Chapter 3 - Soils and Biogeochemistry of Tidal Freshwater Forested Wetlands

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3.1. Introduction

Tidal freshwater forested wetlands are located along the upper tidal reach of coastal rivers. These forests have a highly complex biogeochemistry in large part because they are influenced by coastal tides but are far enough up river that they are not regularly inundated by saltwater. In addition to freshwater tides, these forests also receive water at different frequencies from river floods, saltwater surges, and groundwater sources. The quantity and timing of each source varies seasonally and annually, often resulting in dramatic shifts in soil physiochemical conditions. In the southeastern United States, soil complexity of tidal freshwater forested wetlands is amplified by a variety of site-specific factors such as microtopography, local climate (e.g., wind driven tides), elevation, proximity to river mouth, vegetation type and cover, and physiographic origin. All of these factors result in soil conditions that are highly variable over space and time, and difficult to generalize.

Compared to other wetlands, there have been very few studies on freshwater tidal forested wetlands and among those, very few have evaluated soil conditions in detail. The scarcity of tidal forested research is likely attributed to their narrow range of occurrence. Perhaps the most commonly cited source for general community descriptions in the southeast United States is Wharton et al. (1982) as part of their comprehensive review of bottomland hardwood swamps. More detailed analyses and/or experimental research on edaphic conditions have been conducted along Gulf coast rivers such as the Apalachicola River (Coultas 1984) and Suwannee River (Light et al. 2002) in Florida. Studies on the Atlantic Coast include the

W.H. Conner, T.W. Doyle and K.W. Krauss (eds.), Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States, 65–88.

Pamunkey River, a tributary of the lower Chesapeake Bay in Virginia (Rheinhardt 1991, 1992); the White Oak River (Kelley 1993; Kelley et al. 1990; Megonigal 1996; Megonigal and Schlesinger 2002) and Cape Fear River (Hackney et al. 2002, 2005) in North Carolina; and the Savannah River in South Carolina (Duberstein 2004). Other information comes from numerous county soil surveys produced by the U.S. Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS) (fka, USDA-Soil Conservation Service). Older editions of the USDA-NRCS soil surveys tended to aggregate coastal wetlands and may have contributed to the impression that these were fairly homogeneous soils. However, as newer USDA-NRCS surveys have become available, more intensive classification has occurred (J. DeWit, personal communication; Light et al. 2002), and it has become apparent that floodplain soils are among the most heterogeneous of forested ecosystems.

Because prolonged periods of inundation are common, tidal freshwater wetland soils are noted for being highly anaerobic and containing high amounts of organic matter. Consequently, these soils have the potential to retain nutrients and other material from the inflowing waters through sedimentation, denitrification, plant uptake/detritus storage, sorption and microbial immobilization. Many freshwater tidal wetlands in the southeastern United States accumulate fine sediments in the form of silts and clays which, because of their high sorption affinity, can contribute to the retention of phosphorus and other nutrients (Mitsch and Gosselink 2000). High organic matter contents in tidal freshwater wetland soils also contribute to high cation exchange capacity and the retention of heavy metals and other potential pollutants (Simpson et al. 1983). Where soils are highly reduced, carbon mineralization is limited and dominated by methanogenesis, sulfate reduction, or both in areas where sulfur availability (associated with oceanic water) fluctuates over time (Hackney et al. 2005). As seen through the examples in this chapter, there is considerable temporal and spatial variability in soil processes among and within these tidal swamps.

3.2. Geomorphology

3.2.1. Geological history

The geomorphology of coastlines along both the southeast Atlantic and the Gulf of Mexico has been primarily shaped by multiple changes in sea level that occurred during the late Pleistocene and early Holocene. Following sea level declines in the late Pleistocene (approximately 18,000 years ago), shorelines along the Gulf and southeast Atlantic reached southern extremities that were approximately 100–250 km and 100 km, respectively, seaward of present positions (Hunt 1967). The concurrent increase in channel gradient stimulated the incision of fluvial valleys near coastal areas and, consequently, rivers and floodplains extended to the margins of present-day continental shelves (Smith 1988). Evidence of buried fluvial channels in the Gulf has been found at points 30 m below present sea levels.

Successive glacial and inter-glacial periods during this time caused intense frost and precipitation that led to the rapid transport of surface material to coastal areas. As water levels rose during the post-Wisconsin period of the Holocene, tidal wetlands rapidly expanded into the inundated river systems (Odum et al. 1984). Consequently, as sea levels rose, fluvial valleys were drowned along both coastlines. Along the Gulf coast, the process of sea level inundation within incised fluvial valleys may have been more pronounced to the east of Mobile Bay than to the west (Smith 1988). It might be expected that alluviation would become more pronounced above the zone of oceanic inundation and result in wider alluvial deposition zones there as sea levels rose. However, relatively steep gradients of the Gulf coastal plain may have prevented broad alluvial floodplains from forming with some exceptions such as that of the Mobile-Tensas system.

Along the southeast Atlantic coast most tidal wetlands were formed recently and under the same general process. Unlike some tidal wetlands along the Gulf of Mexico that are declining in size (Louisiana's deltaic wetlands), others along the Atlantic coast are thought to be still expanding. It has been shown that human agricultural activity over the past 300 years has extended the spatial extent of tidal wetlands along the Atlantic coast by increasing the amount of sediment deposition in coastal areas (Frommer 1980). Data collected by Orson et al. (1992) showed that sediment accumulation rates in a Delaware River estuary marsh were 0.04 cm yr^{-1} prior to European colonization. After two centuries of intensive land management (including diked wetlands), tidal marshes became established again and began accumulating sediment at 0.97 cm yr^{-1}. In other estuaries, recent sediment accumulation has likely been tempered by better soil conservation practices (Craft and Casey 2000), the construction of dams (Bednarek 2001), and the channelization of riverbeds that results in the transport of sediments beyond the estuary (Day et al. 2005).

Local relief is lower along the southeast Atlantic coastal plain and the Gulf coast west of Mississippi (<30 m) than along the eastern Gulf (30-90 m) (Walker and Coleman 1987). Differences in relief have probably influenced the width of fluvial valleys in these areas, and according to Walker and Coleman (1987) there are differences between the two coasts in terms of vegetation and landforms. The southeast Atlantic coast can be characterized by salt marshes and tidal creeks whereas the Gulf coast is typified by narrow salt marshes and broad brackish marshes. The southeast Atlantic marshes do not extend between Jacksonville and St. Augustine, Florida, and are replaced there by tidal mangrove swamps. There is no alluviation associated with that coastal zone (Sherman 2005). Walker and Coleman further note that the marsh communities along the southeast Atlantic are more stable than their Gulf coast counterparts due to more rapid subsidence along the Gulf.

One of the most unique ecosystems along the southeast Atlantic coast is that of the St. John's River which runs from south to north on the eastern side of Florida. There is no alluviation associated with the St. John's since it has a very low gradient from its inception to its mouth (Smock et al. 2005). In addition, 59% of the St. John's length is tidally influenced, in contrast to 4% and 12% for the Apalachicola and Suwannee respectively (McPherson and Hammett 1991). Unfortunately, detailed soil information is not available for the St. John's River floodplains.

As alluvial depositional areas such as deltas extend into deeper water, rates of alluviation are slowed and subsidence of the deltaic plain may occur. As a result, stream courses may wander within the plain to areas where gradients are steeper and flow is less impeded (Shirley and Ragsdale 1966). For instance in the Lower Mississippi River Delta, this is the reason that the river course would shift to the southwest without anthropogenic intervention. Rivers of the southeast Atlantic coastal plain may be migrating in a southerly direction as evidenced by high bluffs and wide floodplains on the southern and northern sides of rivers, respectively (Brinson 1990).

3.2.2. Formation

Relative to other tidal wetlands, freshwater forests generally occur at the very fringe of tidal influence. Throughout the coastal plain, the most extensive forests seem to occur along larger rivers that are influenced by a wide tidal range (Duberstein 2004). The salinity of tidewaters is also important, and Rheinhardt (1992) reported that for tidal freshwater forested wetlands along the Pamunkey River, salinity rarely rises above 0.05 ppt except at the extreme downriver extent of the swamp. Extending upriver from tidal salt marshes, freshwater marshes often occur (sometimes extensively) between the salt marshes and tidal forests. Freshwater tidal forested wetlands such as those at the Suwannee River (Light et al. 2002) and Pamunkey River (Rheinhardt 1992) have been found to occur as a mosaic of forest and marsh and the cover of understory vegetation can vary significantly based on canopy closure.

One commonly reported feature of tidal freshwater forested wetlands is the mosaic of hummocks and hollows that occurs along the forest floor. The hummocks are elevated sections or “islands” above the forest floor that consist of a network of interwoven tree roots or moss-covered remnants of tree trunks or large branches (Peterson and Baldwin 2004). The reported heights of these hummocks relative to the surface of the hollows can range between 15-20 cm, and often correspond to the mean water level of the swamp (Rheinhardt and Hershner 1992). Trees in freshwater tidal forested wetlands are frequently limited to those contributing to hummock formations with herbaceous or no vegetation in the hollows. There is also evidence that hydrology related to tidal range and flood duration may influence the relative proportions of hummocks and hollows. Rheinhardt (1992) reported that an ash/blackgum (*Fraxinus*/*Nyssa*) community located along a lower reach of the Pamunkey River was 65.5% hollow compared to only 25.0% for a maple/sweetgum (*Acer*/*Liquidambar*) community ~5 km upriver (where flood durations were shorter). We are aware of no study that has extensively compared edaphic conditions between hummocks and hollows. It is expected that hollows would have more prolonged anaerobic periods and consequently contain higher levels of organic matter; however, Rheinhardt (1992) found that concentrations between hummocks and hollows were comparable in tidal swamps along the Pamunkey River. Nevertheless, given the differences in elevation and hydrology, substantial differences in the soil biogeochemistry of hummocks and hollows are likely.

Given the importance of tidal range to these wetlands, it should be emphasized that the process of sea-level rise is continuing, with rates estimated between 0.1–0.2 cm yr^{-1} worldwide (Gornitz et al. 1982; Warrick and Oerlemans 1990). Rising sea levels may eventually influence all tidal wetlands but some coastal forests may be particularly susceptible (Chapter 1). Rates of sea-level rise along the Gulf of Mexico may be higher than average as indicated by estimates of 1.0 cm yr^{-1} along the Mississippi River Delta (Reed 2002) and 0.23 cm yr^{-1} near Pensacola, Florida (Penland et al. 1987). The impact of increased water levels along coastal river forests may be amplified by altered sediment loads and hydrology from the construction of 20th century dams, agriculture, and development (Walker and Coleman 1987). Consequently, the geomorphic relationship among coastal forest systems, humans, and marine environments remains highly dynamic.

3.3. Soil taxonomic classification

Across the southeast, USDA-NRCS county soil surveys indicate that the following soil orders may be found near river outlets: entisols, inceptisols, histosols, alfisols, mollisols, and ultisols. The presence of some of these orders on a floodplain may seem to conflict with expectations in terms of geomorphic context. The soil orders that might be anticipated to occur on floodplains include those associated with active depositional positions (e.g., entisols and inceptisols). Histosols would also be expected in areas where surface saturation persists for long periods and, as a result, decomposition is inhibited. Entisols and inceptisols exhibit less profile development (i.e., fewer horizons) than some other orders and are said to be young in a geological sense since they are morphologically homogenous in a vertical direction. Conversely, older soils with well developed profiles (e.g., ultisols, alfisols, and mollisols) are usually associated with very stable geomorphic positions such as uplands, as opposed to active alluviation zones. Consequently, the presence of the latter orders on floodplains contributes greatly to soil diversity there. In general, USDA-NRCS surveys indicate that entisols and inceptisols are more common on floodplains along the lower reaches of Gulf coast rivers while ultisols, alfisols, and mollisols, along with entisols and inceptisols, are mapped on the Southeast Atlantic coast. While it is tempting to seek geomorphological reasons for the apparent contrast between the Gulf and Atlantic coasts, this probably indicates nothing more than variation in the level of detail provided in soil mapping among various locations.

As noted, soil maps of lower reaches of rivers emerging along the Atlantic coast often exhibit greater diversity than those of the Gulf coast (e.g., the Mobile and Apalachicola Rivers). Although soil complexes along coastal reaches of the Satilla River (Rigdon and Green 1980) are similar to those described for the Mobile and Apalachicola Rivers, the lower reaches of the Altamaha River (Rigdon and Green 1980) are mapped as aqualfs and aquults (wet soils with a high degree of profile development) and aquents (wet soils with little profile development). Along the Edisto River (Stuck 1982), aquents and aquepts are mapped along the lower reaches of the river; however, udults and aquults occur closer to the coast. The soils of the forested reaches along the Edisto, nearest the coast, are mapped as wet mollisols. Sulfaquents dominate as the forest transitions to marsh nearer the coast. Similarly, although aquents are mapped on the lowest forested reaches of the Greater Pee Dee River (Stuckey 1982), aquults and udults dominate its floodplains a few km upriver from the coast. Soils studied in the tidal forests of the White Oak River estuary (Megonigal and

Schlesinger 2002) were classified as thermic typic medisaprists (Barnhill 1981, 1992) and tidal forests along the Pamunkey River (Rheinhardt 1992) were mapped as ferric mediasaprists of the Mattan series (Hodges et al. 1988).

Along the Gulf coast, soils found along the lower reaches of rivers such as the Mobile (McBride and Burgess 1964; Hickman and Owens 1980) and Apalachicola (Sasser et al. 1994) are predominantly mapped as aquents, aquepts, and saprists. These represent wet mineral (and, in the case of saprists, organic) soils with little horizonation. Further from river channels, udults and aquults are found. As vegetation changes from forest to marsh, soils near the river mouth are typically of the Bohicket series (typic sulfaquent), a wet entisol with high organic matter content.

Published research conducted in freshwater tidal wetlands has generally focused on the forest interior. However, studies that have been more inclusive of the entire tidal forest range have confirmed a wide range of soil characteristics (Doumlele et al. 1984; Light et al. 2002). Light et al. (2002) evaluated 96 soil profiles along an elevation and distance-to-river-mouth gradient for coastal forests along the Suwannee River (including non-tidal floodplains). After analyzing a subset of 67 profiles, they found soils were highly variable and represented 7 orders and 18 taxonomic sub-groups. Visually, soils ranged from upland in appearance to deep mucks. In the upper tidal reaches of the Suwannee River (22–37 km from the river mouth), forest soils were more diverse and included entisols, mollisols, inceptisols and histosols. This portion of the forest was considered to be only partially influenced by tides and the occurrence of highly organic histosols only occurred at the lowest elevations. In the lower tidal swamp reach of the river (<22 km from the river mouth) where tides and salinity are more influential, histosols became much more prevalent and as the elevation of sampled soils decreased there was a general increase in the overall depth of the surficial muck layer (in some cases >2 m). Most forest soils along the lower tidal reach of the Suwannee River were classified as typic or terric haplosaprists (Light et al. 2002). Studying soils in the tidal freshwater forests of the Apalachicola River, Coultas (1984) classified three out of five soil pedons (collected to depths >125 cm) as typic sulfihemists based on existing soil concentrations of sulfur $>0.75\%$ (although surface soil concentrations were typically much lower).

In the southeastern United States, higher proportions of the watersheds of blackwater rivers lie in the coastal plain compared to those of redwater systems which are primarily associated with the piedmont. Blackwater rivers are also characterized by higher organic loads derived from extensive swamps and floodplain forests that dominate the coastal plain landscape

(Schilling and Lockaby 2005). In contrast, redwater rivers often have high grades which impart a greater capacity for carrying sediment (and, consequently, nutrient) loads (Lockaby et al. in press). As a result of the geomorphic differences between the system types, it might be expected that a comparison of redwater vs. blackwater floodplain soils would yield morphological distinctions. However, based on available mapping information, there is little evidence of a taxonomic differential between soils near the outlets of blackwater vs. redwater rivers in the southeast United States. Nevertheless, there are clear distinctions between floodplains of these two river types in terms of biogeochemistry (Schilling and Lockaby 2006).

3.4. Biogeochemistry

3.4.1. Soil organic matter

Soils of freshwater tidal wetlands tend to be highly organic (Table 3.1). In the southeast, organic matter in tidal wetlands is usually highly decomposed muck (saprist soils) as opposed to some coastal wetlands in northern regions where organic matter can often accumulate as fibrist peat (Odum et al. 1984). In terms of concentration, Wharton et al. (1982) listed tidal freshwater swamps among floodplain communities in the southeast as having the highest concentrations of soil organic matter. The high level of organic matter is the result of suppressed decomposition under anaerobic soil conditions and moderate to high autochthonous production within the forest (Wharton et al. 1982). As with most wetlands, the accumulation of organic matter in tidal freshwater swamps is closely linked to hydrology and several authors have reported deep mucky soils occurring in the lowest elevations of the forest (Doumlele 1984; Light et al. 2002). Comparing different forest communities along the Pamunkey River, Rheinhardt (1992) found that an ash-blackgum community had significantly higher organic matter (40.5%) than a maple-sweetgum community (27.8%) located further upriver, and attributed this difference to longer flood durations in the ash-blackgum community.

The physiographic and geomorphic setting of the river also plays a role in soil organic matter content. Estimates from the literature indicate that freshwater tidal forests in blackwater river swamps have higher organic matter concentrations (average 46%) than redwater river swamps (average 31%) (Table 3.1, although see Rheinhardt [1992] and Duberstein [2004]). Reported soil organic matter concentrations often reflect the most highly inundated sections of the forest, however it should be noted that there is

Table 3.1. Soil characteristics and dominant tree communities for freshwater tidal forests in the southeastern United States. Multiple river listings are provided for different sampling sites.

River, Location	River-type	Soil depth (cm)	pH	Organic Matter (%)	Cond. (mmhos /cm)	Textural components clay	Textural components silt	Textural components sand	Tree community	Source
Apalachicola, FL	redwater	8 - 30	5.3	15		46	36	18	Tupelo gum, cypress	Wharton et al. 1982
Apalachicola, FL	redwater	0 - 20	5.6	33	0.1				Mixed hardwood, cypress	Coultas 1984
Apalachicola, FL	redwater	0 - 30	5.5	28	0.9	85	14	1	Mixed hardwood, cypress	Coultas 1984
Apalachicola, FL	redwater	2 - 40	4.9	9	0.2	77	23	0	Mixed hardwood, cypress	Coultas 1984
Apalachicola, FL	redwater	2 - 32	5.8	14	0.1	78	22	0	Mixed hardwood, cypress	Coultas 1984
Apalachicola, FL	redwater	0 - 15	5.7	25	0.2	73	22	5	Mixed hardwood, cypress	Coultas 1984
Choctawhatchee, FL	redwater	8 - 30	5.5	36					Swamp tupelo, bay	Wharton et al. 1982
Escambia, FL	redwater	8 - 30	5.4	15		28	17	55	Sweet bay, cypress, white cedar	Wharton et al. 1982
Pamunkey, VA	redwater	0 - 15	4.8	41					Ash-blackgum	Rheinhardt 1992
Pamunkey, VA	redwater	0 - 15	4.9	25					Maple-sweetgum	Rheinhardt 1992
Patuxent, MD	redwater	0 - 10		44					Shrub	Verhoeven et al. 2001
Savannah, GA	redwater	0 - 13	5.6	77	6.7				Swamp tupelo, tag alder, shrub	Duberstein 2004
Savannah, GA	redwater	0 - 13	5.7	46	3.0				Water oak, swamp bay, tupelo	Duberstein 2004
Aucilla, FL	blackwater	8 - 30	6.3	19		14	13	73	Mixed	Wharton et al. 1982
Sopchoppy, FL	blackwater	8 - 30	5.7	77					Sweet bay, swamp tupelo	Wharton et al. 1982
St. Marks, FL	blackwater	8 - 30	6.2	45		6	21	73	Swamp tupelo, sweet bay	Wharton et al. 1982
Suwannee, FL	blackwater	8 - 30	6.0	40		17	23	60	Swamp tupelo, cypress-ash	Wharton et al. 1982
Suwannee, FL	blackwater	8 - 30	6.4	41		20	30	50	Swamp tupelo, cypress-ash	Wharton et al. 1982
Yellow, FL	blackwater	8 - 30	5.2	53					Sweet bay, white cedar	Wharton et al. 1982

significant spatial variability associated with forest geomorphology. Ridges and other high elevations within the forest (less often reported) will contain more mineral soil conditions and can make generalizations difficult to interpret.

3.4.2. Methanogenesis

Studies examining the redox potential of soils in freshwater tidal wetlands have reported levels capable of supporting methanogenesis (Megonigal and Schlesinger 2002; Hackney et al. 2002, 2005). The emission of methane from wetlands in general is a substantial component of the atmospheric methane budget, and Whalen (2005) has reported that emission rates of 100 mg CH_4 m^{-2} d^{-1} are common for wetland ecosystems. Because of the carbon reserves in freshwater tidal swamps, these wetlands have the potential to produce substantial quantities of methane. A series of studies on methane emissions were conducted at the White Oak River estuary along the east coast of North Carolina. Kelley et al. (1995) found large seasonal changes in methane production in freshwater tidal swamps that did not occur in permanently inundated creek sites nearby. Methane production was mediated by bacterial oxidation at low tide and at the oxygenated sediment surface. While very little CH_4 was produced during the winter season, large fluxes (up to 1000 mg CH_4 m^{-2} d^{-1}) were measured during the growing season. Nearby, non-vegetated sites that were permanently submerged had only moderate increases in CH_4 flux during the summer. Overall methane flux was higher along the vegetated river banks (tidal areas) than in the permanently inundated sites.

Within tidal freshwater swamps, there is evidence that CH_4 emissions have high spatial and temporal variability. Temporal variability of emissions can be attributed to changes in water level associated with tide events (daily fluctuation) and seasonal variability inherent to temperate climate conditions. Kelley et al. (1995) found that the highest daily CH_4 emission rates occurred when water levels coincided with the soil surface (Figure 3.1). They surmised that when tidewaters were below the sediment surface, the aerobic layer at the sediment-water interface increased in area and greater CH_4 oxidation occurred. Alternately, when tidewaters flowed into the swamp and standing water was present, CH_4 emissions declined because of a diffusion barrier caused by the standing water. Furthermore, inflowing tidewaters may provide an aerobic environment for methane oxidation resulting in a reduction of net emissions. Incoming water can also impede methane emission by inundating plant surfaces that normally convey belowground gases to the atmosphere. Many macrophytes provide a

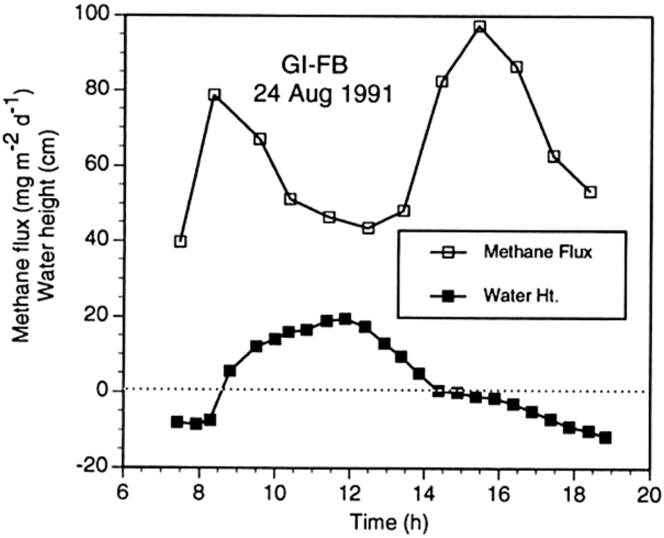

Fig. 3.1. Measured fluxes using static chamber techniques and water level over a 12-h tidal cycle at a tidal swamp in the White Oak River Estuary. Dotted line represents the sediment surface (water height = 0 cm) (Kelley et al. 1995 with permission; copyright (1995) by the American Society of Limnology and Oceanography, Inc.).

conduit for gas exchange between the root zone and the atmosphere through aerenchyma tissue. When plant surfaces become inundated, water can again act as a diffusion barrier for methane emission (Chanton et al. 1992).

Interestingly, it appears that the tidal hydrology of freshwater tidal swamps may suppress CH_4 emissions compared to other forested wetlands. Methane flux rates reported by both Kelley et al. (1995) and Megonigal and Schlesinger (2002) were low compared to ranges reported for swamps in Virginia (83-155 mg CH_4 m^{-2} d^{-1}, Wilson et al. 1989) and Louisiana (146-912 mg CH_4 m^{-2} d^{-1}, Alford et al. 1997) during the growing season. Very low rates were reported by Megonigal and Schlesinger (2002) who estimated a peak monthly mean flux of only 17 mg CH_4 m^{-2} d^{-1} at an upstream tidal forest along the White Oak River although there was considerable spatial variability within sites (one plot had measurements >200 mg CH_4 m^{-2} d^{-1}).

Annual variability of methane production has been reported by Hackney et al. (2005) for tidal swamps and marshes along the Cape Fear River. Using a series of monitoring stations, they have monitored shifts between methanogenesis and sulfate reduction as the primary mode of C mineralization since 2000 (Hackney et al. 2002, 2005). Oceanic saltwater contains higher levels of sulfate so that when tidal ranges expand, microbes that utilize sulfates as an electron acceptor can become prevalent and outcompete methanogens (Mitsch and Gosselink 2000). Examining conditions four times a year, they have found irregular shifts between the two biochemical processes corresponding with upriver influence on the tidal range of oceanic saltwater. When tidal shifts have occurred (as a result of reduced upriver flows during droughts) and the range of saltwater influence has expanded to previously freshwater reaches, they have found that the dominant microbial community readily shifts from methanogenic to sulfate reducing. Other hydrologic differences attributed to tidal range and microtopography can have substantial effects on soil biogeochemistry. Megonigal and Schlesinger (2002) found that for soils in a site further upriver (where soils were exposed from flooding 60% of the year compared to 40% at a site downriver), the hydrologic difference elicited a 27% higher CH_4 oxidation capacity in upper site soils, a vertically deeper occurrence of peak CH_4 production in the soil profile, and an herbaceous stratum that was less flood tolerant.

There is also evidence that the presence and/or quantity of macrophytes in a tidal freshwater swamp may regulate CH_4 emissions. Although flooding may impede some gas emissions via plant tissue to the atmosphere (Chanton et al. 1992), it is likely that these plants still represent an important conduit for CH_4 transmission. Kelley et al. (1995) found discrepancies between two techniques used to estimate CH_4 flux (diffusion models using dissolved methane concentrations v. static chambers) and suggested that the higher rates detected by chambers was attributed to plant-mediated gas transport. Likewise, Megonigal and Schlesinger (2002) explained that the sparse herbaceous cover in their sites (due to dense canopy cover) may have attributed to the low CH_4 emissions detected in their study.

3.4.3. Salinity

Salinity is another critical factor controlling the range and distribution of tidal freshwater swamps. Most of these swamps occur where water salinities are normally below 0.5 ppt however they are exposed to occasional tidal surges that can induce saline conditions in normally freshwater areas (Simpson et al. 1983; Peterson and Baldwin 2004). These surges are caused by storms/hurricanes, tide pushing winds, and upstream droughts that reduce freshwater inputs and increase the tidal extent of saltwater into the forest. Because the frequency and intensity of tidal surges varies over time, it is likely that soil salinity in freshwater wetlands also varies year-to-year. While all tidal swamps are exposed to periodic surges of saltwater, long-term decreases in upstream freshwater flow have the potential to cause more permanent changes to the ecology of the forest. Along the Cape Fear and Suwannee River, researchers have investigated the potential effects of human river management on existing tidal ranges, upriver soil salinities, and the ecology of existing freshwater tidal wetlands (Hackney et al. [2002] and Light et al. [2002], respectively).

In tidal freshwater forests along the Suwannee River (Light et al. 2002) and Apalachicola River (Coultas 1984), the highest salinities within the soil profiles were generally observed below the surface layer and root zone. Along the Apalachicola, tidal forest soils had surface conductivities <1.0 mmhos cm^{-1} (Table 3.1) while subsurface conductivity ranged between 2.9 and 7.7 mmhos cm^{-1}. It is likely that the regular inflow of freshwater effectively flushes occasional salt deposition from the surface. Light et al. (2002) detected a general trend of increasing salinity in Suwannee River tidal forests as transects got closer to the river mouth. In the lower reaches of the tidal forest (those closest to the Gulf) surface soil salinities were often >4 mmhos cm^{-1} and, unlike soils in the upper tidal reaches, the subsurface soils tended to be as saline (or more) than at the surface reflecting the greater exposure to tidal saltwater. It was also noted that swamps in the upper tidal reach that were isolated from regular surface flows but within reach of storm surges, there was not an accumulation of salts in the soils as would be expected. This was attributed to the prevalence of groundwater flushing from a shallow limestone aquifer that is close to the surface.

Soils are considered to be saline if soil conductivity levels exceed 4 mmhos cm^{-1}; however, impacts of salinity on plant growth and species occurrence occur at lower levels. In the Suwannee River floodplains, water tupelo (*Nyssa aquatica* L.) apparently does not occur in lower reaches due to high salinity levels there (> 4 mmhos cm^{-1}). The impacts of high salinity on tree growth and species occurrence are associated with moisture stress imposed by hydrophilic salts near roots. In the cypress/tupelo forests of the lower Mobile Delta, across plots with similar tree ages and past management histories, heights of dominant trees (a well established index of site quality) were considerably lower at soil conductivity levels above approximately 1.75 mmhos cm^{-1} (Figure 3.2). Mobile Delta tree heights averaged 22.3 m on sites with soil conductivities below 1.75 mmhos cm^{-1} and 17.4 m on soils above that conductivity level. Similarly, basal areas averaged 18.7 and 37.1 m^2 ha^{-1} respectively above and below 1.75 mmhos cm^{-1} (Figure 3.3). Low heights and basal areas in the Mobile Delta were associated with average exchangeable Na levels of 817 mg kg^{-1} (upper meter)

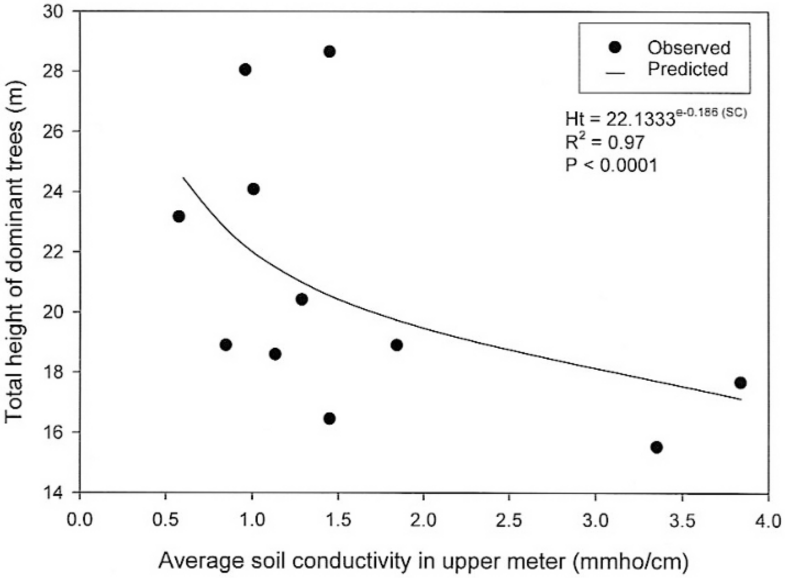

Fig. 3.2. Comparison of total height of dominant trees and soil conductivity in cypress-tupelo stands of the Mobile River Delta (Jim DeWit – unpublished data).

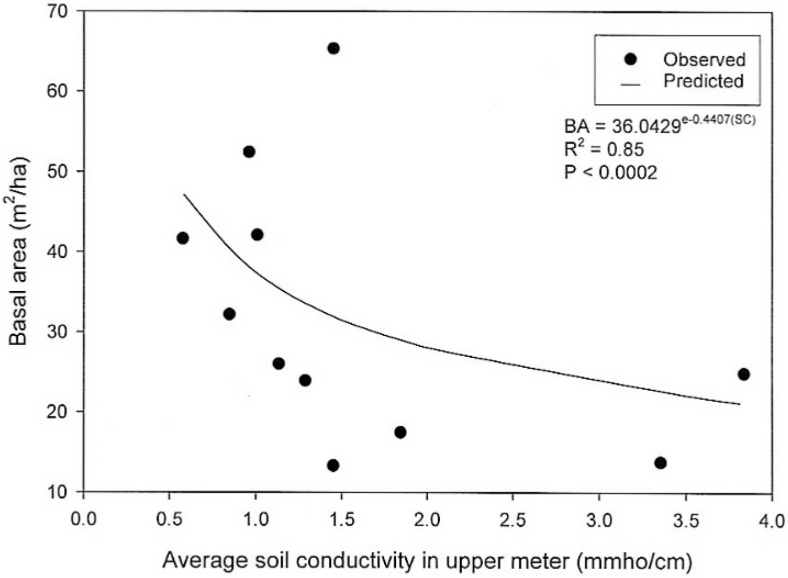

Fig. 3.3. Comparison of total basal area and soil conductivity in cypress-tupelo stands of the Mobile River Delta (Jim DeWit – unpublished data).

while greater heights and basal areas occurred on soils averaging 238 mg kg^{-1} Na. The low basal areas associated with higher conductivity levels in the Mobile Delta are in agreement with the low standing crops of aboveground biomass previously noted in fringe wetland forests such as mangroves (Lugo 1990). In the latter forests, a high root to shoot ratio was often observed.

As would be expected, soil conductivity levels and forest productivity within the Mobile Delta generally rose and fell respectively in a southerly direction toward Mobile Bay (Jim DeWit, unpublished data). However, there is also significant variation locally in the southern portion of the Mobile Delta. DeWit (personal communication) indicates that soils located on natural berms adjacent to the tidal creeks tend to be higher in Na and conductivity and lower in productivity compared to soils in low areas behind the berms. The soils behind the berms often support highly productive tupelo gum stands and can be classed as thapto-histic entisols (entisols that have formed over a buried organic soil). Also, soils adjacent to streams that form within the lower delta are apparently more subject to tidal influences and tend toward higher salinity compared to counterparts adjacent to streams with headwaters further north. Forest productivity is lower on soils adjacent to lower-delta streams vs. soils near streams with watersheds that extend north of the tidal zone.

There is no indication from the limited amount of data available that decomposition in tidal forests is slowed due to salinity or other factors. Actually, decomposition in these systems may be more rapid than elsewhere (see discussion in 3.4.4), an indication of the effectiveness of halophiles and other microbial communities that are adapted to saline environments. Similarly, in the data gathered by DeWit in the Mobile River Delta, no relationship was apparent between soil conductivity and organic matter concentrations in the upper 1 m of soil. This suggests that salinity does not affect soil organic matter accumulation there and that the presence of histosols and near-histosols in coastal forests may be due primarily to wetness as it is in freshwater wetlands.

3.4.4. Soil chemistry and nutrient cycling

Only a handful of studies have reported nutrient concentrations for soils in tidal freshwater swamps in the Southeast (Table 3.2). Compared to other floodplain soils, the concentrations reported in tidal freshwater swamps are much higher. Wharton et al. (1982) explained that one reason for high levels of Ca and Mg may be that several of the blackwater river forests examined have groundwater inputs from spring-fed streams that pass through a

Table 3.2. Soil nutrient concentrations for freshwater tidal forests along different rivers in the southeastern United States. Multiple river listings are provided for different sampling sites.

			Macronutrients							
River, Location	River-type	Soil depth (cm)	S (%)	N (%)	P (ppm)	Ca (ppm)	K (ppm)	Mg (ppm)	Na (ppm)	Source
Apalachicola, FL	redwater	8 - 30			8.0	1676	92	180	53	Wharton et al. 1982
Apalachicola, FL	redwater	0 - 20	3.22	0.76						Coultas 1984
Apalachicola, FL	redwater	0 - 30		0.78						Coultas 1984
Apalachicola, FL	redwater	2 - 40	0.04	0.07						Coultas 1984
Apalachicola, FL	redwater	2 - 32	0.27	0.47						Coultas 1984
Apalachicola, FL	redwater	0 - 15	0.15	0.74						Coultas 1984
Choctawhatchee, FL	redwater	8 - 30			18.0	1213	164	616	664	Wharton et al. 1982
Escambia, FL	redwater	8 - 30			15.2	776	116	456	224	Wharton et al. 1982
Pamunkey , VA	redwater	0 - 15			10.8	806	68			Rheinhardt 1992
Pamunkey, VA	redwater	0 - 15			8.0	578	67			Rheinhardt 1992
Patuxent, MD	redwater	0 - 10		1.21						Verhoeven et al. 2001
Savannah, GA	redwater	0 - 13			62.4	6804	284	1569	889	Duberstein 2004
Savannah, GA	redwater	0 - 13			60.3	3901	239	514	276	Duberstein 2004
Aucilla, FL	blackwater	8 - 30			12.0	2852	35	400	80	Wharton et al. 1982
Sopchoppy, FL	blackwater	8 - 30			29.0	1324	36	628	183	Wharton et al. 1982
White Oak, NC	blackwater	0 - 27		1.20 – 1.74[a]						Kelley 1993
White Oak, NC	blackwater	0 - 27		1.36 – 1.92[a]						Kelley 1993
St. Marks, FL	blackwater	8 - 30			41.2	2462	97	556	145	Wharton et al. 1982
Suwannee, FL	blackwater	8 - 30			55.0	5784	323	870	1011	Wharton et al. 1982
Suwannee, FL	blackwater	8 - 30			46.0	5488	78	499	65	Wharton et al. 1982
Yellow, FL	blackwater	8 - 30			14.4	1404	144	588	196	Wharton et al. 1982

[a] %N range based on soils analyzed at 3-cm depth increment.

limestone substratum. High levels of these same nutrients were also reported for several of the alluvial rivers listed in (Table 3.2). The high concentrations of many listed nutrients may be a reflection of high soil organic matter with nutrients sorbed to or bound within soil organic matter. More research on the form of soil nutrients in these forest soils is needed.

Wharton et al. (1982) and Schilling and Lockaby (2005) have suggested that soils associated with redwater systems are more fertile in terms of base cations than those of blackwater counterparts. These potential distinctions may be due to more active alluviation in the redwater systems. Stanturf and Schoenholtz (1998) also note that alluviation of coastal reaches of redwater river floodplains is usually associated with clayey, Piedmont sediments whereas any alluviation along blackwater rivers is derived from coastal sediments that originate locally. Analyzing tidal swamp soils along the Apalachicola River (a redwater river), Coultas (1984) found the concentration of fine sediment (silts and clay) at or near 100% for most of the pedons examined and Wharton et al. (1982) reported concentrations as high as 82%. In contrast, the concentration of sand in blackwater tidal swamps sampled by Wharton et al. (1982) ranged from 50 to 73%.

Soils in tidal freshwater swamps tend to be acidic with a reported pH range between 4.8 and 6.4 (Table 3.1). This is slightly lower than what is typically reported for alluvial non-tidal swamps (pH of 6 to 7) and comparable to the range reported for cypress domes and other perched-basin swamps with high concentrations of organic matter (Mitsch and Gosselink 2000). In swamps with high organic matter content, soil pH generally reflects the high amount organic acids generated. Our limited literature review (Table 3.1) found that the pH range for redwater swamps (4.8–5.8) is slightly lower than blackwater swamps (5.2–6.4), which is somewhat surprising given the higher organic matter concentrations in the latter. However, pH in blackwater swamps may be moderated somewhat by spring-fed groundwater that commonly emerges from a limerock substratum along the Gulf coast of Florida (Clewell 1991).

Research is also needed to identify important nutrient cycling processes in tidal freshwater swamps. Because prolonged flooding and highly reduced soils are common, denitrification is likely the most important mechanism for N removal from these wetlands. Lockaby and Walbridge (1998) found that denitrification rates reported for southern forested wetlands ranged from 0.5 to 350 kg ha^{-1} y^{-1}. We would expect rates for tidal freshwater swamps to be on the high end of this scale based on the high input of N from rivers.

Based on research related to tidal fluctuations, forest litter accumulation and decomposition in these swamps are likely controlled by tidal fluctuations and movement within the forest. Twilley (1982) described decomposition in selected mangrove systems as rapid with rates being directly related to C:N ratios. Decay rates (k) for mangrove forests in Florida averaged 2.87 (i.e., a turnover time of 0.34 years) (Lugo 1990). Lugo noted that forest floor mass was highly variable in these systems due to fluctuations in water movement. Often, high accumulations could be found at the tidal fringe although very little forest floor mass might accrue in other areas due to the flushing action of the tide. In the latter areas, rapid turnover rates may be primarily due to waterborne export rather than decomposition (Lugo 1990). Similarly, Twilley (1998) states that turnover rates in mangrove systems rise as tidal inundations increase and are directly related to the ecological setting (i.e., rates rank riverine > fringe > basin) and that litter mass loss rates may be greatly accelerated by the presence of crabs. Decomposition rates reported in tidal forests along the South Carolina coast (Ozalp 2005) were rapid (e.g., average k = 1.8, turnover time = 0.55 years). The k rates of Twilley, Lugo, and Ozalp may be compared to an average k from riverine forests of 1.01 (Lockaby and Walbridge 1998).

Plant uptake has been cited as a major pathway for nutrient cycling in tidal freshwater marshes (Simpson et al. 1983) and forested floodplains in the Southeast (Lockaby and Walbridge 1998). In freshwater tidal wetlands, nitrogen and phosphorus are imported into these wetlands largely as inorganic ions and delivered at various degrees by tidal waters, river flooding, groundwater seepage, and precipitation. During the growing season, these nutrients are rapidly assimilated into plant material. After the growing season and plant senescence, much of the nutrients associated with plant detritus are exported from the marsh through tidal flushing. Simpson et al. (1983) reported up to 80 percent of the nitrogen and phosphorus associated with plant litter was lost within 2 months of senescence in a Delaware River tidal freshwater marsh. It is uncertain how much detritus export occurs in tidal swamps but there are circumstances to suggest that the dynamics may be different than marshes. Because of their generally closer proximity to the coast, tidal energies may be higher through marsh systems than the upper tidal reaches occupied by forests. Also, the structure (trees) and microtopography of hummocks and hollows may further dampen tidal energies that would potentially export detritus (although initially, senescent marsh stems would dampen tidal energies as well).

Like N, there is no known account regarding the import-export dynamics of P in tidal freshwater swamps. In acid soils such as those in tidal freshwater swamps, the adsorption of phosphates to aluminum and iron has been identified as a key P retention mechanism (Lockaby and Walbridge 1998). In alluvial rivers where fine sediments occur, we expect that the deposition of P adsorbed to sediment is another key retention process. Although at tidal swamps along the Pamunkey River, Rheinhardt and Hershner (1992) observed that most sediment accumulation seemed to occur along the river edge at low-lying levees.

Studies on nutrient cycling efficiencies in tidal forests are limited to mangrove systems. In his 1990 review, Lugo used mangrove (*Rhizophora* spp.) data to compare nutrient cycling efficiencies (ratio of litterfall mass to nutrients in litterfall, Vitousek 1984) among various mangrove systems. Comparisons indicated that efficiencies of nitrogen cycling were high relative to other forest systems, those of calcium were similar to those of upland forests, and those of phosphorus were highly variable.

3.5. Research needs

It is apparent that relatively little information exists on the soils and biogeochemistry of tidal freshwater forests compared to other wetland types. While saltwater and freshwater tidal marshes have been studied to a much greater extent, we know very little about the regulation of biogeochemical and energy exchange between the sea and coastal forests. The few soil studies in coastal swamps have shown that these areas are highly diverse with variable conditions among and within swamps. A limited number of studies have focused on evaluating potential forest impacts from human induced changes in hydrology (Hackney et al. 2002; Light et al. 2002). While the latter is a very critical topic, information is lacking on fundamental biogeochemical processes such as decomposition, mineralization, and nutrient requirements for these systems. Consequently, a major imbalance is evident when the societal importance of these ecotones (in particular, considering the history of catastrophic events along southeastern coastlines) is compared to our knowledge of how they function.

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