic matter decomposition rates will be higher in the tidally influenced riparian zone, where variable water table position produces alternating patterns of aerobic and anaerobic conditions.

Tidal freshwater systems are dynamic and ecologically important in the landscape; they provide the link between the aquatic (downstream estuary) and upland terrestrial environment. Characterizing riparian zones at the upper boundary of tidal influence will provide insight about how sea level rise and climate change may change their functionality. Collecting these types of data will provide researchers and land managers the knowledge and understanding needed to anticipate and prepare for future change.

## **CHAPTER 2. HYDROLOGIC TRENDS AND BIOLOGICAL RESPONSE IN A TIDAL FRESHWATER FORESTED WETLAND**

### **Introduction**

Tidal freshwater forests play a vital role in the landscape as transitional areas that deliver ecosystem services to the estuary and act as important habitat areas. Prior research on tidal freshwater forested ecosystems has described the range of vegetative communities, role of microtopography, biogeochemical processes, sedimentation patterns, and ecosystem changes associated with salinity intrusion from sea level rise (Hackney et al. 2007; Kroes et al. 2007; Conner et al. 2007; Conner et al. 2009; Duberstein and Conner 2009; Anderson and Lockaby 2011a; Anderson and Lockaby 2011b; Courtwright and Findlay 2011). A majority of the related studies have been conducted in large tidal freshwater forests or swamps that experience daily wetland surface tidal flooding, resulting in tide being the primary driver for hydrologic regime and associated biogeochemical response. Additionally, many of the known tidal freshwater forested wetlands are associated with large rivers along the Southeastern Coastal Plain have been subject to anthropogenic alteration for commercial or municipal use, with examples including the Savannah, Apalachicola, Hudson, Waccamaw and Pee Dee Rivers. Less attention has been paid to low order streams or headwater systems where tidal conditions are present, but surface flooding may not occur on a daily basis. In these reaches, the influence of precipitation, evaporative processes, tide, and freshwater combine to drive hydrologic regime and associated biogeochemical response.

Wetland hydrology, characterized by hydroperiod (the timing, depth, and duration of flooding) is the most important factor governing plant distributions and biogeochemical processes in wetlands (Burke et al. 2003; Mitsch and Gosselink 2007). The zone of transition between a tidal freshwater forest and adjoining non-tidal bottomland hardwood forest is an ideal location to test how water dynamics (i.e., high water table from tidal forcing) influence physical and biological processes in a wetland. Unfortunately, this area has been particularly difficult to identify because 1) it is dynamic and subject to changes in seasonal sea level and river discharge, and 2) uninterrupted forest cover obscures the diminishing tide range (Day et al. 2007; Anderson and Lockaby 2011b). Additionally, these systems have been overlooked because, as Day et al. (2007) states, hydrology for an entire swamp is often inferred from a single water table well or a tide gauge in the adjacent water body. In addition, it is thought that a significant shift from tidal to fluvial hydrodynamics occurs near the limit of tidal excursions (Anderson and Lockaby 2011a), but it has not been well documented. In these transitional systems, levees are at elevations such that high tides do not inundate the surface; tidal flooding occurs seasonally or during spring tide cycles (Wharton et al. 1982). Therefore, few studies have focused on the ecology in the riparian zone transitioning from tidal to non-tidal conditions (Doumlele et al. 1985; Rheinhardt 1992; Rheinhardt and Hershner 1992; Light et al. 2002; 2007 Kroes et al. 2007). This contributes to the uncertainty in how freshwater tides may change riparian ecological processes such as nutrient flux, primary productivity, habitat quality, and biogeomorphic feedbacks.

Hydrologic patterns in the transitional zone are mixed, representing both tidal and non-tidal systems; the influence on hydroperiod from runoff, precipitation,

evapotranspiration, groundwater discharge, and recharge are mediated by the tide (Day et al. 2007). The hydroperiod within tidal forests is highly dynamic on the short-term (in response to daily tide fluctuations), but has a relatively low variability on an annual basis (Rheinhardt and Hershner 1992). Tidal freshwater forests cycle through intense, yet brief patterns of surface or periods of high water table in response to high and low tide cycling. Flooding patterns (i.e., alternating periods of wet and dry and corresponding aerobic and anaerobic soil conditions) influence biological processes in wetland soils including the rate of decomposition (Trettin and Jurgensen 2003). This contrasts to non-tidal bottomland hardwood forests above the tidal influence where hydroperiod is highly variable depending on climate factors (drought or excessive rainfall), water demand from plants, topography, and elevation (Anderson and Lockaby 2011a). Bottomland hardwood forests are seasonally flooded, but it is not uncommon for water table position to decline to depths of one meter or more during the growing season (Harder 2004).

Vegetation communities found in both the upper portions of tidal freshwater forests and bottomland hardwood forests are largely the result of hydrology in the rooting zone. Both forest types are known to be high in species diversity and richness, and it is reasonable to assume that the communities are similar (Baldwin 2007). Freshwater marsh species and common forest herbs typically co-exist in the understory of tidal freshwater forested wetlands (Odum et al. 1984; Doumlele et al. 1985). The composition of the understory is complicated by other factors including light and nutrient availability, but the canopy composition of tidal freshwater swamps is the result of long-term hydrologic conditions, elevation, past land use, and disturbance. Rheinhardt and Hershner (1992) found that canopy composition in a tidal freshwater swamp was related

to the where the water table resided 20% - 80% of the time, rather than the duration or amplitude of tidal flooding. Bottomland hardwood vegetation is adapted for periodic flooding, and tidal freshwater forests exist along a gradient, which can span several floodplain hydrologic zones (Wharton et al. 1982). Floodplain zones (Zone I – open water to Zone V – upland) were defined according to flooding duration and intensity. Tidal freshwater wetlands are found between Zone II (intermittently exposed/ nearly permanent saturation) and Zone IV (seasonally flooded) floodplain designations. Non-tidal bottomland hardwood swamps are typically associated with Zone IV to Zone V (temporarily inundated or saturated) floodplain designations (Wharton et al. 1982). These are the highest areas of the floodplain (terraces or flats) where species are not well adapted to tolerate prolonged flooding.

The objective of this study is to describe processes present at the tidal/ non-tidal boundary of a tidal creek and its related tidal freshwater forested wetland with respect to hydrologic regime (surface water patterns, water table position, and soil moisture), soil oxidation depth, organic matter decomposition, and vegetation patterns. These processes and characteristics will be compared to a non-tidal bottomland hardwood forest. The primary null hypothesis is, the soil moisture regime of a tidal freshwater forested wetland is not wetter than a non-tidal bottomland hardwood forest. Alternatively, the soil moisture regime within the tidally influenced forest is wetter than the non-tidal forest. Specifically, the hydroperiod of the tidal freshwater forest is a function of the tidal regime in Huger Creek, and the hydroperiod of a non-tidal bottomland forest is primarily dependent upon stream stage and precipitation. Additionally, a shift from tidal to fluvial hydrodynamics will be evident as the tide diminishes, and will affect the soil moisture

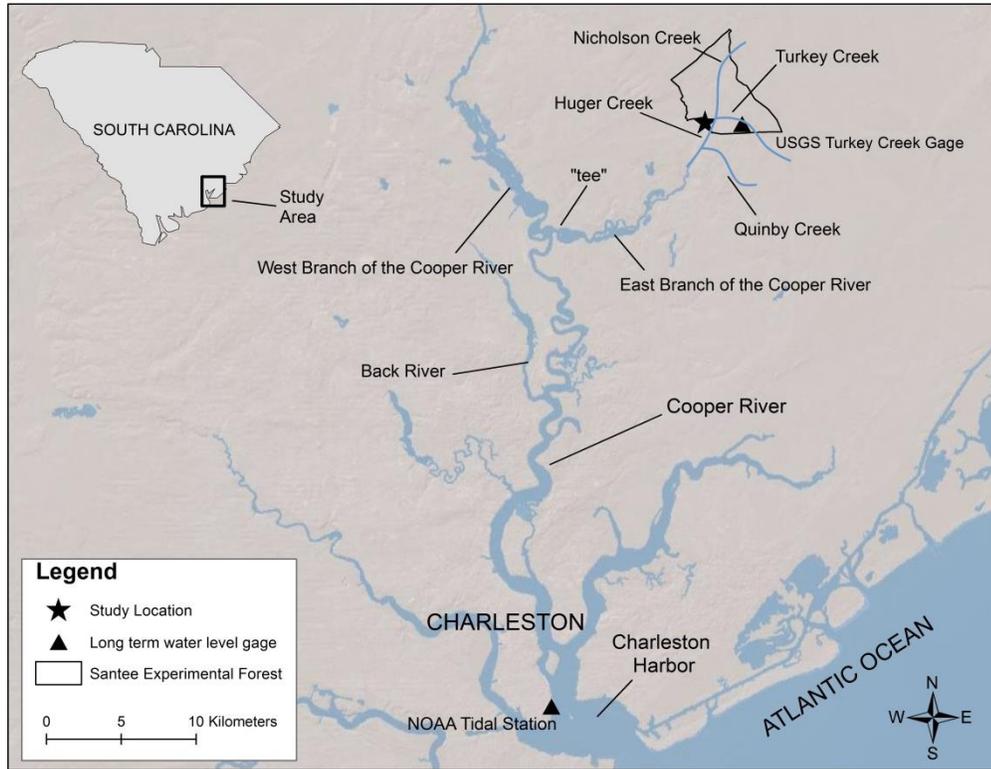
regime in the riparian wetland. The shift in hydrodynamics will be reflected in the biological response variables along the diminishing tidal gradient and will contrast to the non-tidal system. This study has broad implications for understanding the hydrodynamics in the transition zone between a tidal freshwater forested wetland and non-tidal bottomland hardwood forest. The collection and interpretation of these data are fundamental to understanding how bottomland hardwood forests will respond to sea level rise.

## **Methods**

### *Study Area*

The study site is the Huger Creek watershed located in the Santee Experimental Forest (hereafter SEF), within the Francis Marion National Forest, located 60 km northeast of Charleston, South Carolina (Figure 3). Huger Creek is a fourth order stream tidal freshwater stream draining approximately 23,521 ha. Two third order watersheds, Nicholson and Turkey Creek form Huger Creek at their confluence. Huger Creek flows southwest, is joined by Quinby Creek forming the East Branch of the Cooper River. The East Branch joins the Cooper River, and eventually discharges into the Charleston Harbor estuary.

The 13 km reach of the East Branch of the Cooper River, from the “tee” up to the confluence of Quinby Creek is described as a tidal slough (Conrads and Smith 1997). The floodplain along the entire length of the East Branch of the Cooper River is vegetated by freshwater marsh. Huger Creek is approximately 5.5 km long; the lower 2.8 km is freshwater marsh, with forested cover along its upper 2.7 km reach. The width of



**Figure 3. Location of study area within the Santee Experimental Forest (SEF forest boundary denoted by black outline) in proximity to the Atlantic Ocean, major tributaries of the Cooper River basin, and locations of long – term water level gages**

the stream channel narrows and becomes anastomosed within the forested reach, and small vegetated islands are common.

The Huger Creek floodplain is flat and wide, 500 m at the widest point, but not equal in width on both sides. The study sites for this project were established on the narrow (100 m wide) southern side of the floodplain at the base of a hillslope where an old tramline existed (Figure 4). The majority of the riparian zone is not inundated daily by either high tide because water level does not exceed creek bank levees; some surface flooding may occur in the lowest portion of the surface area. It is likely that high tides enter the floodplain via scour channels and water table rises vertically from tidal forcing causing ponding in hollows and backswamp areas of the floodplain (personal

observation). The riparian zone study plots range between 0.93 and 4.9 m, and are referenced to the North American Vertical Datum of 1988 (NAVD88).

The climate in the region is humid – subtropical with long hot summers and mild humid winters (Dai et al. 2011). A cold, dry continental air mass is predominant during the winter months with prevailing winds from the northeast. During the summer, winds prevail from the southwest which bring moisture from the Bermuda High in the Atlantic and warm air from the Gulf of Mexico (SC DNR 2013). The long-term (1946 - 2007) mean ambient air temperature is 18.5 °C and the average annual rainfall is 1370 mm, with most rainfall occurring during the summer months (Dai et al. 2011). Seasons are delineated as winter (December – February), spring (March – May), summer (June – August), and fall (September – November) (NWS 2013). The growing season in coastal South Carolina is long, lasting from 15 March – 15 November (Harder 2004).

Soils in the floodplain are dominated by the Meggett series (Meggett Loam [Mg] and Meggett Clay Loam [Mp], taxonomically described as “fine, mixed, active, thermic Typic Albaqualfs” (NRCS 2013; Long 1980). Characterized as deep, nearly level, and clayey; Meggett soils have a low hydraulic conductivity, low specific yield, and high water retention capacity. The soil texture is fine sandy loam to sandy clay in the upper 8 to 40 cm, with masses of oxidized and reduced peds common from seasonal high or low water table. Depth to the argillic horizon (clay) ranges from 8 – 40 cm and extends to approximately 127 cm. Below the argillic horizon, the texture is sandy clay, here oxidized and reduced peds are common (NRCS 2013).

Vegetation within the riparian zone is associated with the bottomland hardwood forest type described as deciduous, or mixed deciduous/ evergreen, closed-canopy forest

communities occupying terraces and levees within riverine floodplains (Wharton et al. 1982). The canopy of the bottomland hardwood type includes sweetgum (*Liquidambar styraciflua*), spruce pine (*Pinus glabra*), laurel oak (*Quercus laurifolia*), water oak (*Quercus nigra*), live oak (*Quercus virginiana*), swamp chestnut oak (*Quercus michauxii*), sugarberry (*Celtis laevigata*), American elm (*Ulmus americana*), red maple (*Acer rubrum*), swamp tupelo (*Nyssa biflora*), and baldcypress (*Taxodium distichum*). Frequent sub-canopy and shrub species include ironwood (*Carpinus caroliniana*), swamp (stiff) dogwood (*Cornus foemina*), possumhaw (*Ilex decidua*), wax myrtle (*Morella cerifera*), dwarf palmetto (*Sabal minor*), swamp bay (*Persea palustris*), and highbush blueberry (*Vaccinium corymbosum*) (Wharton et al. 1982; Harms et al. 1998; FNAI 2010).

The present canopy composition of the SEF forest has been influenced by anthropogenic alterations related to agriculture, timbering, and natural disturbance. Prior to colonization of the region during the late 1600s, the landscape was a mosaic of pine-hardwood flatwoods and bottomlands characteristic of the region. Beginning in the early 1700s, the landscape and drainages were altered for rice and food crop agriculture that continued throughout the 19<sup>th</sup> century (Czwartacki and Trettin 2013). Large rice fields occupied the broad floodplains; streams were channelized and impounded to provide flood irrigation to crops. After the collapse of the rice industry in the late 1800s, plantation lands were timbered until 1933, which ceased with the acquisition of the land by the U.S. Federal Government. The SEF was established four years later to serve as a research unit focusing on the silvaculture of loblolly pine (*Pinus taeda*).

Over the last several decades, natural afforestation has taken place, but through episodic disturbance. Hurricane Hugo, a category 4 hurricane, made landfall on September 21, 1989 near Bulls Bay, South Carolina. It was estimated that over 80% of the canopy was destroyed, and all the major forest stand types within the SEF were affected; in the bottomlands, 86% of trees were either broken or uprooted (Hook et al. 1991). Despite the widespread destruction, Song et al. (2012) state that the composition and dominance of canopy species remained similar pre and post hurricane damage.

### ***Study Site Monitoring Infrastructure and Design***

Field monitoring stations were established within the Huger Creek stream channel and along transects within the riparian zone. An additional non-tidal transect was established at Turkey Creek approximately 2.3 km upstream the head of the tide. Monitoring transects were labeled Lower Lower Tidal (LLT), Lower Tidal (LT-1-LT-3), Middle Tidal (MT-1-MT-3), Upper Tidal (UT-1-UT-3), and Non-Tidal (NT-1-NT-3). The naming convention reflects both an increasing distance upstream, and increasing distance from the stream channel towards the wetland interior. The site design assumed that a decreasing wetness gradient would exist moving upstream, corresponding with the decreasing tidal influence in Huger Creek. The non-tidal replicate site was chosen to identify differences in stream flow dynamics, floodplain processes, and vegetation communities between the tidal reach and the upstream non-tidal reach.

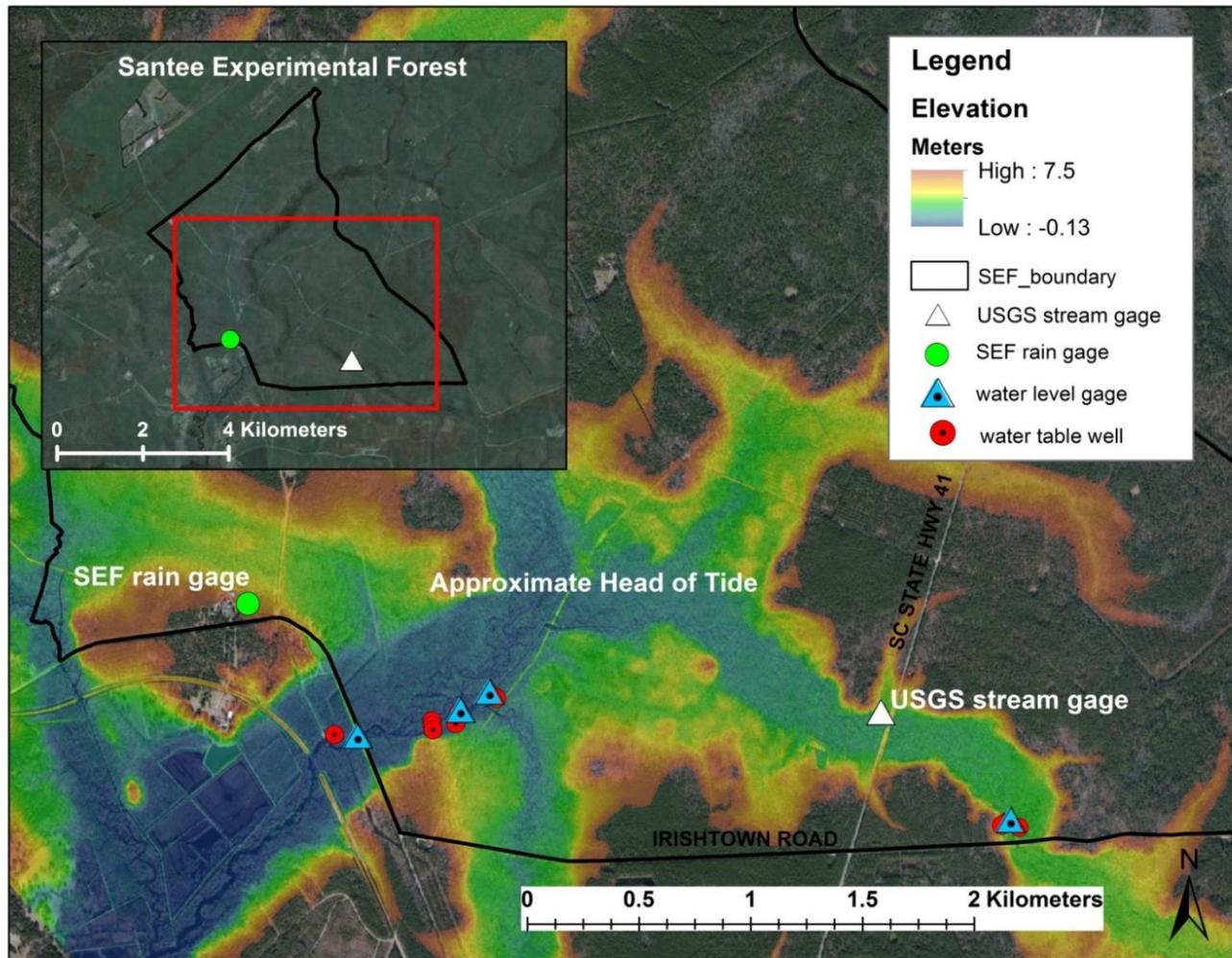


Figure 4. Aerial imagery (Source: ESRI World Imagery, <http://services.arcgisonline.com/arcgis/services>; Projection: NAD1983 UTM Zone 17N) and a LiDAR DEM (Light Detecting and Ranging Digital Elevation Model) (Source: <http://cybergis.unc.edu/santee/LiDARData.html>, Photo Science, Inc.) showing elevation in the floodplain and the location of monitoring sites, the Santee Experimental Forest rain gage, and USGS stream gage (no. 02172035) used in this study

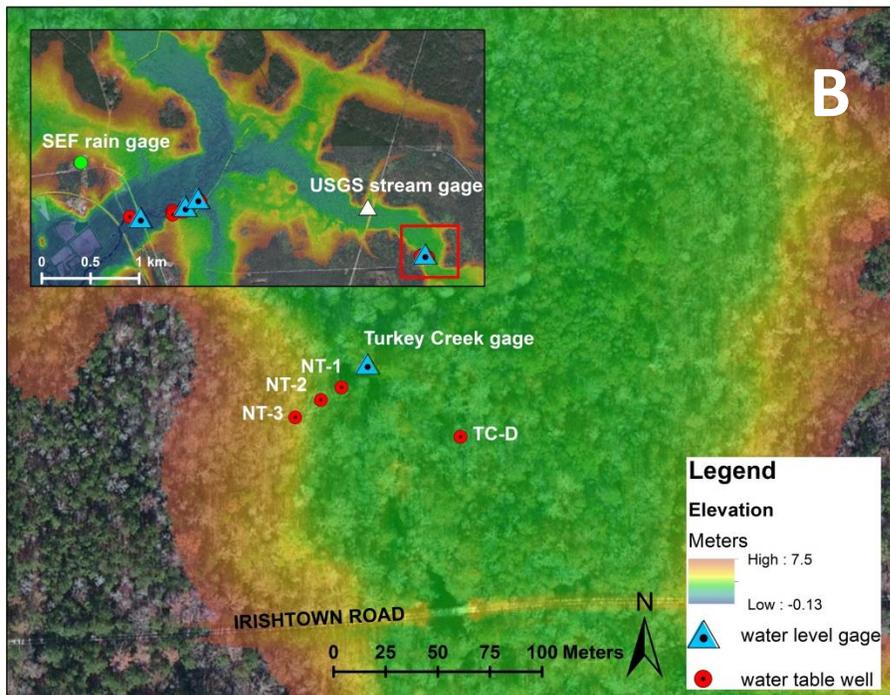
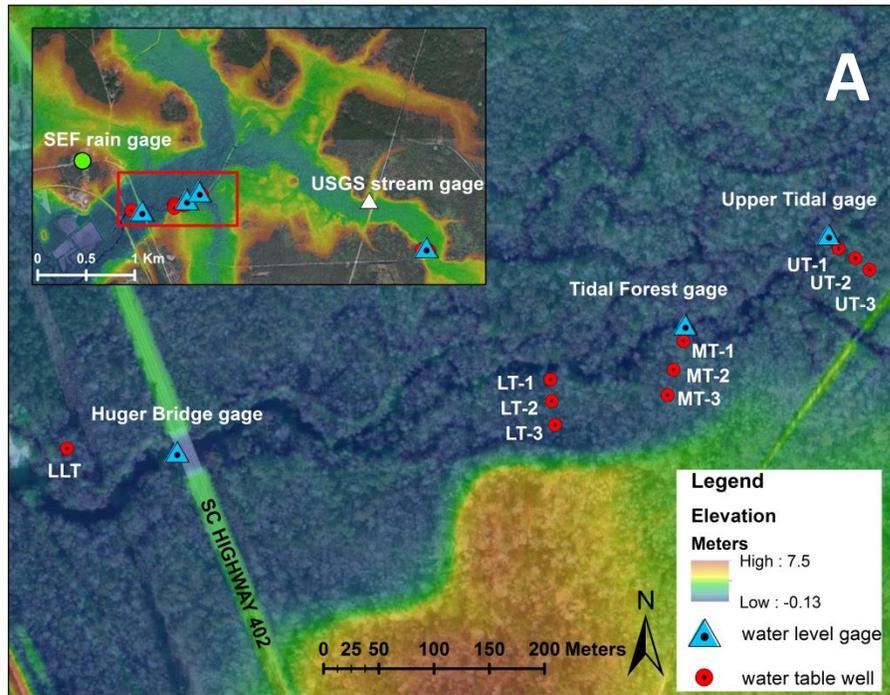


Figure 5. Aerial imagery (Source: ESRI World Imagery, <http://services.arcgisonline.com/arcgis/services>; Projection: NAD1983 UTM Zone 17N) and a LiDAR DEM (Light Detecting and Ranging Digital Elevation Model) (Source: <http://cybergis.uncc.edu/santee/LiDARData.html>, Photo Science, Inc.) showing elevation in the floodplain and hydrologic monitoring transects in the tidally influenced Huger Creek study site (A) and in the non-tidal Turkey Creek site (B). Locations of USGS stream gage and SEF rain gage are shown in inset maps.

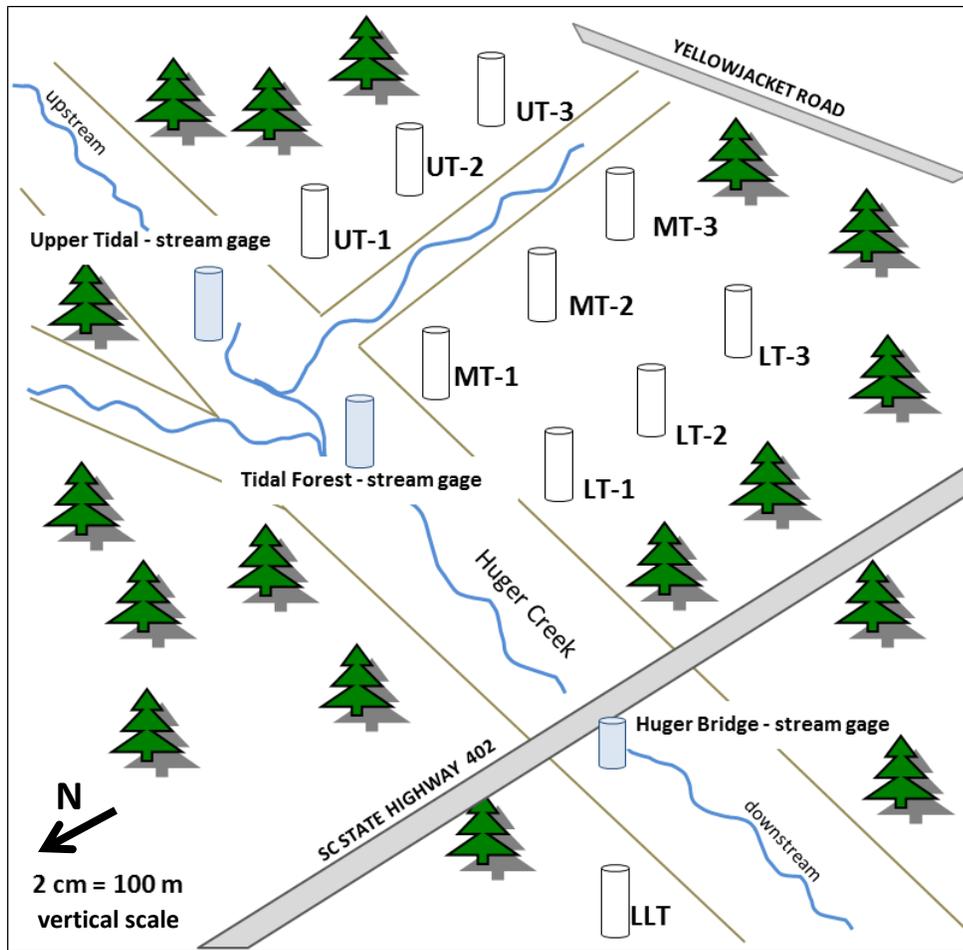


Figure 6. Conceptual diagram of hydrologic monitoring transect layout in the tidally-influenced study area. Approximate longitudinal (upstream/downstream) scale: 2cm = 100 m, transverse horizontal distance not to scale.

Each site was represented by a water table well and a vegetation plot. Selected sites also included a soil moisture monitoring plot and below ground organic matter decomposition plot. Ground elevation was determined using a bare earth, Light Detecting and Ranging (LiDAR) Digital Elevation Model (DEM) (Photo Science, Inc., 2007) and verified in the field with a (Spectra Physics, Laserplane 650) laser level. All elevations (wetland surface and streambed) were determined in relation to the North American Vertical Datum of 1988.

## ***Hydrologic Monitoring***

### *Surface Water*

Stream stage of Huger Creek was measured using automatic logging water level sensors at two locations. The Huger Creek Bridge gage (HCB<sub>r</sub>) was installed beneath an overpass on South Carolina State Highway 402. A second gage (Tidal Forest) was installed 450 m upstream from the bridge at the middle tidal (MT) transect. Manual staff gages were located at the bridge gage, tidal forest gage, and upper tidal transect to collect manual readings where automatic loggers were absent, for calibration, and verification of automatic logger output. Stream gage housing constructed of 5.8 cm diameter, perforated polyvinyl chloride (PVC) functioned as a stilling well for a pressure transducer (WL-16; Global Water, Inc., Gold River, CA) suspended inside that was set to collect stage at fifteen-minute intervals. At the Turkey Creek site, the stream gage was set to collect stage at thirty-minute intervals. The Huger Bridge gage was established in February 2011, with other gages (Tidal Forest and Turkey Creek) coming online in July 2011. A USGS managed stream gage (Gage no. 02712035) was located on Turkey Creek beneath an overpass at South Carolina State Highway 41 (Figure 4). Its location was between the tidal and non-tidal sites and collected stream stage and discharge. Data were available for download via the National Water Information System web interface (USGS 2013).

### *Water Table*

Wells constructed of vented, 3.8 cm diameter PVC installed to a depth of 2 m to enable measurement of water table position. A 10.2 cm diameter, open bucket hand auger was used to dig wells, and soil profile data were recorded during installation (See APPENDIX A for detailed soil profile descriptions). Wells were backfilled with coarse-

grained sand, the upper 10 – 20 cm of the annular space was sealed with a bentonite cap, and backfilled with native material. Wells LLT, LT-1, LT-2, LT-3, MT-1, MT-2, UT-1 and NT-1 were instrumented with pressure transducers (Levellogger Gold, Solinst Ltd., Georgetown, Ontario, Canada) set to record data at thirty-minute intervals. All water table readings were corrected for barometric pressure effects on-site with a Solinst Barologger (Solinst Ltd., Georgetown, Ontario, Canada). Raw data were transformed into depth below surface and m NAVD88 for analysis. Wells that did not contain pressure transducers were measured during field visits with a Solinst water level tape (Solinst Ltd., Georgetown, Ontario, Canada). During site visits, wells were downloaded and manually measured with a Solinst water level tape to verify water depth below surface.

Well installation took place during May and June 2011, with the exception of LLT, which was installed in February 2012. LLT was not associated with an established transect, and was located approximately 100 m downstream from the bridge stream gage site (Figure 5A). The LT transect was located 343.5 m upstream from the HCBr stream gauge, with wells installed at distances of 3.5 m (LT-1), 24.3 m (LT-2), and 49.5 m (LT-3) from the stream channel. The MT transect was located 123.3 m upstream from LT, with wells installed at distances of 5.9 m (MT-1), 34.5 m (MT-2), and 55.4 m (MT-3) from the stream channel. The UT transect was located 167.8 m upstream from MT, with wells installed at distances of 5.9 m (UT-1), 18.4 m (UT-2), and 34.3 m (UT-3) from the stream channel.

Wells within the Turkey Creek drainage were installed at distances of 5.3 m (NT-1), 17.8 m (NT-2), and 34 m (NT-3) from the main channel (Figure 5B). In a previous

study on the Turkey Creek watershed (Renaud 2008), a water table well (TC-D) was installed within a depression in the riparian zone on the opposite side of the floodplain. Water table level has been collected on an hourly basis since 2006. These data were used for comparison to look at water table trends since 2009, and to serve as a comparison for collected at Turkey Creek during this study.

### *Soil Moisture*

Soil moisture was measured in plots at sites LT-1, LT-2, MT-1, MT-2, UT-1, and NT-1. The chosen locations for soil moisture plots reflected the predicted decreasing wetness gradient corresponding with diminishing tidal influence, and a replicate non-tidal plot. Decagon ECH<sub>2</sub>O - EC-5 soil moisture sensors (Decagon Devices, Inc., Pullman, WA) were installed horizontally into the undisturbed soil matrix at depths of 25 cm, 50 cm, and 75 cm below ground surface. The sensors were not individually calibrated to the Meggett soil type, and the default factory calibration was used. Sensors measured the dielectric constant of the soil, which is a strong indicator of soil water content (Cabos and Chambers 2010). Sensors were programmed to measure volumetric water content (VWC m<sup>3</sup> water m<sup>-3</sup> soil) on an hourly basis. Measuring VWC m<sup>3</sup> m<sup>-3</sup> at specific depths was to allow for the comparison of volumetric water content with changes in depth to water table from tide fluctuations, seasonal variation, or rainfall.

## ***Biological Response Monitoring***

### *Soil Oxidation Depth*

Iron rods (rebar) were inserted into the undisturbed soil to a depth of 140 cm at the same locations as the soil moisture plots. Rods were read several times throughout the study, and were used as an indicator of soil oxidation depth (McKee 1979; Bridgham et al. 1991). The portions of the rods that were exposed to oxidative conditions were rusted and orange-red in color, and portions that were exposed to anoxic conditions were a flat inky black. Using a measuring tape, the depth to oxidation was recorded. The iron rod readings were used in conjunction with depth to water measurements and soil moisture to infer the mean oxidation status and soil moisture regime at each site.

### *Organic Matter Decomposition*

Wood decomposition was measured every 60 days for a period totaling one year. Decomposition plots were located at sites LT-1, LT-2, MT-1, MT-2, UT-1, NT-1, and NT-2. Similar to the soil moisture and iron rod plot locations reflected the predicted decreasing wetness (tidal) gradient with the reasoning that hydroperiod would affect the decomposition rate (Baker et al. 2001; Ozalp et al. 2007). Commercially obtained craft sticks (Horizon Group, USA) were 1.91 cm x 15.24 cm wood (Species unknown). The choice of craft sticks provided a homogenous medium to evaluate decomposition as mass lost over time. Sticks were air dried at 60 °C for 24 hours, then weighed and labeled with a plot and replicate number. For each plot, there were five replicates for each sixty-day period of decomposition. Sticks were installed to a depth of 15 cm below ground surface by digging a small trench, and inserting sticks vertically into the undisturbed soil matrix. The purpose of vertical positioning reduced the occurrence of water ponding on top of

sticks. Every 60 days, sticks were retrieved from the field. In the lab, sticks were cleaned of soil and debris, and air-dried at 60°C until a constant weight was achieved. Data were recorded as percent mass remaining relative to initial mass, and compared as mass remaining over time at each site.

### *Vegetation Composition*

Vegetation was measured in three 100 m<sup>2</sup> plots along each transect. A 10 x 10 m (0.01 ha) vegetation plot was established adjacent to each well location. Vegetation was separated into three strata, overstory trees, understory shrubs and saplings, and a ground cover component. The overstory included all woody vegetation measuring  $\geq 2.5$  cm dbh (diameter at breast height, or 1.4 m above ground). Understory vegetation consisted of saplings and shrubs  $\geq 30$  cm in height and  $< 2.5$  cm dbh. The ground component included all herbaceous species, vines, and woody vegetation  $< 30$  cm tall. Within the overstory, the diameter at breast height of every species was measured to the nearest mm and recorded. In the understory and ground components, all species within in the 100 m<sup>2</sup> plot were recorded and assigned a numeric cover class 1 to 10, as defined by the North Carolina Vegetation Survey, with 1 = trace to 10 = 95% cover (Peet et al. 1996). Numeric cover class represented a range, for example, Cover class 3 = 1 - 2%, for data analysis, cover class was transformed into percent cover and the midpoint was used. Vegetation was qualitatively described by overall species richness (total number of species), and quantitatively described by transect basal area, and relative values of density, dominance, and frequency (Doumlele et al. 1985). The relative variables were summated to derive an importance value for each species.

### ***Precipitation and Potential Evapotranspiration***

Thirty-minute rainfall data for 2011 and 2012 were collected via tipping bucket (Texas Electronics, TE525WS) and transmitted to a Campbell Scientific CR10X data logger at the Santee Experimental Forest Headquarters weather station. All data were verified or corrected with manual rain gauge readings at the time of each download. Potential evapotranspiration (PET) was calculated using the Thornthwaite equation and estimates of PET on a monthly basis:

$$PET = 16L_d \left( \frac{10T_a}{I} \right)^a$$

where  $PET$  = monthly potential evapotranspiration (mm),  $L_d$  = length of day factor,  $T_a$  = mean monthly ambient air temperature ( $^{\circ}\text{C}$ ),  $I$  = annual heat index,  $a$  = empirical coefficient (Brooks et al. 2003). Mean ambient air temperature was collected for North Charleston, South Carolina from the National Weather Service Forecast Office (NWS 2013).

### **Data Analyses**

All hydrologic (stream stage and water table) and soil moisture data collected with automatic data loggers were summarized in a continuous time series. Time series hydrographs were visually inspected for erroneous data points, including malfunctioning loggers and storm events that resulted in rapid well flooding. These data points were manually removed. The initial examination of time series data revealed trends reflecting seasonal variation and the hydrologic response to local climate events such as storms and extended periods of dry conditions. Water table data were described in two different ways, 1) depth to water, to describe site – specific physiological condition, and 2) water

table elevation (m NAVD88) to describe inter site hydrological conditions. Depth to water table (cm below ground surface) was used in conjunction with iron rod measurements, vegetation analysis, and organic matter decomposition rates. With surface water and shallow groundwater compared to a common datum, their relationship and connectivity could be evaluated. Statistical analyses on water table and stream stage elevation were performed using Minitab version 16.2.3 (Minitab, Inc. 2012) with a  $\alpha = 0.05$  level of significance. Simple linear regression was performed on stream stage data to compare relationships to seasonal sea level and rainfall. To compare overall groundwater trends, mean daily water table elevation was calculated, and individual sites along each transect were aggregated. This allowed for the comparison of water table elevation along the longitudinal (decreasing tidal) gradient and between tidal versus non-tidal regime. To detect differences between transects one-way analysis of variance (ANOVA) with Tukey's groupings and two sample t-tests were employed. One-way ANOVA was performed on the iron rod data and on organic matter decomposition rates to see if the mean soil oxidation depth and mean mass remaining after 240 days of decomposition were related to changes in hydrologic regime.

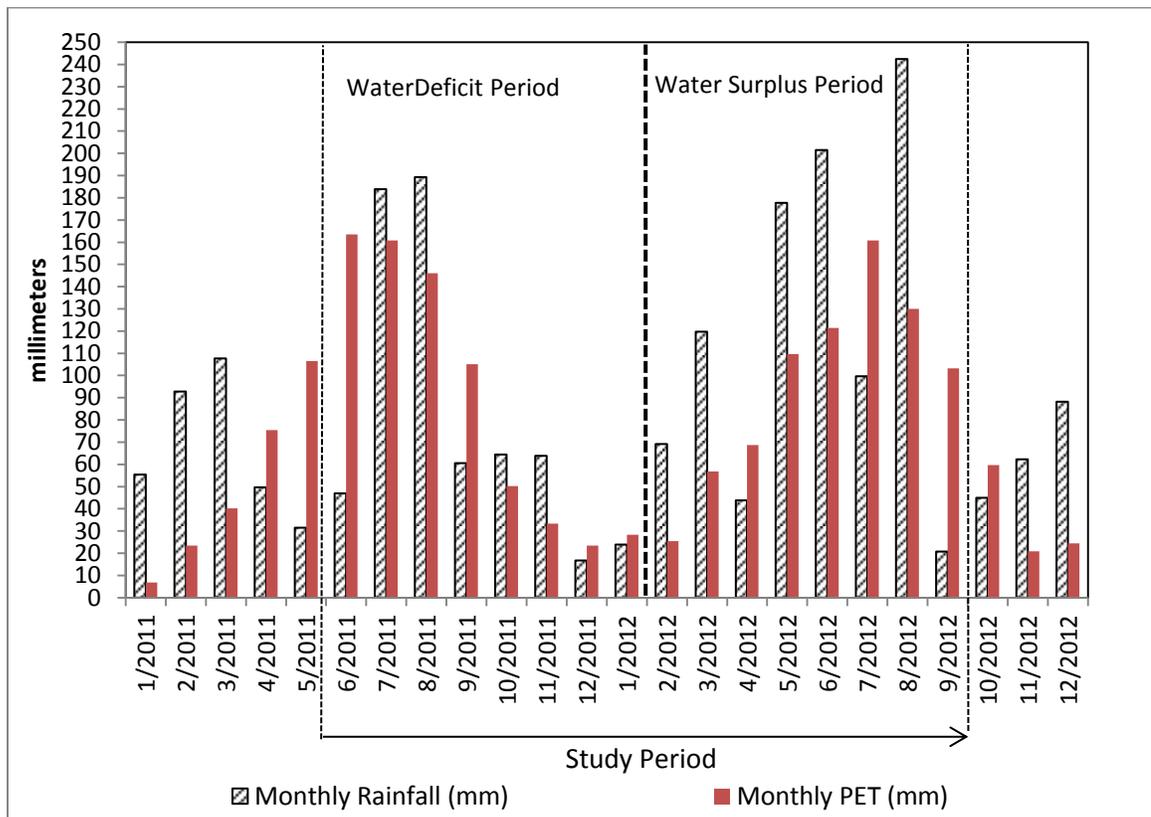
Vegetation was summarized by transect location to compare changes in vegetative community along the decreasing tidal gradient. One-way ANOVA was performed on mean species richness, overstory basal area, overstory density, and mean diameter at breast height for canopy species. Non-parametric Kruskal – Wallis rank sum tests (Minitab, Inc. 2012) were conducted to see if tree communities differed in flooding tolerance along the tidal gradient and to compare the wetland indicator status of species among sites.

## Results

### *Hydrologic Results*

#### *Precipitation and Potential Evapotranspiration*

The total rainfall during 2011 and 2012 was 963 millimeters and 1194 mm respectively, and the average monthly rainfall during 2011 and 2012 were 80 mm and 99 mm, respectively. Both the total annual rainfall and average monthly rainfall were below the 63-year, annual average of 1370 mm, and monthly average of 114 mm (Dai et al. 2011).



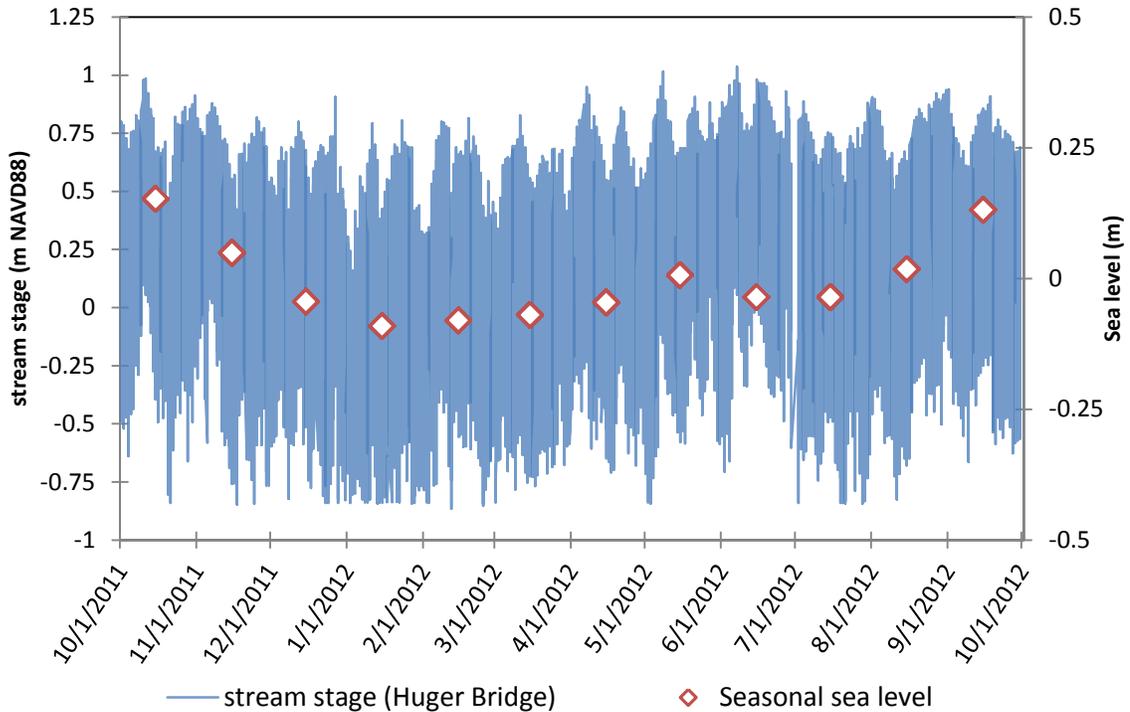
**Figure 7. Monthly rainfall totals collected at the Santee Experimental Forest Headquarters and potential evapotranspiration. Study period indicates period of water table monitoring, June 2011 through September 2012. Dotted line at February 2012 delineates study period into deficit period (June 2011 – January 2012) and surplus period (February 2012 – September 2012) based on the comparison of monthly rainfall to monthly potential evapotranspiration.**

Potential evaporation in 2011 and 2012 was 935 mm and 909 mm, respectively. Overall, rainfall exceeded PET during both years, but for purposes of analysis, the study period was broken up into a deficit period and a surplus period (Figure 7). This was based on monthly rainfall and PET totals, which appeared to influence both stream stage and water table position. During the deficit period, PET (711 mm) exceeded precipitation (650 mm). During the surplus period, precipitation (974 mm) exceeded PET (776 mm). The deficit period (June 2011 - January 2012) spanned both the growing season and dormant season, and included the summer, fall, and winter seasons. The surplus period, (February 2012 – September 2012) also spanned the dormant and growing season and included all seasons.

During 2011 and 2012, rainfall totals were highest during the growing season, a typical pattern observed in humid sub-tropical climates (Dai et al. 2011). During 2011, the wettest months were July and August, with each experiencing successive days of thunderstorms, resulting in monthly rainfall totals of 184 mm and 189 mm respectively. There were two large storm events during August, the first, occurred on 8/6/2011, where 59 mm of rainfall fell in the span of 3 hours; the second, occurred on 8/13/2011 with 39 mm of rainfall in one hour. During 2012, storm events were spread out throughout the growing season. Successive rain days between 6/10 – 6/13/12 totaled 151 mm, and caused overbank flow in the stream channel within Turkey Creek.

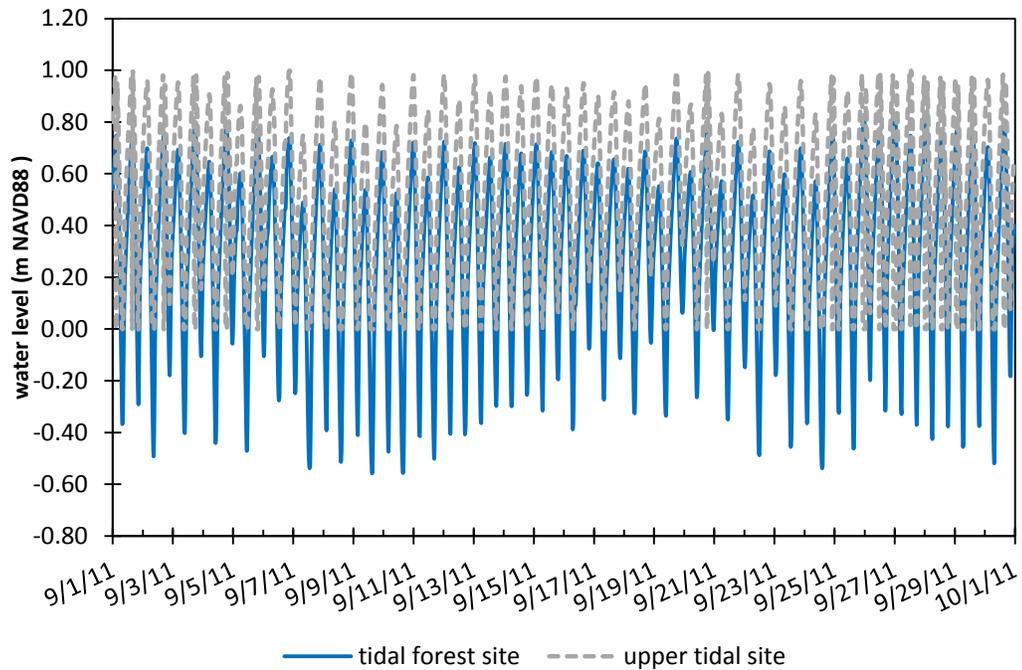
### *Huger Creek Stream Stage and Tide Range*

Huger Creek was dominated by semi-diurnal oceanic tides, with an observable current reversal at the bridge gauging station. During each 24-hour period, there were two high and two low tides of unequal amplitude. The mean stage at the bridge gauging site was 2.04 m (referenced to the streambed), (or 0.23 m) referenced to North American Vertical Datum of 1988. The mean tidal range was 1.28 m, with corresponding tidal amplitudes (m above or below mean tide level) of +0.82 m and -1.09 m. The tide cycle averaged 12.5 hours and lagged approximately 4.5 hours behind the Charleston tidal gauging station (NOAA gage no. 8665530) (NOAA 2013a). Tidal range in the creek was predominantly affected by seasonal sea level, and monthly lunar cycles. Sea level near Charleston, South Carolina was at the lowest during the winter months (November – January) and generally increased until October (NOAA 2013b). The Huger Creek hydrograph followed this general pattern (Figure 8). Additionally, throughout the monthly lunar cycle, Huger Creek experienced increased range and amplitude corresponding with the new and full moon and diminished tide range and amplitude, corresponding with the first and last quarter moons.



**Figure 8. Hourly tide levels for water year 2012 (1 October 2011 – 30 September 2012) at the Huger Creek bridge stream gauging site, and average seasonal sea level cycle for Charleston, South Carolina (Gage no. 8665530) (NOAA 2013b).**

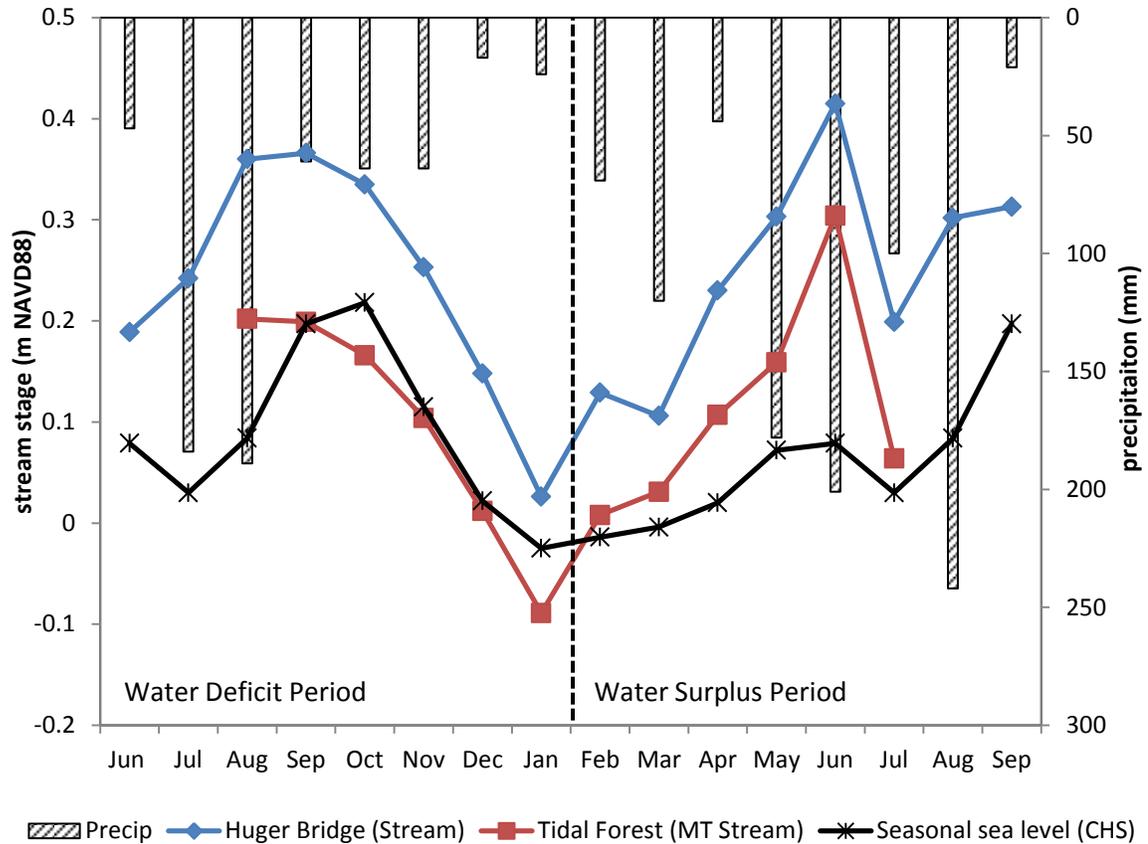
The mean stage at the upstream, tidal forest gauging site was 0.90 m, or 0.20 m NAVD88. The mean tide range was 1.15 m, with corresponding tidal amplitudes of + 0.26 m and - 0.86 m. The high tide peaked within the same 15-minute period as the bridge site, and current reversal occurred immediately upon the changing tide. For the majority of the study period, the primary source of water was from the flood tide. The USGS stream gauge at Turkey Creek (02172035) reported zero discharge from June 2011 to March 2012 (USGS 2013). Intermittent stream discharge was reported during the spring and summer seasons of 2012, but only in response to runoff from storm events. At low tide, the creek completely drained, revealing a coarse sandy substrate at the tidal forest and upper tidal stream gage sites.



**Figure 9. Hourly tide levels for September 2011 at the Huger Creek tidal forest stream gaging site (represented by blue line) and estimated tide levels at upper tidal stream site (gray dotted line).**

Using water level data from the tidal forest gauging site and creek bed elevation, the tidal range for the non-gaged upper tidal site was estimated (Figure 9). The estimated tide range at the upper tidal site was 0.80 m, or 0.82 m NAVD88. The upper limit of tidal excursions was approximately 600 meters upstream from the upper tidal site in response to increasing streambed elevation..

Stream stage in Huger Creek was predominantly controlled by seasonal sea level and monthly lunar or tides, but precipitation patterns played a role in regulating water level.



**Figure 10. Seasonal sea level at the Charleston outer coast, mean stream stage of Huger Creek at Bridge and Tidal forest stream gauging stations during water table monitoring period (June 2011 – September 2012). Dotted line splits study period into water deficit and water surplus periods.**

Mean monthly water levels in Huger Creek generally followed the pattern of seasonal sea level (Figure 10). Deviation occurred during months where total rainfall exceeded 150 mm and appeared to be most evident during July and August of 2011 and during the 2012 growing season. Regression analysis indicated that during the dry period changes in seasonal sea level explained 66% ( $R^2 = 0.663$ ) of the variation in stream stage, but during the wet season that number fell to 49% ( $R^2 = 0.4869$ ). During months where the precipitation was high, the tide range in Huger Creek appeared to be dampened.

### *Turkey Creek Stream Stage and Water Table*

Stream stage at Turkey Creek was dependent upon precipitation, and the creek was dry for 146 days (40%) of the yearlong monitoring period. When water was present during the deficit period, the stage averaged 0.40 m ( $\pm 0.09$ ) (standard deviation). When water was present during the surplus period, the average stage was 0.75 m ( $\pm 0.15$ ). Stream stage patterns were similar to the USGS site in response to rainfall. Stream stage was higher at the USGS (approximately 0.76 m) during the deficit period, and  $>1$  m during the surplus, presumably because the channel was narrower than the upstream Turkey Creek site. Surface water stage in Turkey Creek and water table levels in the riparian zone were closely tied to precipitation (Figure 11). The water table remained deep during most of the fall and winter seasons of the deficit period, and varied during throughout the growing season of the surplus period from the combined factors of increased rainfall and evapotranspiration. Significant differences were observed ( $p = 0.039$ ) between average water table position during the deficit period (1.83 m NAVD88/ 1.73 m bgs (below ground surface)) and the surplus period (2.23 m NAVD88/ 1.32 m bgs). Comparisons to long-term water table trends at the well on the adjacent side of the floodplain showed a similar pattern (Figure 12). In previous years, the water table position was higher relative to the ground surface, especially during the dormant period.

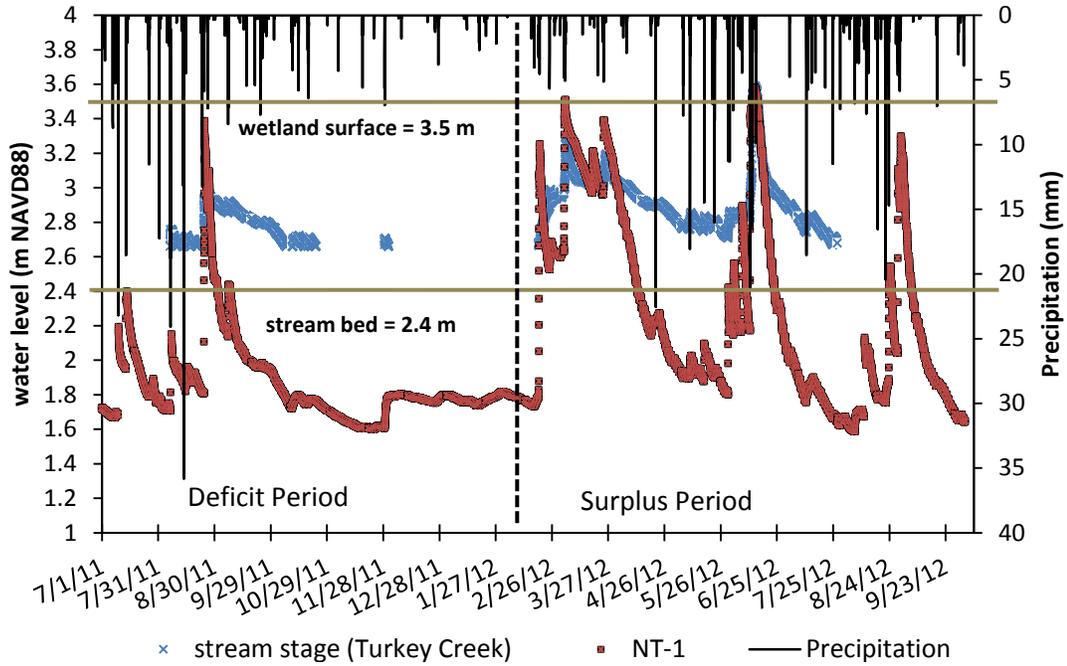


Figure 11. Hydrograph of Turkey Creek and well site NT1 with precipitation on the secondary y-axis. Both water table and stream stage show a large response from rainfall events and evaporative processes.

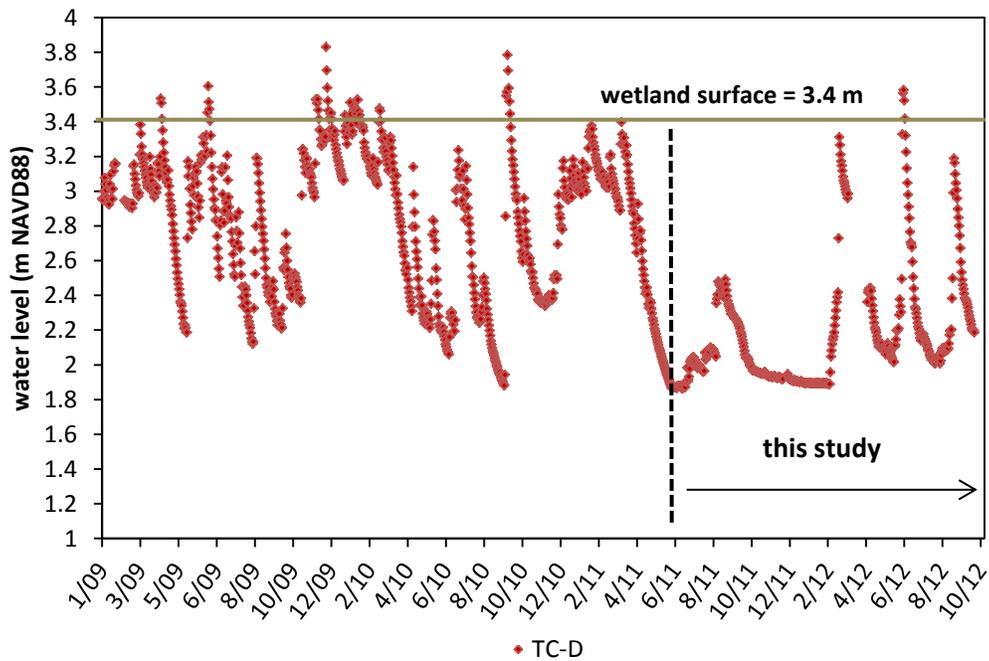
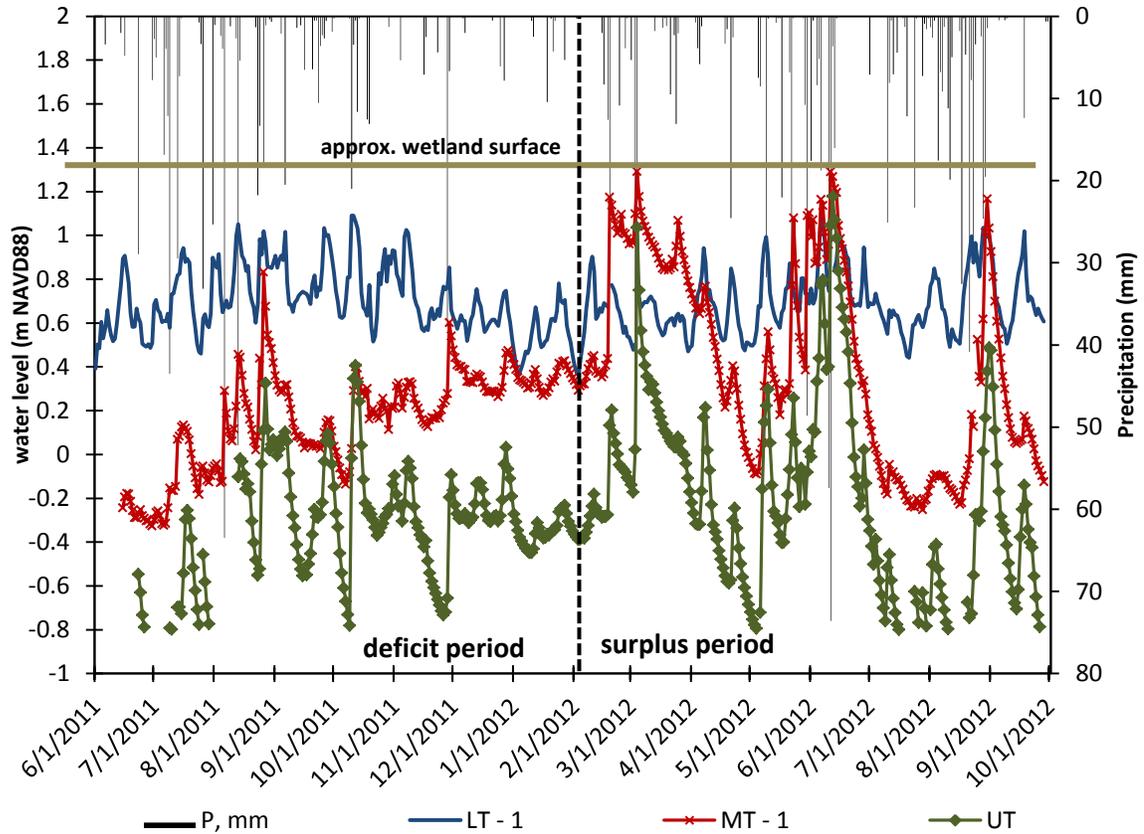


Figure 12. Long-term hydrograph of water table elevation at well site TC-D, dotted line shows Turkey Creek riparian zone at the Turkey Creek depression well.

Rainfall events  $\geq 10$  mm generated a rise in water table, and larger events caused brief near surface water table conditions (August 2011, March 2012, and June 2012). The largest response to rainfall (1.12 m) occurred on 13 June 2012 after 4 days of rain totaling 151 mm; this caused the entire floodplain to be inundated for several days.

#### *Overall Water Table Trends*

Water table position in the tidal reach was on average closer to the surface and showed a reduced response to rainfall events. Water table elevation generally increased in depth along the decreasing tidal gradient (Figure 13). The hydrograph of LT-1 shows that throughout the study period there was little variability ( $< 1$  m) in water table position. Water table elevations trended near the surface, and showed only minimal response to rainfall events. LT-1 appeared to decline in elevation during January and February 2012 corresponding with the low seasonal sea level (Figure 10). Hydrographs of MT-1 and UT-1 showed a deeper water table overall, and a greater response to rainfall. Figure 13 also shows that water table elevations were on average lower during the deficit period than the surplus period.



**Figure 13. Water table elevation at creek bank wells (LT-1, MT-1, and UT-1) over the entire study period with precipitation plotted on the secondary y-axis, horizontal line splits study period into water deficit and water surplus periods.**

Significant differences in water table elevation were observed between the deficit period and the surplus period along MT ( $p = 0.010$ ), but not at LT ( $p = 0.686$ ) or UT ( $p = 0.349$ ) (Table 3). Hydroperiod along LT was largely a function of the tidal creek. The position of the water table at UT-1 was on average below the elevation of the adjacent stream channel. Monthly mean water table elevation (m NAVD88) and m below ground surface (m bgs) with standard deviations (APPENDIX B) and long-term hydrographs and for each site (APPENDIX C) are found in the appendices.

**Table 3. Average water table elevation (m NAVD88)/ depth to water (m below ground surface) during deficit and surplus periods, the asterisk denote values that were significantly different.**

<b>Transect</b>	<b>Study Period</b>	<b>Deficit</b>	<b>Surplus</b>
Lower tidal	0.69 m/ 0.45 m bgs*	0.68 m/ 0.45 m bgs	0.70 m / 0.44 m bgs
Middle tidal	0.16 m/ 1.08 m bgs*	-0.06 m/ 1.30 m bgs*	0.38 m/ 0.87 m bgs*
Upper tidal	-0.26 m/ 1.54 m bgs*	-0.33 m/ 1.61 m bgs	-0.20 m/ 1.48 m bgs
Non-tidal	2.03 m/ 1.54 m bgs*	1.83 m/ 1.73 m bgs*	2.23 m/ 1.32 m bgs*

Water table response to rainfall events was varied in the tidal reach and contrasted to the non-tidal site. Generally, the response was dampened in the tidal reach because the tidal creek was influencing hydroperiod, and as the tide range decreased in the creek, the water table response to rainfall increased. Water table response to the June 2012 storm event was varied among sites. The LT hydrographs rose on average 12 cm, compared to 24 cm at MT, 81 cm at UT (Figure 13). In contrast to the non-tidal site, the floodplain surface was never inundated from overbank flooding.

*Surface Water – Water Table Interactions*

Water table hydrographs suggest that the tidally influenced riparian zone of Huger Creek followed an irregular tidal flooding pattern. In the immediate area of the monitoring transects, wetland surface was not flooded by tide water because the maximum creek stage never overtopped the natural levee; instead, the water table rose vertically in response to tidal forcing. Only one side of the tidal riparian zone was instrumented with water table wells, so information about the entire floodplain was

lacking. However, the ground elevation and levee height was slightly lower on the northern side of the floodplain, and regular flooding may occur during perigean spring tides (Figure 4). All of the monitoring sites at Huger Creek were adjacent to the tidal creek, but tidal forcing expressions in the water table were spatially and temporally variable. Tidal water table forcing was observed at LLT, LT-1, LT-2, LT-3, MT-1, and UT-1, but only LT-1 consistently showed influence throughout the study period.

Hydrographs at LT-1 showed semi-diurnal fluctuations that coincided with high and low tide cycling (Figure 14). The water table elevation peaked approximately 15 minutes after the maximum stream stage in the channel and experienced a 2.5-hour delay in leaving. Tide driven diurnal water table fluctuations had average daily amplitudes of +0.36/ -0.41 m and +0.19/ -0.13 m, relating to higher high water and lower low water respectively. Evidence of tidal influence in interior wetland hydrographs (LT-2 and LT-3) were associated with spring tide forcing, and mediated by other factors such as evapotranspiration, groundwater discharge, topographic position, and hydraulic damming from interflow and runoff originating in the adjacent upland. Interior wells along MT and UT lacked tidal pulsing completely.

There appeared to be a direct relationship between the mean stream stage in Huger Creek (m NAVD88), and the degree of water table tidal forcing. During periods of higher water (seasonally or precipitation) the connectivity (defined as water table tidal forcing) between Huger Creek and the riparian zone increased. Generally, when the mean surface water elevation in Huger Creek reached bank full  $\geq 0.7$  (m NAVD88) interior wells (LT-2 and LT-3) showed tidal forcing. Below this threshold, tidal forcing

beyond the creek bank was largely absent, but Huger Creek still functioned as a freshwater reservoir to those portions of the tidal reach.

Figures 14 and 15 present hydrographs along the LT transect during October 2011 (deficit period/ growing season/ high sea level) and February 2012 (surplus period/ dormant season/ low sea level). They demonstrate the spatial and temporal variability of tidal influence. In figure 14, a spring tide coincided with 24 mm of rainfall. Water table along the length of the transect rose to near saturated conditions. When water table elevation and surface water elevation neared 1 m NAVD88 (approximately 10 – 20 cm bgs), it produced tidal forcing into the wetland interior. When surface water elevations declined in response to the monthly lunar cycle, tidal forcing in the wetland interior diminished. During the days prior to the rainfall and spring tide, the overall gradient moved from the bank to the interior and from the upland to the interior. As the water table receded, the gradient shifted and drained towards the creek bank.

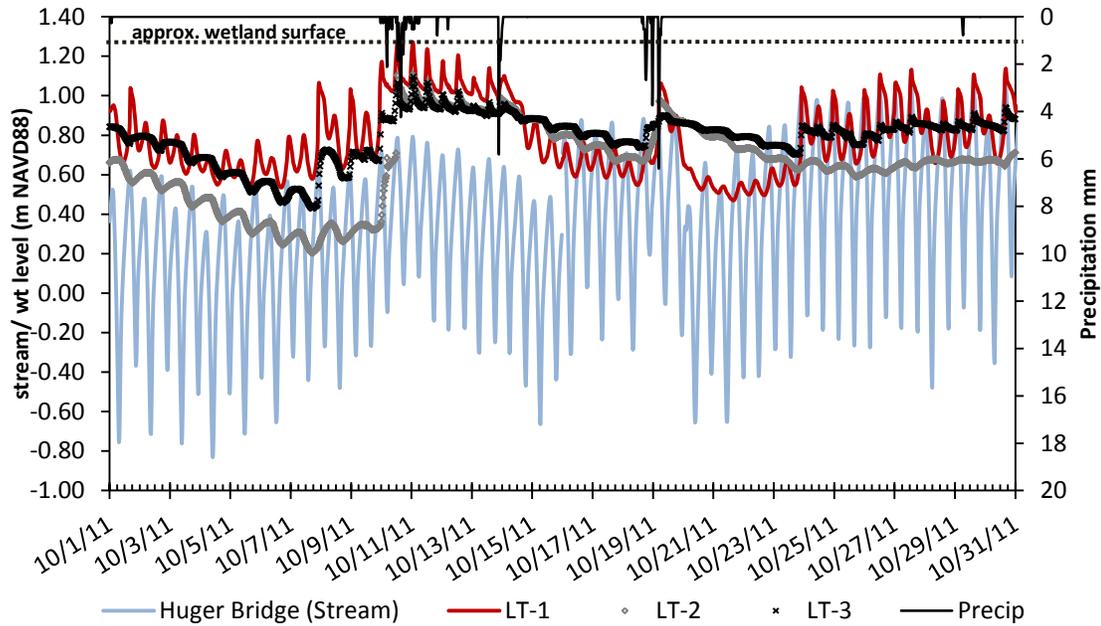


Figure 14. Hydrograph of water table elevation and stream stage in Huger Creek along the LT transect during October 2011 (deficit period, growing season, and high sea level) with rainfall plotted on secondary y-axis. Rainfall on 10/10/11 coincides with spring tide.

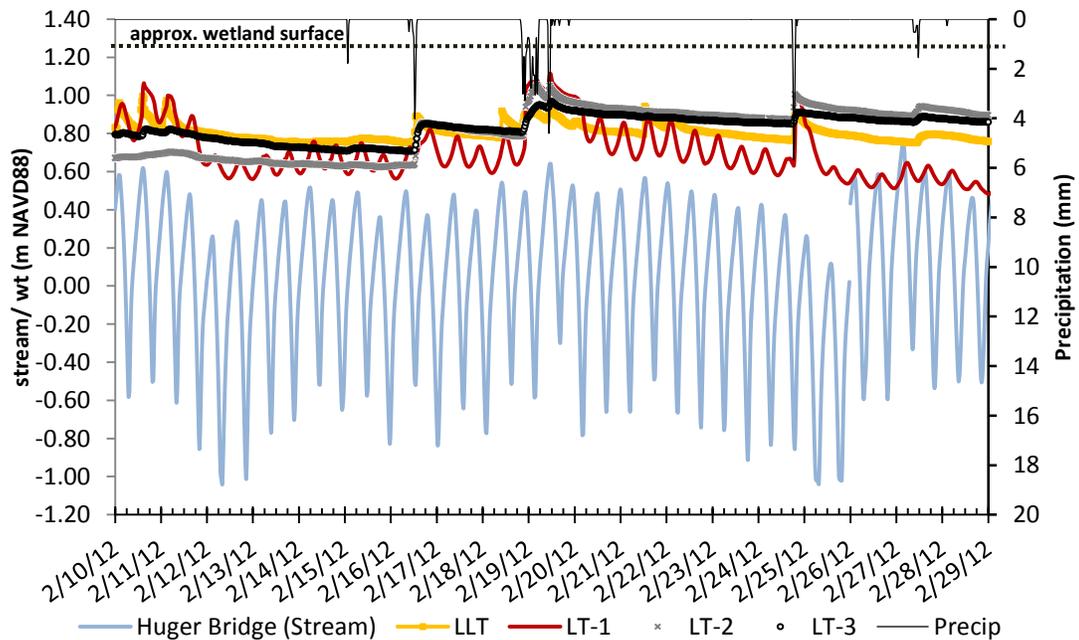


Figure 15. Hydrograph of water table elevation and stream stage in Huger Creek along at LLT and the LT transect during February 2012 (surplus period, dormant season, low sea level) with rainfall plotted on secondary y-axis. The dotted line represents the approximate wetland surface 1.24 m NAVD88 at the LT transect.

During February, tidal forcing was only apparent in LT-1, and with smaller amplitude (Figure 15). The water table response at LLT was unexpected (lack of daily tide signal) which may be due to the differences in soil texture. The upper 30 cm of the soil profile at LLT had a mucky consistence with a high density of root material with clay below. Observations indicate that this area was continually saturated or near saturated, which contrasted to the rest of the floodplain soils in this study. The water table responded to rain events, but the mean water level in Huger Creek was not high enough to produce tidal forcing in the water table, even during the spring tide on 2/21/2012.

The water table at MT-1 was deeper relative to ground surface, showed reduced tidal pulsing, and a larger response to precipitation and evapotranspiration (Figure 16A). This suggests that climate effects of ET and rainfall were the predominant controllers of water table position, but the presence of tide mediated climate extremes. Daily tide driven water table fluctuations averaged  $< 10$  cm, and were mixed with an evapotranspiration signal. Precipitation during February caused a steep rise in water table, presumably due to antecedent soil conditions. The evapotranspiration signal was absent in February, and tidal forcing was reduced (Figure 16B). Reduced tidal pulsing was due to the lower sea level (smaller tides) during the winter. AT UT-1, the water table was considerably deeper, by approximately 0.50 m, and the water table response to rain and evapotranspiration was similar to the non-tidal site. When the water table position was below the streambed, there was essentially no tidal communication between the water table and Huger Creek. The stream reach was channelized at this location, which may have limited the connection between the riparian zone and the creek. There were however, brief instances where the water table experienced tidal forcing (Figure

17A). This was apparent after large rain events when the water level in the wetland was within 1 m to the surface.

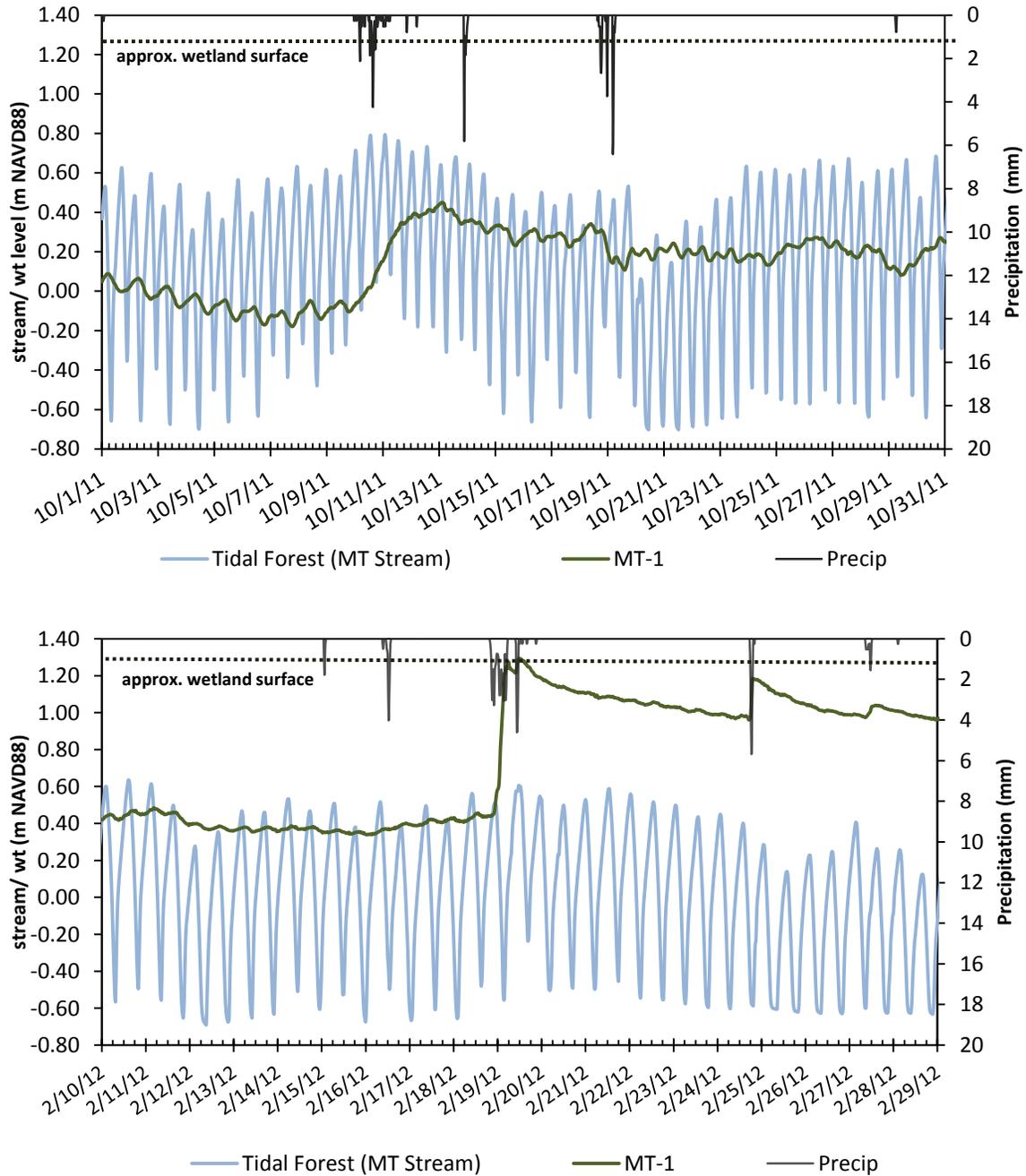
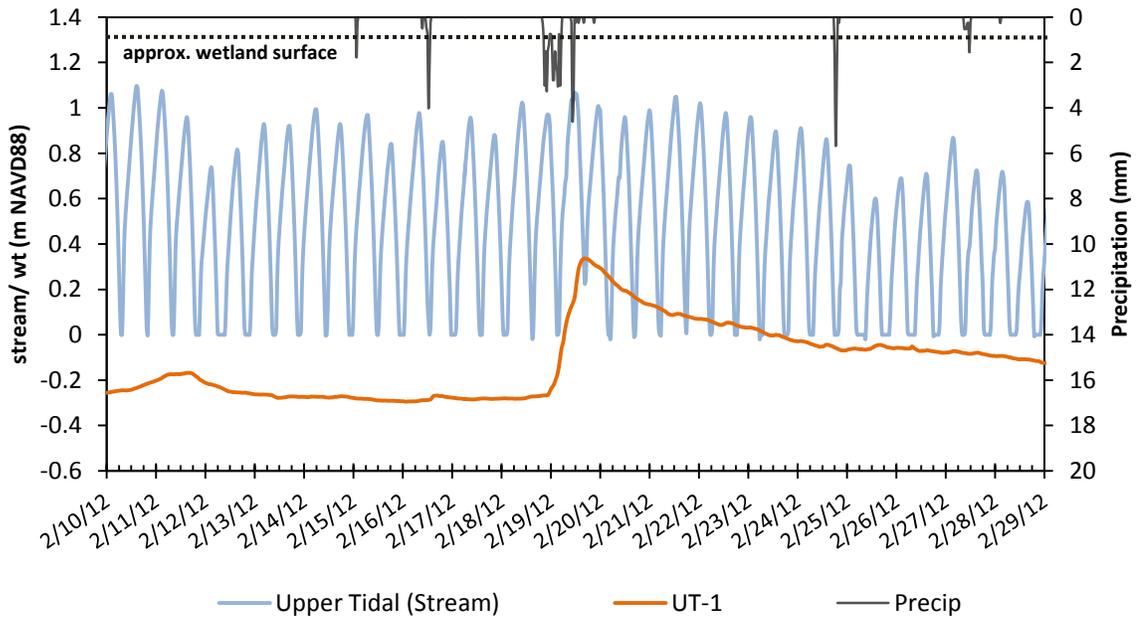
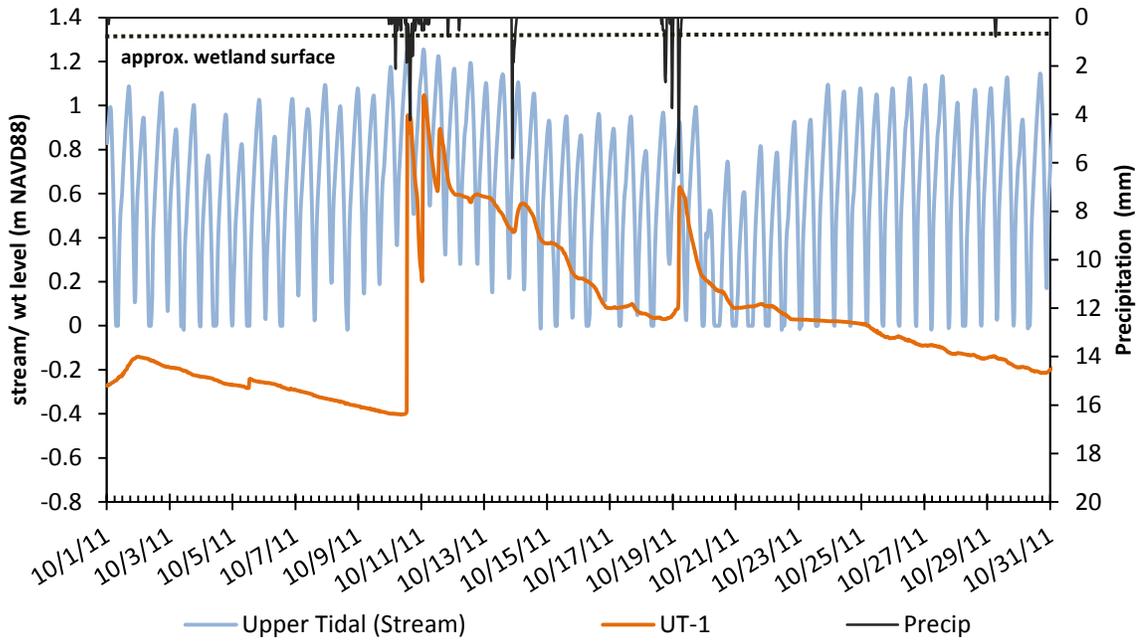


Figure 16 A and B. Hydrograph of water table elevation and stream stage in Huger Creek at tidal forest gauging site and well MT-1 during October 2011 (16A) and during February 2012 (16B)



**Figure 17 A and B. Hydrograph of water table elevation and stream stage in Huger Creek at upper tidal stream gauging site and well UT-1 during October 2011 (17A) and during February 2012 (17B)**

### Soil Moisture

It is suspected that that standard factory calibration was not suitable for the high clay content of the Meggett soils. Within individual plots, sensors reported a wide range of readings (from saturated conditions to dry) that did not agree with water table data at the same site. Therefore, data obtained from the soil moisture sensors could not be used to interpret soil moisture at depth.

### Biological Response Results

#### Soil Oxidation Depth

The seasonal mean depth to oxidation generally corresponded with the predicted wetness gradient (Figure 18). Rods were read four times during the study on 28 July 2011, 8 February 2012, 26 April 2012, and 3 October 2012.

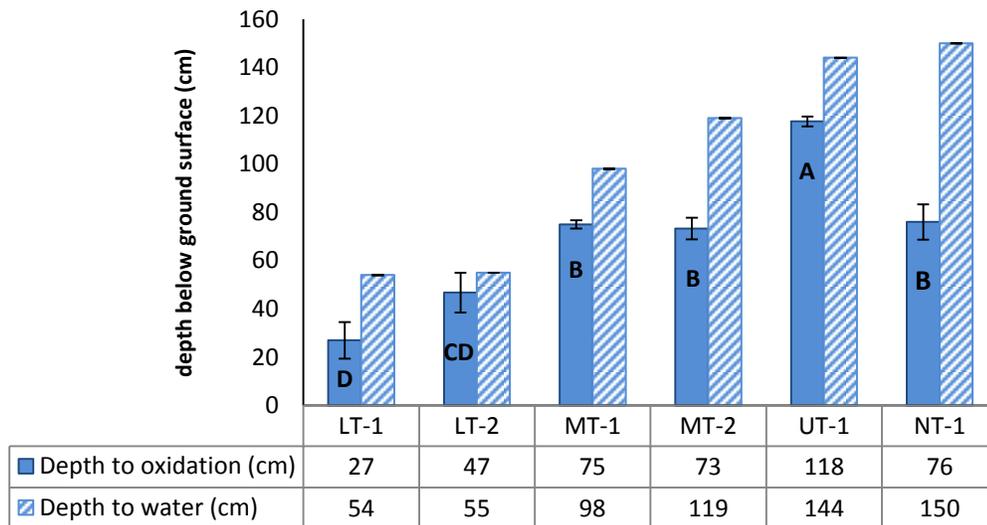


Figure 18. Average depth to oxidation from readings as read from iron rods and mean depth to water during period of observation (cm bgs) with standard error. Letters represent Tukey's groupings and bars that do not share a letter indicate that oxidation depths were significantly different.

The average depth to water between each reading was recorded and compared to iron rod readings. As expected, the mean depth to oxidation was similar within each transect and increased in depth as the tidal gradient decreased.

### *Organic Matter Decomposition*

After 240 days, the wooden sticks at Turkey Creek lost significantly ( $p = 0.000$ ) more mass (49%) than the sticks at Huger Creek (32%), but differences in water regime could not explain the variation. No significant differences were observed between the two sites LT-1 and NT-2 with highly contrasting water regimes (Figure 19). The best predictor of organic matter decomposition was time, and not water regime. A steady decline in mass occurred over time within the tidal and non-tidal reaches, which appeared to be independent of mean depth to water (Figure 20).

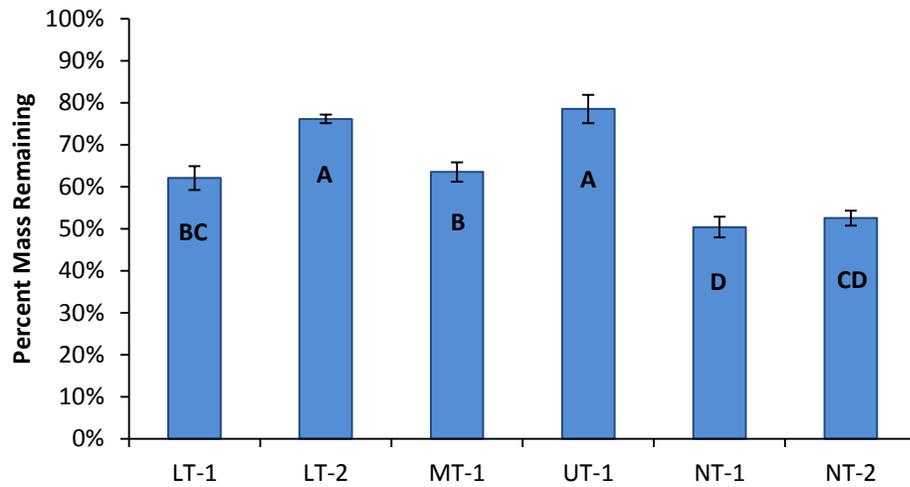


Figure 19. Mean percent mass remaining from organic matter decomposition after 240 days with standard error. Letters represent groupings using the Tukey's comparisons. Bars that do not share a letter are significantly different.

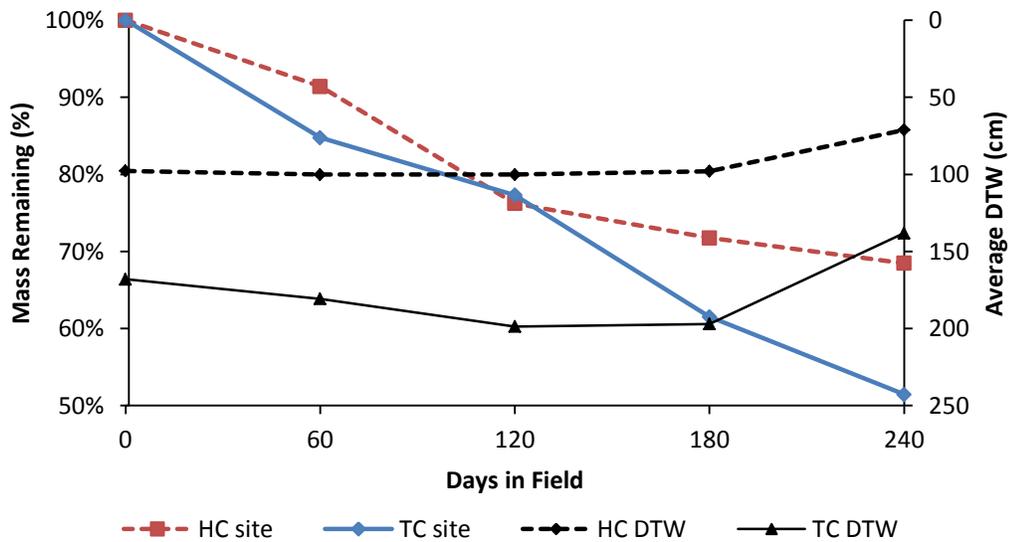
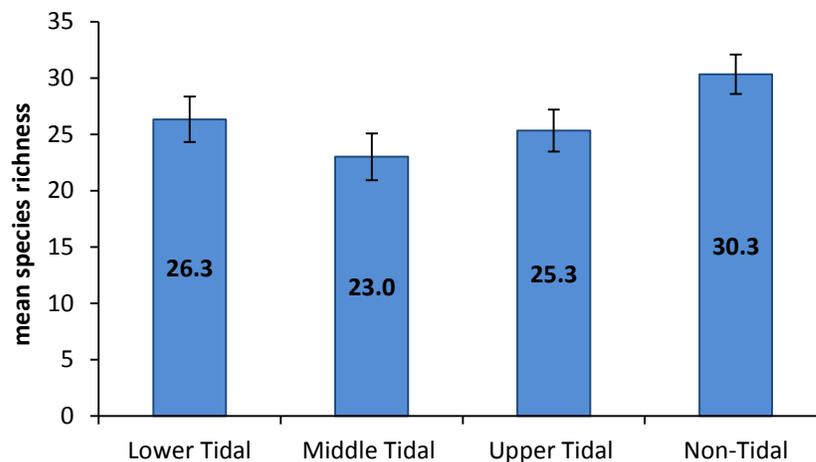


Figure 20. Mean percent mass remaining from organic matter decomposition over time at Huger Creek (tidal) and Turkey Creek (non-tidal) decomposition plots after 240 days, average depth to water at entire site plotted on secondary y-axis.

### *Vegetation Community Composition*

There were 63 species identified in the canopy, shrub, and ground strata across the four sites sampled at Huger and Turkey Creeks. Vegetation communities in the riparian zones of both the tidal and non-tidal reaches were similar in composition and richness. Several species were common to all plots regardless of topographic position and water regime. There were no significant differences found in average species richness between Huger and Turkey Creek ( $p = 0.093$ ) or along the decreasing tidal gradient ( $p = 0.134$ ) (Figure 21). Overstory species were summarized by density, basal area, mean dbh, importance value, and water logging tolerance (Table 4). The waterlogging tolerance ranges (Hook 1984; Theriot 1993) were used to determine if there were more trees in the “most” or “moderate” waterlogging tolerance category in the tidal reach compared to the non-tidal reach, or if a gradient of wetness tolerance was present along the tidal gradient.



**Figure 21. Mean species richness along montitoting transects LT, MT, UT, and UT; bars represent standard error.**

**Table 4. Composition and structure of overstory plots showing basal area, density, mean diameter at breast height, and mean depth to water. Importance values were calculated by using relative values of dominance, density, and frequency and sum to 300 for each site. Waterlogging tolerance values are also presented.**

	Waterlogging tolerance <sup>1</sup>	Huger Creek			Turkey Creek	
		Lower tidal	Middle tidal	Upper tidal	Non tidal	
Basal area (m <sup>2</sup> ha <sup>-1</sup> )		17.99	19.08	42.89	31.85	
Density (no. ha <sup>-1</sup> )		2600	1409	2565	2132	
Mean DBH (cm)		9.8	13.7	16.3	13.4	
Mean WT depth (m)		0.47	1.09	1.44	1.5	
Importance Value						
Species name	Common name					
<i>Carpinus caroliniana</i>	ironwood	weak	110.0	93.0	35.7	76.8
<i>Liquidambar styraciflua</i>	sweetgum	moderate	30.8	57.0	13.7	25.4
<i>Cornus foemina</i>	swamp dogwood	moderate	27.1			24.5
<i>Quercus laurifolia</i>	laurel oak	weak	26.4	11.6	73.9	7.4
<i>Ulmus americana</i>	American elm	moderate	25.4	19.6	8.9	7.343
<i>Quercus michauxii</i>	swamp chestnut oak	weak	16.3	16.3	30.9	11.9
<i>Fraxinus pennsylvanica</i>	green ash	moderate	15.6	26.2	26.6	16.2
<i>Nyssa aquatica</i>	water tupelo	most	13.8			7.6
<i>Fraxinus caroliniana</i>	Carolina ash	most	13.1			
<i>Acer rubrum</i>	red maple	moderate	8.0		16.4	
<i>Nyssa biflora</i>	swamp tupelo	most	7.1	11.5		
<i>Celtis laevigata</i>	sugarberry	weak	6.4			
<i>Carya sp.</i>	hickory	not listed			16.4	25.3
<i>Diospyros virginiana</i>	persimmon	moderate				27.5
<i>Ilex decidua</i>	possumhaw	moderate			5.6	15.6
<i>Ilex opaca</i>	American holly	weak		48.3	19.4	13.5
<i>Morus rubra</i>	red mulberry	weak				9.3
<i>Nyssa sylvatica</i>	blackgum	weak			10.5	
<i>Pinus sp.</i>	pine species	not listed			14.2	8.8
<i>Pinus taeda</i>	loblolly pine	moderate				7.8
<i>Quercus sp.</i>	oak species	not listed		16.3		
<i>Quercus pagoda</i>	cherrybark oak	weak			11.6	6.0
<i>Quercus nigra</i>	water oak	weak to mod			7.9	
<i>Ulmus alata</i>	winged elm	weak			8.2	8.9
Total			300.0	300.0	299.9	300.0

<sup>1</sup>Waterlogging tolerance from Hook (1984) except *C. foemina* from Theriot (1993) and *Q. nigra* from Denslow and Battaglia (2002)

Six canopy species occurred in all plots. Ironwood, (*Carpinus caroliniana*), sweetgum (*Liquidambar styraciflua*), laurel oak (*Quercus laurifolia*), green ash (*Fraxinus pennsylvanica*), swamp chestnut oak (*Quercus michauxii*), and American elm (*Ulmus*

*americana*) comprised over 60% of total canopy importance value. The overstory stratum at both the lower tidal and middle tidal Huger Creek was dominated by ironwood and sweetgum. Swamp dogwood (*Cornus foemina*) and American elm were co-dominants at the lower tidal site. American holly (*Ilex opaca*) and green ash were co-dominants at the middle tidal site. American holly was not observed at the lower tidal site. The canopy of the upper tidal site was dominated by laurel oak, ironwood, and swamp chestnut oak. The non-tidal site was dominated by ironwood, with persimmon (*Diospyros virginiana*), swamp dogwood, and sweetgum as co-dominants.

The tidal sites had average density of 2191 stems ha<sup>-1</sup>, with stand basal area between 18 – 43 m<sup>2</sup> ha<sup>-1</sup>. The non-tidal site had a stem density of 2132 stems ha<sup>-1</sup> and stand basal area of 32 m<sup>2</sup> ha<sup>-1</sup>. Trees were numerous, but had small diameters, which provides rationale for the high density to low basal area ratio. There were no significant differences observed between mean tree diameters (p = 0.488) or mean basal area (p = 0.493) among sites.

**Table 5. Overstory species water logging tolerance values by transect. The percent most or mod is was derived by dividing the total listed species by the total number of most or moderately tolerant trees at each transect.**

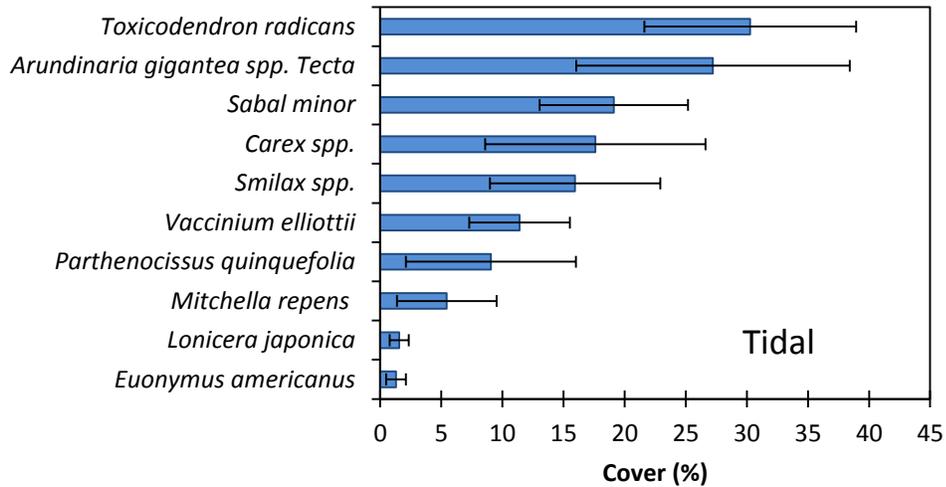
	Lower tidal	Middle tidal	Upper tidal	Non tidal
MOST	3	1	0	1
MODERATE	5	3	5	7
MODERATE/ WEAK	0	0	1	0
WEAK	4	4	7	7
NOT LISTED/ UNKN	0	1	2	2
Total Listed Species	12	8	13	15
<b>% MOST or MOD</b>	<b>67%</b>	<b>50%</b>	<b>38%</b>	<b>53%</b>

There was no significant difference in the median number of species considered “most” or “moderate” in waterlogging tolerance between Huger and Turkey Creek, ( $H = 0.101$  df = 1;  $p = 0.748$ ) or along the tidal gradient ( $H = 3.00$ ; df = 3;  $p = 0.392$ ).

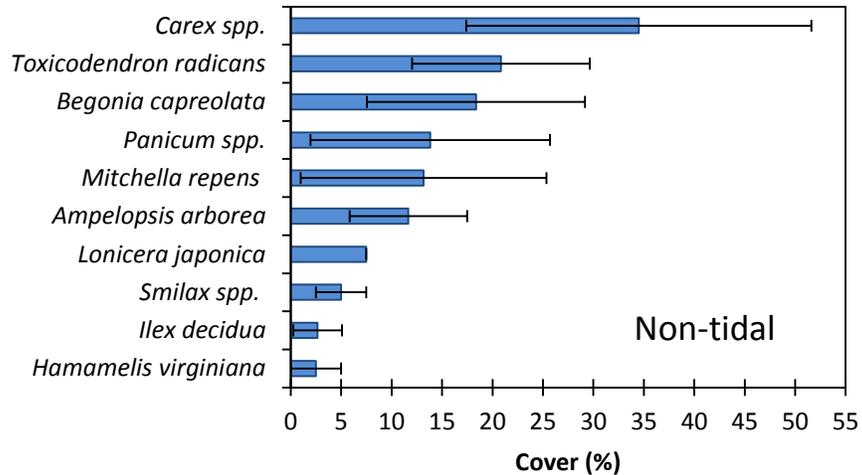
Understory species classified as shrubs (<2.5 dbh,  $\geq 30$  cm tall) included both shrub species and saplings identified in the overstory. Ten species contributed to the highest average percent cover (combined shrub and ground strata) at the Huger and Turkey Creeks (Figures 22 and 23). At all sites, this stratum was sparse and achieved an overall low percent cover. Therefore, cover values were combined with the ground strata for analyses. The dominant shrub species within the tidal reach were switch cane (*Arundinaria gigantea* spp. *Tecta*), dwarf palmetto (*Sabal minor*), Elliott’s blueberry (*Vaccinium elliotii*), and greenbrier (*Smilax* spp.). The shrub strata at the non-tidal transect was largely absent, and comprised mostly of the sapling age class of overstory species present in the canopy. Switch cane was not observed at the non-tidal transect. The composition of the herbaceous stratum had the greatest diversity overall, but was sparse at the upper and middle tidal sites. Ground cover was similar in the tidal and non-tidal reaches; poison ivy (*Toxicodendron radicans*) and sedges (*Carex* spp.) had the highest ground cover overall. Panic grass (*Panicum* spp.) was common in the non-tidal reach.

Individual species’ wetland indicator status was used to determine the proportion of hydrophytes present along each transect and between the tidal and non-tidal sites. The wetland indicator status was developed for use in wetland delineation to indicate an individual species’ preferred habitat, and the frequency of it occurring in a wetland or an upland habitat (USDA 2012). The definitions are OBL (obligate wetland – almost

always in wetlands), FACW (facultative wetland – usually in wetlands), FAC (facultative – commonly in wetlands, but also uplands), FACU (facultative upland – occasionally in wetlands, but usually upland) and UPL (obligate upland – almost always in uplands) (USDA 2012).



**Figure 22.** Top ten shrub and ground component species contributing the highest average percent cover at the nine Huger Creek (tidal) vegetation plots, bars represent standard error.



**Figure 23.** Top ten shrub and ground component species contributing the highest average percent cover at the three Turkey Creek (non-tidal) vegetation plots, bars represent standard error.

At the tidal sites, 30% of the species that comprised the highest percent cover were facultative wetland species, compared to only 10% at the non-tidal site. A small patch of soft stem bulrush (*Schoenoplectus tabernaemontani*) a species commonly found in freshwater marsh habitat, was identified at the lower tidal site, but had overall low cover. The complete list of species identified in the tidal and non-tidal riparian zones can be found in APPENDIX D. Within all strata, the percentage of obligate and facultative wetland species decreases appears to decrease with the decreasing tidal gradient, and then increases again at the non-tidal site (Table 6). The results of the Kruskal – Wallis Test indicated that there was no significant difference in the median number of OBL or FACW listed species between tidal and non-tidal regime, ( $H = 2.34$ ;  $df = 1$ ;  $p = 0.126$ ) or along the tidal gradient ( $H = 6.57$ ;  $df = 3$ ;  $p = 0.087$ ).

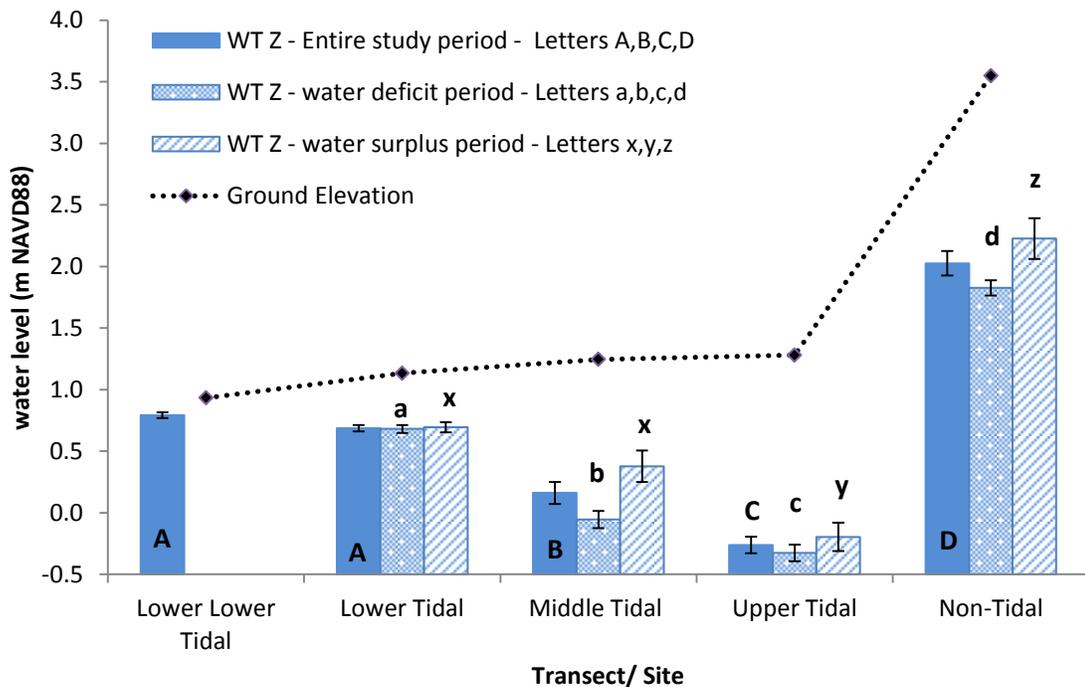
**Table 6. Wetland indicator status values for all species (overstory, shrub, and ground) identified at each transect. The percent OBL or FACW was derived by dividing the total listed species by the total number of OBL or FACW listed plants at each transect.**

	Lower tidal	Middle tidal	Upper tidal	Non tidal
OBL	6	1	0	1
FACW	14	10	11	13
FAC	13	12	16	17
FACU	2	2	4	6
NOT LISTED/ UNKN	2	9	7	14
Total Listed Species	35	25	31	37
<b>% OBL or FACW</b>	<b>57%</b>	<b>44%</b>	<b>35%</b>	<b>38%</b>

## Discussion

### *Hydrology*

The mean water table position was consistently higher relative to the ground surface in the lower tidal portion of the Huger Creek watershed, and was consistently deeper at locations with less tidal influence from Huger Creek and increasing distance upstream (Figure 24). The data suggest that the sites lower lower tidal (LLT) and lower tidal (LT) represented the tidally dominated portion of the riparian zone, middle tidal (MT) was mixed (tidal/ fluvial) and was the convergence tidal/non-tidal zone, and upper tidal (UT) was weakly tidal.



**Figure 24.** Mean water table elevation at each site during entire study period, deficit period, (June 2011 – January 2012) and surplus period (February 2012 – September 2012) with standard error. Dotted line represents ground elevation at each monitoring site; bars that do not share a letter grouping are significantly different

This was perhaps due to the presence of the artificially deepened and straightened drainage running east of Huger Creek between UT and MT. The water table at UT-1 only showed tidal evidence during spring tide cycles, and when hydraulic head in the wetland was at a higher elevation than the streambed. Water table position and response to rainfall and evapotranspiration at UT and NT (the Turkey Creek transect) were similar, suggesting that the study design accurately captured the tidal/non-tidal transition zone. The hydroperiod in the tidally influenced riparian zone sharply contrasts with the hydrologic patterns observed at the non-tidal reference site. Because this study was conducted during a period of lower than average rainfall, the differences between the two systems were very pronounced. The data also suggest that portions of the tidally influenced riparian zone function as a groundwater discharge area, and portions of the non-tidal site function as a groundwater recharge area.

Water levels in the tidally influenced riparian zone were closely related to both the mean surface water stage and tide range in Huger Creek. Both the tide range and mean stage were greater in the lower reach of Huger Creek when compared to the middle and upper reaches. During the study period, the channel adjacent to LLT and the LT transect never drained completely, suggesting that Huger Creek functioned as a reservoir supplying a constant source of water to the wetland through the daily tide cycle. Wetland hydroperiod was tied to the daily, monthly, and seasonal periodicity of Huger Creek rather than to climate factors of rainfall and evapotranspiration. Although data were only collected during the surplus period for LLT, due to the topographic position of the site, and the similar pattern observed in hydrographs, it is assumed that the relationship would be similar to the lower tidal hydrographs. These results agree with findings by

Rheinhardt and Hershner (1992) who suggested that hydroperiod in tidal freshwater swamps is dynamic on the short-term (daily high and low tide cycling) in response to water table forcing, but relatively low on the long-term (monthly/seasonal/yearly) due to the constant source of water. On the long-term, LT-1 followed a seasonal pattern corresponding with seasonal sea level. An overall deeper water table and reduced tidal forcing was observed during the months of January and February 2012 when sea levels were seasonally low, and contrasted to observations in October 2011 when sea level was highest. These results agree with Anderson and Lockaby (2011b) who observed a similar pattern in their study finding a close relationship to sea level and water table hydrographs; they suggested that tidal connectivity was related to changes in seasonal sea level.

Tidal pulsing in water table hydrographs was most apparent in the wells located directly adjacent to the creek, and tidal forcing of the water table was strongest in the lowest portions of the study area. The tidal forcing of the groundwater was similar to descriptions in other studies (Rheinhardt and Hershner 1992; Kroes et al. 2007; Anderson and Lockaby 2011b). Daily water table fluctuations were present in the creek bank well (LT-1), corresponded with high and low tide cycling in Huger Creek, and displayed a bimodal monthly pattern according to lunar phase. The constant supply of water did not allow the water table to decline below the mean low stage in Huger Creek, which led to high water table conditions throughout the study period. This combined with the reduced response to precipitation and evapotranspiration, explained the low long-term variability of water table position at LLT and LT, and provided evidence that Huger Creek was the primary determinant of hydroperiod.

Along the tidal forest and upper tidal reach of Huger Creek, (adjacent to the MT and UT transects) the mean surface water stage was lower and tidal range was reduced. Beginning at the MT transect, the water table was deeper below ground surface compared to LT and LLT sites and showed considerable variation between the deficit and surplus periods (approximately 0.5 m) despite a small increase (< 5 cm) in ground elevation. Here, hydroperiod was influenced by a combination of factors including mean surface water stage, tide range, seasonal sea level, precipitation, and evapotranspiration. For a majority of the study period, the channel drained completely on the outgoing tide because flow from the upland was absent. Without a constant supply of water from the stream or rainfall, water table levels declined to depths deeper than downstream, but still showed evidence of daily tidal forcing and bi-modal monthly lunar cycling. Compared to LT, the response to precipitation was larger and daily tide forcing was mixed with evapotranspiration and groundwater recharge. In the upper tidal stream reach, the tide range averaged 0.80 meters, but did not produce daily water table forcing. It is suspected that the altered stream channel and artificial levee somewhat isolated the floodplain, and reduced the connectivity to the stream channel. The upper tidal reach the stream appeared to be in a losing condition for a majority of the study period because hydrographs indicated that the water table adjacent to the channel was lower in elevation than the stream channel bed. Water table position was highly dependent on precipitation; rainfall caused a steep rise in water table and drained very quickly. Daily water table forcing was absent, except when the water table was within 1 meter to the surface. The hydrograph appeared to somewhat follow the bi-modal lunar cycle (see Figure 13).

A shift from tidal dominated to fluvial dominated dynamics was observed between the lower and middle tidal transects at the Huger Creek site. These findings contrast with a previous study that observed this shift approximately 11 km below the tidal/non-tidal convergence zone, (Anderson and Lockaby 2011b) and highlights that tidal conditions may prevail very close to transition zone. There is paucity of literature describing the hydrology within the tidal/non-tidal forested transition zone, and it is common that the hydrologic regime of an entire tidal system is inferred from a single water table well or by monitoring an in-channel tide gauge (Rheinhardt and Hershner 1992; Rheinhardt 1992; Kroes et al. 2007; Anderson and Lockaby 2011 a, b; Courtwright and Findlay 2011). The use of only one water table well, may not represent the hydrology of the entire riparian zone due to the high degree of heterogeneity inherent to tidal freshwater forested wetlands. Because transects were located close to one another and included both known tidal and non-tidal sites, this study was able to examine surface water – ground water interaction at a fine resolution.

When comparing the tidal and non-tidal systems with respect to how surface water and water table levels responded to climate, the response was markedly different. Water table position, response to precipitation and evapotranspiration observed at the non-tidal site was comparable to other studies conducted within the Turkey Creek watershed (Harder 2004; Garrett 2010). Harder (2004) and Garrett (2010) found that water table levels receded to depths over 1 meter during the growing season, due to evapotranspiration demands. Garrett (2010) stated that that rainfall events  $\geq 10$  mm produced a rise in water table. In this study, similar observations were recorded at both NT-1 and TC-D. These findings contrast to the hydroperiod along the LT transect where

long-term water table averaged 0.5 m bgs and rainfall response was reduced because of the tidally mediated hydroperiod. This agrees with findings by Rheinhardt and Hershner (1992) who stated that rainfall did not produce an observable effect in water table position due to the presence of tide.

The water table response to rainfall and evapotranspiration increased along the decreasing tidal gradient, but still contrasted with the non-tidal system. A large storm event (10-13 June 2012, 151 mm precipitation) did not inundate the wetland surface in the tidal reach, but flooded the non-tidal site for 3 days. At the Turkey Creek site, small amounts of rainfall during the growing season caused a large rise in water table followed by a sharp decline from evapotranspiration, reflecting the small specific yield of the Meggett soils. This observation may be a function of topographic position or due to the strong tidal gradient, but it also may be due to differences in soil texture between the two sites. Soil texture was not directly measured, but based on the field descriptions it appeared that soils had a higher sand content in the tidal reach compared to the high clay content in the non-tidal reach. Soils in the tidal reach were stratified in layers of sand and clay, and contrasted to the predominantly clayey soils in the non-tidal reach (Appendix A). These small differences may have affected infiltration rate, the soils in the tidal drained initially faster, and excess water was likely removed from the system due to the hydraulic gradient present during the outgoing tide.

Descriptions of soils in tidal freshwater forests typically include a mucky consistence with high organic matter content in the rooting zone and clay increasing at depth. The high organic matter is due to the prolonged saturated conditions caused by regular tidal flooding patterns. The floodplain surface along most monitoring transects

within the tidal reach was firm and lacked prominent microtopography (personal observation). A firm wetland surface is typical in systems experiencing an irregular tidal flooding pattern, and has been observed in other studies with similar flooding regimes (Rheinhardt 2007). In addition, the creek bank levees adjacent to transects were high enough to prevent either daily high tide from flooding the wetland surface. It is suspected that the presence of sloughs connected portions of the wetland that were far distances from the main channel. Sloughs can act as conduits for water to travel, and increase riparian connectivity (Kroes et al. 2007). Where sloughs were common, (near LT-3 and LLT) the soil consistence was mucky in the rooting zone, and hummock and hollow topography was visible. Although the tidal reach did not undergo a regular flooding regime, Rheinhardt (2007) suggested that the mean depth to water table in the rooting zone is a better predictor of wetness than the duration or depth of flooding.

#### *Organic Matter Decomposition*

The results of the organic matter decomposition experiment demonstrate that at this scale, the differences in physiochemical properties (soil moisture regime or oxygen content) between the tidal and non-tidal sites were not great enough to facilitate faster decomposition in the tidal reach. Previous studies in tidal freshwater forests found higher decomposition rates at sites that alternated through periods wet and dry when compared to those that were continually saturated (Ozalp et al. 2007; Courtwright and Findlay 2011). Similar results were observed in a decomposition study in non-tidal Southeastern bottomland hardwood forests, adding that decomposition rates were also slowed by dry conditions (Baker et al. 2001). Collectively, these findings suggested that difference in

soil edaphic condition is the greatest factor influencing the rate or extent of decomposition.

The floodplains of Huger Creek and Turkey Creek did not experience any surface flooding during the 240-day decomposition period, which contrasts the regular tidal flooding regime in the aforementioned tidal studies. The sticks were buried to a depth of 15 cm to compensate for the differing hydrologic regime, but the water table did not reach within the 15 cm zone often enough to replicate a daily wetting and drying cycle. The iron rod data agree with these findings showing that soils were oxidized at minimum depth of 27 cm in the lower tidal reach and as deep as 118 cm in the upper tidal reach (Figure 19). However, the water table rose to near the surface at LT-1 approximately two times a month, corresponding with spring tides, and on three occasions, the water table at Turkey Creek rose to just below the surface in response to storm events. At Turkey Creek, near surface (< 20 cm) saturation lasted from 1 to 3 days exposing the sticks anaerobic conditions. These brief flooding patterns (high water table conditions) may have stimulated the decomposition process, and provide some explanation as to why no significant differences were observed between the lower tidal (wettest) Huger Creek site and Turkey Creek. Brief flooding was shown to stimulate decomposition in a similar bottomland hardwood study (Baker et al. 2001). The differences in decomposition rates were likely due to other site-specific factors such as differences in soil properties and decomposer communities.

### *Vegetation*

The vegetation occupying the riparian zone of Huger and Turkey Creeks were similar with respect to composition and structure. Our results indicate that the vegetation was insensitive to the tidal and non-tidal designations of the floodplain because significant differences were not detected in the flooding tolerance of overstory species, and there was not a significantly higher percentage of OBL or FACW indicator species in the tidal reach compared to the non-tidal reach. Forest composition was similar to the tidal and non-tidal reference sites in a bottomland hardwood resilience study conducted on the Santee Experimental Forest (Czwartacki and Trettin 2013).

Previous studies in tidal freshwater forests (See Table 1) have identified a wide range of vegetative communities, which correspond to floodplain hydrologic zones, ranging between Zone II to Zone IV (Wharton et al. 1982; Theriot 1993). Vegetation communities in Zone II and III floodplains are adapted to near or fully saturated conditions in the rooting zone from regular tidal flooding patterns. Since the Huger Creek riparian zone did not undergo a regular flooding pattern, the overall regime was drier. Species assemblages were associated with the Zone III and IV bottomland hardwood communities from Wharton et al. (1982), wet flat hardwoods described by Harms et al. (1998), and other tidal freshwater forests where the water regime was described as irregular tidal, upper tidal, or fluvial (Rheinhardt and Hershner 1992; Light et al. 2007; Kroes et al. 2007; Duberstein and Conner 2009).

The range of stem densities within the tidal riparian zone was similar to other tidal forest studies (Doumlele et al. 1985; Rheinhardt and Hershner 1992; Kroes et al. 2007; Anderson and Lockaby 2011a), but the basal area was lower. The range of stem densities

and basal area within the non-tidal riparian zone were similar to findings from Denslow and Battaglia's (2002) bottomland hardwood study. The below average basal area in the tidal reach was most likely due to the presence of many small diameter trees as the forest is still undergoing recovery from Hurricane Hugo (Song et al. 2012). Rheinhardt and Hershner (1992) observed two distinct forest communities in swamps along the Pamunkey River, Virginia, finding that small differences in the depth to water table controlled canopy composition. Our observations did not agree with their findings as ironwood and sweetgum were the most important canopy species along all transects even where depth to water differed greatly.

The canopy composition of tidal freshwater forested wetlands occurring in South Carolina are commonly described as being dominated by baldcypress, water tupelo, swamp tupelo, red maple, and Carolina ash (Conner et al. 2007). In this study, all of the dominant species, with the exception of baldcypress, were identified within the tidal reach, although, none were dominant. Baldcypress were present in both the tidal and non-tidal reaches but did not appear in the sample plots. The relative elevation of the forest floor to mean high water relates to the absence of baldcypress. The mean high water in Huger Creek was 0.80 m NAVD88, and the elevation of the forest floor was between 1.19 and 1.3 m NAVD88. In floodplains where the forest floor is higher than the mean high water, bottomland hardwood communities are more common than cypress-tupelo stands (Day et al. 2007). In the non-tidal reach, the presence of the baldcypress along the Turkey Creek drainage suggests that the relative distance between mean high water and the forest floor was shorter. Vegetation data were not collected at the LLT site, but it is suspected there would be more baldcypress due to the lower topographic

position. A secondary reason for the absence of baldcypress is due to a massive landscape clearing during the 1700s for rice agriculture , and timbering operations during the early 1900s (Czwartacki and Trettin 2013). Stands of baldcypress were observed in along tidal reach near Limerick Plantation, as these trees were not removed because they were located on rice dikes.

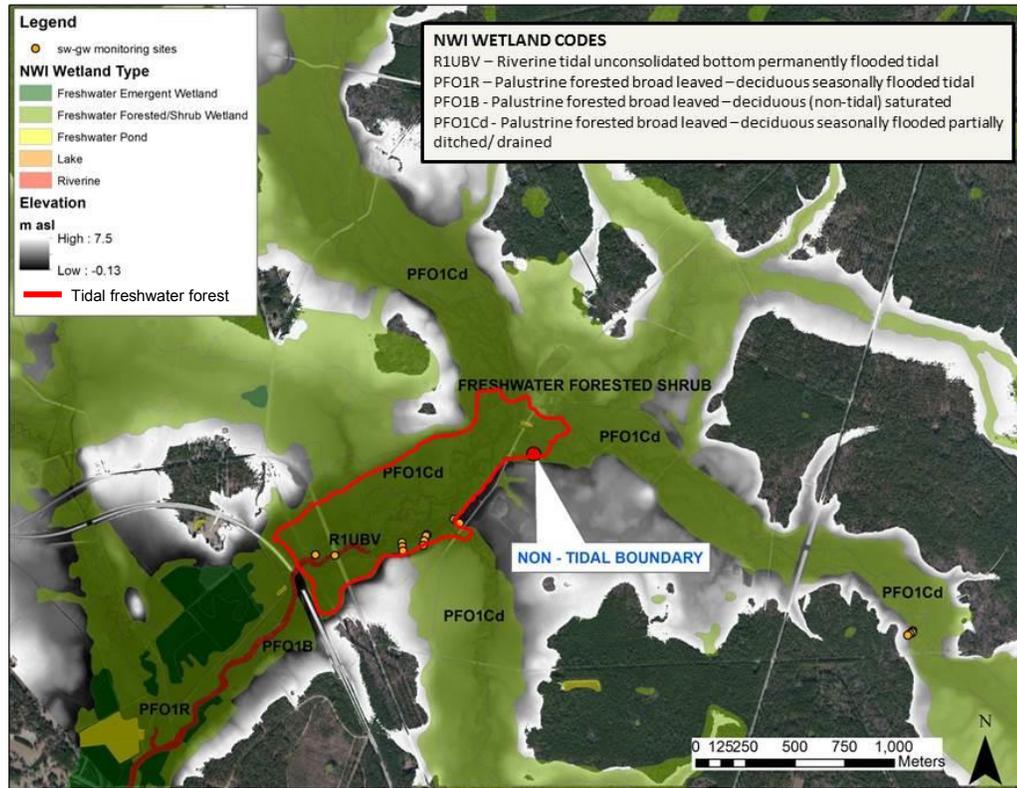
Since canopy composition is related to long-term conditions, and is highly influenced by past disturbance regimes, it was expected that the understory would be more sensitive to water regime. Because tidal freshwater forested wetlands are ecotones, tidal freshwater marsh species and typical forest herbs usually co-exist in the understory. The understories have been described as dense, and high in species richness due to an open canopy and hummock and hollow topography (Rheinhardt 2007; Baldwin 2007). As a whole, the understory was diverse, (Tidal = 30 species, Non-tidal = 42 species) but sparse in total percent ground cover. These data suggest that the depth to oxidation in the rooting zone was not different enough to promote multiple hydrologic microsites, which drive high species diversity and density commonly observed in the understory of in tidal freshwater forests (Baldwin 2007). Collectively, the vegetation data suggest a subtle gradient of wetness existed from lower tidal to upper tidal. These observations may be due to the relative small size of the system compared to previous studies, and that these observations were within the forest continuum rather than along the marsh forest border.

The hydrologic data in this study show distinct differences in hydroperiod and water regimes at Huger Creek and Turkey Creeks. Despite contrasting hydrology, the vegetation and organic matter decomposition results were nuanced. Detailed soil profile data were not collected, but all soils were mapped in the Meggett Series (Long 1980) and

our general descriptions (Appendix A) agree with Meggett soil profile data collected by Harder (2004) and described in the Soil Survey of Berkeley County, SC (Long 1980). Meggett soils are described as having fine sandy loam in the upper 30 cm with increasing clay at a 50 – 70 cm depth. They are poorly drained, clayey, and possess a high water retention value. Even during an unsaturated state and oxidized conditions, water is held tightly in the pore spaces and available to plants. This suggests that even though large differences were observed in hydrologic regime, both between the tidal and non-tidal sites, and along the tidal gradient, the clay soil retained enough water to mute the responses of the biological metrics we selected to measure.

#### *Wetland Mapping*

The data from this study suggest that the Santee Experimental Forest contains over 70 ha of seasonally flooded- tidal freshwater forested wetland. In the current, 2013 National Wetland Inventory (NWI), only a small portion of Huger Creek is considered riverine permanently flooded – tidal, with the riparian zone is listed as a partially ditched or drained palustrine forested wetland (PFO1Cd) (US FWS 2013). In its current state, there is no differentiation between the tidal (Huger Creek) and non-tidal (Turkey Creek) riparian zones (Figure 25).



**Figure 25. Aerial map with National Wetland Inventory layer showing wetland designation codes, note that entire study area is designated as PFO1Cd – palustrine forested (broad-leaved deciduous) partially ditched or drained with no differentiation between tidal and non-tidal zone**

The NWI classification scheme relies on heavily on vegetative communities, which this study has demonstrated are largely insensitive to the tidal and non-tidal designations of the riparian zone. Because this was the first detailed hydrologic conducted study within the Huger Creek watershed, the hydrology was largely unknown. This study has produced new data and suggesting that Huger Creek is classified as a riverine tidal unconsolidated bottom permanently flooded tidal wetland (stream channel), and a palustrine forested seasonally flooded-tidal wetland (riparian zone) (Cowardin et al. 1979).

Observations suggest that reliance on the National Wetland Inventory system may omit important transitional areas due to the coarseness of the classification scheme. Accordingly, the mapping of tidal freshwater forested wetlands is inaccurate and inconsistent, especially in the upper reaches. Tidal freshwater swamps (and forests) have been described as the sentinels of sea level rise; therefore, increased mapping efforts are needed to accurately assess this important resource (Cormier et al. 2012). The use of a LiDAR DEM (Light Detecting and Ranging Digital Elevation Model) data greatly enhances the ability of researchers and managers to discern hydrologic pathways present in floodplain systems.

## **Perspectives**

### ***Hydrologic gradients along the forested tidal/ non-tidal transition zone***

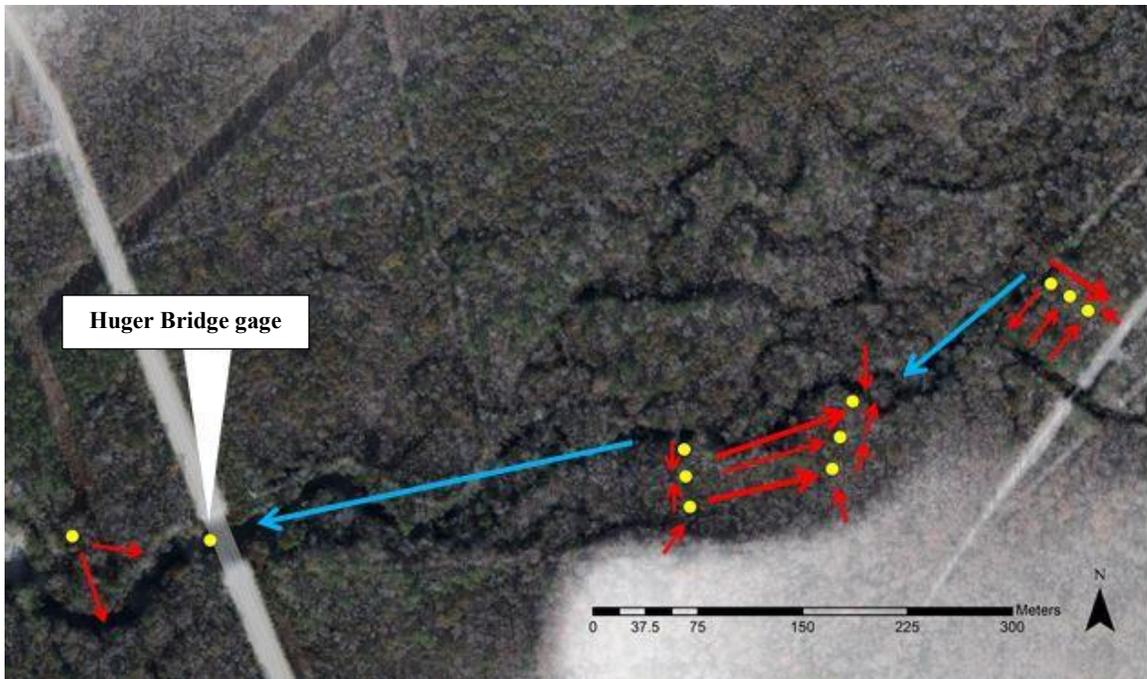
Hydrology has been described as the overarching driver of biogeochemical feedbacks in wetlands; the hydrology determines the physiochemical environment, which drives biological processes that alter hydrology (Mitsch and Gosselink 2007). This study has described the water dynamics present in a tidally influenced riparian wetland and compared them to a topographically similar non-tidal riparian wetland. The non-tidal section may represent a donor wetland because its hydroperiod was dependent upon precipitation inputs, and appeared to function as a groundwater recharge area. The tidal section may represent a conveyer wetland because the wetland received water from the upland and donated water to the downstream (Brinson 1993). The lower tidal reach may also be a significant groundwater discharge area, as shallow water tables limit groundwater recharge (Callahan et al. 2011).

The grid infrastructure of multiple surface water and water table monitoring sites employed in this study allowed for the creation of hydraulic head maps that included all surface water and water table monitoring sites within the tidal reach. Water table and stream surface elevation in the tidal reach (See Figure 6) were mapped at a single time point. This demonstrated the dynamic hydrologic gradient present in the tidal reach, and has implications for sea level rise. Hydraulic head was mapped on a normal day, 3 November 2011 (Figure 26), and after a large storm event, 13 June 2012 (Figure 27).

**Table 7. Hydraulic gradient point measurements in meters above NAVD88 at surface water and water table monitoring sites on 11/3/2011 (deficit period – dry conditions) and 6/12/2012 (surplus period – flooded conditions)**

Site	Hydraulic Head (m NAVD88)	
	Dry	Wet
	Conditions	Conditions
	11/3/2011	6/13/2012
LLT		0.89
Huger Bridge - SG	-0.41	0.03
LT-1	0.79	0.76
LT-2	0.74	1.00
LT-3	0.86	0.94
Tidal Forest - SG	-0.02	0.52
MT-1	-0.34	1.15
MT-2	-0.18	1.14
MT-3	0.00	1.11
Upper Tidal - SG	0.40	0.90
UT-1	-0.21	1.02
UT-2	Dry	1.01
UT-3	Dry	1.27

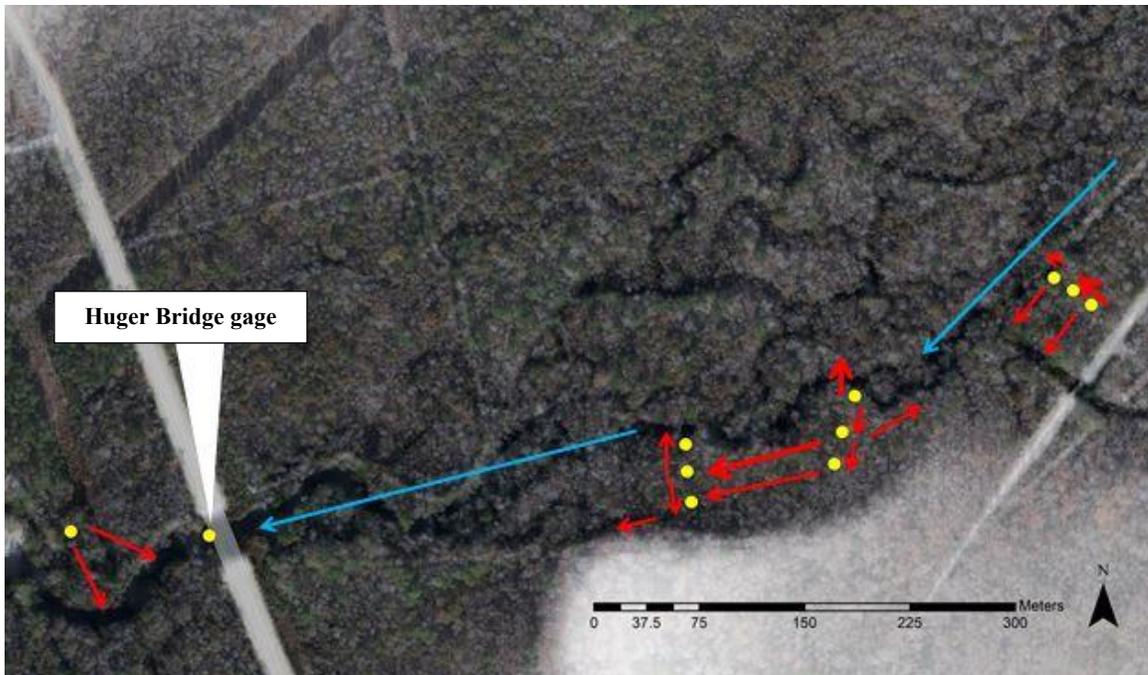
The water dynamics depicted in Figure 26 represent the normal conditions observed throughout the entire study period. Measurements for LLT were not available, but it is assumed that the gradient always followed the stream due to the low topographic position. The predominant hydrologic gradient of the water table was upstream, and the tidal creek functioned as a freshwater reservoir (Figure 26). The lower tidal transect consistently drained towards the middle, which indicated that water was stored in this portion of the wetland, and provides reasoning as to why the water table consistently resided near the surface even during periods of low rainfall and seasonal sea level. Small differences in floodplain topography relate to water storage and potential groundwater discharge and recharge areas (Courtwright and Findlay 2011). Along with vegetation, these small floodplain features influence hydrology, and are capable of transmitting water across the floodplain (sloughs) for storage (back-swamps and flats) (Kroes et al. 2007). Comparing the deficit period to the surplus period revealed that the tidal riparian zone functioned differently depending on antecedent soil conditions. For a majority of the deficit period, the gradient between sites was steep between the lower, middle, and upper tidal transects, and flatter during the surplus.



**Figure 26. LiDAR imagery with surface water and water table monitoring sites, arrows indicate the direction of hydraulic gradient in the wetland and stream on 3 November 2011, during the outgoing tide, graph represents predominant gradient during deficit period. Sizes of arrows indicate relative magnitude of head drop.**

LT-3 was also influenced by the toe slope and slough at the southwestern corner. This area remained consistently wetter than LT-1 and LT-2. The small channel adjacent to UT filled with tide water during the high tide functioning as a lateral well. The losing stream condition was also apparent at UT-1.

Hydraulic head measurements were also inspected on 13 June 2012, after a significant period of rainfall (151 mm), which provide contrast to the “normal” dry gradient (Figure 27). Under flooded conditions, the overall gradient was flat, and the wetland discharged into the water body. This pattern highlights that more water was stored in the interior wetland especially at LT-3. Should these same measurements have been taken during an incoming tide, it is suspected the gradient between LT-1 and the stream, and MT-2 and the stream would be reversed.



**Figure 27. LiDAR imagery with surface water and water table monitoring sites, arrows indicate the direction of hydraulic head in the wetland and stream on 13 June 2012, during the outgoing tide, graph represents predominant gradient during flooded conditions in the surplus period. Sizes of arrows indicate relative magnitude of head drop.**

Because no monitoring sites were installed directly below LT, the exact water level in the swamp was unknown, but observing hydrologic patterns at LT-3 may provide insight. This area of the floodplain was rich with micro relief compared to other portions of the floodplain. Soils here were mucky and often saturated. The area directly southwest appeared to have similar relief, thus hydroperiod may be similar. This suggests that the lower tidal riparian zone may function as a headwater source for the estuary during periods of low river flow. Hydraulic head along LT remained at a near surface elevation on an annual basis, regardless of the hydrologic deficit observed in the first half of this study. It follows that when water is discharged to the creek and released from the riparian zone, it carries nutrient laden sediments to downstream aquatic environments.

### *Implications for sea level rise*

With continued sea level rise, tidal freshwater forested wetlands will experience increased tidal flooding and longer inundation patterns, and it follows that adjoining non-tidal bottomland hardwood forests will soon experience persistent freshwater tides. Tree mortality from increased tidal flooding in upland reaches may be very significant due to the dendritic nature of low order streams. Because there is more interface between the floodplain and stream in the upper reaches, there is a greater possibility that the flooding duration will be longer harming species not adapted to prolonged inundation patterns. There is an abundance of literature documenting the effects of salinity intrusion from sea level rise on tidal freshwater forested wetlands, (Hackney et al. 2007; Krauss et al. 2009; Doyle et al. 2010; Cormier et al. 2012; Craft 2012; Noe et al. 2012; Williams et al. 2012) but due to the landscape position and modifications on the Cooper River, increased freshwater tidal flooding may be a more likely scenario for the Huger Creek.

Anthropogenic modifications (dam discharges) which affect flows in the West Branch of the Cooper, and ultimately the Cooper River main stem, may prevent saline waters from encroaching upstream into the East Branch of the Cooper. Daily dam discharges serve two functions, 1) to generate hydroelectric power, and 2) to protect a drinking water reservoir by regulating salinity levels (Bradley et al. 1990). The current saltwater/freshwater (1 ppt) dividing line is located at river kilometer 24 at the seaward shoreline of the Back River (Figure 3) at its confluence with the Cooper River (SC DNR 2012). However, in salt wedge estuaries like the Cooper River, downstream alterations such as the planned deepening of Charleston Harbor may cause the tidal prism to move upstream. The headwater sources for the East Branch of the Cooper are contained in the

Francis Marion National Forest, and freshwater discharge is dependent upon precipitation and groundwater discharge. During periods of low precipitation (as demonstrated in this study), flows can be completely absent and high tide becomes the primary source of stream flow. Unless dam discharges are increased to compensate for the excess water, there is a potential for the freshwater/saltwater wedge to move further up the Cooper River.

Sea level along the South Carolina coast has risen an average of  $3.15 \text{ mm yr}^{-1}$  (NOAA 2013) as measured at Charleston, South Carolina (NOAA gauge no. 8665530) ] from 1921 - 2006. Sea level is predicted to rise between 0.5 to 1.4 meters by 2100 (Rahmstorf 2007). It would take an increase of approximately 0.25 to 0.5 m of water to overtop the levees along the lower tidal reach in the Huger Creek. This value may be even less during perigean or normal spring tide conditions. The entire wetland would not be flooded, but hummock and hollow topography could become more pronounced from both ponded conditions and water scouring the wetland surface (Rheinhardt and Hershner 1992; Conner et al. 2007). Increased microtopography would change the physiochemical state of the wetland, and for a time, there could be an increase the primary productivity, carbon export, and nutrient cycling. However, as flooding became deeper and more prolonged, the woody vegetation could experience mortality, and productivity rates would decrease. Because a majority of the Huger Creek floodplain is considered hydrologic Zone IV (Wharton et al. 1982) with weakly to moderately flood tolerant tree species (Hook 1984; Theriot 1993), this transition may occur quickly when compared to a Zone II or III floodplain. Forested wetlands play a vital role in the hydrologic cycle, and large-scale losses of woody vegetation would alter the water balance at the catchment

scale. Forested areas mitigate runoff from rainfall events or floods by storing large amounts of water and have the potential to improve water quality. The presence of tree roots reduces erosion rates, which slows the rate of sediment delivery to waterways. Their structure also provides shelter and breeding grounds for waterfowl among many other species. The loss of woody vegetation, especially in the tidal zone could have detrimental effects to estuary health that we are just beginning to understand.

The upland terrestrial boundaries of tidal freshwater forested wetlands are vulnerable to sea level rise and should be considered part of the coastal zone. Titus and Richman (2001) estimate that over 58,000 km<sup>2</sup> of land has topography under the 1.5 m contour (relative to mean sea level) line along the Atlantic and Gulf coasts. If sea level rises to its maximum estimated level over the next 100 years, much of the coastal zone, including the tidal freshwater forested wetlands will be inundated. Even without these projections, recent research suggests that tidal freshwater forests are accreting sediment at 1.3 – 2.2 mm yr<sup>-1</sup> (Craft 2012) which is below the average rate of sea level rise along the South Carolina coast (NOAA 2013c). Tidal freshwater forests are undergoing reversed succession as the forest slowly transforms into emergent (freshwater or brackish) marsh (Conner et al. 2007). It is unknown if the process can be reversed. Unfortunately, the current use of landscape simulation models are insufficient for predicting what changes will occur as sea level continues to rise. “Bathtub” models do not take into account the heterogeneous feedbacks present in these systems, and may underestimate their importance. Models need to be revised to quantify the potential losses or gains of ecosystem services and habitats as it relates to sea level rise. Despite the threat of rising sea level, urbanization in the coastal zone continues to increase in the United States;

accurate assessments of wetland habitats are needed by land managers so the best strategies for development are implemented.

### **Summary and Recommendations**

The main objective of this study was to characterize the hydrologic regime and resulting biological response in a tidal freshwater forested wetland along a decreasing tidal gradient, and compare it to a similar non-tidal site. The overarching research question was to determine if a tidal freshwater forested wetland was wetter than a non-tidal bottomland hardwood forest, assuming similar topographic position. The sites used in this study were Huger Creek, a tidally influenced 4<sup>th</sup>-order watershed and Turkey Creek, a non-tidal tributary. Both sites were located in the Santee Experimental Forest, a U.S. Forest Service research unit, contained within the Francis Marion National Forest, northeast of Charleston, South Carolina. The stream and riparian zone of each site were instrumented with a network of surface water, water table level monitors, and soil moisture plots to detail hydrologic patterns. Depth to oxidation, organic matter decomposition rates, and riparian vegetation was collected to determine if the differing hydrologic regime (longitudinal decreasing tidal gradient and tidal versus non-tidal regime) affected biological processes in the riparian zone.

The hydrologic data from this study supported the overall hypothesis that the tidal freshwater forested wetland is wetter than the non-tidal bottomland hardwood swamp. Tidal forcing was prevalent in the water table hydrographs. The tidal regime of the Huger Creek was irregular (seasonally flooded tidal) because creek levees were high enough to prevent high tides from flooding the wetland surface. However, because only

one side of the floodplain was instrumented (due to accessibility and time constraints) it is unknown if the irregular pattern was consistent within the entire tidal reach. Minor surface flooding from spring high tides was observed in water table hydrographs at the lowest elevation site (LLT), but this was not observed at other sites. In addition, the study period coincided with two years of below average rainfall, which affected the amount of freshwater input discharged from the upland. For the majority of the study period most water in the stream channel was tide water. This may have affected the tidal regime of the forest, and it should be visually re-assessed during high tides during a normal year. During the first summer and fall of this study there was no discharge recorded at the Turkey Creek USGS stream gauge. Spring rainfall temporarily restored flow in Turkey Creek, and water table levels at the middle and upper tidal transects were higher relative to ground surface. For these reasons, the study period was delineated into a deficit and surplus period.

The results of this study emphasize that the hydroperiod in the lower portion of the Huger Creek riparian zone was dominated the ebb and flow of the tidal creek. The large tidal range caused Huger Creek to function as a freshwater reservoir made available to the riparian zone through the daily tide cycle, and kept water table position near the surface on an annual basis. A daily tidal signature of 0.10 – 0.30 m was apparent in the water table closest to the creek back at the lower tidal transect, and tidal hollow flooding was observed in the interior wetland. Although tidal water table forcing was not observed on a daily basis in the interior of the wetland, the time between high tide cycles was brief, and the wetland could not drain. The long-term hydroperiod was relatively stable, <1 m variation when comparing across the deficit and surplus periods, and the tide

muted the water table response to precipitation and evapotranspiration. Additionally, water table levels at LLT and LT-1 closely followed the seasonal sea level patterns and bi-modal monthly lunar tide patterns in Huger Creek.

As the tide range in the creek diminished upstream, a change in hydrodynamics, from tidal to fluvial dominated occurred. Tidal groundwater pulsing was reduced at the middle tidal transect to  $< 0.10$  m, showed an increased response to precipitation and evapotranspiration, and were variable between deficit and surplus periods. At the upper tidal site, the water table was below the streambed for a considerable portion of the study period, and the stream was in a losing condition. Evidence of tidal forcing was only present after large rain events that caused the water table to become  $\leq 1$  m to the ground surface. Hydrographs at upper tidal and the non-tidal site were similar with respect to depth to water and response to precipitation and evapotranspiration.

The shift from tidal to fluvial dominated processes occurred near the middle transect approximately, 500 meters below the head of the tide. These results contrasted to other findings where the shift from tidally dominated to fluvial dominated processes occurred several kilometers downstream. Additionally, the observed differences in hydroperiod between the deficit and surplus periods in this study suggest that this point is seasonally variable and influenced by upland creek discharge and precipitation.

The biological metrics of vegetation and organic matter decomposition did not reveal a gradient of wetness corresponding with the decreasing tidal range of Huger Creek. Depth to oxidation agreed with tidal regime, but not distinct enough to show differences in vegetation composition and decomposition rates between the tidal and non-tidal sites. This resulted in a failure to reject the null hypothesis that organic matter

decomposition (total mass remaining over time) would be greater where fluctuating water table (i.e.: tidal forcing) produced alternating periods of wet and dry. The riparian vegetation was overall insensitive to the tidal and non-tidal designations of the floodplain, again resulting in a failure to reject the null hypothesis that vegetation would have discrete differences in response to the changes in hydrologic regime. The results of the tests did not detect significant differences in flooding tolerance of canopy species, nor wetland indicator status of riparian vegetation to indicate “wetter” species existed in the tidal reach. These results emphasize that vegetation patterns along gradients are nuanced and the effects of past land use history (agriculture and timbering) or disturbance (Hurricane Hugo) likely influenced riparian vegetative communities. Presumably, the differences in flooding patterns were not intense enough to alter biological metrics at this, however, the hydrologic data emphasize that the tidal and non-tidal riparian zones are hydrologically distinct, and should be evaluated separately.

In this study, the use of multiple stream stage and water table well monitoring sites has demonstrated the complexity of surface water and groundwater interaction across spatial and temporal scales. It reinforces that the hydrologic regime and upper-forested boundary in tidal freshwater forested wetlands is dynamic due to seasonality and local climate effects. This dataset combined with previous studies in the upper portions of Turkey Creek will enhance our knowledge of lowland forest ecology as we are propelled into a period of changing climate and rising sea levels.

### ***Future Work***

This project generated a fine resolution hydrologic dataset on a previously unstudied portion of the Santee Experimental Forest. Due to time constraints of the current study, only a small area could be surveyed. The following are recommendations for further study on the Huger Creek site, which will enhance our knowledge about tidally influenced freshwater ecosystems. The location and current state of the tidally influenced riparian zone would make an excellent reference site to study tidal freshwater forest ecology. Huger Creek is not currently undergoing conversion to an oligohaline marsh, and it is protected from development because a substantial portion is contained within the Santee Experimental Forest. In developing the site, the tidal gradient monitoring should extend across the floodplain. This study showed how the floodplain was asymmetrical, and only the narrow side could be studied. Instrumenting both sides of the floodplain will provide a clearer picture of the water balance and provide insight to the complete tidal regime within the riparian zone. Vegetation surveys should be more intensive and include leaf area index (LAI) and measures of primary production. In addition, the surveys should encompass a larger area, include the opposite side of the floodplain, and be extended to include regions of downstream tidal freshwater marsh. Marsh surveys could be extended farther downstream to include the oligohaline and mesohaline parts of the estuary.

Surface water and groundwater chemistry measurements would augment the current dataset. Comparing surface water and groundwater conductivity measurements could determine the degree of connectivity (hydrologic vs. hydraulic connection) between the tidal creek and the riparian zone. A second recommendation is to construct a

water budget for the Huger Creek watershed. There are no published accounts of water budget estimations for tidal freshwater forested systems. Nested piezometers installed next to the existing water table wells could determine the vertical hydrologic gradient, and the installation of rain gages and sap flow (evapotranspiration measurements) instrumentation would detail site-specific conditions for the watershed. Velocity measurements could easily be taken and referenced to the existing stream gage at the SC State Highway 402 overpass. Finally, a general recommendation is to increase mapping efforts using LiDAR data. This will help refine the known aerial estimates of tidal freshwater forested wetlands in the Southeastern United States, and provide information about the status of this valuable ecosystem.

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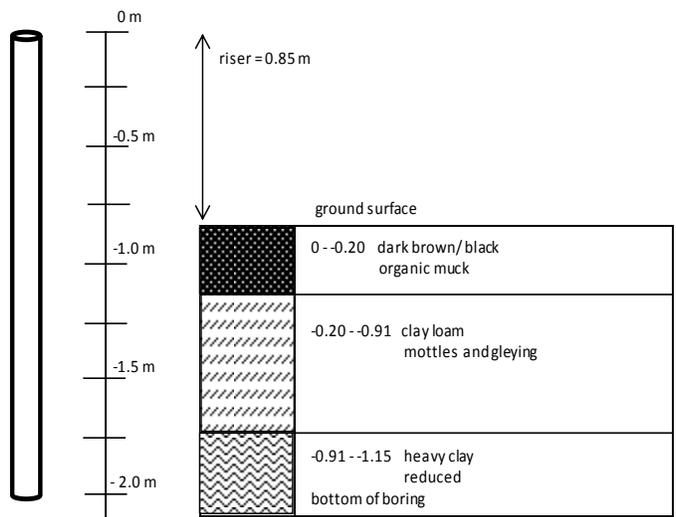
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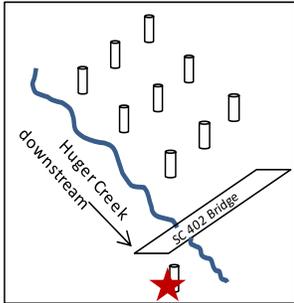
# APPENDIX A. WELL CONSTRUCTION DIAGRAMS AND SOIL PROFILE DESCRIPTIONS

Well construction diagram and soil profile for well LLT-1

Well material : 2.0 in PVC, total length 6.73 ft (2.05 m)  
 Soil map unit: Mp - Megget clay loam  
 Ground elevation : 0.933 m (NAVD 88)  
 Date installed : 02/03/2012

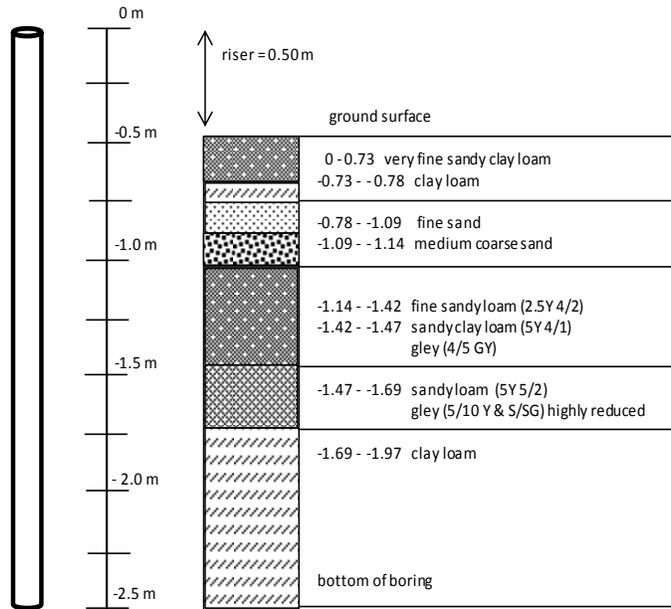


Vertical scale: 1 cm = 0.25 m

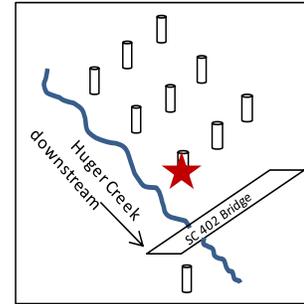


Well construction diagram and soil profile for well LT-1

Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Megget loam  
 Ground elevation : 1.24 m (NAVD 88)  
 Date installed : 5/17/2011

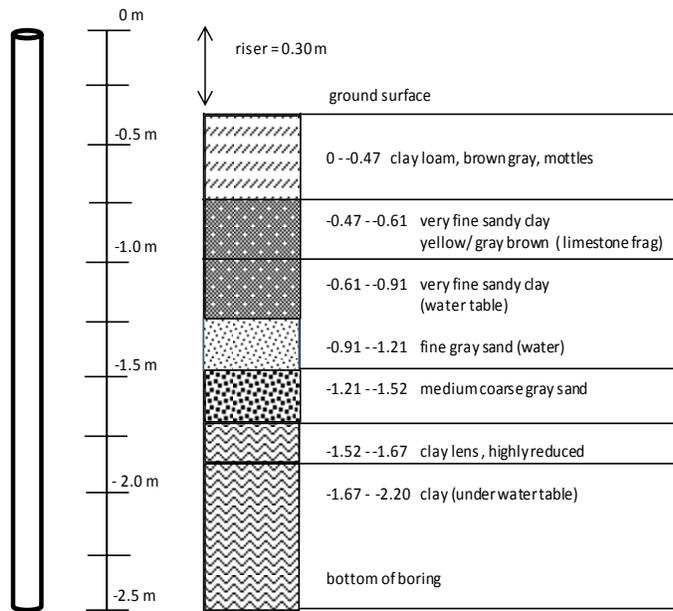


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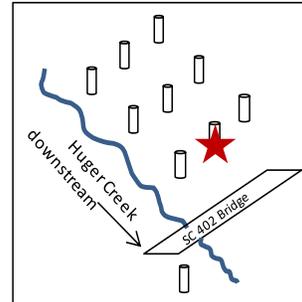


Well construction diagram and soil profile for well LT-2

Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Meggett loam  
 Ground elevation : 1.12 m (NAVD 88)  
 Date installed : 06/10/2011

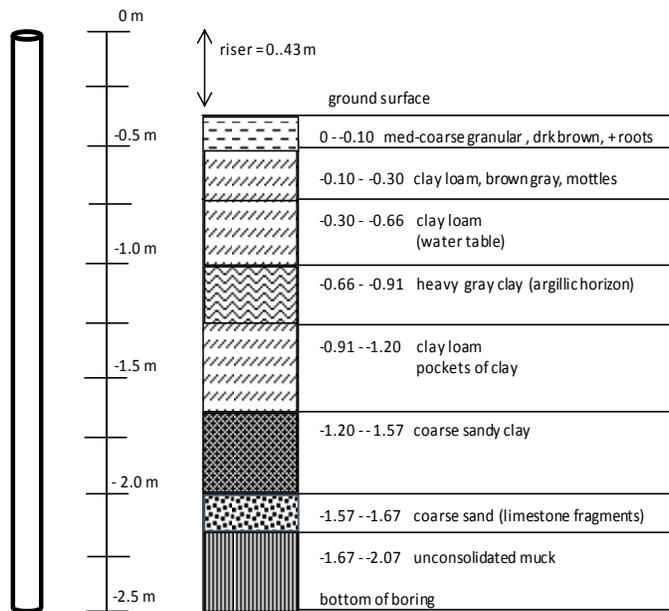
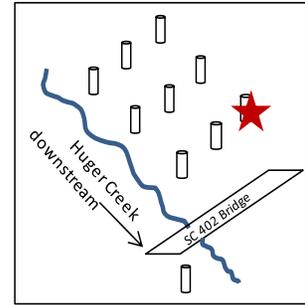


Vertical scale: 1 cm = 0.25 m



Well construction diagram and soil profile for well LT-3

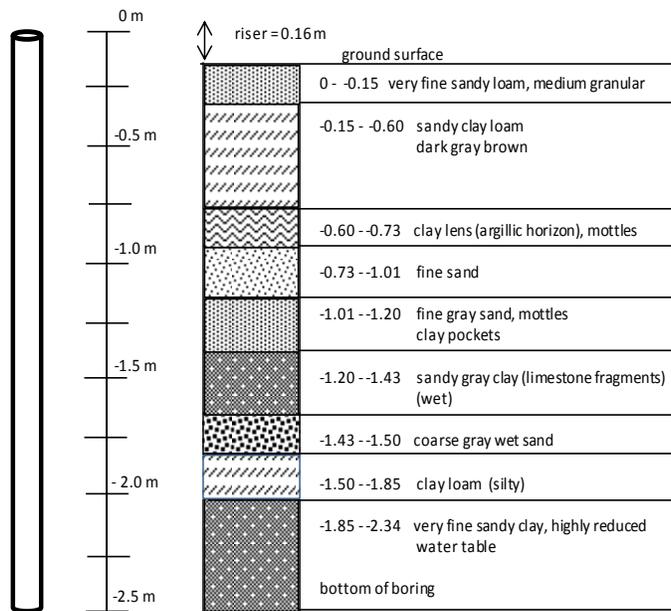
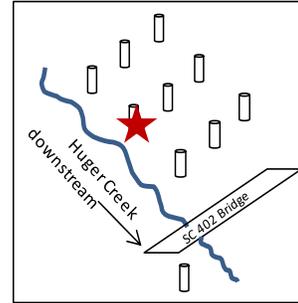
Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Meggett loam  
 Ground elevation : 1.04 m (NAVD 88)  
 Date installed : 06/14/2011



Vertical scale: 1 cm = 0.25 m

Well construction diagram and soil profile for well MT-1

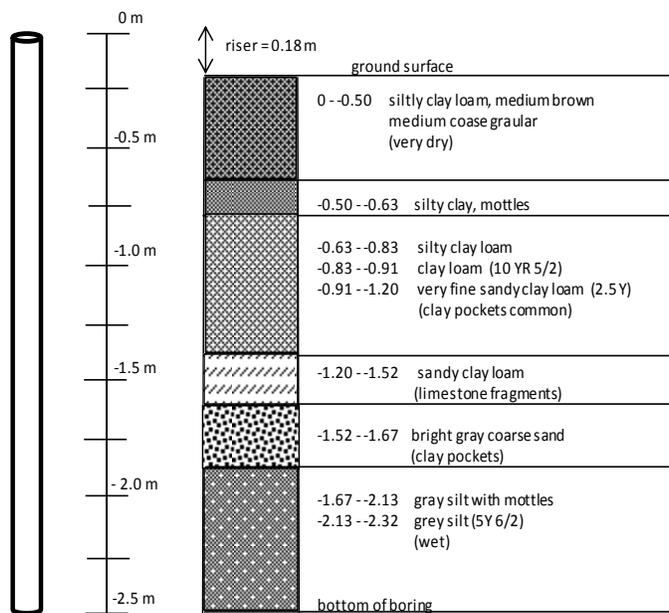
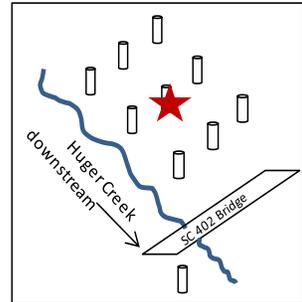
Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Meggett loam  
 Ground elevation : 1.26 m (NAVD 88)  
 Date installed : 06/10/2011



Vertical scale: 1 cm = 0.25 m

### Well construction diagram and soil profile for well MT-2

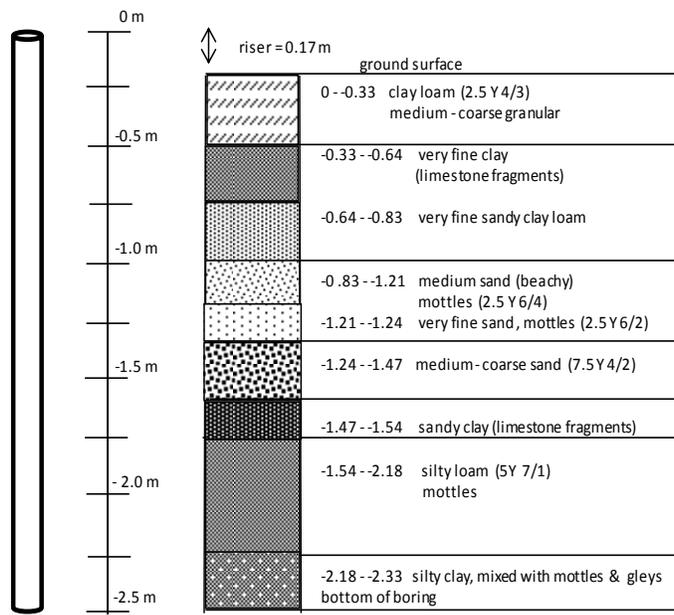
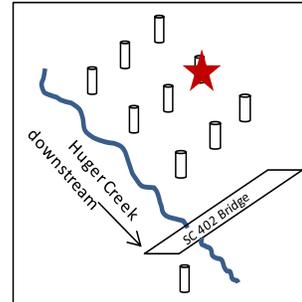
Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Meggett loam  
 Ground elevation : 1.26 m (NAVD 88)  
 Date installed : 06/10/2011



Vertical scale: 1 cm = 0.25 m

### Well construction diagram and soil profile for well MT-3

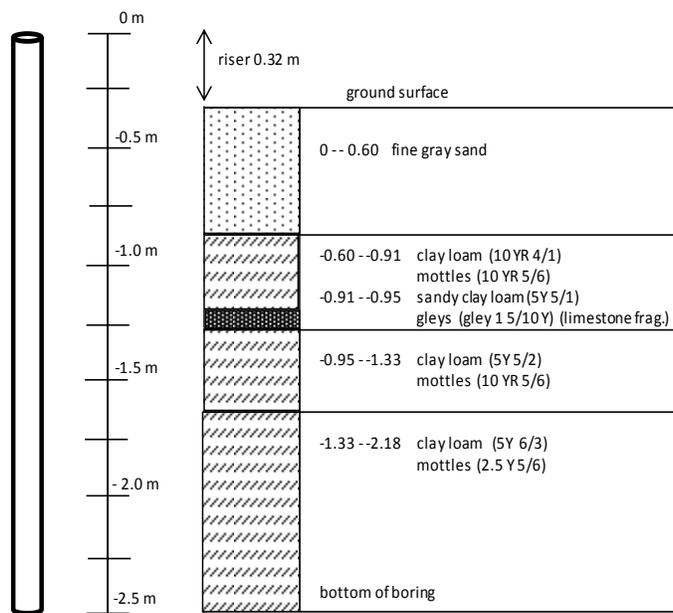
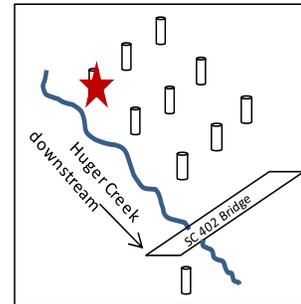
Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil map unit: Mg - Meggett loam  
 Ground elevation : 1.22 m (NAVD 88)  
 Date installed : 06/16/2011



Vertical scale: 1 cm = 0.25 m

Well construction diagram and soil profile for well UT-1

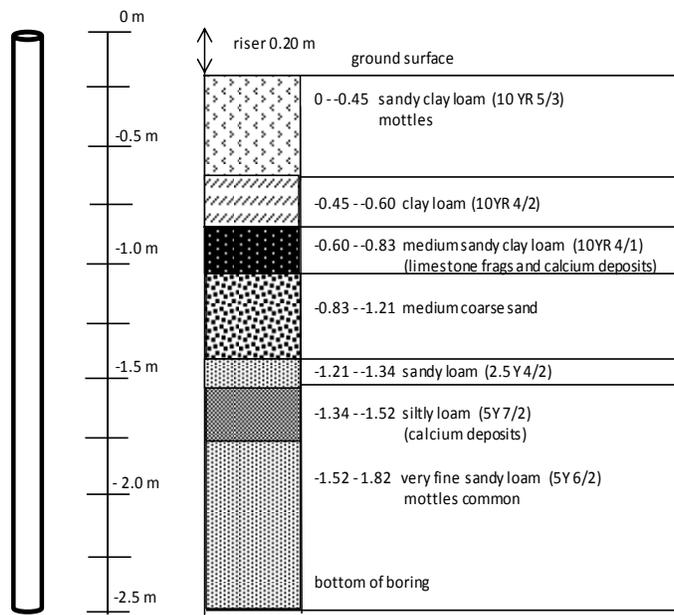
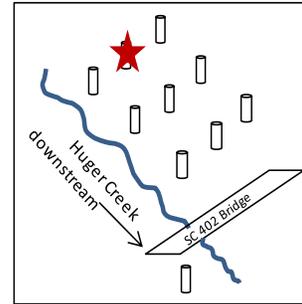
Well material : 1.5 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 1.28 m (NAVD 88)  
 Date installed : 05/17/2011



Vertical scale: 1 cm = 0.25 m

### Well construction diagram and soil profile for well UT-2

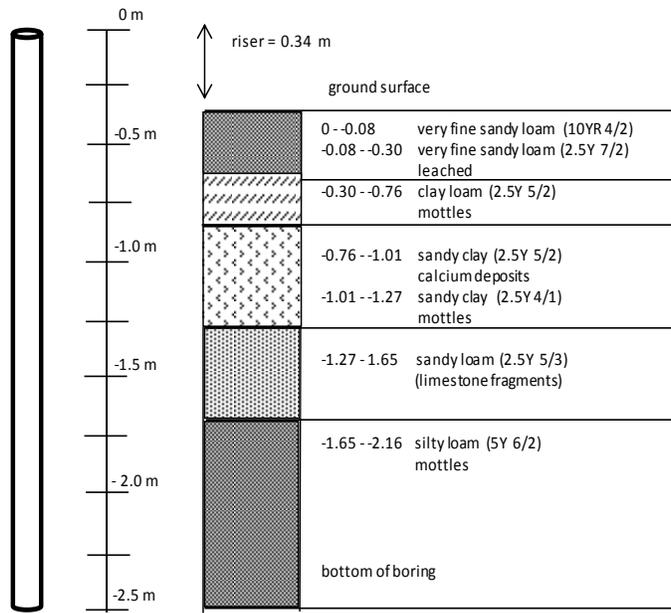
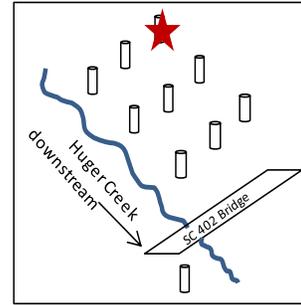
Well material : 2.0 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 1.36 m (NAVD 88)  
 Date installed : 06/21/2011



Vertical scale: 1 cm = 0.25 m

Well construction diagram and soil profile for well UT-3

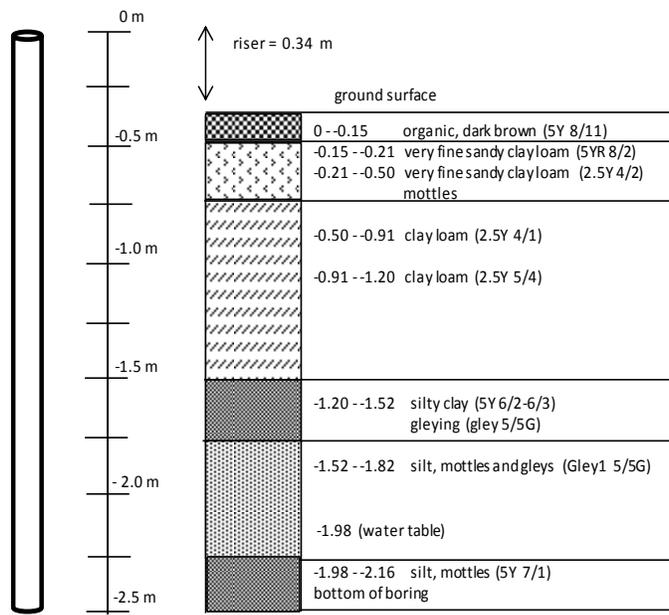
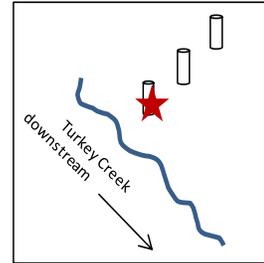
Well material : 2.0 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 1.36 m (NAVD 88)  
 Date installed : 06/27/2011



Vertical scale: 1 cm = 0.25 m

Well construction diagram and soil profile for well NT-1

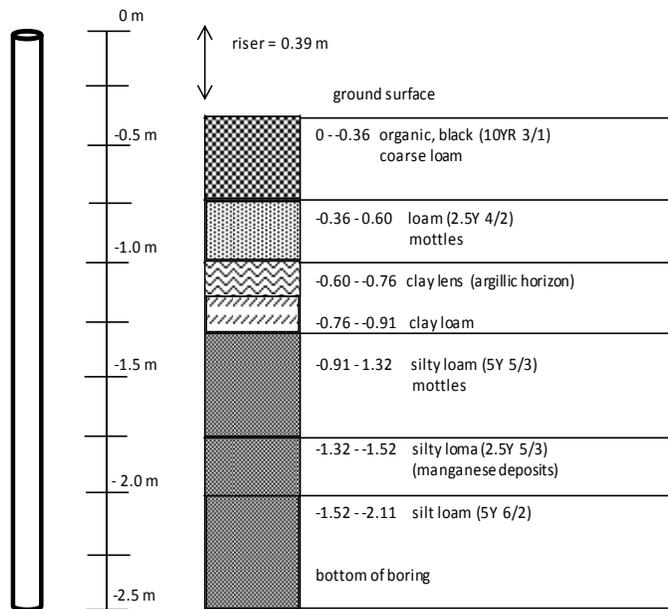
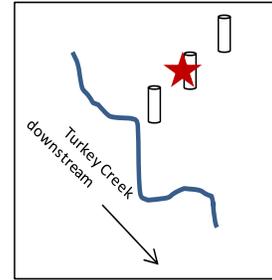
Well material : 2.0 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 3.55 m (NAVD 88)  
 Date installed : 06/27/2011



Vertical scale: 1 cm = 0.25 m

Well construction diagram and soil profile for well NT-2

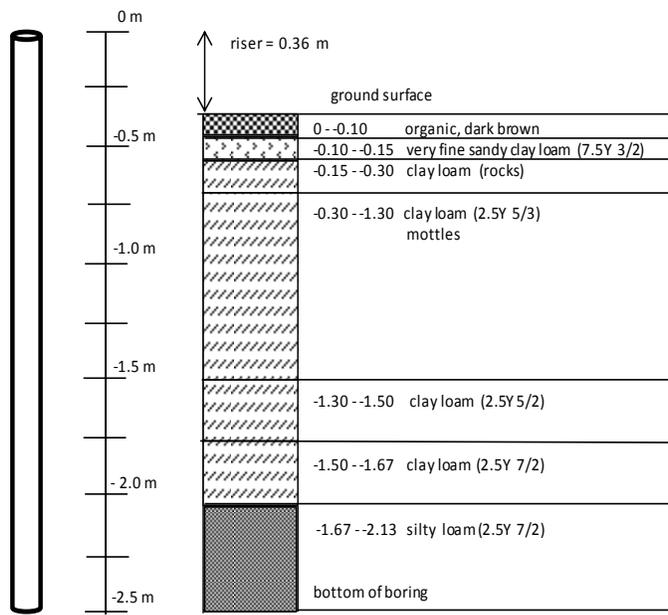
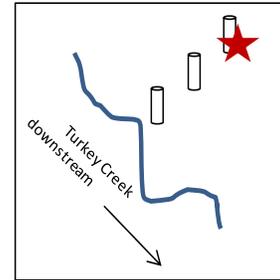
Well material : 2.0 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 3.82 m (NAVD 88)  
 Date installed : 06/27/2011



Vertical scale: 1 cm = 0.25 m

### Well construction diagram and soil profile for well NT-3

Well material : 2.0 in PVC, total length 8.25 ft (2.51 m)  
 Soil Map Unit: Mg - Meggett loam  
 Ground elevation : 4.85 m (NAVD 88)  
 Date installed : 06/27/2011

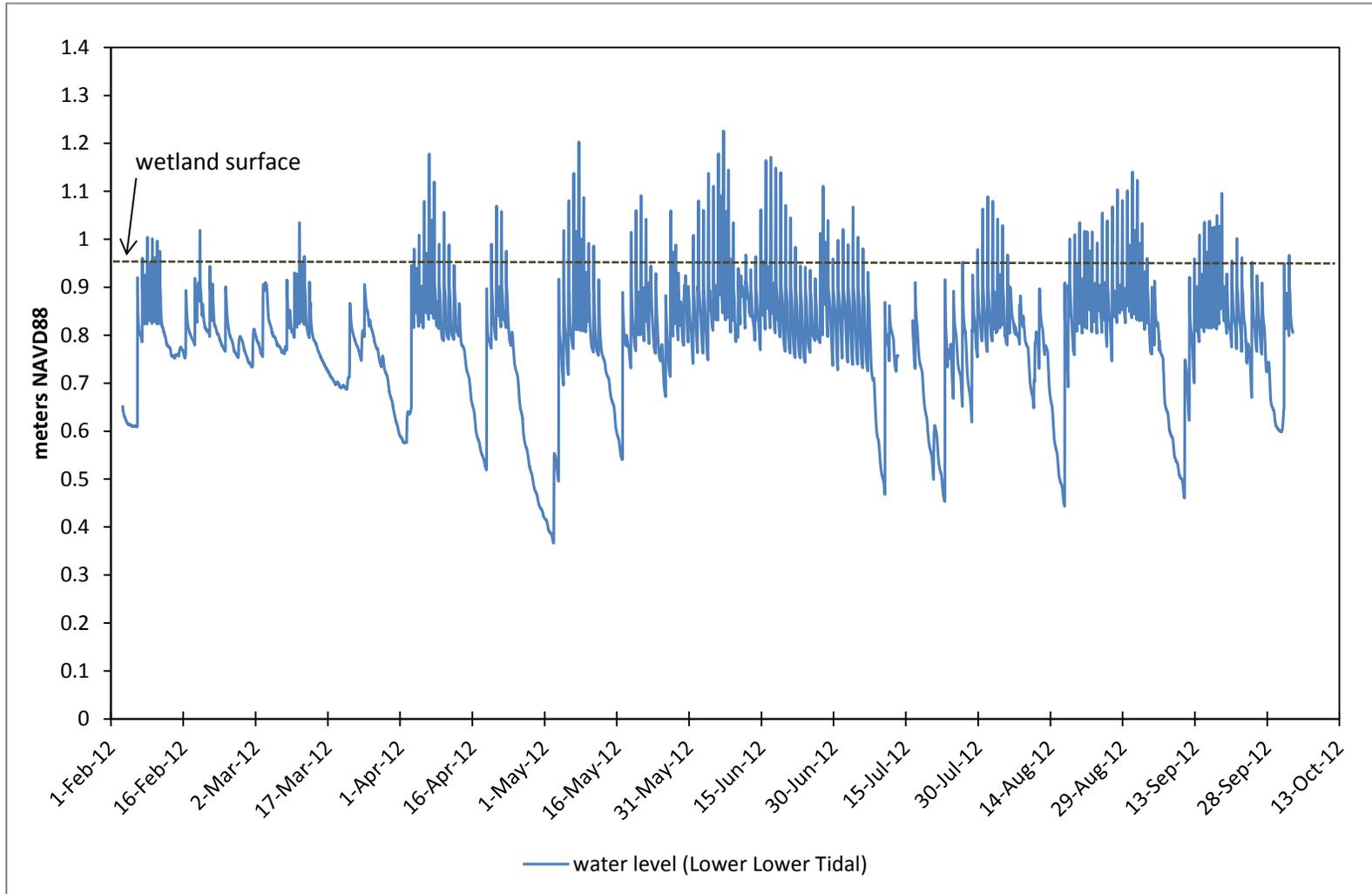


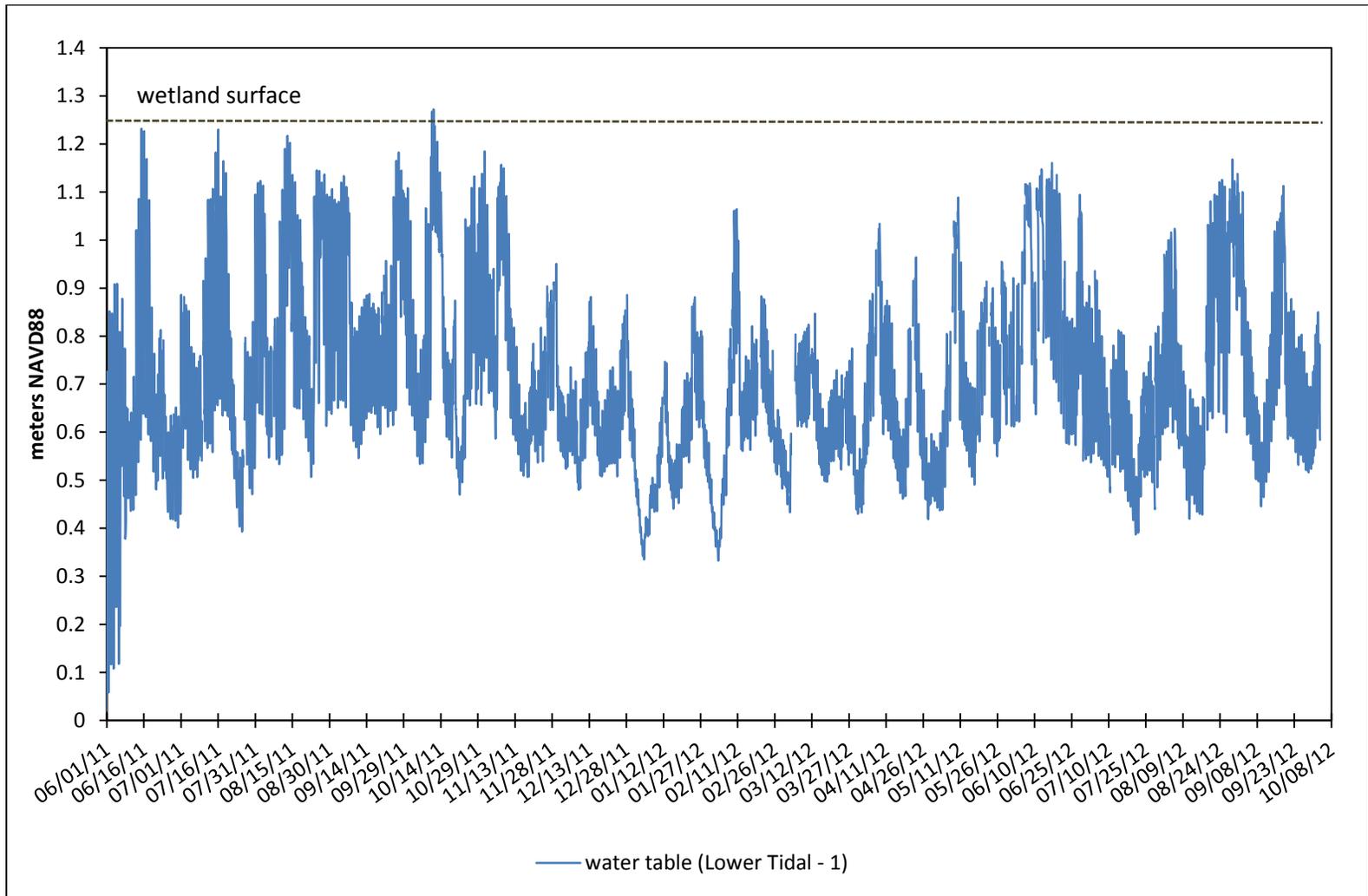
**APPENDIX B: MEAN MONTHLY WATER TABLE ELEVATION AND DEPTH TO WATER WITH STANDARD DEVIATION BY SITE**

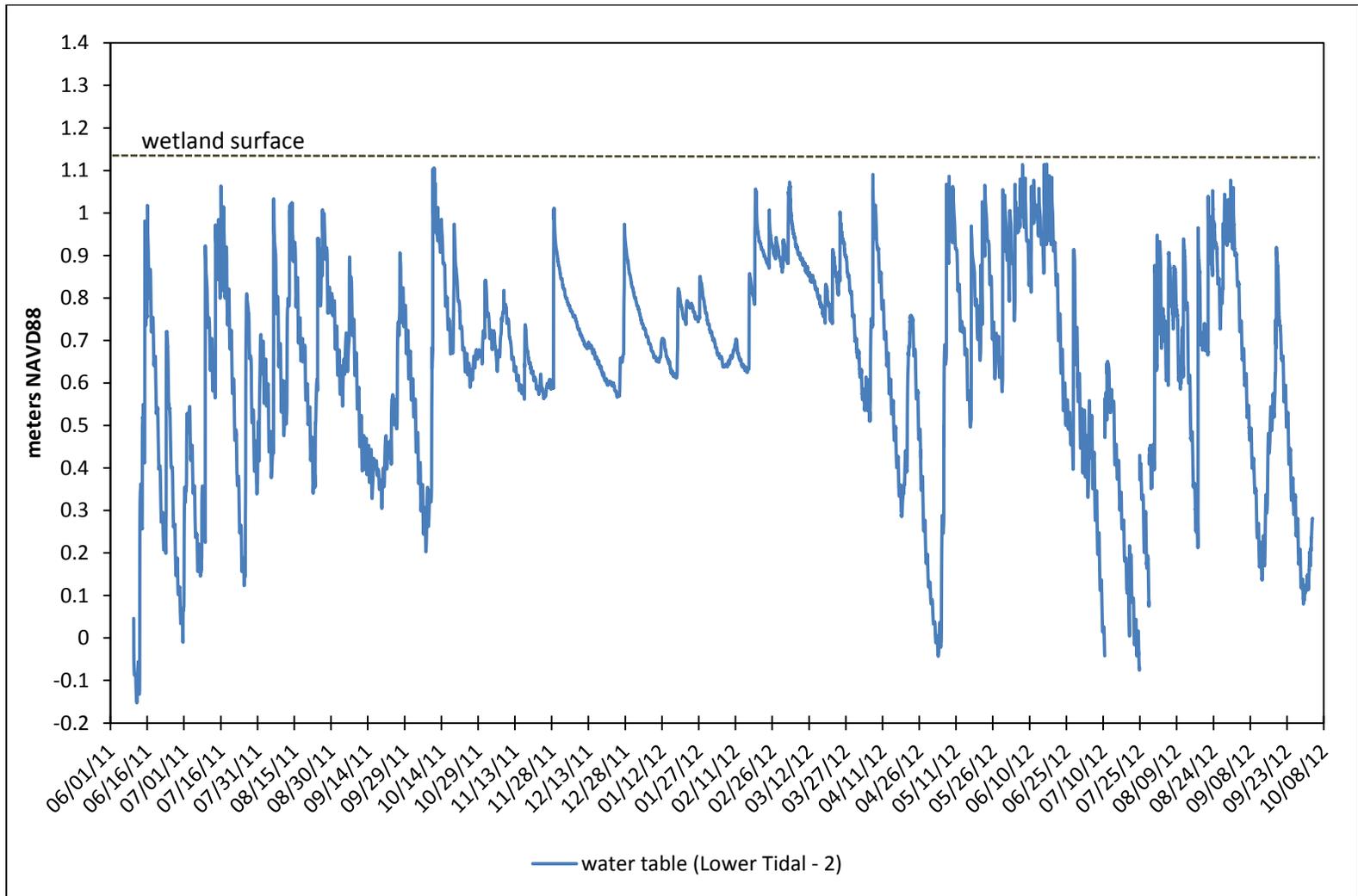
<b>WATER TABLE ELEVATION (m NAVD)</b>									
	<b>Period</b>	<b>LLT</b>	<b>LT-1</b>	<b>LT-2</b>	<b>LT-3</b>	<b>MT-1</b>	<b>MT-2</b>	<b>UT-1</b>	<b>NT-1</b>
Jun-11	Deficit		0.604 (0.13)	0.365 (0.30)	0.483 (0.27)	-0.261 (0.05)	-0.536 (0.08)	-0.675 (0.11)	1.686 (0.01)
Jul-11	Deficit		0.694 (0.13)	0.541 (0.23)	0.647 (0.21)	-0.107 (0.14)	-0.472 (0.16)	-0.491 (0.46)	1.686 (0.01)
Aug-11	Deficit		0.825 (0.13)	0.698 (0.16)	0.796 (0.10)	0.235 (0.24)	-0.099 (0.30)	-0.052 (0.39)	1.879 (0.18)
Sep-11	Deficit		0.810 (0.11)	0.562 (0.15)	0.733 (0.10)	0.142 (0.11)	-0.239 (0.15)	-0.204 (0.23)	2.082 (0.41)
Oct-11	Deficit		0.786 (0.16)	0.660 (0.20)	0.791 (0.12)	0.146 (0.16)	-0.196 (0.20)	-0.242 (0.29)	2.082 (0.14)
Nov-11	Deficit		0.738 (0.13)	0.676 (0.09)	0.791 (0.06)	0.248 (0.11)	-0.278 (0.08)	-0.377 (0.23)	1.769 (0.05)
Dec-11	Deficit		0.626 (0.06)	0.714 (0.10)	0.758 (0.07)	0.358 (0.06)	-0.115 (0.09)	-0.223 (0.09)	1.637 (0.04)
Jan-12	Deficit		0.546 (0.11)	0.725 (0.06)	0.773 (0.05)	0.348 (0.05)	-0.051 (0.05)	-0.348 (0.06)	1.783 (0.01)
Feb-12	Surplus	0.798 (0.07)	0.630 (0.14)	0.785 (0.14)	0.801 (0.07)	0.627 (0.34)	0.315 (0.48)	-0.192 (0.17)	1.774 (0.02)
Mar-12	Surplus	0.784 (0.06)	0.610 (0.08)	0.856 (0.07)	0.851 (0.05)	0.952 (0.12)	0.950 (0.06)	0.179 (0.27)	2.127 (0.47)
Apr-12	Surplus	0.745 (0.15)	0.655 (0.13)	0.560 (0.22)	0.674 (0.19)	0.422 (0.24)	0.258 (0.36)	-0.354 (0.23)	3.123 (0.18)
May-12	Surplus	0.779 (0.13)	0.710 (0.13)	0.685 (0.31)	0.729 (0.26)	0.382 (0.34)	0.233 (0.45)	-0.210 (0.31)	2.382 (0.29)
Jun-12	Surplus	0.876 (0.04)	0.833 (0.13)	0.845 (0.18)	0.868 (0.08)	0.817 (0.33)	0.797 (0.35)	0.383 (0.43)	1.957 (0.11)
Jul-12	Surplus	0.751 (0.10)	0.611 (0.07)	0.323 (0.19)	0.495 (0.24)	-0.129 (0.10)		-0.614 (0.14)	2.653 (0.48)
Aug-12	Surplus	0.826 (0.10)	0.768 (0.17)	0.764 (0.16)	0.813 (0.13)	0.086 (0.40)		-0.402 (0.37)	1.801 (0.12)
Sep-12	Surplus	0.786 (0.01)	0.700 (0.13)	0.470 (0.24)	0.636 (0.22)	0.186 (0.28)		-0.356 (0.31)	2.001 (0.44)
<b>AVE WT Z (m)</b>		0.793	0.697	0.639	0.727	0.278	0.044	-0.261	2.026
<b>N</b>		8	16	16	16	16	13	16	16
<b>STD Dev</b>		0.04	0.09	0.16	0.11	0.33	0.45	0.27	0.4
<b>STD Err</b>		0.02	0.02	0.04	0.03	0.08	0.13	0.07	0.1
<b>Ground Elev. (m NAVD)</b>		0.933	1.24	1.12	1.04	1.26	1.23	1.38	3.55
<b>AVE DTW (m)</b>		0.14	0.543	0.481	0.313	0.982	1.186	1.641	1.524

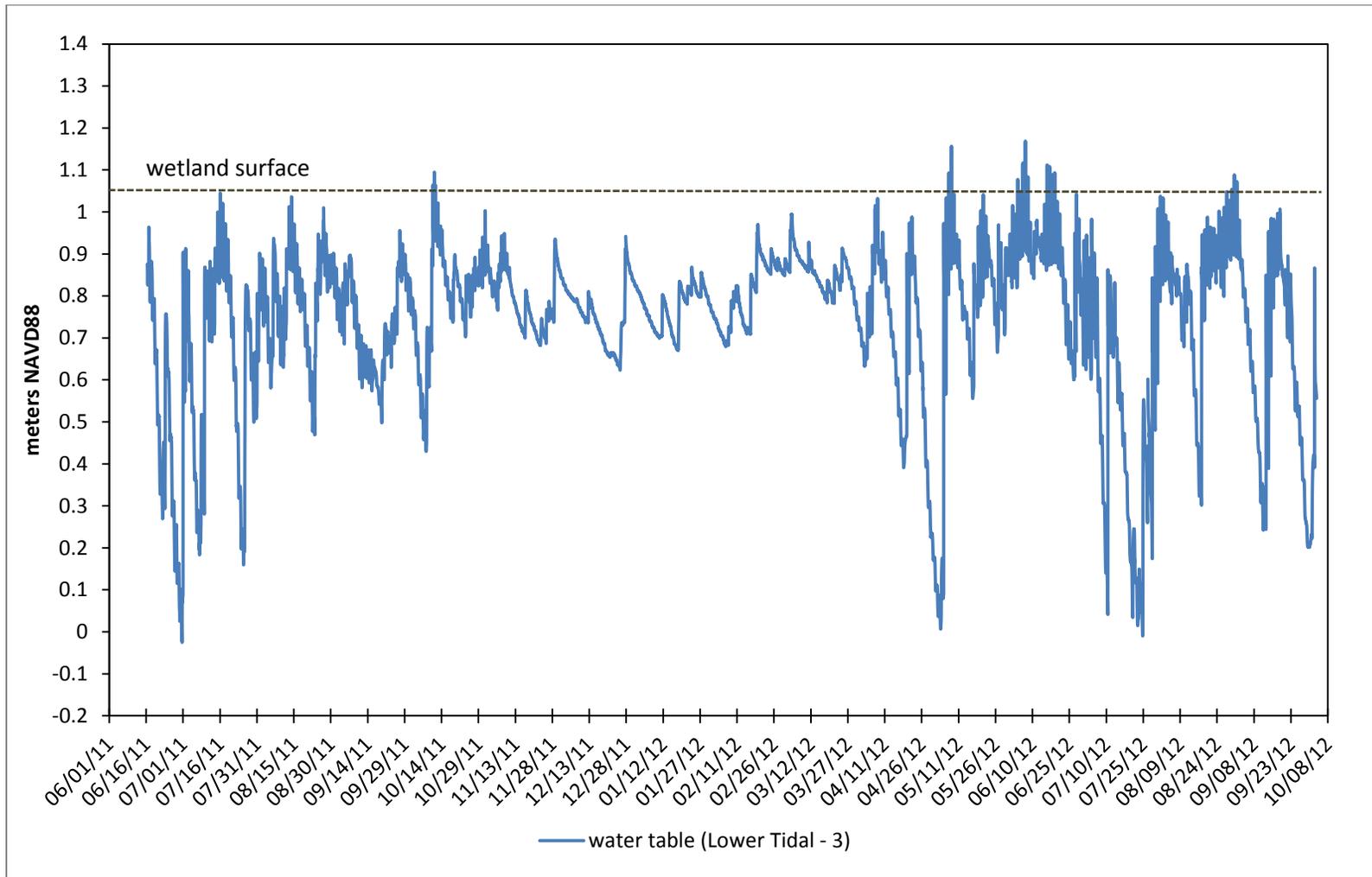
DEPTH TO WATER (m bgs)									
	Period	LLT	LT-1	LT-2	LT-3	MT-1	MT-2	UT-1	NT-1
	Jun-11	Deficit	0.636 (0.13)	0.755 (0.30)	0.557 (0.27)	1.521 (0.05)	1.766 (0.08)	1.955 (0.11)	1.864 (0.01)
	Jul-11	Deficit	0.546 (0.13)	0.579 (0.23)	0.393 (0.21)	1.367 (0.24)	1.702 (0.16)	1.771 (0.46)	1.864 (0.01)
	Aug-11	Deficit	0.415 (0.13)	0.422 (0.16)	0.244 (0.10)	1.025 (0.24)	1.329 (0.30)	1.332 (0.39)	1.671 (0.18)
	Sep-11	Deficit	0.430 (0.11)	0.558 (0.15)	0.307 (0.10)	1.118 (0.11)	1.469 (0.15)	1.484 (0.23)	1.468 (0.41)
	Oct-11	Deficit	0.454 (0.16)	0.460 (0.20)	0.249 (0.12)	1.114 (0.16)	1.426 (0.20)	1.522 (0.29)	1.468 (0.14)
	Nov-11	Deficit	0.502 (0.13)	0.444 (0.09)	0.249 (0.06)	1.012 (0.11)	1.508 (0.08)	1.657 (0.23)	1.781 (0.05)
	Dec-11	Deficit	0.614 (0.06)	0.406 (0.10)	0.282 (0.07)	0.902 (0.06)	1.345 (0.09)	1.503 (0.09)	1.913 (0.04)
	Jan-12	Deficit	0.694 (0.11)	0.395 (0.06)	0.267 (0.05)	0.912 (0.05)	1.281 (0.05)	1.628 (0.06)	1.767 (0.01)
	Feb-12	Surplus	0.135 (0.07)	0.610 (0.14)	0.335 (0.14)	0.239 (0.07)	0.633 (0.34)	0.915 (0.48)	1.472 (0.17)
	Mar-12	Surplus	0.149 (0.06)	0.630 (0.08)	0.264 (0.07)	0.189 (0.05)	0.308 (0.12)	0.280 (0.06)	1.101 (0.27)
	Apr-12	Surplus	0.188 (0.15)	0.585 (0.13)	0.560 (0.22)	0.366 (0.19)	0.838 (0.24)	0.972 (0.36)	1.634 (0.23)
	May-12	Surplus	0.154 (0.13)	0.530 (0.13)	0.435 (0.31)	0.311 (0.26)	0.878 (0.34)	0.997 (0.45)	1.490 (0.31)
	Jun-12	Surplus	0.057 (0.04)	0.407 (0.13)	0.275 (0.18)	0.172 (0.08)	0.443 (0.33)	0.433 (0.35)	0.897 (0.43)
	Jul-12	Surplus	0.182 (0.10)	0.629 (0.07)	0.797 (0.19)	0.545 (0.24)	1.389 (0.20)	1.894 (0.14)	0.897 (0.48)
	Aug-12	Surplus	0.107 (0.10)	0.472 (0.17)	0.356 (0.16)	0.227 (0.13)	1.174 (0.40)	1.682 (0.37)	1.749 (0.12)
	Sep-12	Surplus	0.147 (0.11)	0.540 (0.13)	0.650 (0.24)	0.404 (0.22)	1.074 (0.28)	1.636 (0.31)	1.549 (0.44)
	<b>AVE DTW (m)</b>	0.14	0.543	0.481	0.313	0.982	1.186	1.541	1.524
	<b>N</b>	8	16	16	16	16	13	16	16
	<b>STD Dev</b>	0.04	0.09	0.16	0.11	0.33	0.45	0.27	0.4
	<b>STD Err</b>	0.01	0.02	0.04	0.03	0.08	0.13	0.07	0.1
	<b>Ground Elev. (m NAVD)</b>	0.933	1.24	1.12	1.04	1.26	1.23	1.38	3.55
	<b>AVE WT Z (m)</b>	0.793	0.697	0.639	0.727	0.278	0.044	-0.161	2.026

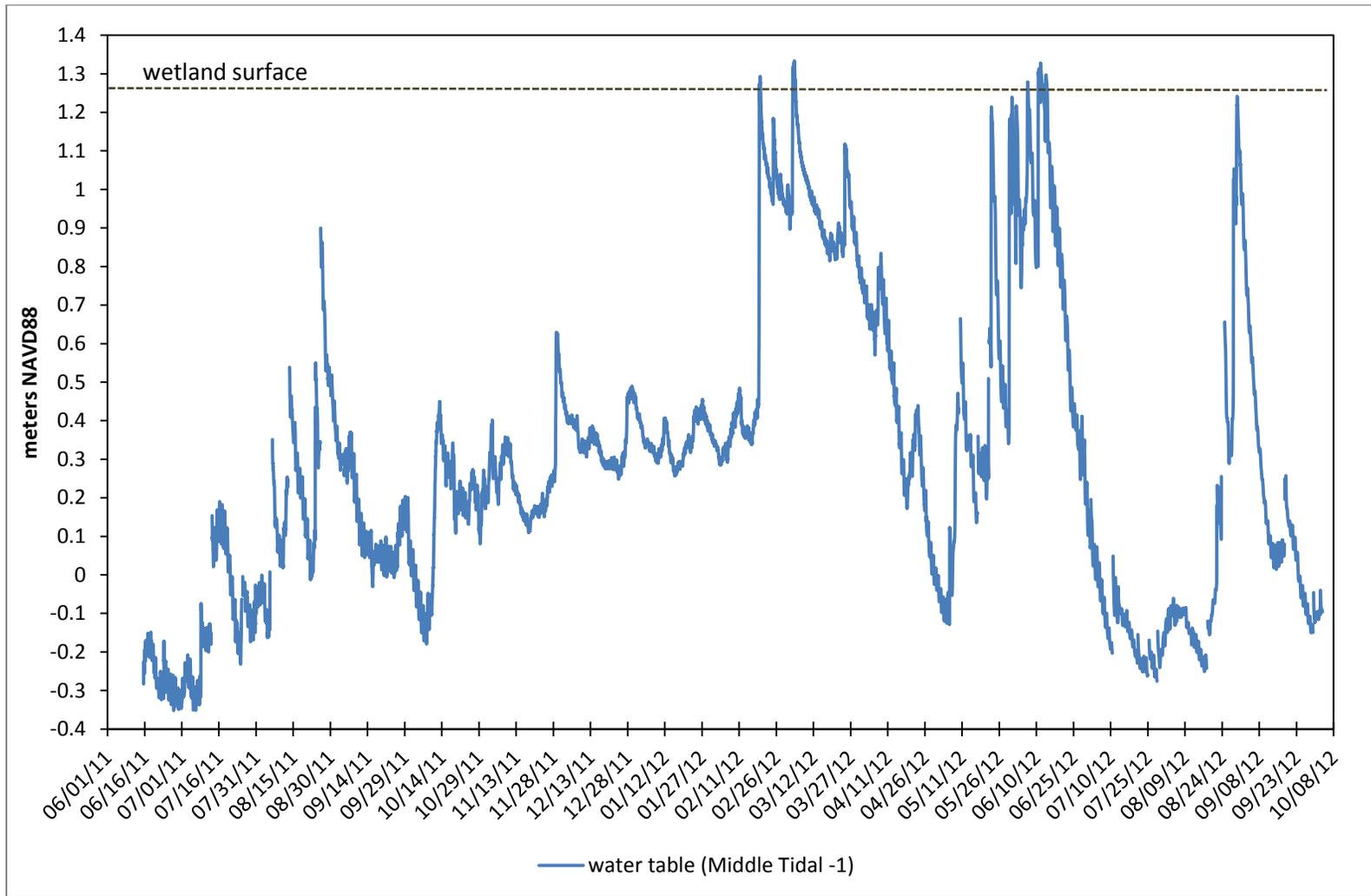
### APPENDIX C: LONG TERM WATER TABLE HYDROGRAPHS BY SITE

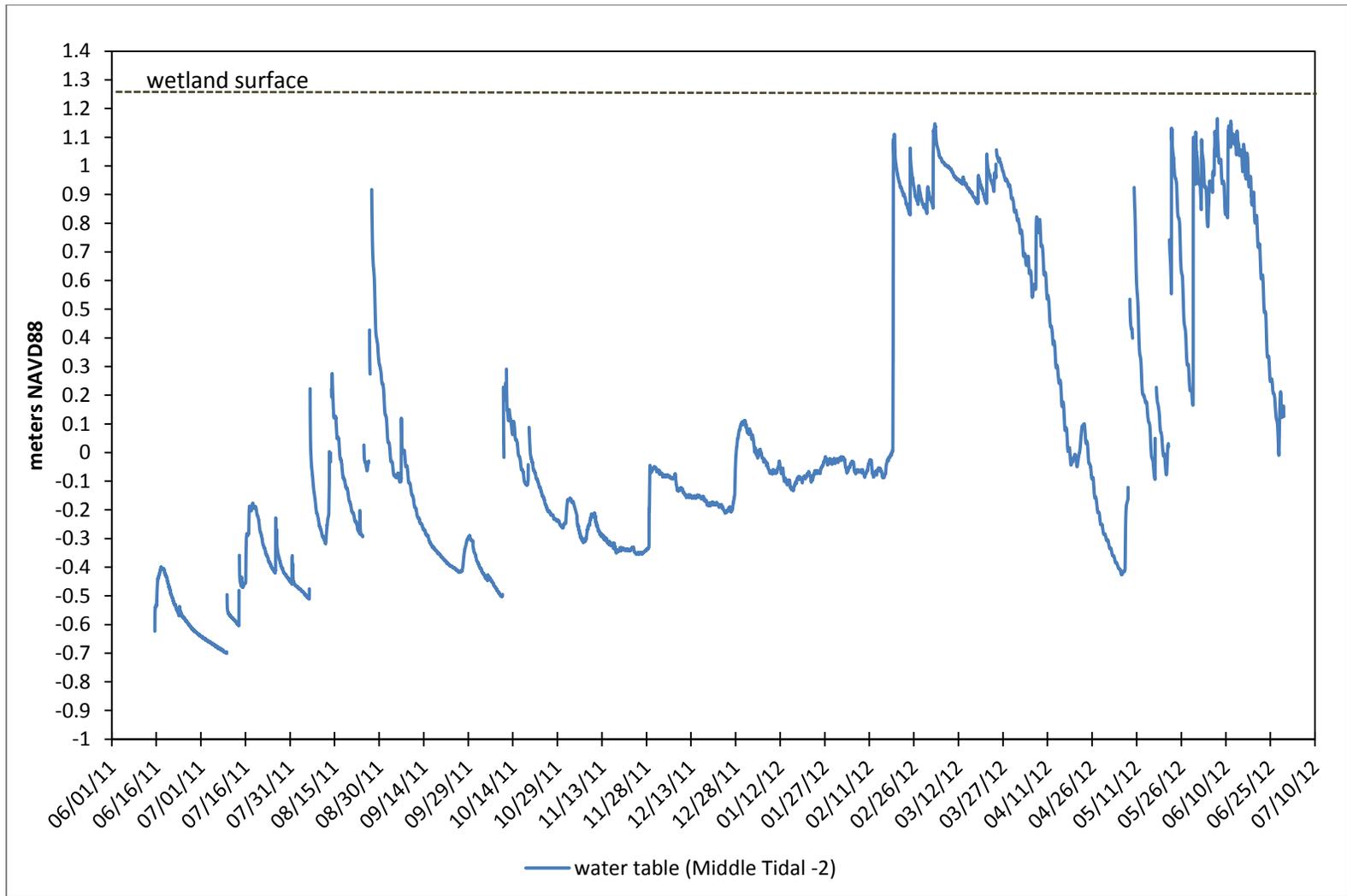


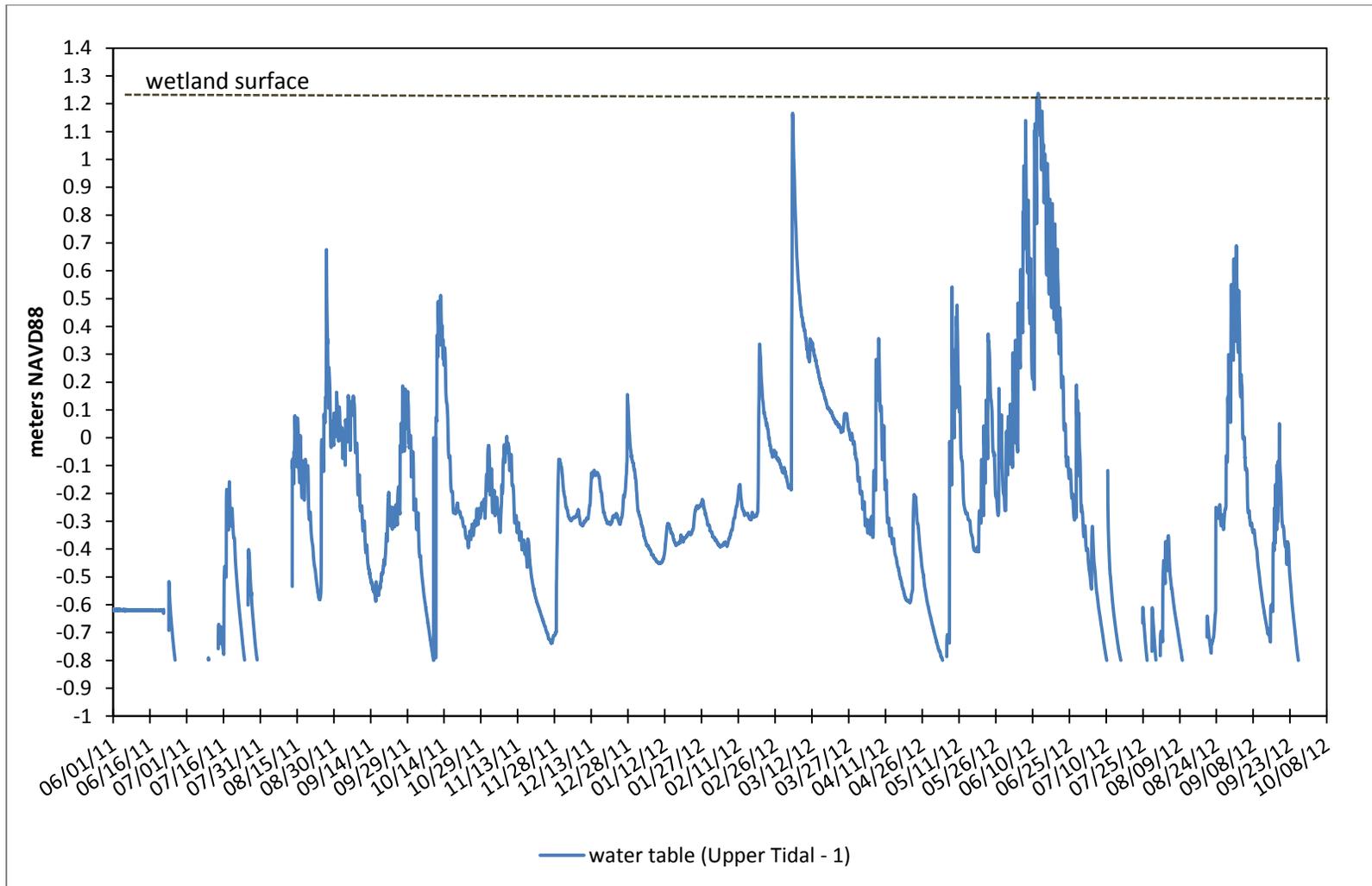


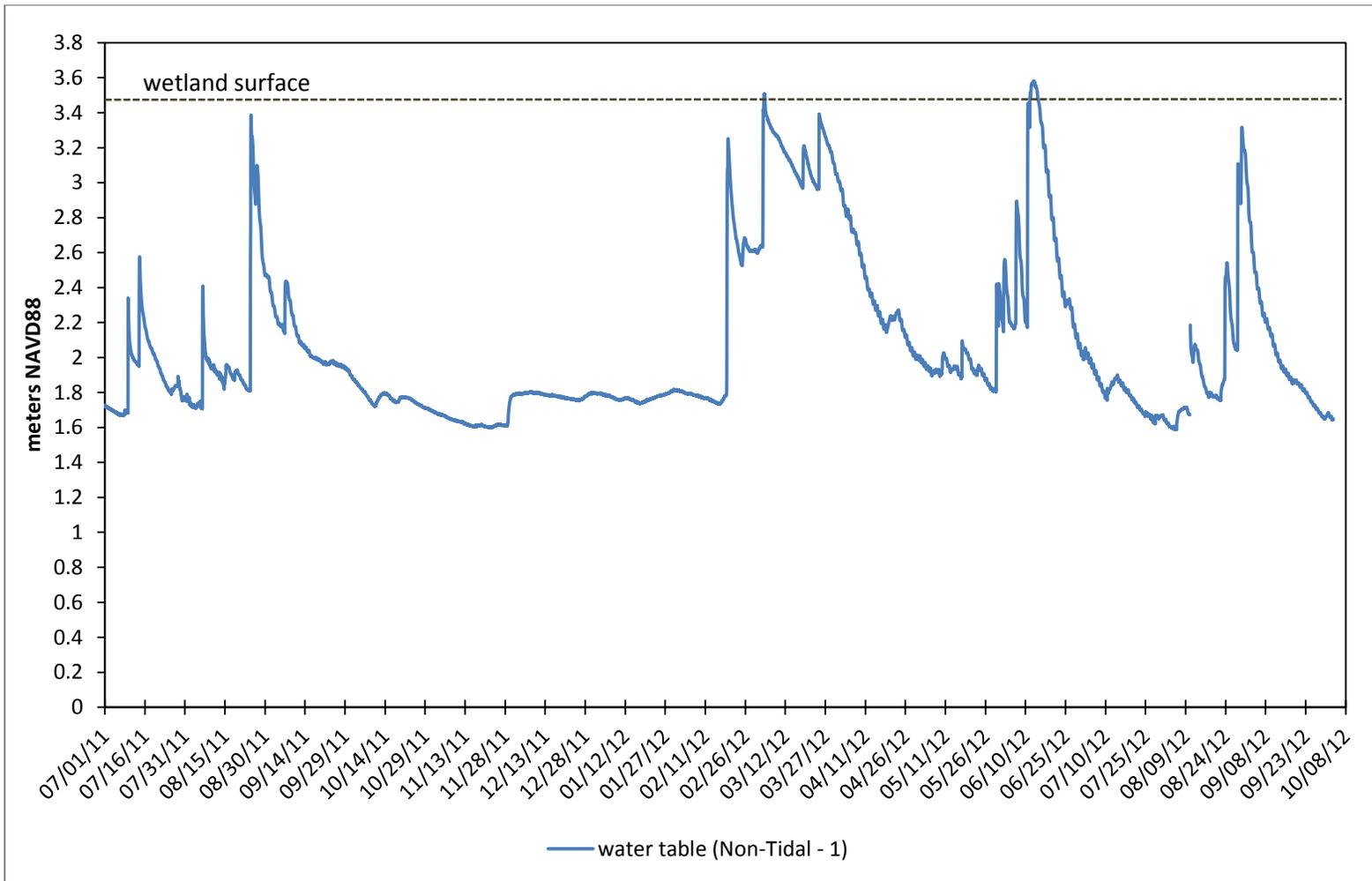












**APPENDIX D. SCIENTIFIC AND COMMON NAMES WITH WETLAND INDICATOR STATUS OF PLANT SPECIES IDENTIFIED IN THIS STUDY**

Scientific Name	Common Name	Wetland Indicator Status	Wetland			
			Lower Tidal	Middle Tidal	Upper Tidal	Non-Tidal
<i>Acer rubrum</i> L.	red maple	FAC	x		x	
<i>Ampelopsis arborea</i> (L.) Koehne	Peppervine	FAC	x	x	x	X
<i>Arundinaria gigantea</i> (Walter) Muhl. ssp. <i>tecta</i> (Walter) McClure	switch cane	FACW	x	x		X
<i>Begonia capreolata</i> L.	cross vine	FAC	x	x	x	X
<i>Berchemia scandens</i> (Hill) K. Koch	Alabama supple jack	FACW	x		x	X
<i>Boehmeria cylindrica</i> (L.) Sw.	smallspike false nettle	FACW	x			
<i>Callicarpa americana</i> L.	American beautyberry	FACU				X
<i>Campsis radicans</i> (L.) Seem. ex Bureau	trumpet creeper	FAC		x	x	X
<i>Carex</i> L.	Sedge	NL*	x	x	x	X
<i>Carpinus caroliniana</i> Walter	Ironwood	FAC	x	x	x	X
<i>Carya</i> sp.	Hickory	NL*				X
<i>Celtis laevigata</i> L.	Sugarberry	FACW	x			
<i>Chimaphila maculata</i> (L.) Pursh	striped wintergreen	NONE				X
<i>Cornus foemina</i> P. Mill	swamp dogwood	FACW	x			X
<i>Diospyros virginiana</i> L.	Persimmon	FAC				X
<i>Euonymus americanus</i> L.	American strawberry bush	FAC	x	x	x	X
<i>Fraxinus pennsylvanica</i> Marshall	green ash	FACW	x	x	x	X
<i>Fraxinus caroliniana</i> Mill.	Carolina ash	OBL	x			
<i>Galium triflorum</i> Michx.	fragrant bedstraw	FACU				X
<i>Gelsemium sempervirens</i> (L.) W.T. Aiton	Carolina jessamine	FAC	x	x	x	X
<i>Hamamelis virginiana</i> L.	American witch-hazel	FACU				X
<i>Hydrocotyle umbellata</i> L.	marsh pennywort	OBL	x			
<i>Ilex decidua</i> Walt.	Possumhaw	FACW	x		x	X
<i>Ilex opaca</i> Aiton	American holly	FAC	x	x	x	X
<i>Liquidambar styraciflua</i> L.	Sweetgum	FAC	x	x	x	X
<i>Lonicera japonica</i> Thunb.	Japanese honeysuckle	FAC	x	x	x	X
<i>Lyonia lucida</i> (Lam.) K. Koch	fetterbush lyonia	FACW	x	x	x	X
<i>Mitchella repens</i> L.	partridgeberry	FACU	x	x	x	X
<i>Morus rubra</i> L.	red mulberry	FACU				X

<i>Nyssa aquatica</i> L.	water tupelo	OBL	x				x
<i>Nyssa biflora</i> Walter	swamp tupelo	OBL	x	x			
<i>Nyssa sylvatica</i> Marshall	black gum	FAC				x	
<i>Osmunda cinnamomea</i> L.	cinnamon fern	FACW	x	x			x
<i>Panicum</i> L.	panic grass	NL*	x	x	x		x
<i>Parthenocissus quinquefolia</i> (L.) Planch.	Virginia creeper	FACU	x	x	x		x
<i>Pinus</i> sp.	pine sp.	NL*	x	x	x		x
<i>Pinus taeda</i> L.	loblolly pine	FAC					x
<i>Quercus pagoda</i> Raf.	cherrybark oak	FACU				x	x
<i>Quercus laurifolia</i> Michx.	laurel oak	FACW	x	x	x		x
<i>Quercus marilandica</i> Münchh.	blackjack oak	NONE					x
<i>Quercus michauxii</i> Nutt.	swamp chestnut oak	FACW	x	x	x		x
<i>Quercus nigra</i> L.	water oak	FAC				x	
<i>Rubus</i> L.	Blackberry	NL*				x	x
<i>Sabal minor</i> (Jacq.) Pers.	dwarf palmetto	FACW	x	x	x		x
<i>Sanicula canadensis</i> L.	black snakeroot	FACU			x	x	x
<i>Schoenoplectus tabernaemontani</i> (C.C. Gmel.) Palla	soft stem bulrush	OBL	x				
<i>Smilax</i> L.	Greenbrier	NL*	x	x	x		x
<i>Taraxacum officinale</i> F.H. Wigg.	common dandelion	FACU					x
<i>Toxicodendron radicans</i> (L.) Kuntze	poison ivy	FAC	x	x	x		x
<i>Trachelospermum difforme</i> (Walter) A. Gray	climbing dogbane	FACW				x	
<i>Ulmus alata</i> Michx.	winged elm	FACU				x	x
<i>Ulmus americana</i> L.	American elm	FACW	x	x	x		x
<i>Vaccinium elliotii</i> Chapm.	Elliott's blueberry	FACW	x	x	x		x
<i>Viburnum</i> L.	viburnum sp.	FACU					x
<i>Viola</i> L.	Violet	NL*			x	x	x
<i>Vitis rotundifolia</i> Michx.	muscadine	FAC	x	x	x		x
	Clover						x
	Moss		x				
	pea vine				x	x	x
	unknown aquatic		x				
	unknown flowering herb				x		x
	unknown shrub				x		
	unknown vine						x

\*NL (not listed unless specific species)