VEGETATION DYNAMICS OF A TIDAL FRESHWATER MARSH: LONG-TERM AND INTER-ANNUAL VARIABILITY AND THEIR RELATION TO SALINITY

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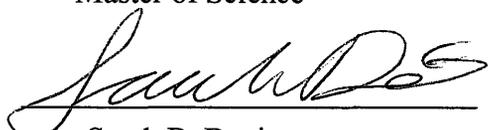Master of Science

By

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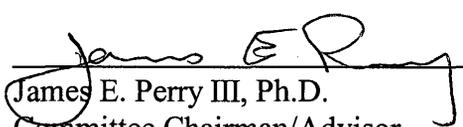
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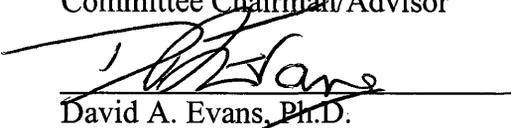This thesis is submitted in partial fulfillment of
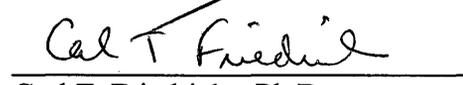
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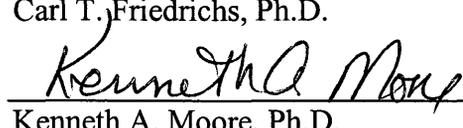
Master of Science

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ABSTRACT

Tidal freshwater marshes (TFMs) support diverse plant communities and provide valuable habitat for commercially important fish and wildfowl populations. Relative sea level rise (RSLR) may cause declines in this diversity due to increased salinity and altered hydroperiod, but little research has focused on the community level response of TFM plants to increased salinity. Sweet Hall Marsh, a component of the Chesapeake Bay National Estuarine Research Reserve in Virginia, is a model system for studying long-term variation in vegetation dynamics of a TFM. Vegetation studies completed in 1974 and 1987 provided historical data. The second of the two studies quantitatively documented a change in vegetation from the first study, suggesting a shift towards an oligohaline marsh. In 2003, previous study areas were re-sampled using the same methods. Percent cover, stem density, and frequency were measured at each of 76 1-m^2 plots throughout the growing season. Dominant species, based on aboveground biomass collected in August 2003, were *Zizania aquatica* (266g/ m^2) and *Peltandra virginica* (71.3g/ m^2). Comparison across the three studies of importance values of perennial species found an increase in salt-tolerant perennials (including *S. alterniflora* and *E. palustris*) and a decrease in freshwater perennials (including *P. virginica* and *C. stricta*). Highly variable river salinities suggest that salinity stress associated with storm events and years with low freshwater flow might be contributing to the plant community change. A follow-up study in July 2004 showed a reversal in the vegetation trend; the relative importance of salt-tolerant species had decreased while the importance of some freshwater species increased from July 2003 to July 2004. Implications are that both the perennial and annual components of the vegetation community of this TFM may be highly variable from year to year. Therefore, the increase in salt-tolerant perennials observed in 2003 may have been a response to short-term climatic factors rather than long-term salinity increases. Results support the hypothesis that TFMs may remain vegetated with increasing salinities associated with rising sea levels and highlight the need for yearly monitoring of vegetation community.

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INTRODUCTION

In post-European settlement of America, wetlands were generally viewed as a waste of valuable land and as refuges for mosquitoes, snakes, and criminals. These opinions resulted in large-scale efforts to dike, drain, and fill marshes and swamps across the country (Dahl and Allord 1999). Efforts were largely successful; more than ½ of the wetlands in the United States have been destroyed (Mitsch and Gosselink 2000). Virginia's wetlands faired slightly worse; 42% of the original wetlands remain today (Dahl 1990). Beginning in the 1970s, society's views of wetlands shifted (Dahl and Allord 1999). Wetlands are now considered valuable for aesthetic, economic, and ecological reasons (Mitsch and Gosselink 2000). The primary ecological values of wetlands include protection of human communities from floods, removal of sediments and toxins from water, and provision of hotspots of biodiversity (Mitsch and Gosselink 2000). Of the many wetland types, coastal marshes are particularly valuable. Coastal marshes protect waterfront homes from erosion, provide nurseries for fish and feeding grounds for wildfowl (Mitsch and Gosselink 2000).

Coasts with high tidal ranges and relatively flat coastlines, such as the Atlantic and Gulf Coasts, have large expanses of marshes. Marshes are also common in estuaries, such as the Chesapeake Bay. Depending on the salinity of the water surrounding them, coastal marshes are referred to as salt marshes (those with average salinities greater than 5 psu), oligohaline marshes (or 0.5 to 5 psu), or tidal freshwater marshes (those with

freshwater but tidally influenced) (Odum et al. 1984). Salt marshes are the most common

of these types and are found on ocean coasts as well as in high salinity regions of

estuaries (Mitsch and Gosselink 2000). Because of the harsh growing conditions (high

salinity and anaerobic soil), only a few species of plants survive in salt marshes and a

single plant, *Spartina alterniflora*, often dominates.

In some estuaries, far upstream from the ocean, there are regions where the water

is mostly fresh (less than 0.5 psu) but the river remains tidal (Odum et al. 1984). In this

tidal freshwater region, marshes are vastly different from salt marshes. Monotypic stands

of *Spartina alterniflora* are replaced by a diverse mosaic of loosely associated species

that change throughout the growing season (Odum et al. 1984). Tidal freshwater marshes

(TFMs) provide essential nurseries and spawning grounds to fish, such as Striped Bass

(*Morone saxatilis*) and critically important habitat for migratory waterfowl. They are

also important refuges for numerous rare and endangered plants (Odum et al. 1984.

Coastlines with large marsh areas are typically where the strongest effects of

relative sea level rise (RSLR) are expected (EPA 2002). In the Chesapeake Bay, for

example, relative sea level is rising (EPA 2002) and may be causing the water level to

increase and the saline water to reach further upstream (Hull and Titus 1986, Gates

1993). The response of salt marshes to RSLR is well documented (see, for example, Day

and Templet 1989, Patrick and Delaune 1990, Warren and Niering 1993, Morris et al.

2002). Despite the tremendous importance of TFMs and the likelihood that the plant

diversity may plummet with rising salinities, their potential response to RSLR is not well

understood. A series of manipulative and greenhouse studies to look at individual species

response to increased salinity have been completed (Flynn et al. 1995, Baldwin et al.

2001, Howard and Mendelssohn 1999a, 1999b, 2000). Recently, a landscape level study showed that coastal marshes, including TFMs, appear to shift up and downstream with yearly salinity fluctuations (Higinibotham 2004). Except for previous work upon which this study builds (Perry and Hershner 1999), no marsh level study of the response of a TFM to RSLR has been completed.

This study took advantage of the availability of historic vegetation and river salinity data and new vegetation community analysis and salinity data collected at Sweet Hall Marsh, a TFM in the Chesapeake Bay, to look for long-term vegetation change within one marsh's plant community and possible correlation of that change with RSLR. The goals of the study were to (1) compile river and porewater salinity data, (2) document the current vegetation community at Sweet Hall Marsh, (3) compare the vegetation community with historic data, and (4) look for possible correlations between long-term vegetation changes and salinity increases associated with relative sea level rise.

BACKGROUND

I. Introduction to Tidal Freshwater Marshes

Tidal freshwater marshes (TFMs) are found upstream of tidal salt marshes and downstream of non-tidal freshwater wetlands (Odum et al. 1984). Their position within the estuary allows the marshes to experience lunar tides while maintaining low salinities (Odum et al. 1984). The average annual salinity in the tidal freshwater range of an estuary is less than 0.5 psu (Odum et al. 1984). In the United States, TFMs are commonly found along the Middle and Southern Atlantic coasts as well as on the Gulf Coast (Mitsch and Gosselink 2000).

Typically, tidal freshwater portions of the estuary support higher diversity than saltier regions (Odum et al. 1984, Lalli and Parsons 1997, Perry and Atkinson 1997). Such extremely high plant diversity may result from the presence of many micro-habitats as well as the intermediate levels of disturbance and stress (Grime 1979). The diverse plant communities in TFMs are composed of broad-leafed herbs, grasses, rushes, and shrubs (Odum et al. 1984). The diverse habitats in turn create good spawning grounds and habitat for juvenile and adult fish (Mitsch and Gosselink 2000). Additionally, a tremendous diversity and quantity of bird species utilize TFMs (Mitsch and Gosselink, 2000).

TFM soils are composed primarily of silts and clays (Mitsch and Gosselink 2000). The organic content varies depending on location within the marsh (10-15% at stream

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bank to 30-45% in high marsh areas) (Simpson et al. 1983) but is generally lower than that of salt marshes (Mitsch and Gosselink 2000). Neubauer (2002) found the percent organic content of the soils at Sweet Hall Marsh varied between approximately 15 and 25%.

II. Vegetation Assemblages of Tidal Freshwater Marshes

Typical plant assemblages for TFMs are difficult to define because of the high diversity and the dramatic seasonal change within the marsh (Odum et al. 1984). Dominant plant communities undergo seasonal variation from bare mud in the winter to dense, tall-vegetated cover in late summer (Odum et al. 1984). Despite difficulty in defining vegetation community types within TFMs, several schemes have been created. Odum (1984) synthesized these works and created eight loosely defined plant assemblages that occur within the United States: (1) Spatterdock Community Type, (2) Arrow-arum/Pickerelweed Community Type, (3) Wild Rice Community Type, (4) Cattail community Type, (5) Giant Cutgrass Community Type, (6) Mixed Aquatic Community Type, (7) Big Cordgrass Community Type, and (8) Bald Cypress/Black Gum Community Type.

Another distinction found in the literature is between low and high marsh plants. Plant typically found in the low marsh, where lower relative elevation means higher frequency and depth of inundation by estuarine water, are more tolerant to anaerobic soils. In a study of several TFMs associated with the Delaware River, Simpson et al. (1983) suggested that the low marsh is dominated by succulent perennials, such as

Nuphar lutea (formerly *N. advena*) and *Pontederia cordata*. The high marsh, which is less frequently inundated due to its higher relative elevation, is characterized by a high diversity of perennials (*Peltandra virginica, Acorus calamus, Sagittaria latifolia, Typha* sp.) and annuals *(Bidens laevis, Impatiens capensis*, and *Polygonum arifolium)*. Additionally Simpson et al. (1983) describe a zone of annuals, dominated by *Polygonum punctatum* and *Amaranthus cannabinus*, on the stream bank.

Field observations at Sweet Hall Marsh by Doumlele (1976) found two associated groups (*P. virginica, Leersia oryzoides, P. cordata,* and *P. punctatum* association and an *I. capensis, P. arifolium,* and *Carex stricta* association). These associations are not described by Odum (1984) and both fit into the high marsh classification according to Simpson et al. (1983). This illustrates the difficulty of defining strict plant associations within TFM community.

III. Role of Seed Banks within Tidal Freshwater Marshes

Several studies suggested that, at least at the Trenton Marshes in New Jersey, seed banks of TFM's are representative of the current vegetation cover (Leck and Graveline 1979, Parker and Leck 1985, Leck and Simpson 1987). Leck and Simpson (1995) found that perennial species were not well represented in the seed bank though they are consistently an important part of the vegetation community. Among the annual species, some of the seeds are transient (germinating each year) while others persist in the seed bank for multiple years (Leck and Simpson 1987). The seeds of typical of TFM plants generally have the ability and mechanism to facilitate dispersal (Parker and Leck 1985). Because of its position within an estuary with strong riverine and estuarine flow, Sweet

Hall Marsh is likely to have potential seed and propagule sources from upstream freshwater as well as salt marshes.

IV. Vegetation Change within Tidal Freshwater Marshes

Dominant vegetation in TFMs has been shown to change seasonally (Doumlele 1981, Simpson et al. 1983, Odum et al. 1984), annually (Leck and Simpson, 1995 and Whigham and Simpson 1992), and perhaps over decadal time scales (Perry and Hershner 1999). Inter-annual variation is most likely a response to yearly variation in climatic and habitat conditions (Grime 1979). Many factors can lead to both long-and short-term vegetation change including (1) alteration of the habitat from natural or human caused events, (2) changes in abiotic conditions (such as changes in salinity, hydroperiod, or nutrient enrichment) that may create a more suitable (or less suitable) habitat for certain species, and (3) invasion of new species (van der Valk, 1981). Changes in salinity, elevation, and grazing will be addressed in more detail here because they are likely sources of change (especially long-term change) at Sweet Hall Marsh.

1. Inundation and Salinity Tolerances of Plant Species

In freshwater environments the ion concentration inside a plant's cells is greater than the external ion concentration allowing the cells to develop turgor. If external ion concentrations increase, individual cells will rapidly dehydrate and water will leak from the roots (Larcher 1995). Additionally, if taken up by the plant, high concentrations of

sodium ions may interfere with phosphorous metabolism resulting in decreased growth and possibly death (Raven et al. 1992). The salinity tolerance of a species depends on its ability to regulate its internal ion concentration and/or its ability to sequester excess ions within leaves (Hale and Orcutt 1987).

By definition, wetland plants are adapted to survive in hydric soils (Mitsch and Gosselink 2000). Inundation of soils leads to anoxia inhibiting aerobic respiration within the plant's roots (Mitsch and Gosselink 2000). Hydrophytes have several morphological and physiological mechanisms for transporting oxygen to their roots including adventitious roots, arenchyma, hypertrophied lenticels, and pressurized gas flow (Mitsch and Gosselink 2000). Different species of hydrophytes are capable of withstanding different degrees and durations of inundation, thereby creating the hydric gradients frequently found in wetlands (Mitsch and Gosselink 2000). For example, the ability of a plant species to transport oxygen to its roots through pressurized gas flow is correlated with the water depth in which the species can survive (Brix et al. 1992).

Odum et al. (1984) suggested that even occasional pulses of high salinity water would prevent some plant species from utilizing a habitat. Those extremely salt intolerant plants include *N. lutea*, *Iris versicolor*, and *Zizaniopsis mileacea* (Odum et al. 1984). A recent comprehensive review of the salinity and inundation tolerances of tidal freshwater plant species has not been completed. However, several experiments have tested effects of salinity increases and inundation changes on oligohaline marsh plants (see Baldwin and Mendelssohn 1998). Experimentally increasing inundation decreased species richness and favored perennial species over annual species (Baldwin et al. 2001). Flooding significantly reduced seedling emergence in all species observed except *P.*

virginica and *L. oryzoides* (Baldwin et al. 2001) suggesting that *P. virginica* and *L. oryzoides* are likely to be found in a TFM with high levels of inundation. Alternatively, one would expect *P. punctatum* and *Sagittaria* sp. to decrease with increased inundation (Baldwin et al. 2001). *Schoenoplectus tabernaemontani* (synonymous with *Scirpus validus* and *Scirpus tabernaemontani* (USDA-NRCS 2004)) and *Sagittaria lancifolia* showed high tolerance to increased flooding and increased salinity (Baldwin and Mendelssohn 1998). Where applicable, the results of these studies have been incorporated into Table 1 (for salinity tolerances) and Table 2 (for inundation tolerances), which document the general salinity and inundation tolerances of the 30 most common species found in Perry's 1987 study of Sweet Hall Marsh (Perry and Hershner 1999). There is some disagreement between the sources as to the salinity tolerance of some species. Most relevant, *L. oryzoides*, which is considered very intolerant to elevated salinity levels (Odum 1984) was found, in a manipulative laboratory study, to survive a high salinity event (9.4 psu for up to one month) (McKee and Mendelssohn 1989). Disagreements may arise because different populations of the same species may vary in their tolerances (Hester et al.1996, Howard and Mendelssohn 1999), because some species may respond differently to gradual, extended salinity increases versus quick, short salinity pulses, or as a result of compounding factors (such as differences in hydroperiod) (McKee and Mendelssohn 1989). More field and lab studies are necessary to accurately determine the salinity tolerance of fresh and oligohaline marsh plants.

Plant Species	Fresh	Oligo	Meso	Comments	Sources
Amaranthus cannabinus					5, 6
Bidens laevis					1, 4, 6
Carex hyalinolepis					1
Carex stricta					1
Echinochloa walteri					1,4,5,6
Eleocharis palustris					3
Eleocharis quadrangulata				No data	
Hibiscus moscheutos					4, 5, 6
Impatiens capensis				Conflicting data	1, 5
Kosteletzkya virginica					4
Leersia oryzoides				Conflicting data	4, 5
Mikania scandens				No data	
Murdannia keisak				No data	
Peltandra virginica				Conflicting data	1,4,5,6
Phragmites australis					4, 5, 7
Pluchea odorata				No data	
Polygonum arifolium					4, 6
Polygonum punctatum					1,2,4,5,6
Polygonum sagittatum				No data	
Pontederia cordata					1, 2, 6
Rumex verticillatus					

Plant Species	Fresh	Oligo	Meso	Comments	Sources
Sagittaria lancifolia	▬▬▬▬				2, 3
Schoenoplectus americanus	▬▬▬▬▬▬▬				3, 4, 5
Schoenoplectus tabernaemontani	▬▬▬▬▬▬▬				2,4,5,6
Spartina alterniflora	▬▬▬▬▬▬▬▬				5, 6, 7
Spartina cynosuroides	▬▬▬▬▬▬▬				1, 4, 6
Teucrium canadense			– – –	Conflicting data	1, 5, 6
Thelypteris palustris	▬▬			No data	
Typha angustifolia	▬▬▬▬▬▬▬				1,4,5,6,7
Zizania aquatica	▬▬▬▬▬				1,4,5,6

Table 1: Salinity Tolerances of Common TFM Plants

This figure lists published salinity tolerances of species important to study (either because of presence in 2003 vegetation community or decline since previous studies). Sources: Source 1 is PLANTS Database from the USDA (PLANTS, 2003), Source 2 is Baldwin and Mendelssohn, 1998, Source 3 is Howard and Mendlessohn, 1999, Source 4 is Table 4 from Odum, 1984, Source 5 is Table 5 from Odum 1984, Source 6 is Anderson et al., 1968, and Source 7 is Konisky and Burdick, 2004.

Plant Species	Low Marsh	Mid Marsh	High Marsh	Source
Amaranthus cannabinus		*		2
Bidens laevis		*	*	1
Carex hyalinolepis				
Carex stricta				
Echinochloa walteri		*		2
Eleocharis palustris				
Eleocharis quadrangulata				
Hibiscus moscheutos		*	*	1
Impatiens capensis		*	*	1
Kosteletzkya virginica				
Leersia oryzoides	*	*	*	1, 2, 3
Mikania scandens				
Murdannia keisak				
Peltandra virginica	*	*	*	1, 2
Phragmites australis		*		
Pluchea odorata				
Polygonum arifolium			*	1
Polygonum punctatum		*	*	1, 2, 3
Polygonum sagittatum			*	1, 3
Pontederia cordata	*	*		1
Rumex verticillatus				
Sagittaria lancifolia				
Schoenoplectus americanus				
Schoenoplectus tabernaemontani				
Spartina alterniflora				
Spartina cynosuroides	*	*	*	1, 2

Plant Species	Low Marsh	Mid Marsh	High Marsh	Source
Teucrium canadense				
Thelypteris palustris				
Typha angustifolia		*	*	1
Zizania aquatica	*	*	*	1, 2

Table 2: Inundation Tolerances of Common TFM Plants
Lists general elevations in which species are found in tidal freshwater marshes. Sources: Source 1 is based on Figure 13 from Odum, 1984. Based on a qualitative review of figures based on elevation studies reported in Perry, 1991. Source 3 is Baldwin et al. 2001.*Sagittaria* sp.

2. Effects of Vertebrate Grazing on Plant Community

Numerous vertebrate grazers are found in TFMs including ducks, geese, and muskrats (Odum et al. 1984). Vertebrate grazers have been shown to impact the biomass and plant community composition within tidal marshes (Evers et al. 1998).

Muskrats (*Ondatra zibethicus*) are commonly found in Sweet Hall Marsh (personal observation). Muskrats build lodges of plant material, dig tunnels through the marsh, and feed on stems, roots, and rhizomes of marsh plants (Connors et al. 2000). Few studies have been completed on the effects of muskrats grazing on marshes (Connors et al. 2000). Connors et al. (2000) found no significant impact of muskrats on plant species richness or diversity, despite reductions in aboveground biomass. Muskrats make extensive systems of tunnels, which are typically 13-15 cm in diameter and up to 13 m long (Connors et al. 2000), and burrows which may influence water movement within the marsh (Simpson et al. 1983).

Studies focusing on avian grazers have focused mainly on geese feeding in salt marsh communities. Those studies have found that grazing can reduce above and belowground biomass (depending on the species of goose) and change plant community composition (Zacheis et al. 2001). Evers' (1998) study from *Sagittaria* sp. dominated marshes suggests that goose grazing would lead to higher proportion of less desirable food species, such as *P. punctatum*, and lower proportion of succulent species, such as *Sagittaria* sp.

In an exclosure study, Baldwin and Pendleton (2003) found that the presence of vertebrates grazers impacted percent cover and biomass in the low lying marsh, but did not affect species richness in either the low marsh or the high marsh. The plant

community found in the high marsh area of Baldwin and Pendleton's (2003) study site is similar to the plant community found in Sweet Hall Marsh.

3. Marsh Surface Elevation Change

Marsh surface elevation (relative to sea level) is a function of local subsidence, accretion rates, and the rate of relative sea level rise (RSLR). The absolute surface elevation of a marsh is a function of the rate of sediment accretion minus the rate of subsidence (Cahoon et al. 1995).

A marsh's ability to vertically accrete is related to production of organic matter and the deposition of sediment on to the marsh surface (Neubauer et al. 2002). Therefore, changes in accretion rate can be due to changes in vegetation or factors controlling sediment deposition. The type and density of vegetation within the marsh affect the rate of allochthonous sediment deposition and the quantity organic matter preserved in the sediment (Pasternack and Brush 2002). In addition to the effectiveness of sediment trapping by plants, sediment deposition rates are determined by sediment availability and duration of inundation (Friedrichs and Perry 2001, Pasternack and Brush 2002). Changes in current and/or wave velocities, land use changes, and the location of the turbidity maximum can alter the concentration (and therefore availability) of sediment in the water (Friedrichs and Perry 2001). The length of time that the marsh is flooded (which is determined by tidal cycle, stream flow, and relative elevation of the marsh) also affects the accretion rate with increased inundation typically leading to increased accretion rates (Friedrichs and Perry 2001). Neubauer et al. (2002) estimated that the accretion rates in a low marsh area of Sweet Hall to be approximately 8.5 mm/yr. In a

portion of the marsh with higher surface elevation (and therefore shorter inundation period), Campana (1998) estimated the accretion rates were 4.9 to 6.0 mm/yr.

Subsidence is the cumulative effect of sediment compaction (shallow subsidence), groundwater withdrawal, and tectonic activity (Mitsch and Gosselink 2000). Subsidence can cause a decrease in absolute surface elevation, though it may not result in a decrease of surface elevation relative to sea level (see discussion below and Friedrichs and Perry 2001). The Department of Environment Quality considers subsidence a medium level threat to Virginia's coastal wetlands (DEQ 2000). The local rate of subsidence at Sweet Hall Marsh may be higher than the regional average, however, because significant groundwater withdrawal occurs in the vicinity (Perry and Hershner, personal communication).

Over geologic time, the sea level has risen and fallen many times due to changes in the size of the ocean basins, the temperature of the water, and the amount of water held in ice (Brown et al. 1998). There is consensus among scientists that the rate of eustatic sea level rise is increasing (Titus and Narayanan 1996). Titus and Narayanan (1996) predicted that there is a 50% chance that global sea level will rise by 45 cm and a 1% chance that it will rise 112cm by the year 2100. The Chesapeake Bay region's average rate of sea level rise has been about 1 meter per 1,000 years during the last 5,000 years (EPA 2002). And in the York River Estuary, in particular, long-term tidal gauge records show the sea level rising at a rate of 3.95mm/year (+/- 0.27mm/year).

In an area experiencing RSLR, marsh surface elevation can remain stable relative to the sea level because of a positive feedback cycle between length of time the marsh is flooded and inorganic sediment deposition (see Friedrichs and Perry 2001, for review).

This cycle allows the accretion rate to increase thereby compensating for the combined effects of subsidence and sea level rise. Rates of accretion found by Neubauer et al (2002) and Campana (1998) suggested that this will be the case at Sweet Hall Marsh. If accretion rates cannot keep pace with RSLR, the relative elevation of the marsh surface may decrease.

Significant research has focused on the impact of RSLR on coastal salt marshes. These studies have shown that given enough sediment deposition, marsh surface elevation (relative to the sea level) can remain stable (Patrick and Delaune 1990, Morris et al. 2002). Alternatively, loss of wetlands, vegetation community change, and/or plant death are seen in areas with low sediment availability and RSLR (Day and Templet 1989, Warren and Niering 1993). If the marsh is able to maintain its relative surface elevation, then a long-term shift in vegetation community would not be expected (Warren and Niering 1993).

V. Model Tidal Freshwater Marsh System: Sweet Hall Marsh

Compared with salt marsh systems, the effects of RSLR on TFMs have been studied less and are more complicated (Perry and Hershner 1999). This is because, in addition to raising the water level, relative sea level rise may increase the salinity of estuaries (Hull and Titus 1986, Gates 1993). Studies of the Delaware and San Francisco Bays concluded that saltwater will intrude further upstream as sea level rises (Hull and Titus 1986, King et al. 1989).

Sweet Hall Marsh, a component of the National Estuarine Research Reserve System, provides a unique opportunity to study the potential impacts of rising sea levels

on a TFM. The marsh's position at the lower boundary of the tidal freshwater portion of the river means that the potential affects of saltwater intrusion on the vegetation community may be observed at Sweet Hall Marsh before they are observed in upstream marshes. Also, groundwater withdrawal in the region may accelerate the rate of subsidence in the region thereby increasing inundation (Perry and Hershner, personal communication). However, Sweet Hall Marsh's position just above the estuarine turbidity maximum may allow for increased sediment deposition as salinity increases and turbidity maximum shifts closer to the marsh (Friedrichs, personal communication). This observation, coupled with reported accretion rates at Sweet Hall Marsh exceeding the local rates of RSLR, suggested that possible changes in vegetation community associated with RSLR at Sweet Hall Marsh will most likely be due to changes in salinity, rather than hydroperiod.

In addition to two studies of accretion rates (Neubauer 2002, Campana 1998), a long-term data set documenting the vegetation patterns at Sweet Hall Marsh has already been collected. Two previous studies completed in 1974 and 1987 provide historical quantitative vegetation data for the site (Doumlele 1981, Perry and Hershner 1999). The second of the two studies quantitatively documented a change in vegetation from the first study. The results suggested a shift towards an oligohaline marsh with the frequency and density of salt-tolerant plants increasing. Several factors may have contributed to the changing plant community, including salinity increases associated with RSLR or short-term climatic factors (such as drought or to storm-associated extreme high tide) (Perry and Hershner 1999). The study raises the question, was the presence of more salt-tolerant plant species indicative of a long-term change in plant composition or a short-term

change that has since disappeared? To address this question, we assessed the current

vegetation community at Sweet Hall and compared it to the historical communities.

Which led to the following hypothesis:

> Null Hypothesis 1: If species composition of Sweet Hall Marsh is not
> significantly different from the species composition in 1974, then the
> previously documented vegetation change may not have been indicative of
> a long-term shift.

> Null Hypothesis 2: If species composition of Sweet Hall Marsh is not
> significantly different from the species composition in 1984, then the
> vegetation community may not have continued to shift.

> Alternate Hypothesis: If species composition of Sweet Hall Marsh has
> continued to change, then there may be a long-term shift in vegetation
> community.

If the null hypothesis is rejected (i.e. there appears to be a long-term trend in vegetation

change at Sweet Hall) then the next important research question is, "What is causing the

vegetation shift?" We chose to test the hypothesis that salinity increases associated with

relative sea level rise may be causing a shift in vegetation community. To address that

hypothesis we looked at porewater salinity at Sweet Hall Marsh, long-term trends in

Pamunkey River salinity data, and natural history of the plant species. Natural history of

the species provided approximate salt-tolerance levels. The hypotheses to address

possible causes of the shifting vegetation are:

> Null Hypothesis: If species composition has changed, but not towards more salt-
> tolerant species, then the vegetation community change is not due to sea level
> rise.

Alternative Hypothesis: If species composition has shifted towards more salt-tolerant species and the river salinity data shows an increasing trend, then the vegetation shift could be due to relative sea level rise.

This study will also examine the role of inter-annual variation in the plant community at Sweet Hall Marsh through a limited follow-up study in July 2004. The hypothesis associated with that study are:

Null Hypothesis: If the magnitude of changes in importance value from July 2003 to July 2004 are on the same order as (or greater than) changes from 1974 or 1987 to 2003, then inter-annual variation, rather than long-term trends, may be the causing documented community change.

Alternative Hypothesis: If the magnitude of changes in importance value from July 2003 to July 2004 are smaller than changes from 1974 or 1987 to 2003, then long-term shifts may be the causing documented community change.

METHODS

I. Site Description

The study took place in a 440 ha tidal freshwater marsh, Sweet Hall Marsh, which is one of the four components of the Chesapeake Bay National Estuarine Research Reserve in Virginia (CBNERRVA) (Perry and Hershner 1999). Sweet Hall Marsh is the lowest salinity marsh of the four components of the CBNERRVA. It is located on the Pamunkey River 83 km upstream from the mouth of the York River (Figure 1).

II. River and Porewater Salinity

The Chesapeake Bay Program has occasionally collected salinity data from a station downstream of Sweet Hall Marsh from 1984 to present. Samples are collected on an irregular schedule, usually once or twice a month. No samples were collected from November 1986 through March 1987. Mean salinity during the growing season (May-September) was determined from recorded surface values. The salinity data collected by the Chesapeake Bay Program was plotted and a straight line fitted to the plot. A regression was used to test for increasing salinity levels at the station. Additionally, in 2001 CBNERRVA established a permanent water quality station at Sweet Hall Marsh that records river salinity every 15 minutes

Figure 1: Sweet Hall Marsh is located on the Pamunkey River in King William County, Virginia USA approximately 25 river km upstream from the confluence of the Pamunkey and Mattaponi Rivers. Locations of transects 1-4 are depicted in the bottom figure. NOAA nautical chart no.12243.

daily. CBNERRVA salinity data was used to determine minimum, maximum, and average salinity for 2003 and 2002.

To test soil water salinity, 15 porewater samplers ("sippers") were placed adjacent to selected permanent plots within the marsh. Sippers collect porewater through a 5 cm porous PVC window at a depth of 3 to 8 cm below the sediment surface. Sipper placement was chosen to represent the dominant vegetation assemblages within the marsh. Vegetation assemblages were determined using Correspondence Analysis from the data collected in May 2003. In June, sippers were placed at the selected sites and then allowed to equilibrate for one month. For comparison, each month a profile of the porewater was collected at one plot selected based on the plant community. A profile consists of three sippers each of which collect porewater at a different depth below the sediment's surface (3 to 8 cm, 23 to 28 cm, and 43 to 48 cm). The salinity of each porewater sample was determined to 0.01 ppt in the lab using a salinity probe from an YSI 6000 (or 6600) datasonde (Yellow Springs Instruments, Yellow Springs, Ohio).

II. Vegetation

Transects used by Perry and Hershner (1999) were re-sampled (Figure 2). Three of the four transects used by Perry and Hershner were also used by Doumlele (1981). Each of the transects start at the creek bank and stretch landward to an adjacent shrub/forested wetland. The entire length of each transect was divided into 10 meter sections. A 1 m^2 plot was located within each 10 meter section at a random distance

(between 0 and 9 meters) along and away from the transect resulting in the sampling of 76 1- m^2 plots. Permanent plots were marked by short wooden stakes with flagging and sampled once a month, in the middle of the month, from May through October 2003.

Methods for collecting vegetation data within the permanent plots followed Perry and Hershner (1999) and Doumlele (1981). Percent cover (the area within each plot that each species covers (Table 3) and stem densities (the number of stems rooted within the plot) were recorded for each species present within the plots. Frequency was indirectly measured through percent cover data. To determine frequency, the number of plots a species occurred in (for each month) was divided by the sum of all of the species occurrences for that month.

Relative frequency, relative density, and relative dominance were calculated using the following formulas:

$$\text{Relative frequency} = \frac{\text{Species frequency}}{\text{Sum of frequency values for all species}} \times 100$$

$$\text{Relative density} = \frac{\text{Number of individuals of the species}}{\text{Number of individuals of all species}} \times 100$$

$$\text{Relative dominance} = \frac{\text{Species coverage}}{\text{Sum of coverage values for all species}} \times 100$$

Range of Cover	Mid-Point
96-100	97.5
76-95	85.0
51-75	62.5
26-50	37.5
6-25	15.0
1-5	2.5
Trace	0.5

Table 3: Cover Class Scale
  Vegetation cover scale used for 2003 and 2004 vegetation surveys.

Species importance values, which are the sum of the parameters, were used to rank species. Several different methods of calculating importance values were used. As in previous studies, an annual mean IV was calculated by determining an IV for each month (using previously state formulae) and then averaging across the months. Additionally, an annual mean IV was calculated for each species for each transect (so each species has four IVs), and for each plot (so each species has 76 IVs). For example, to calculate IVs for Transect 1 the following formulae were used:

$$\text{Relative frequency} = \frac{\text{Species frequency}}{\text{Sum of frequency values for all species on Transect 1}} \times 100$$

$$\text{Relative density} = \frac{\text{Number of individuals of the species}}{\text{Number of individuals of all species on Transect 1}} \times 100$$

$$\text{Relative dominance} = \frac{\text{Species coverage}}{\text{Sum of coverage values for all species on Transect 1}} \times 100$$

Patterns of vegetation and location associations were explored using Correspondence Analysis (in Matlab 6.0) with IVs based on transect. Species diversity was calculated using the Shannon index (Shannon and Weaver 1949, Doumlele 1976, 1981). Species richness and evenness were calculated according to Pielou (1969).

Comparisons between the three studies were made using the annual mean IVs for the entire marsh and the change in annual mean IV. To standardize the data across the three studies, IVs based on May-September data were used for all comparisons between

each of the studies. Only IVs from the ten most important species were available from the 1974 data set.

A Wilcoxon sign-ranked test was used to test for changes in the importance values of the to ten species from 1974 and 1987 and their paired values from 2003.

Sorenson's index of similarity (Kontkanen 1957) was used to determine the association between the flora of this study and each of the previous studies:

QS= $\underline{\quad 2c \quad}$ X 100

$\qquad$ a + b

where, QS=Sorenson's Index of Similarity

a=number of species found in previous study,

b=number of species found in this study, and

c=number of species common between the studies.

In July, traditionally the peak of the biomass for *P. virginica* (Perry 1991), biomass was collected from 1/16 m^2 plots located 1 meter south of each permanent plot. In August, the historical peak biomass for the entire community (Perry 1991), biomass was collected from ¼ m^2 plots located 1 meter north of each permanent plot. Biomass was measured by cutting all of the above ground plant material that was rooted within the selected plot. The plants cut from each biomass plot were separated by species, placed in a drying oven at 60° C until their weight stabilized (indicating drying was complete). A final dry weight was recorded for each species in each of the 76-biomass plots. Dominant species were defined as the species' weights whose sums make up greater than 50% of the total biomass.

To explore the role of inter-annual variation in the plant community, a follow-up study was completed in July 2004. The transects were revisited, new plots established following the same procedures, and frequency, percent cover, and stem density data was collected in each of those plots. Species' rate of seasonal change, based on monthly percent cover values, were plotted to determine how representative a one month sampling scheme would be of an annual importance value for a species.

RESULTS

I. River and Porewater Salinity

Salinity at a Chesapeake Bay Program station (station code RET4.1, located 9.8 km downstream from the study site) was highly variable from 1984 to 2002 (Figure 3). Average surface salinity at the station was 4.1 psu, with a frequent low of 0.0 psu and a high of 15.8 (on September 16, 2002). Salinity was extremely high (mean=10) in 2002 (the year before the major portion of this study) and fairly high (mean=6.3) in 1986 (the year before the previous study). The annual variation in river salinity meant that regression analysis had a low R-squared value (0.09) and was not significant (p=0.23). Variation in river salinity appears to be correlated with freshwater flow at the USGS stream gauging station on the Pamunkey River (Figure 2). Based on this correlation, we can speculate that salinity during the growing season was about average in 1973 and 1974, well above normal in 1986 and about normal in 1987, and very high in 2002 and very low in 2003.

Salinity measurements from 2002 and 2003, recorded at Sweet Hall Marsh, also illustrate the extreme variability in river salinity (Figure 4). In 2003 the average salinity of the Pamunkey River at Sweet Hall Marsh was 0.2 psu. Peak river salinity (11.7 psu) during that year occurred during Hurricane Isabel, on September 18 (Figure 5).

Figure 2a: Plot of daily mean streamflow at USGS Station #01673000 in Hanover, Virginia correlated with salinity recorded at Sweet Hall Marsh by CBNERRVA in 2002.

Figure 2b: Plot of mean streamflow during the growing season (May-September) at USGS station #01673000 in Hanover, Virginia from 1942 through 2002. The maximum growing season mean streamflow was 1843 ft^3/s in 2003 and the minimum was 120 ft^3/s in 2002. Streamflow in the other years relevant to this study was: 615 ft^3/s in 1973, 777 ft^3/s in 1974, 333 ft^3/s in1986, and 721 ft^3/s in 1987.

Total precipitation (at weather station in West Point, VA) was 172 cm (67.5 inches) in 2003, which was 57 cm (22.4 inches) above normal (NOAA, 2002). Precipitation from the previous year, which may affect availability and viability of seeds, was 103 cm (40.51 inches). However, much of the rain fell late in the year and, therefore, the river salinity was unusually high during the growing season (Figure 6). Average salinity at Sweet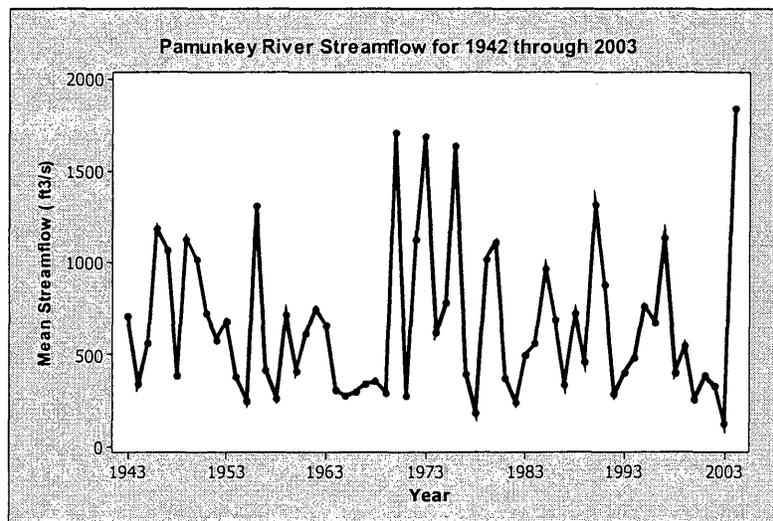 Hall for 2002 was 5.9 psu with a peak of 15.7 psu on October 8. When the data points were averaged across each day, there were only thirteen days in 2002 (all in December) when the salinity was at or below the tidal fresh designation (0.5 psu).

Porewater salinities from each site ranged between 0.10 psu and 1.00 psu and averaged 0.31 psu for July through September 2003 (Figure 7). Mean salinities for the shallow porewater sippers for each transect were: T1=0.35 psu, T2=0.53 psu, T3=0.19 psu, T4=0.38 psu. Based on this limited sample, the porewater appears to be less variable than the river salinity. The data from the set of profile sippers suggested that salinity increases with depth (for profile sippers, mean of shallow=0.405 psu, of mid-depth=1.16 psu, and deep=1.14 psu).

Figure 3: Long-term Salinity Data
Regression analysis of mean salinity during the growing season (May-September) for
1985 through 2002 shows no significant linear pattern. These measurements are based on
data from Chesapeake Bay Program station RET4.1 which is 9.8 km downstream of
Sweet Hall Marsh on the Pamunkey river. Error bars show one standard deviation above
and below growing season mean.

Figure 4: Monthly Average Salinity from 2002 and 2003
Monthly salinity averages collected from CBNERRVA fixed station located at Sweet
Hall Landing. Averages were calculated from salinity measurements taken every 15
minutes with a YSI 6600 (or YSI 6000) multi-parameter data sonde. Error bars represent
one standard deviation above and below the mean value.

Pamunkey River Salinity:
Daily Averages from 2003

Figure 5: Daily Average Salinity for 2003
River salinity at Sweet Hall Marsh during 2003. September spikes are associated with tropical storms and Hurricane Isabel. Average daily salinity was calculated from salinity data collected every 15 minutes by YSI 6600 (or YSI 6000) multi-parameter datasonde deployed at Sweet Hall Landing by CBNERRVA.

Figure 6: Daily Average Salinity for 2002
River salinity at Sweet Hall Marsh during the entire year of 2002. Average daily salinity was calculated from salinity data collected every 15 minutes by YSI 6600 (or YSI 6000) multi-parameter datasonde deployed at Sweet Hall Landing by CBNERRVA.

<u>II. Vegetation</u>

1. Characterization of Vegetation Community

The vegetation community of Sweet Hall Marsh is characterized by a diverse plant community made up of loose associations of herbaceous species. Thirty vascular plant species, representing 14 families, were identified within the 76 permanent and 76 biomass plots in 2003. These include one fern, 18 monocotyledones, and 11 dicotyledones (Table 4). Table 5 lists all species fo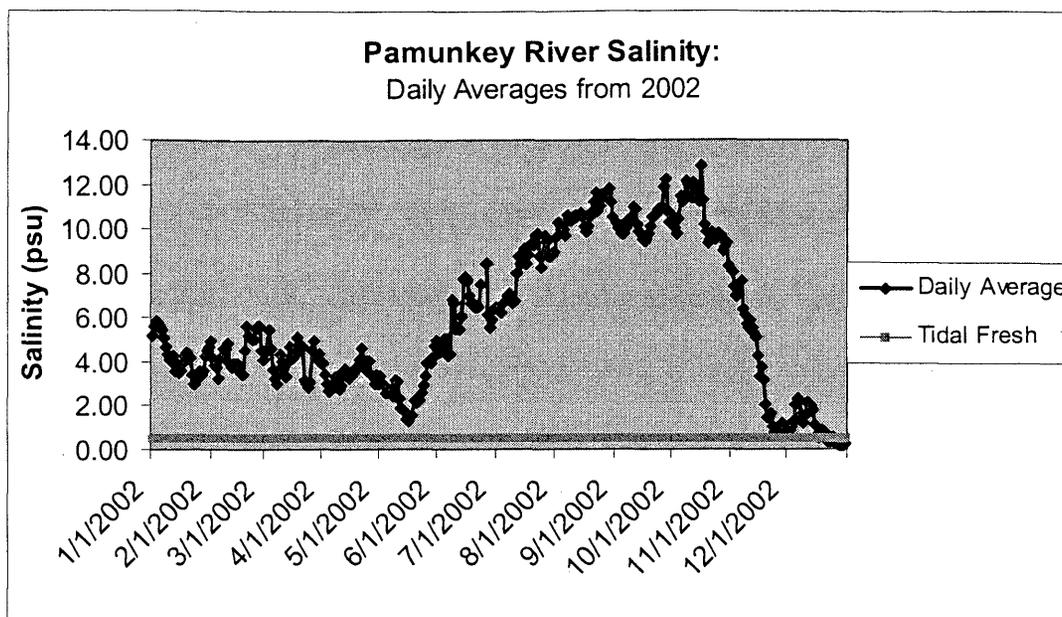und within the permanent plots and their corresponding relative density, relative cover, relative frequency, and annual mean Importance Value (IV). *Peltandra virginica*, an emergent macrophyte common in TFMs, had the highest importance value. *Phragmites australis*, an invasive species known to form monotypic stands, ranked 8th. A species common to more saline marshes, *Spartina alterniflora*, has the 9th highest importance value.

Based on aboveground standing biomass collected in July and August, the dominant species within the sampling area were *Zizania aquatica* and *P. virginica*. In July, which has been shown to be time of peak biomass for *P. virginica* (Doumlele 1981, Perry and Hershner 1999), *P. virginica* made up 23.3% of the aboveground biomass with an average of 205.3 g/ m^2. *Z. aquatica* made up 28.6% of the aboveground biomass with an average of 251.3 g/ m^2. The same two species comprised greater than 50% of the biomass in August, with *Z. aquatica* making up 44.1% (266g/ m^2) and *P. virginica* making up 11.2% (71.3g/ m^2). P.virginica and *Z. aquatica* also had the highest annual mean IVs in 2003.

Porewater and River Salinity:
2003 Growing Season

Figure 7: River and Porewater Salinites during 2003 Growing Season
Plot of monthly average salinity during the 2003 growing season collected at the
CBNERRVA fixed monitoring station with a YSI 6600 (or YSI 6000). Porewater is the
average salinity from water collected once each month from thirteen shallow porewater
sippers placed throughout the study site. Error bars represent one standard deviation
above and below the mean salinity. Porewater data from profile sippers is not shown.

Table 4: Species List
All species found within the permanent and/or biomass plots in 2003 listed with their complete nomenclature, common name, and whether the species is an annual (A), perennial (P), or can occur as either A or P. Rank, based on annual mean IV from 2003 is also listed. Species found only in biomass plots were not assigned a rank and are demarcated with a *.

Group	Family	Species	Common Name	A/P
Ferns	Thelypteridaceae	*Thelypteris palustris* Schott.	eastern marsh fern	P
Monocots	Alismataceae	*Sagittaria lancifolia* L.	bulltongue	P
	Araceae	*Peltandra virginica* (L.) Schott	arrow arum	P
	Commelinaceae	*Murdannia keisak* (Hassk.) Hand.-Maz.	asian day lily	P
	Cyperaceae	*Carex stricta* Lam.	upright sedge	P
		Carex hyalinolepis Steud.	shoreline sedge	P
		Eleocharis palustris (L.) Roemer & J.A. Schultes	common spikerush	P
		Eleocharis quadrangulata (Michx.) Roemer & J.A. Schultes	squarestem spikerush	P
		Schoenoplectus tabernaemontani (K.C. Gmel.) Palla	softstem bulrush	P
		Schoenoplectus americanus (Pers.) Volk. ex Schinz & R. Keller	chairmaker's bulrush	P
	Juncaceae	*Juncus sp*		
	Poaceae	*Echinochloa walteri* (Pursh) Heller	coast cockspur grass	A
		Leersia oryzoides (L.) Sw.	rice cutgrass	P
		Phragmites australis (Cav.)Trin. ex Steud.	reed grass	P
		Spartina alterniflora Loisel	smooth cordgrass	P
		Spartina cynosuroides (L.) Roth	big cordgrass	P
		Zizania aquatica L.	wild rice	A
	Pontederiaceae	*Pontederia cordata* L.	pickerelweed	P
	Typhaceae	*Typha angustifolia* L.	narrowleaf cattail	P
Dicots	Amaranthaceae	*Amaranthus cannabinus* (L.) Sauer	tidalmarsh amaranth	A
	Asteraceae	*Bidens laevis* (L.) B.S.P.	smooth beggartick	A
		Mikania scandens (L.) Willd.	climbing hempvine	P
		Pluchea odorata (L.) Cass.	sweetscent	A
	Lamiaceae	*Teucrium canadense* L. var. canadense	Canada germander	P
	Malvaceae	*Hibiscus moscheutos* L.	crimsoneyed rosemallow	P
		Kosteletzkya virginica (L.) K. Prel ex Gray	virginia saltmarsh mallow	P
	Polygonaceae	*Polygonum arifolium* L.	tearthumb	A
		Polygonum punctatum Ell.	dotted smartweed	A
		Polygonum sagittatum L.	Arrowleaf tearthumb	A
		Rumex verticillatus L.	swamp dock	P

Species	Rel Density	Rel Cover	Rel Frequency	IV
Peltandra virginica	13.82	24.99	18.83	57.63
Zizania aquatica	16.31	20.47	16.26	53.04
Eleocharis palustris	31.83	4.2	2.76	38.79
Echinochloa walteri	10.37	7.21	10.43	28.01
Polygonum punctatum	4.12	3.95	12.42	20.5
Bidens laevis	3.55	9.27	6.46	19.28
Phragmites australis	3.5	3.65	5.98	13.13
Spartina alterniflora	3.79	2.79	2.18	8.76
Amaranthus cannabinus	1.57	0.81	5.6	7.98
Typha angustifolia	1.22	2.41	2.19	5.82
Eleocharis quadrangulata	5.2	0.02	0.17	5.4
Polygonum arifolium	0.46	0.78	3.27	4.5
Schoenoplectus tabernaemontani	1.3	0.48	2.06	3.84
Spartina cynosuroides	0.67	0.72	1.56	2.95
Hibiscus moscheutos	0.41	0.54	1.67	2.61
Carex hyalinolepis	0.35	0.56	1.5	2.41
Leersia oryzoides	0.41	0.15	1.55	2.12
Pontederia cordata	0.22	0.33	1.43	1.98
Pluchea odorata	0.23	0.18	1.33	1.74
Sagittaria lancifolia	0.18	0.14	0.46	0.78
Rumex verticillatus	0.09	0.14	0.35	0.58
Juncus sp	0.18	0.03	0.29	0.5
Schoenoplectus americanus	0.07	0.01	0.39	0.47
Polygonum sagittatum	0.02	0.02	0.24	0.27
Murdannia keisak	0.01	0.02	0.17	0.2
Thelypteris palustris	0.01	0.01	0.17	0.19
Unknown	0.08	0.04	0.05	0.18
Unknown monocot	0.04	0.05	0.06	0.14
Mikania scandens	0	0	0.12	0.12
Kosteletzkya virginica	0	0.01	0.06	0.07

Table 5: Relative Values for Species
All species found within the study plots in 2003 with their corresponding relative density, relative cover, relative frequency, and annual mean importance values.

Succulent or very small species may be under-represented by aboveground dry biomass weights so rank order of species based on IVs does not directly correspond to species' rank based on biomass (Table 6).

In order to concentrate analysis on the dominant vegetation, eleven rare species were removed from the analysis. Rare species were defined as those with an IV of less than one (Figure 8). The other nineteen species found within the permanent plots were defined as common (IV>1) and included in all analysis. IVs (calculated by plot) and their corresponding estimate of error are graphed in Figure 9.

Ordering of the species using Correspondence Analysis (based on annual mean importance values calculated for each species for transect) suggested some grouping of species, with a strong connection between *Carex hyalinolepis* and *Spartina cynosuroides* (Figure 10). Correspondence Analysis also suggested that the vegetation communities vary between transects, with Transects 2 and 4 being the least similar. The connection between *C. hyalinolepis* and *S. cynosuroides* was strongly linked to Transect 4. *S. alterniflora* (and *E. palustris* which it was loosely associated with) appear to be more dominant in Transects 1 and 2 (Figure 11).

Species	July	August
Peltandra virginica	205.39	71.32
Zizania aquatica	251.30	266.56
Eleocharis palustris	42.82	18.22
Echinochloa walteri	83.68	39.46
Polygonum punctatum	6.36	6.25
Bidens laevis	57.62	49.47
Phragmites australis	100.44	51.75
Spartina alterniflora	65.05	25.06
Amaranthus cannabinus	0.61	3.51
Typhà angustifolia	46.95	30.85
Eleocharis quadrangulata	0.06	0
Polygonum arifolium	0.32	1.83
Schoenoplectus tabernaemontani	4.18	6.18
Spartina cynosuroides	7.22	22.85
Hibiscus moscheutos	4.42	2.00
Carex hyalinolepis	1.31	3.54
Leersia oryzoides	0.63	1.70
Pontederia cordata	0.46	2.77
Pluchea odorata	0.27	0.32
Sagittaria lancifolia	0.08	0
Rumex verticillatus	0.65	0
Juncus sp	0	0
Schoenoplectus americanus	0.21	0
Polygonum sagittatum	0	0
Murdannia keisak	0	0.10
Thelypteris palustris	0	0
Unknown	0	0
Unknown monocot	0	0
Carex stricta	0	1.3
Mikania scandens	0	0
Kosteletzkya virginica	0	0
Teucrium canadense	0	0

Table 6: Species List with Biomass Values
All species found within the biomass and permanent plots in 2003 listed with their corresponding biomass from July and August 2003 (in grams per m^2). Species are listed in rank order based on annual mean IVs, with *Peltandra virginica* having the highest IV and a rank of 1.

Annual Mean IV

Figure 8: Common Species Determined and Defined
Annual mean importance value for 2003 season for each species found within the
permanent plots. The nineteen species with IV >1 are defined as "common species" and
will be used in data analysis.

Annual Mean Importance Values:
For 2003 by Plot

Figure 9: Importance Values by Plot
Bar graph of annual mean importance values +/- one standard deviation. IVs shown here
were calculated for each plot and are therefore slightly different from IVs calculated for
entire study site. This calculation method allowed error estimates to be determined. For
comparison with previous studies, IVs were calculated for entire study site and error
values cannot be determined.

Symmetric Plot

Figure 10: Grouping of Species by Correspondence Analysis

Bi-plot resulting from sorted transects and species IV completed using correspondence analysis. Species whose names are underlined are considered tolerant to higher salinities (of mesohaline or high oligohaline levels), others are either less tolerant of salt or their tolerance is unknown. Species' positions that are demarcated with a plus sign (+) have increased in importance. Those with a triangle (▼) have decreased in importance. Species whose importance has been variable across the three studies are fairly stable are marked with a square (■).

Figure 11a: Data Matrix Before Analysis
Color enhanced image of the common species' annual mean importance value for each transect before sorting with Correspondence Analysis.

CA for Common Species IV from 2003: Grouped by Transect

Figure 11b: Grouping of Transects by Correspondence Analysis
Color enhanced image of the sorting resulting from Correspondence Analysis of the common species' annual mean importance value for each transect. Lighter colors indicate higher importance values.

2. Comparison with historic vegetation community

Species diversity values were similar for each of the vegetation studies (Table 7). Richness values were much lower for 2003 than for previous studies. The richness equation (N/√S, where N=number of species found and S=number of st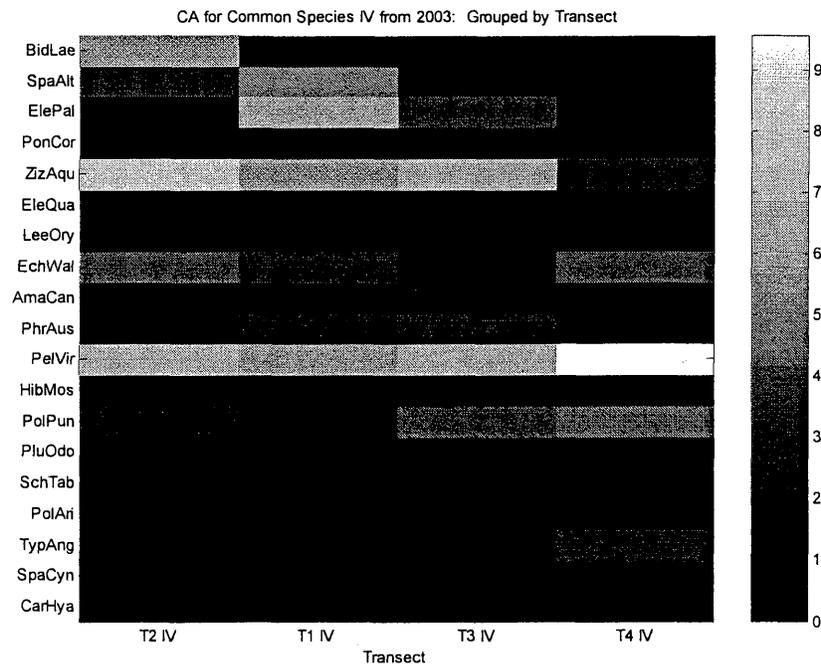ems found) is dependent on sample size therefore the lower values may indicate only that more stems were counted. There were more plots sampled in 2003 than in the previous studies, so this increase in total stem count was expected.

Sorenson's Index of Similarity was 64% similar for 2003 and 1987 studies and 60% similar for the 2003 and 1974 studies. Descriptive comparison of the IVs from each of the three studies suggested that there is no overarching trend that includes all species (Figure 12). Rather, some species have shown dramatic declines while others have shown dramatic increases in IV. Wilcoxon Sign-Ranked tests comparing the IV of the top ten species from 1987 with their paired values from 2003 indicated that there was no all-inclusive trend (p-value=0.10). However, comparison of the 1974 and 2003 paired values suggests that over the longer time scale there is a change in the community (p-value=0.01). Nine of the top ten species from 1974 decreased in importance or were absent from the study plots in 2003.

To create a more specific picture of long-term trends in importance we focused on the perennial species (Figure 13).

		May	June	July	August	September
Species Diversity	1974	0.870	0.882	0.980	1.035	1.008
	1987	1.004	0.805	1.067	1.135	0.745
	2003	1.015	1.099	1.141	1.135	0.933
Species Evenness	1974	0.615	0.631	0.693	0.676	0.697
	1987	0.687	0.723	0.671	0.724	NA
	2003	0.735	0.796	0.816	0.812	0.816
Species Richness	1974	0.768	0.619	0.614	0.768	0.661
	1987	0.499	0.591	0.732	0.651	NA
	2003	0.270	0.286	0.304	0.349	0.440

Table 7: Species Diversity Indices
Species Diversity, Evenness, and Richness for each of the three studies. Fields for which no data was available are designated with a NA. Species evenness and richness data were not available for September 1987. Species Diversity = H, Species Evenness = H/ log S, and Species Richness S/Square Root of N, where H is the Shannon-Weaver Diversity Index, S is number of species sampled, and N is the number of individual stems.

Figure 12: Importance Values From Three Studies Compared
Bar graph of annual mean importance values (based on May-September data) for each of the three studies. Only species which were common (IV>1) in 2003 are shown and they are plotted in order of importance for that year. Species noted with an * were reported in the 1974 study, but no importance value was available because they were not among the ten most important species that year.

Change in IV:
Perennials with Change >1

Figure 13: Change in Importance Values of Common Perennials

Illustrates the amount of increase or decrease in importance value from previous studies to 2003 study. Only perennial species with a change greater than 1 are shown. Species which were present in the 1974 study, but for which no data is available are indicated with an asterisk (*).

0

Species tolerant of salinity stress, such as *Typha angustifolia, S. alterniflora, P. australis,* and *Eleocharis palustris* (see Table 1), increased in IV and, therefore, grouped together on the right side of the graph. Interestingly, *S. cynosuroides* and *C. hyalinolepis* are again grouped together, but, because they decreased in importance, they are grouped on the left side of the graph (opposite from the other salt-tolerant species). In previous studies these species were found to be associated with areasof higher elevation (such as levees) (Doumlele 1981, Perry 1991). The other plants that decreased in importance were *Leersia oryzoides, P. virginica, Carex stricta,* and *Pontederia cordata,* all of which are intolerant of salinity stress.
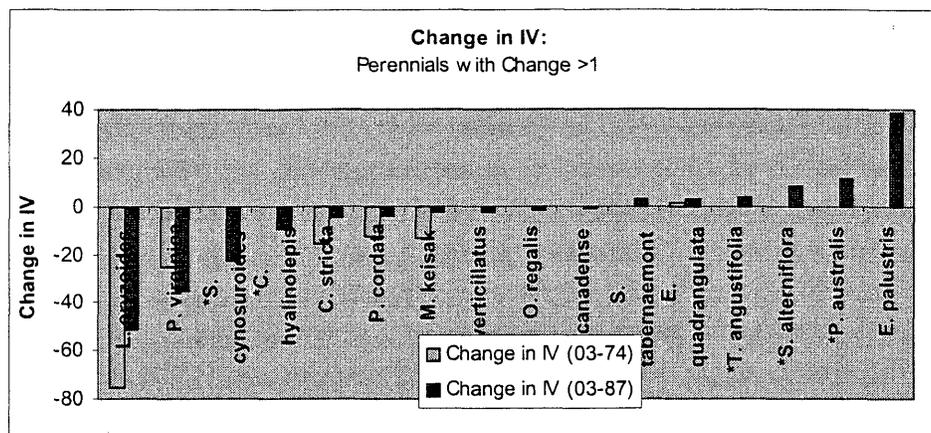
Numerous species were documented in previous studies that were not found in the permanent plots of this study; including two species (*C. stricta* and *Impatiens capensis*) that were among the ten most important plants in 1974 and four plants which were "common" (with IV >1) in 1987 (*C. stricta, Bidens coronata, Osmundo regalis,* and *Teucurium canadense*). Alternatively, *Koseteletzkya virginica, Pluchea odorata,* and *Sagittaria lancifolia* were found in 2003, but were not documented in the study plots in previous studies.

3. 2004 Vegetation Study

Importance Values varied from July 2003 to July 2004 for both perennial and annual species (Figure 14). *E. palustris* and *S. alterniflora* decreased substantially from the previous study; *E . palustris'* July IV decreased from 53.93 to 21.29, *S. alterniflora* decreased from 8.93 to 0, and *P. virginica* decreased from 68.80 to 51.38. *C. stricta, L. oryzoides* and *P. punctatum* increased from 2003 to 2004; *C. stricta* increased

from 0 to 2.67, *L. oryzoides* increased from 1.96 to 15.87, and *P. punctatum* increased

from 16.67 to 27.12. Rate of seasonal change of the species in 2003 was explored by

graphing the percent cover values for each month (Figure 15).

Figure 14: Change in Importance Value from 2003 to 2004
This graph illustrates the amount of increase or decrease in importance from July 2003 to July 2004. Only perennial species with a change greater than one are shown.

Figure 15: Rate of seasonal change during 2003 for *Eleocharis palustris, Peltandra virginica, Spartina alterniflora*, and *Zizania aquatica*. "Total Cover" is the sum of the cover values for that species in all plots. Some species, such as *S. alterniflora*, exhibit gradual seasonal changes while other species, such as *P. virginica* show sharp declines. *Z. aquatica*'s steep decline from September to October is probably due to Hurricane Isabel. One month sampling as used in 2004 study may be less accurate for species with sharp seasonal changes.

DISCUSSION

I. River and Porewater Salinity

The inter-annual variation in salinity levels and the limited extent of the data set

obscure any gradual increase in river salinity that may be associated with relative sea

level rise. Fluctuation in the freshwater flow in the Pamunkey River appears to cause

variability in salinity at Sweet Hall Marsh. The limited salinity data does make it clear,

however, that the Sweet Hall Marsh plant community is living in an environment with

highly variable, and sometimes very high, salinities. In future studies new data collection

techniques, such as the continuous fixed station monitoring completed at Sweet Hall

Marsh by CBNERRVA, may help distinguish between subtle long-term trends and

extreme inter-annual variation.

The 2002 and 2003 data from the CBNERRVA fixed station illustrate that the

salinity is variable from year to year as well as for shorter (often storm-associated)

events. In 2002 the Sweet Hall experienced an extended high salinity event (Figure 6).

Annual average salinity was 5.9 psu. The extended high salinity event lasted throughout

the growing season therefore the plants needed to grow and reproduce in relatively high

salinities. In contrast, in 2003 the low average annual salinity (0.2 psu) would be suitable

for freshwater plants. Even in 2003, however, there was a short-term spike in the salinity

associated with Hurricane Isabel when the salinity rose rapidly (from 0.3 to 11.7 psu on

September 18, 2003) and then quickly returned to fresh levels (<0.5 psu) two days later.

Glycophytes may able to survive such a short-term event (Howard and

Mendelssohn 1999b), though field observations at Sweet Hall Marsh one week after

Hurricane Isabel suggest that most of the vegetation was not able to survive the high

salinity. October's sampling supported these observations; a total of 2,235 live stems

were counted during the September 2003 sampling (September 15-16) compared to only

913 live stems counted in October 2003 (October 14-17). In a field experiment, McKee

and Mendelssohn (1989) also observed a rapid increase in water column salinity (to15

psu) that resulted in death of freshwater plants.

Greenhouse studies that explored differing effects of the rate of increase of

salinity, duration of exposure to high salinities, and the final salinity. These studies

suggested that the final salinity (i.e. peak water column salinity), rather than the rate of

increase in salinity, determines whether a species can survive (Howard and Mendelssohn

1999a, 1999b, Howard and Mendelssohn, 2000). The rate of increase in salinity tested

in the experiment, however, was much slower (most extreme treatment was 0 to 12 g/L

increase over 3 days) (Howard and Mendelssohn 1999a, 1999b), than the increase

documented at Sweet Hall Marsh in 2003 (0.2 to 11.7 psu over 16 hours) or than

previously documented during a hurricane in Louisianna (approximately 0.5 to 10-15 g/L

over 4 hours) (Jackson et al. 1995). Howard and Mendelssohn (1999a) also suggested

that increasing the time a plant typically found in oligohaline marshes (in this case,

Eleocharis palutris, *Panicum hemitomon*, *Sagittaria lancifolia*, and *Schoenoplectus*

americanus) is exposed to high salinities, the more severe its growth suppression will be.

The limited porewater sampling completed with this study suggested that the

porewater salinity may be less variable than the river salinity (Figure 7). Earlier studies

of higher salinity marshes and subtidal sediments also found that interstitial salinities, especially in fine-grained sediments, were more constant than the overlying waters (Lindberg and Harriss 1973, Hackney and de la Cruz 1978, Chapman 1981).

Porewater salinity data collected at Sweet Hall in 2003 were different from expected in two respects. Previous porewater studies completed in tidal salt marshes have found porewater salinities to be higher than that of the overlying water column (Lindberg and Harris 1973, Hackney and de la Cruz 1978). Porewater salinities at Sweet Hall appeared to be the same or lower than river salinities (Figure 7). However, previous studies measured salinity just over the marsh surface, while in this study, the river salinity measurements were made within the upper layers of mainstem of the Pamunkey River. Lindberg and Harris (1973) also reported that porewater salinities decreased with depth (presumably because of the increased influence of groundwater). The limited porewater data collected at Sweet Hall, suggested that at Sweet Hall salinity may increase with depth (from 0 to 48 cm). Several factors may explain the possible salinity gradient; including higher rates of flushing of very shallow layers because of channels along the plant's roots or dilution by rainwater, or higher density of salt water coupled with the loosely consolidated soils of TFMs.

II. Vegetation

1. Characterization of Vegetation Community

The vegetation community at Sweet Hall Marsh in 2003 was typical of a tidal freshwater marsh community; it was floristically diverse and composed of loose associations of species. It was atypical, however, because of the strong presence of some

very salt-tolerant plant species (such as *Spartina alterniflora*) and the lack of some common freshwater species (most notably *Nuphar lutea*) (Odum et al. 1984).

Sweet Hall Marsh may also differ from other TFMs because of the importance of perennial species. The Trenton Marshes in New Jersey, a well-studied TFM, was dominated by annual species (Whigham and Simpson 1992, Leck and Simpson, 1995). Whigham and Simpson (1992) calculated an annual to perennial ratio for multiple years using above-ground biomass (from a limited number of plots within a small study area). Their ratios were somewhat variable and, in late summer, were sometimes as high 5:1. Rough calculations of the annual to perennial ratio for Sweet Hall Marsh (using biomass data collected in August 2003) indicates that it may be a more perennially dominated system with a ratio of 5:8.

2. Comparison with Historic Vegetation Community

The Wilcoxon-sign ranked test showed that the vegetation community as a whole did change during the 30-year span of the data set. Descriptive comparisons also support that the vegetation community changed (Figure 12). Some species show extreme variability within the three studies, which may be the result of a long-term trend towards a new plant community. Alternatively, the variability may be an indicator that the vegetation community is extremely variable from year to year.

Although the composition of the vegetation community changed since 1974 and, at least descriptively, since1987, the total productivity likely remained relatively constant. In a small study of long-term biomass changes in a TFM, Whigham and Simpson (1992) also found that while the components of the community changed

tremendously from year to year, the total biomass was fairly constant. DeBerry and Perry (2004) found similar results when comparing biomass and species composition of created and natural reference marshes. The consistency may arise because while different species may be utilizing the resources, the quantity of resources, especially nutrients, light, and space, are remaining relatively constant.

Several factors, including the variability in the community composition and the importance of perennial species in Sweet Hall Marsh, suggested that for this study the analysis of the vegetation should be separated into annual and perennial species. Another factor was that earlier studies of TFMs found high levels of inter-annual variation in the importance of specific plants, with annual species especially showing high levels of variation (Whigham and Simpson 1992, Leck and Simpson 1995). It is reasonable to assume that annual species dependence on seed viability and germination (both of which are strongly influenced by inundation and salinity (Baldwin et al., 2001, Chambers et al., 2003)) results in more population variability than would be expected within the perennial components of the community. Grime (1979) suggested that in perennially dominated systems, year-to-year changes in environmental conditions would cause variation in the more ruderal, or pioneer, species but not necessarily in the long-lived perennials.

As expected, annual species showed tremendous variation between the studies. Of the annual species that show dramatic changes, *Zizania aquatica* and *Echinochloa walteri* show the most obvious increases. *Z. aquatica* is an obligate wetland plant that is frequently found in freshwater marshes in the Eastern United States (Boland and Gurk 1992). It is an annual grass with fibrous roots (Boland and Gurk 1992) and is considered to be tolerant of only low levels of salinity (Anderson et al. 1968, Odum et al. 1984,

USDA-NRCS 2004). *E. walteri* is an annual grass which is found in wetlands and waste

areas in the southeastern United States (Radford et al. 1968). It is considered a fresh and

oligohaline species (Anderson et al. 1968, Odum et al. 1984, USDA-NRCS 2004). We

expected, and found, that these annual species were successful in 2003 because of the low

salinities associated with the high precipitation. In 2002 when salinities were well above

5 psu during the growing season, however, we speculate that these annual species were

less common and show reduced vigor.

Perennial species did show a distinct pattern (Figure 13). Species not tolerant to

salinity, *Leersia. oryzoides*, *Peltandra virginica*, *Carex stricta*, and *Murdannia keisak*,

decreased from 1974 and 1987 to 2003 while salt-tolerant species increased in IV. These

data, coupled with other's findings (Perry 1991, Perry and Hershner 1999) appeared to

suggest that there is a long-term trend towards more salt-tolerant species. If higher

salinities are present, the shift in plant species that results could be caused by multiple

factors; freshwater plants can be killed directly by the salt because of the osmotic

imbalance (Hale and Orcutt 1987, Raven et al. 1992, Larcher 1995) or because of the

higher sulfide levels in the soil (Chambers et al. 2003). The death of the intolerant

species may allow species that are generally poor competitors (e.g. *S. alterniflora*) to

establish within the marsh (Konishy and Burdick 2004). Alternatively, the disturbance in

the community could allow more invasive species, such as *P. australis*, to spread

(Chambers et al. 2003). Changes in salinity can also indirectly create changes in the

plant community. For example, the competitive advantage could be changed because the

increased amount of ocean water could change the nutrient availability with the system

(Emery et al. 2001).

Regardless of the mechanism causing the shift, the relatively high importance of salt-tolerant species in 2003 is especially interesting considering the low salinity levels in the river during that year. The perennial species which decreased in importance (*L. oryzoides*, *P. virginica*, *C. stricta*, and *Pontederia cordata*) are typical of freshwater wetlands. *Peltandra virginica*, *C. stricta*, and *P. cordata* are generally considered to be intolerant of elevated salinities (Anderson et al. 1968, Odum et al. 1984, Baldwin and Mendelssohn 1998, USDA-NRCS 2004) (Table 1). *L. oryzoides* is commonly found in freshwater wetlands, though greenhouse experiments have shown that it can withstand high salinity levels (up to 9.4 psu for one month) (McKee and Mendelssohn 1989) and is able to recover from high salinity events to a limited extent (Flynn et al. 1995). Their presence in this study supported our hypothesis that perennial species are less sensitive to yearly variations and therefore may be good predictors of long-term change.

The apparent trend towards more salt-tolerant species was also supported by determing which species disappeared or were functionally absent from the community in 2003. *Carex stricta, Bidens coronata, Impatiens capensis, Teucrium canadense, M. keisak, P. cordata*, and *L. oryzoides* were once common in Sweet Hall Marsh, but have disappeared or nearly disappeared. With the possible exception of *L. oryzoides*, these species are considered freshwater plants or are typically found only in freshwater wetlands (though Anderson (1968) suggests that *T. canadense* may tolerate mesohaline salinities) (Anderson et al. 1968, Odum et al. 1984, USDA-NRCS 2004). This suggests that these species are barely able to survive at Sweet Hall Marsh, perhaps because of the occasionally high salinities. The rarity of *C. stricta* (which was not found in the permanent plots, but was observed within the marsh) and complete absence of *I. capensis*

are especially interesting considering that they were both among the top ten most important species in 1974. Both species are considered tolerant of only fresh water.

Examination of the species which "appeared" in 2003 but were not present in earlier studies further supported the hypothesis that the plant community may be shifting towards more salt-tolerant species. *Kosteletzkya virginica* is tolerant of mesohaline waters. Similarly, *S. lancifolia*, is considered tolerant of oligohaline waters and is often seen growing in association with other salt-tolerant species (Howard and Mendelssohn 1999a, 1999b, Howard and Mendelssohn 2000). No pertinent natural history was found for *P. odorata*.

Two salt-tolerant plants, *Spartina cynosuroides* and *Carex hyalinolepis*, decreased in importance from the previous study. These species form one of the stronger plant associations described by the Correspondence Analysis. Perry (1991) found that these plants were most frequently found in areas of higher elevation, such as on creek-side levees. The decrease in these species, therefore, may be a response to an overall decrease in elevation or to decreased prevalence of levees. Levees form in marshes because of increased sediment deposition at the creek bank (Friedrichs and Perry 2001). Levees may be a dynamic feature at Sweet Hall Marsh. The decrease in *S. cynosuroides* is especially interesting because it had increased substantially from 1974 to 1987. Perry and Hershner (1999) hypothesized that this increase may have been indicative of the increasing salinity levels at Sweet Hall. It is also possible, however, that this increase was due to change in elevation levels within the marsh. The decrease in IV of *S. cynosuroides* and *C. hyalinolepis* from 1987 to 2003 could indicate that even perennial species are dynamic from year to year.

Analysis of only the perennial species suggested that the plant community is shifting towards a more salt-tolerant state. This, however, does not show the distribution of the species within the marsh. Field observations suggested that the salt-tolerant species are not spread evenly throughout the marsh. Nor are they concentrated in just one region of the marsh. Rather, there are "islands" of salt-tolerant species (most notably the *S. alterniflora-E. palustris* association revealed by the Correspondence Analysis) within the mosaic of more typically freshwater species. These islands share other characteristics of salt marshes, including more consolidated substrate and presence of sulfides (the latter detected by odor). Experimental manipulation by Baldwin and Mendelssohn (1998) suggested an explanation for these islands; species that were otherwise able to cope with low-level salinity stress were not able to survive the same level of salinity stress if it was coupled with an additional stressor, such as simulated grazing (Baldwin and Mendelssohn 1998). It is possible, therefore, that the freshwater plants at Sweet Hall were able to survive despite the salinity stress, but in the isolated areas where another stressor was introduced (such as within a muskrat eat out area or an area of intense wrack deposition), the freshwater species died. The freshwater species may have then been replaced by salt-tolerant species—hence the patches of salt-tolerant plants.

Alternatively, these patches could be areas where interstitial salinities are higher perhaps due to differences in groundwater flow. Porewater samples collected within these areas, however, do not have higher salinities than samples taken outside of these regions.

3. 2004 Vegetation Community

The follow-up study in 2004 explored the magnitude of inter-annual variation in IV and the hypothesis prompted by Baldwin and Mendelssohn (1998), that disturbed areas will shift to more salt-tolerant species. We recognize that sampling for only one month was not adequate to create a complete picture of vegetation dynamics of a TFM because of the seasonal variation in importance of different species (as documented by Doumlele 1981, Odum et al. 1984, and Simpson et al. 1983). The means and standard deviations of the variation between July 2004 and July 2003 were similar to the variations seen between annual IVs of the earlier studies and greater than that found between adjacent months in 2003. For species with gradual seasonal changes sampling for only one month may be fairly accurate depiction of yearly importance value. One month sampling may be less accurate for species with sharp seasonal changes, such as *P. virginica*.

Despite these limitations, the July 2004 data did show that there was large variation in the relative importance of both perennial and annual species from July 2003 to July 2004. Therefore yearly variation in perennial species may be more dynamic than previously expected. If this is true, perennial species may not be as reliable indicators of long-term change within the vegetation community as we had hypothesized.

The 2004 data also showed that the trend among the perennial species towards more salt-tolerant species observed in 2003 was not continued in 2004 (Figure 14). Perennial species that had previously decreased, hypothetically because of their intolerance to high salinities, including *C. stricta* and *L. oryzoides* were much more common in 2004. While the very salt-tolerant association of species, *S. alterniflora* and

E. palustris, was nearly absent from the marsh. The "island" of salt-tolerant vegetation which Transect 1 bisected disappeared in 2004.

It is possible that the high numbers of salt-tolerant perennials found in 2003 were an indicator of the previous year's high salinities, rather than an indicator of a long-term trend. Inland freshwater wetland vegetation community have been shown to be more related to historic plant assemblages than current environmental conditions (Seabloom et al. 2001). If TFMs are also strongly influenced by past vegetation assemblages (and therefore past environmental gradients), it is reasonable to hypothesize that the salt-tolerant vegetation observed in 2003 was still a response to the 2002 high salinity year, rather than a long-term affect of relative sea level rise.

Sweet Hall may shift between two states, a fresh state and a salty state, just as non-tidal marshes alternate between flooded and draw-down states (van der Valk 1981). In this way, it is understandable that what we captured in 2003 (and perhaps in 1987) was a snapshot of the vegetation community in a salty phase. With relative sea level rise, the marsh will likely spend more time in the salty state. This may cause the extirpation of strictly freshwater species (indeed, this has been the case at Sweet Hall since 1974 when strictly freshwater species such as *Nuphar lutea* were not found in the marsh). However, because of the strength of the tidal and riverine flow and the presence of freshwater marshes upstream, it is likely that the seed source for freshwater plants will still be present.

These results compliment a wealth of research that has focused on whether tidal freshwater marshes can transition to mesohaline marshes or whether they will revert to open water or mudflat (McKee and Mendelssohn 1989, Flynn et al. 1995, Howard and

Mendelssohn 1999a and 1999b, Howard and Mendelssohn 2000). Greenhouse experiments suggest that tidal freshwater marshes may be able to remain vegetated (McKee and Mendelssohn 1989). Though later studies found that recovery was limited when high salinities were maintained for long time periods (3 months) (Howard and Mendelssohn 1999a, 1999b). The findings in 2003 and 2004 at Sweet Hall suggest that the freshwater vegetation is able to recover even from long duration (1 year) high salinity events (such as the 2002 event), but the recovery may take more than one growing season.

Landscape level studies (Visser et al. 2002, Higinbotham 2004) also support the hypothesis that in response to surface salinity increases, freshwater marsh vegetation communities can rapidly shift towards more salt-tolerant communities rather than becoming open-water systems (Higinbotham et al. 2004). Higinbotham et al. (2004) found that the boundaries of the fresh-to-brackish and brackish-to-salt marsh types shifted over a 50-year span. Rather than consistently shifting upstream, as would be expected in response to relative sea level rise, the marsh types shifted up and down stream perhaps in response to precipitation (and therefore salinity) (Higinbotham 2004). What we documented in Sweet Hall may have been a similar shift but captured at a smaller scale. Our results further suggest that the marsh's response to slight salinity increases associated with RSLR may be obscured by the much wider salinity variations due to yearly climatic differences. To be able to tease out the differences between inter-annual variation and long-term change, more frequent studies are needed.

CONCLUSIONS

This study found long-term and inter-annual variation within the annual and perennial components of the plant community at Sweet Hall Marsh. Comparison of 2003 vegetation data with previous studies suggested a shift towards more salt-tolerant vegetation. Salt-tolerant vegetation dominated in some isolated patches, perhaps in response to death of freshwater vegetation in those areas due to salinity stress and an additional stress (such as grazing). River salinity data confirmed that plant community is occasionally exposed to high salinities, but it did not reveal evidence of increasing salinities due to relative sea level rise. A follow-up vegetation survey suggested that large numbers of salt-tolerant species found in 2003 may have been a short-term response to the high-salinity event in 2002, rather than an effect of gradual salinity increases associated with RSLR. Data also suggested that annual and perennial components of the vegetation community are highly variable on an inter-annual scale. The vegetation community may alternate between salt and fresh states in response to inter-annual variation in river salinity. The dynamic nature of the vegetation community of TFMs as shown in this study and Higinbotham et al's (2004) landscape level study lead us to speculate that TFMs will remain vegetated with rising sea levels. To document responses to RSLR, continuous salinity measurements and more frequent (yearly) vegetation surveys are necessary to separate long-term trends from inter-annual variation. More frequent studies will also allow monitoring of the possible spread of *Phragmites australis*. The National Estuarine Research Reserve has established emergent plant

biomonitoring at other reserve sites; Sweet Hall would be an excellent site for expansion

of this program.

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