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A Study of Sediment Accretion Dynamics in Mature and Restored Tidal Freshwater Forested Wetlands in the James River Watershed using Surface Elevation Tables and Marker Horizons

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A thesis submitted in partial fulfillment in the requirements for the degree of Master of Science in Environmental Studies at Virginia Commonwealth University.

by

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Abstract

A STUDY OF SEDIMENT ACCRETION DYNAMICS IN MATURE AND RESTORED TIDAL FRESHWATER FORESTED WETLANDS IN THE JAMES RIVER WATERSHED USING SURFACE ELEVATION TABLES AND MARKER HORIZONS

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Sediment accretion and elevation change in tidal forests, and the corresponding ability of these wetlands to keep pace with sea-level rise (SLR), represent data gaps in our understanding of wetland sustainability. Surface Elevation Tables and marker horizons were installed in three mature tidal forests and a restored tidal marsh, allowing us to measure elevation change, accretion, and subsidence. Additionally, we measured predictor variables to test for their significance in explaining accretion and elevation change rates. Mean accretion at our sites was 11.67 +/- 3.01 mm yr^{-1} and mean elevation change was -20.22 +/- 8.10 mm yr^{-1}, suggesting subsidence occurring beneath the sites. Processes contributing to accretion and elevation change at our sites may be driven by hydrologic patterns. Comparing our elevation trends with SLR trends suggests that our study sites may not keep pace with SLR. However, we may be observing short-term oscillations that do not indicate true long-term trends.

Introduction

Climate change resulting from anthropogenic activity is the source of many global phenomena that will have direct impacts on not only the human race, but on the entire biosphere. An especially significant consequence of climate change is the acceleration of eustatic sea-level rise, which, when augmented by regionally variable land subsidence, will be a determinant factor dictating the survivability of tidal wetlands (Goodwin et al., 2001; Craft et al., 2009; Torio & Chmura, 2013). Tidal freshwater forested wetlands, specifically those in the James River watershed, may be especially vulnerable to sea-level rise, in part due to their position at the head of estuaries. Whereas downstream marshes have space to migrate inward in the face of sea-level rise, many tidal forests in the James River watershed are already pressed against the fall line, limiting their ability to migrate inland. If tidal forests in the James River watershed are to survive sea-level rise, they will need to grow vertically at a pace faster than that of sea-level rise.

Understanding how tidal freshwater forested wetlands will respond to sea-level rise is critical, as these ecosystems provide many unique ecosystem services. While their position at the head of estuaries potentially imperils tidal forests in the James River watershed, it also allows them to protect connected downstream waterways from upstream nutrient loading (Craft, 2012; Ensign et al., 2014). Additionally, tidal forests are also well positioned to supply organic material to feed connected downstream trophic webs (Craft, 2012). Tidal forests abate large amounts of floodwater, protecting connected urban environments from storm events, and provide unique wildlife habitat to numerous species. Despite benefiting directly and indirectly from many of these ecosystem functions, we understand relatively little about the accretion and elevation change dynamics in tidal freshwater forested wetlands (Craft, 2012; Ensign et al., 2014).

Historically, tidal wetlands have been able to keep pace with rising sea levels through self-regulating mechanisms of vertical sediment accretion and elevation change (Morris et al., 2002). Vertical sediment accretion is the vertical gain in surface elevation as a function of sediment deposition factors measured against removal processes (Morris et al., 2002; Neubauer et al., 2002; Neubauer, 2008; Weston et al., 2011). Shallow subsidence is elevation loss due to subsurface processes, such as root decomposition, compaction, and soil shrinkage. Elevation change is the total net vertical gain or loss, including both surface accretion/erosion and belowground processes, such as soil expansion and root growth, as well as subsidence factors (Stagg et al., 2016).

Deposited sediment is a combination of organic matter (OM) and mineral sediment, with relative contributions dependent on region, location within the watershed, proximity to a sediment source and estuarine turbidity maxima, net primary productivity (NPP) within the wetland, hydrology and tidal regime. Sediment deposition can be derived from allochthonous sources, such as riverborne suspended sediment and associated OM, and from autochthonous production of OM that is dependent on the dominant macrophyte community and NPP within the wetland (Cahoon & Reed, 1995; Neubauer et al., 2002; Neubauer, 2008). Aboveground vegetation is responsible not only for in situ production of OM, but also trapping riverborne sediment and slowing down flow velocities, allowing suspended sediment to settle on the wetland surface (Gleason et al., 1979; Mudd et al., 2010; Baustian et al., 2012).

Alternatively, sediment respiration, compaction, and shrinkage can affect subsidence, limiting net elevation change (Cahoon, 2006; Cahoon et al., 2011). Salinity can reduce NPP in freshwater wetlands, limiting in situ production of the OM component to elevation change and reducing aboveground surface vegetation critical in sequestering riverine sediment (Spalding & Hester, 2007; Weston et al., 2011; Noe et al., 2013; Morrissey et al., 2014). Additionally, increases in salinity and subsequent increased decomposition rates of organic matter due to introduction of sulfates may contribute to shallow subsidence (Craft, 2007; Ensign et al., 2014). Root decomposition can also be accelerated by salt-water intrusion and can limit net elevation gains (Cahoon, 2006; Craft, 2007; Crawford et al., 2007; Kirwan & Guntenspergen, 2012).

Rapid changes in hydrology, such as those caused by storm events, influence elevation change. Storms influence both surface and subsurface processes, and effects can be both short-term and long-term (Cahoon, 2006). Subsidence and compaction must be considered when measuring storm surge effects, as sub-surface subsidence often offsets partially or completely the surface accretion resulting from a storm surge (Cahoon, 2006). A more recent study examined the impacts of hurricane-induced sedimentation in the marshes of Barataria Bay in the Louisiana Mississippi River delta, concluding that accretion resulting from storm deposition was often offset by subsequent compaction under the weight of the newly deposited sediment (Baustian & Mendelssohn, 2015). The 2015 study also identified a possible feedback mechanism wherein storm induced sediment deposition and associated nutrient introduction could increase localized NPP. Such a feedback may have long-term effects on elevation change, as greater aboveground vegetation density could improve aboveground sediment sequestration, increasing positive elevation growth in the long-term (Baustian & Mendelssohn, 2015). Wetlands in which elevation growth is OM limited will require enhanced NPP and/or mineral sediment deposition to offset OM decay rates (Kirwan et al., 2013).

Complex interactions of many elements mean that tidal wetlands will be variably sensitive to relative sea-level rise (RSLR) based on number of factors. Position within the watershed, land use, proximity to sediment sources, dominant plant communities and their tolerances to inundation and salinity, NPP within the wetland, suspended sediment concentration available to sequester, and relative elevation interact to govern wetland accretion and elevation change dynamics (Chmura & Hung, 2004; Neubauer, 2008; Ensign et al., 2014). As a result of these complex dynamics, it is difficult to confidently predict how a particular wetland might respond to sea-level rise. Thus, the ability of tidal wetlands to keep pace with accelerated sea-level rise is uncertain (Neubauer, 2008; Weston et al., 2011).

Studies in Marsh Ecosystems

The majority of elevation change and accretion studies in coastal wetlands have focused on marsh ecosystems. Saltwater, freshwater, and brackish marshes have all been relatively well studied, and it has been suggested that marsh response to sea-level rise varies, with brackish marshes faring better than fresh and salt-marshes (Craft et al., 2009). Some research proposes that, while salt marshes existing at a lower, "optimal" elevation may have a higher NPP and be more frequently flooded, salt marshes with higher present elevation yet lower NPP may be more stable in the face of SLR (Cahoon & Reed, 1995; Morris et al., 2002). More recently, studies of elevation change and accretion in New Hampshire salt marshes focused on positive feedback mechanisms wherein wetlands increase pace of vertical growth in response to increased flooding. Kirwin et al. (2016) argue that marsh vulnerability to SLR may be exaggerated, and that most marshes tend to keep pace with SLR. Their results suggest that low marsh sites, through positive feedback mechanisms, accrete sediment significantly faster than high marsh sites, particularly where suspended sediment concentration (SCC) >~30 mg/L and where tidal flux > 1 meter (Kirwan et al., 2010, 2016). With more frequent flooding and subsequent sediment deposition expected with predicted accelerated rates of SLR, accretion and resultant elevation change occurring at present-day low marsh sites may be more representative of how wetlands will respond to future rates of SLR than present-day high marsh sites (Kirwan et al., 2016).

Studies in Tidal Freshwater Forested Wetlands

Accretion and elevation change dynamics remain poorly studied in tidal freshwater forested wetlands (Craft, 2012; Ensign et al., 2014). There exists more tidal freshwater forested wetland acreage in the United States than tidal freshwater marsh; these ecosystems may be especially vulnerable to RSLR (Ensign et al., 2014).

The limited literature that does exist suggests that tidal freshwater forested wetlands accrete sediment at a slower rate than freshwater and brackish marshes (Craft, 2012; Ensign et al., 2014; Stagg et al., 2016). Such trends suggest the likely reduction in tidal freshwater forested wetland acreage along with the expansion of oligohaline and brackish marsh that exhibit higher accretion rates than current RSLR (Craft, 2012, Ensign et al., 2014). A recent study examining elevation change along a gradient from tidal freshwater forested wetland to tidal freshwater marsh found that many of the tidal forest study sites were resilient to SLR, though some only marginally so (Stagg et al., 2016). While these data contrast with the Craft (2012) and Ensign et al. (2014) studies, where tidal forest study sites did not appear to keep pace with SLR, the Stagg et al. (2016) study does support that oligohaline marshes are more resilient than tidal forests and will likely expand. The lower accretion rates typical of tidal forests, coupled with expansion of brackish and oligohaline marsh, would mean that many tidal forests would need to migrate upland to survive sea-level rise. If development and coastal squeeze prohibit that retreat, we will lose ecosystem services specific to tidal freshwater forested wetlands as they transition to marsh habitats (Pezeshki et al., 1990; Craft, 2012; Torio & Chmura, 2013).

Studies in Restored Wetlands

In addition to threatening tidal freshwater forested wetlands directly through logging, and indirectly through fragmenting paths of landward migration, accelerated development along the eastern seaboard will put increasing pressure on all coastal wetlands. Mitigation banking works to reproduce lost wetland acreage on an acre-to-acre ratio, but fails to take into account ecosystem services specific to particular wetland types, as well as failing to address the potential for lost functionality and inferior accretion dynamics in these reproduced ecosystems. The ability of constructed and restored forested wetlands to replace ecosystem services lost through anthropogenic wetland disturbance remains low (Crawford, 2002; Crawford et al., 2007).

Recent meta-analyses suggest that salt marshes are keeping pace with sea-level rise, and that most instances where they do not are typically a result of anthropogenic activity modifying sedimentation dynamics, such as with impaired wetlands undergoing restoration (Kirwan et al., 2016). However, in the case of salt marshes, even restored marshes generally accrete at a rate fast enough to keep pace with sea-level rise. A study of constructed *Spartina alterniflora* marshes showed that, while C accumulation was greater in mature marshes, sediment deposition rates were generally equal in both mature and restored marshes once the *Spartina alterniflora* was established (Craft et al., 2003). Elevation change and accretion dynamics in restored tidal forests, however, are understudied. Subsidence factors in restored Atlantic white cedar forests have been studied, with results indicating that root decomposition in restored sites occurs at faster rates than in reference (mature) sites (Crawford et al., 2007). As root decomposition has been shown to be an important factor limiting OM accumulation and preservation of coastal wetlands (Craft, 2007), this evidence suggests a need for thorough studies of elevation change dynamics in restored tidal forested wetland types.

The Scope of Our Study

At the VCU Rice Rivers Center in Charles City County, Virginia, we have a unique opportunity to compare accretion and elevation change rates in a restored wetland with those at a reference site, the adjacent mature tidal freshwater forested wetland at Harris Creek. Subsequently, our four study locations at Kimages Creek, Harris Creek, Presquile National Wildlife Refuge, and James River National Wildlife Refuge represent a novel opportunity to address data gaps concerning the accretion and elevation change in mature tidal freshwater forested wetlands and in a restored wetland that originally existed as a tidal freshwater forested wetland. These data gaps have important implications for wetland sustainability, as well as policy implications for best management and future restoration of this wetland type.

In order to address these data gaps, we have installed 18 Surface Elevation Tables (SETs) coupled with feldspar marker horizons in the restored wetland along Kimages Creek and within the mature tidal forest along Harris Creek at the VCU Rice Rivers Center, as well as within tidal forests in Presquile and James River National Wildlife Refuges. These SETs represent the first to be installed in tidal freshwater forested wetlands in the lower Chesapeake Bay and James River watersheds. In addition to allowing us to begin monitoring elevation change in tidal freshwater forested wetlands in this area, our array of SETs will be an important contribution to the NOAA Chesapeake Bay Sentinel Sites Cooperative dataset. Subsequently, this long-term study will build on the existing body of knowledge to advance the future development of protocol for wetland creation and restoration, as well as best management practices to promote the conservation, preservation, sustainability and ecological integrity of mature tidal freshwater forested wetlands.

Objectives

Sediment accretion and the corresponding ability to keep pace with RSLR in both mature tidal freshwater forested wetlands and restored wetland sites represent significant data gaps in the current body of literature pertaining to wetland sustainability, and have important policy implications concerning the best management of wetlands. In order to address these data gaps, this study measured contemporary elevation change and accretion rates in three mature tidal freshwater forested wetlands, and a tidal freshwater marsh currently undergoing restoration. The objectives of this study were as follows:

1. Assess the sustainability of tidal freshwater forested wetlands within the lower Chesapeake Bay and James River watersheds by measuring and comparing contemporary rates of sediment accretion and elevation change in three tidal freshwater forested wetland sites along the James River in Virginia; Harris Creek at VCU's Rice Rivers Center, Presquile National Wildlife Refuge (PNWR), and James River National Wildlife Refuge (JRNWR).
2. Assess the success and sustainability of the Kimages Creek wetland restoration in the context of elevation change and sediment accretion by measuring contemporary elevation change and accretion rates in the restored site and comparing the mean elevation change

and accretion rates with those at corresponding locations in an adjacent benchmark site, the mature tidal freshwater forested wetland at Harris Creek.

3. Take measurements of primary components in tidal freshwater wetland ecosystems governing accretion rates and their relative significance in dictating accretion and elevation change. This was addressed via aboveground vegetation surveys, measuring suspended sediment concentration within the channels, measuring tidal inundation height, and measuring distance from our sites to the sediment source.

Methods

Site Descriptions

Presquile National Wildlife Refuge

Presquile National Wildlife Refuge is a 1,329-acre island on the James River in Chesterfield County, Virginia. Presquile was previously a mass of land surrounded by an oxbow in the James River, becoming an island in 1934 when the Army Corp of Engineers cut the channel at the southern edge of the island to facilitate navigation (Figure 1). In 1952, the U.S. government took control of Presquile and the Department of the Interior created the refuge on the island (Parker & Wyatt, 1975).

Tidal freshwater forested wetlands dominate Presquile, covering around 60% of the refuge. Frequent inundation and saturated conditions perpetuate the existence of OM-rich, mucky soils consistent with anoxic substrate. Previous studies have stated that dominant plant species in the Presquile NWR tend to be flood-tolerant trees, with *Fraxinus spp.* (ash) and *Nyssa sylvatica* (black gum) being the most widely distributed dominant species (Parker, 1977). *Acer rubrum* (red maple), *Taxodium distichum* (bald cypress) and *Nyssa aquatica* (tupelo gum) are also relatively abundant species (Parker, 1977; USFWS, 2013).

Due to the dense tree canopy, shrubs and herbaceous understory species are more sparsely distributed. *Saururus cernuus* (lizard's tail), *Peltandra virginica* (arrow arum), *Polygonum arifolium* (hastate tearthumb) were common, and *Murdannia keisak* (Asian spiderwort) was especially abundant in our surveys.

James River National Wildlife Refuge

Located along the James River in eastern Prince George County, Virginia, James River National Wildlife Refuge (JRNWR) is a 4,325-acre refuge that was established in 1991. Relatively close in proximity to both the Rice Rivers Center and Presquile National Wildlife Refuge, the tidal freshwater forested wetlands in JRNWR closely mimic those of Harris Creek and Presquile (Figure 1). Dominant species of vegetation in the tidal forest include *Fraxinus pennsylvanica* (green ash), *Nyssa sylvatica* (black gum), *Taxodium distichum* (bald cypress), and *Acer rubrum* (red maple) (USFWS, 2013).

Harris Creek

Just to the west of the Kimages Creek drainage on the VCU Rice Rivers Center property in Charles City County, Virginia, the adjacent Harris Creek is a mature tidal forest that represents a reference and benchmark against which the Kimages Creek wetland restoration success is measured (Figure 2). Dominated by *Acer rubrum* (red maple), *Fraxinus pennsylvanica* (green ash), *Taxodium distichum* (bald cypress), Harris Creek exemplifies an ideal result of the Kimages Creek wetland restoration.

Kimages Creek

Kimages Creek is a second order coastal plain stream in a previously forested tidal freshwater wetland, but has been logged at least twice, once in 1862 and again in 1927, and was dammed in 1927 to create Lake Charles, a 72-acre impoundment. After VCU took possession of the property to create the Rice Rivers Center field station, the dam was naturally breached by a storm event in 2006, and then partially removed in 2010 to reestablish tidal communication between Kimages Creek and the James River (Bukaveckas & Wood, 2014). The wetland currently exists as a freshwater tidal marsh and is maintained as a restored wetland; dominant species include *Typha angustifolia* (narrow leaf cattail), *Murdannia keisak* (Asian spiderwort), *Polygonum sagittatum* (arrowleaf tearthumb), *Leersia oryzoides* (rice cutgrass), *Juncus effusus* (softrush), *Pontederia cordata* (pickerelweed), and *Sagitaria latifolia* (broadleaf arrowhead). Natural woody recruitment includes *Acer rubrum* (red maple), *Liquidambar styraciflua* (sweet gum), *Salix nigra* (black willow), and *Platanus occidentalis* (sycamore) (Bukaveckas & Wood, 2014).

Surface Elevation Tables

We used Surface Elevation Tables (SETs) to make repeated measures of surface elevation relative to fixed benchmarks within our study sites. Beginning in May 25, 2016, we installed an array of 18 SET benchmarks (see *Installation Procedure* pg. 63) in our study locations at Kimages Creek, Harris Creek, Presquile National Wildlife Refuge, and James River National Wildlife Refuge (Figures 1 and 2). The SET arm, the mobile portion of the SET system, was taken to each benchmark and attached to the receiver with clamps before taking measurements. We used a modern SET instrument consisting of two primary components; a vertical base that attaches to the benchmark receivers, and a horizontal arm through which run 9 pins. The arm was leveled along two axes before taking pin measurements (Figure 3).

Each SET was measured every 2 months (Appendix Table A). SET measurement procedures involved placing a portable aluminum platform across the SET location, also termed the "sampling station," supported on either side by two plastic step stools. Attached to the legs of the step stools were treated wood boards to prevent them from sinking into the soft substrate.

This mobile platform allowed us to measure the SETs without impacting the substrate surface within the sampling stations.

The SET arm was attached to the benchmark receiver, and locked into position before being leveled across two axes. This procedure was repeated for each of the 4 cardinal orientations we established at each SET. We took an elevation profile at each orientation by lowering the 9 pins to the sediment surface, and then locking the pins into place with clips. At this point, we measured the distance (mm) from the top of each pin to the top of the SET arm using a standard metal meter stick (Figure 4). We took the mean of the 9 pin measurements of each orientation as a subsample mean, for a total of (n=4) subsample means for each SET location per sampling round (Boumans & Day, 1993; Childers et al., 1993).

These subsample means were used for our within-wetland analyses, in which we tested for significant variation in elevation change between SETs using a single factor ANOVA (Boumans & Day, 1993; Childers et al., 1993). The grand mean (mean of the four subsample means) was used for our across-wetland analyses, in which we tested for significant variation in elevation change rates across the 4 wetlands using a single factor ANOVA (Boumans & Day 1993; Childers et al., 1993). We avoided using individual pin measurements as sub-samples as this may have resulted in spatial autocorrelation due the close proximity of adjacent pins.

When making pin measurements, the sediment surface was defined visually as the point where the bottom of the pin just made contact with the sediment surface. When twigs or leaf litter impeded contact between the pin and the substrate surface, it was moved by hand, the exception being when such organic matter was decomposing to the point of becoming part of the substrate matrix, in which case it was treated as the sediment surface. If larger objects, such as logs, or impressions/ divots appeared under an orientation in between sampling rounds, measurements proceeded as usual and a note was taken. When, due to timing, we sampled a SET while inundated by tidal flooding, the pin was lowered while, with one finger, we felt for the sediment surface. When, by tactile feeling, we could confirm that the pin made contact with the substrate surface, the pin was locked in place.

For the duration of this study, SETs were only measured by Ron Lopez. Human error was determined and corrected for by taking multiple test measurements at SETs 1 and 2 before officially beginning our measurements. Dr. Scott Neubauer was present for and assisted with these tests. In between each measurement, we completely dismantled and re-assembled the SET instrument and attached it to the benchmark. Error was reduced to +/- 2 mm by the time we finished our last test measurements at SET 2.

Marker Horizons

Feldspar marker horizons (MH) are often used in conjunction with SETs to determine (in addition to total elevation change relative to the SET benchmark) surface accretion and shallow subsidence occurring below the MH and above the bottom of the SET benchmark pipe, known as the zone of shallow subsidence (Figure 3)(Cahoon & Turner, 1989**;** Cahoon & Reed, 1995; Ensign et al., 2014; Stagg et al., 2016). Marker horizons were established at the zero measurements of the respective SETs and were measured at the same times as the corresponding SETs thereafter (Appendix Table A). This sampling regiment allowed us to compare accretion (MH) and elevation change (SET) from a baseline measurement; the zero measurement of the SET.

Before installing marker horizons, we established a 9 m^2 sampling station around the SET benchmark. A scaffolding of 1.27 cm PVC was erected around each sampling station, 9 m^2 in area, sitting roughly 30 – 60 cm off the ground, and with one border facing approximately north. The scaffolding was meant to deter hunters, researches, and students from stepping in the sampling station, and increase sampling station visibility. We established four marker horizons at each sampling station, generally in the corners of the sampling station (Figure 5). In circumstances where a log, hummock, or other obstacle prohibited corner installation, we offset the marker horizon by 50 cm in one direction along one edge of the sampling station perimeter.

We used G-200 EU white feldspar clay to establish our marker horizon plots. Plots were .25 m^2 and approximately 1 cm thick (Noe & Hupp, 2012). In some areas where *Murdannia keisak* mats were especially prominent, we used more clay to ensure the sediment surface was covered. As the clay absorbs moisture, it firms up to create durable marker horizons, and once they were buried by sedimentation we were able to measure accretion occurring on top of the horizons by coring though them and measuring from the bright white horizon created by the clay to the sediment surface (Cahoon and Turner, 1989; Noe & Hupp, 2012; Ensign et al., 2014).

The marker horizon measurement procedure was modified from the USGS-defined protocols and based on input from Dr. Nathanial Weston of Villanova, who employed a similar method of cryo-coring to measure marker horizons. This modified protocol used a 6.35 mm I.D. length of copper tubing with a .30 caliber bullet soldered to one end, hereafter termed the "bullet." The casing and gunpowder were removed from the bullet prior to soldering. The bullet was inserted into the marker horizon to a depth of approximately 7.62 – 10.16 cm. The funnel apparatus consisted of a 7.62 cm diameter, 0.46 m length segment of PVC with a plastic nipple at one end. The plastic nipple was attached to a 6.35 mm O.D. length of vinyl tubing. Connections were sealed with plumbers cement and silicone.

Facing north, we sampled each marker horizon from left to right and top to bottom, thus avoided coring the same area twice. With the bullet in the marker horizon plot, we inserted vinyl tubing from the funnel apparatus into the bullet. We then placed a standard plastic funnel into the top of the PVC funneling apparatus and began pouring in liquid N_2 from a 5 liter dewar. As the nitrogen burned off, we added more until the copper was frosted over. At this point, we pulled the core, which when extracted resembled a "sediment popsicle" around the bullet.

With a "good horizon," the sediment surface and the bright white marker horizon would be well defined, allowing us to take mm-accuracy measurements of sediment accretion on top of the marker horizon. We took 4 measurements (mm) around the circumference of the core using a standard metal meter stick, the mean of which was the marker horizon subsample for that plot. Ron Lopez took all measurements. We attempted to core each of the 4 marker horizons at each SET location per round of sampling, however there were instances where we did not pull a measurable core for each marker horizon. The mean of all measurable marker horizons at a SET location was taken as the MH sample for that round of sampling.

Aboveground Vegetation Density

Aboveground vegetation density surveys were conducted at peak growing season in late July through early August 2016, and again in late January through early February 2017. These surveys consisted of species identification, percent cover, and stem count. Our surveys were non-destructive, as we did not remove any vegetation from the plots. Maintaining the integrity of the vegetation present was important for mitigating disturbance of the natural sediment retention within the sampling station, as aboveground vegetation has been shown to be a critical component in sequestering riverine sediment (Gleason et al., 1979; Baustian et al., 2012).

In order to randomize our samples, we divided our sampling stations into eight 1 m^2 subplots, not including the center subplot where the SET benchmark lies. When facing north, the subplots (henceforth referred to as "plots") were numbered 1-8, moving from left to right, and top to bottom (Figure 6). We then used a random number generator to select three replicate plots for each sampling station to be surveyed. The same three plots from each sampling station were sampled during the summer and winter surveys. Each survey began with species identification.

Percent coverage was quantified by looking down at the plot, visually splitting the plot into 4 quadrats, and approximating what total percentage of the plot was covered by vegetation of each species. Percent bare ground and percent open water were also recorded.

Stem count was also specific to species, however all species were included for our predictor variable, which was defined as stem density per unit area (stem count per 1 m^2). Ron Lopez conducted all vegetation surveys.

Total Suspended Solids

Concentrations of suspended sediment in Kimages Creek, Harris Creek, Powells Creek, and the southeastern creek at Presquile were measured via sampling for Total Suspended Solids (TSS). TSS provides a measure of sediment present in each creek channel and available to be deposited at our sampling stations. During our study period, we sampled for TSS twice, during the third and fourth SET sampling periods. Sampling occurred while we sampled the SETs in a particular channel location, typically taking the upstream samples after finishing the upstream SET(s), and taking the downstream samples as we left the creek channel.

We chose this sampling method as it is a standard and proven protocol for quantifying TSS in streams, and we were equipped to process the samples in the Environmental Analysis Laboratory at VCU. By taking replicates at the far northern and southern reaches of our sampling stations along a given creek, we could generate a mean TSS quantity that was representative of suspended sediment concentration throughout the channel.

We sampled each stream by dipping 1-liter, wide-mouth, lab quality, opaque Nalgene bottles into the stream surface at randomly selected locations just north of the northernmost SET(s) and just south of the southernmost SET(s) in that particular wetland for each sampling round. Three replicate samples were taken in the upstream and three in the downstream, for a total of (n=6) samples per wetland for each sampling round. The samples were filtered within 6 hours of being taken from the stream. After filtering the samples, the Nalgene bottles were thoroughly rinsed with D.I. water and dried in preparation for further sampling.

Prior to filtering samples, we prepared the fiberglass filter paper (Pall Corp.) by rinsing them with three 30 ml washes of D.I. water while in a vacuum flask. When water was no longer dripping from the filter, we removed it with forceps, placed it in an open petri dish, and placed the dish in a drying oven at 70°F for 48 hours. The filter papers were removed and placed in a desiccator for 1 hour to cool. The papers were weighed with a scientific scale, the weights recorded, and then they were placed back in the oven for another hour. The papers cooled for 1 hour in a desiccator, and they were re-weighed. If the change in weight was less than 4% of the initial weight, the papers were labeled, stored in the desiccator, and their final weights recorded.

Upon returning from field collection, samples were filtered through the aforementioned filter papers using a vacuum flask. After agitating the sample thoroughly within the Nalgene bottle, a measured portion was poured into a graduated cylinder before being run through the vacuum flask. A portion between a minimum of 50 ml and a maximum of 200 ml of each sample was used, and the volume was recorded. After the sample was run through, three 30 ml washes of D.I. water were run through the sample. This procedure was repeated for all samples.

In between individual samples, all vessels and components of the vacuum flask were rinsed with D.I. water. When the filter paper with the filtered sample residue was dry, it was removed from the vacuum flask with forceps, placed in its petri dish, and the dish was placed in a drying oven at 70°F for 48 hours. The filter papers with residue were removed, the dishes closed, and placed in a desiccator for 1 hour to cool. The filtered samples were weighed with a scientific scale, the weight recorded, and then they were placed back in the oven for another hour. The filtered samples cooled for 1 hour in a desiccator, and they were re-weighed. If the change in weight was less than 4% of the initial weight, the filter papers with sample residue were labeled, stored in the desiccator, and the final weights recorded.

The difference between the weight of the filter with the dried sample residue and the clean filter gives us the per unit volume measure of TSS in mg/L using the formula:

$$TSS = \frac{(A - B)1000}{C}$$

A = Weight of filter + residue (mg)

B = Weight of filter (mg)

C = Volume of sample (ml)

We took the mean of each set of 3 replicates from the northern and southern extents of the sampling range, giving us two subsample means for each stream per round of sampling. The average of those two subsample means was used as our sample for that round of measurements.

Tidal Inundation Extent

In order to quantify a measure of opportunity for suspended sediment to be delivered to study sites via tidal inundation, we measured the height of the tide at our sampling stations following high tide. A greater water column coming into contact with aboveground vegetation offers a greater opportunity for TSS to both exist above the sediment during a given tide, and to be subsequently trapped by contact with the vegetation or to settle on the surface due to the slowing of flow velocity of the water (Mudd et al., 2010).

Measurements were taken during the third and fourth rounds of SET sampling (Table 1). We used four 1.27 cm PVC posts in each sampling station, standing approximately 1 meter above sediment surface, to measure the tidal inundation height. Following the high tide, we measured from the sedimentation line left on the PVC to the sediment surface (cm) using a standard metal meter stick. Ron Lopez took all measurements. In most instances, we could read a very clear sediment line on the PVC, and we took the average of the 4 measurements as our sample mean. For the second round of measurements, we prepped the PVC by roughing a vertical strip along the length of the PVC with a file. This allowed sediment to easily adhere to the PVC on the clean, roughened strip, and enabled us to easily read the second round of measurements.

Distance to Sediment Source

ArcMap GIS software was used to identify and calculate the nearest distance from each SET to its respective creek channel center. This distance represents a spatial measure of how far sediment needs travel from sediment source to our sites, and serves a proxy for hydroperiod, as distance from the creek impacts the length of time the sites are flooded during tides. ArcMap

GIS software provides a simple and accurate method for locating and calculating the distance from each SET to the nearest point on the creek bank and creek center.

Using a Trimble GPS Series 7 unit to create a shapefile with geo-referenced points of each of the SET benchmarks, we projected that shapefile over a spatially referenced aerial photograph of the study locations. By projecting the geo-referenced SET locations together with the aerial photography, we created a spatially correct geo-referenced map.

We then digitized lines representing creek bank and creek center on the map for each of the 4 creek systems. By using the ArcMap "Near Tool" for all of the SETs in each wetland relative to the respective digitized lines representing the creeks, the software automatically locates the nearest distance between the SETs and the creek, outputting the distance in meters.

Sea-Level Rise Data

As we were not able to locate long-term water level data from functioning tide gauge stations in the nearby upstream or downstream vicinity of our study locations, we used monthly average SLR data obtained from the NOAA tide gauge at Sewells Point, Norfolk, Virginia. Sewells Point lies approximately 66 river miles downstream of the VCU Rice Rivers Center. NOAA used least squares linear regression to generate a SLR trend of 4.61 +/- 0.23 mm yr^{-1} (Boon, 2005). In order to generate a more localized trend of water level change, we used data from the pier at the VCU Rice Rivers Center (15 minute intervals, NAVD 88 datum) to generate trends of SLR along the James River in the immediate area of our study sites. While the Rice Rivers Center tide gauge is closer in proximity to our sites than the NOAA Sewells Point tide gauge, the Rice Rivers Center data is of a much shorter temporal scale than the 120 year dataset from Sewells Point, so we chose to compare our elevation change rates and accretion rates with SLR trends generated from both datasets.

Statistical Analyses

All analyses were performed in R with an α level of 0.05. Rates of change for all SETs (elevation change) and marker horizons (accretion) were generated via least squares linear regression, as per standard protocols for studying elevation rate differences based on non-temporal effects such as wetland location or type (Callaway et al., 2013). These rates would be used for all subsequent analyses of variation in elevation change rates and accretion rates. Welch's T-Test was used to test for significant differences between accretion and elevation change rates across all study sites. We used a single factor ANOVA to test for significant variation among mean accretion rates across all wetland sites. We also used a single factor ANOVA to test for significant variation among mean elevation change rates across all wetland sites. Single factor ANOVA was used to test for variation in mean elevation change rates within each wetland due to SET location, as well. Multiple regression was used to test for the relative statistical significance of the potential drivers of elevation change rates and accretion rates. Least squares linear regression was used to generate SLR trends following protocols set forth by NOAA (Boon, 2005).

Results

Within forested wetlands in the lower James River watershed, accretion rates (Mean=1.98, SD=1.73) varied significantly from elevation change rates (Mean=-3.16, SD=3.90)(Welch's T-test, $F(23.44)=-5.12$, $p=0.00003$ (Table 1). Generally, accretion rates trended towards the positive, while elevation change rates trended towards the negative. The exception occurred in Harris Creek, where the majority of the sampling stations exhibited both positive accretion and elevation change rates (Table 2).

We found statistically significant variation among mean elevation change rates across all wetland sites (ANOVA, $F(3,4)=4.49$, $p=0.01$)(Figure 7). Mean accretion rates among wetland sites did not vary significantly from one another (ANOVA $F(3,14)=1.02$, $p=0.41$)(Figure 7). Rates based on 4 repeated measurements for SETs and 3 repeated measurements for MHs, taken at 2-month intervals over an 8-month period, were projected to an annual rate (Figure 8).

Kimages Creek

Kimages Creek elevation change trends were negative over the course of this study, with a total mean elevation change rate of -35.32 mm y^{-1} (Table 2, Figure 8). The majority of SETs at Kimages Creek and the other wetland locations showed a similar pattern in elevation gain and loss; after the zero measurement, the second measurement was relatively high, followed by a subsequent drop in elevation over the final two measurements (Figures 9 and 10, respectively).

Overall trends for elevation change at Kimages Creek were negative, with SETs 6 and 11 showing the highest rates of elevation decrease (Figure 9). Elevation change rates within Kimages Creek varied significantly (ANOVA, $F(5,18)=36.36$, $p<0.0001$) with SET 6 and SET 11 varying significantly from SETs 4, 5, 7, and 12 (Figure 11). The rates of loss at SETs 6 and 11 were the most rapid rates of elevation loss observed at any of the sites during the study period. Annual mean rate of elevation loss for SETs 6 and 11 combined was -71.49 mm yr^{-1}, when SETs 6 and 11 were excluded from mean elevation rate of all Kimages Creek SETs, the annual mean elevation rate was -12.40 mm yr^{-1}.

Overall Kimages Creek had the greatest subsidence rates of any of the studied wetland ecosystems. All accretion rates at Kimages Creek, with the exception of those measured at the SET 7 marker horizons, showed a positive trend (Figure 10). The average rate of accretion across the Kimages Creek sampling stations was 12.12 mm y^{-1}.

Harris Creek

Overall, Harris Creek had the highest rates of positive elevation change of any of the wetland study locations (Table 2). Total mean elevation change at Harris Creek was 2.4 mm yr^{-1}. The only two SETs that showed an elevation loss at Harris Creek during the study period were SETs 1 and 2, which occur at the southern extent of the eastern bank of Harris Creek (Figure 12). Elevation change rates did not vary significantly from one another within Harris Creek (ANOVA, $F(5,18)=0.47$, $p=0.80$). The marker horizons in Harris Creek all showed positive rates of accretion, with a total mean accretion rate of 12.6 mm yr^{-1} (Figure 13).

Presquile National Wildlife Refuge

Presquile NWR exhibited patterns of positive accretion and negative elevation change. Elevation change rates in Presquile all showed a negative trend, with elevation at each SET dropping at a similar rate, and a total mean trend of -27.06 mm yr^{-1} (Figure 14). Elevation change did not vary significantly between sampling stations within Presquile (ANOVA, $F(2,9)=0.55$, $p=0.60$). SET 9, the northernmost sampling station in Presquile, showed a negative trend in accretion for the duration of the study (Figure 15). SETs 8 and 10, lying south of SET 9, showed positive accretion rates (Figure 15). Mean accretion at Presquile was 3.69 mm yr^{-1}.

James River National Wildlife Refuge

All sampling stations in James River NWR had a negative trend in elevation change and a positive trend in accretion rates (Figures 16 and 17, respectively). Mean elevation change in JRNWR was -20.89 mm yr^{-1}, and mean accretion was 18.29 mm yr^{-1} (Table 2). Elevation change varied significantly among SETs within James River NWR (ANOVA, $F(2,9)=4.96$, $p=0.04$)(Figure 18).

Multiple Regression of Predictor Variables

Across all sites, accretion was best explained by vegetation density ($p=0.02$) and TSS ($p=0.04$), followed by distance to sediment source ($p=0.25$) and tidal inundation height ($p=0.76$), however, our model only explained 14% of variability ($F(4,13)=1.68$, $p=0.21$, $R^2=0.14$). Elevation change rates across all sites were not well explained by any of our variables. There were no significant explanatory variables of accretion rates or elevation change rates in our multiple regressions when limited to only Harris Creek and Kimages Creek data.

Wetland Elevation Change Compared with RSLR

Total mean accretion rates at our wetlands showed positive aboveground sediment accumulation, with a total mean accretion rate of 11.67 mm yr^{-1} across all sites. Elevation change rates, with the exception of Harris Creek, were negative, with a total mean elevation change rate of -20.22 mm yr^{-1} (Table 2). These tidal wetlands are sequestering and accreting sediment on the soil surface; however, elevation is being lost at a faster rate through subsurface processes (Figure 8). With the exception of Harris Creek, accretion within all wetland sites is not occurring at a rate fast enough to offset elevation loss.

NOAA estimates, based on the 120-year dataset from the tide gauge at Sewells Point, Virginia, that RSLR for coastal Virginia is 4.61 +/- 0.23 mm y^{-1} (Figure 19). Our rates of change suggest that, while accretion is occurring at rates fast enough to keep pace with RSLR in this area, total negative elevation change suggests that these wetlands do not keep pace with local RSLR.

Using least squares regression to calculate a trend for water level change at the Rice Rivers Center pier, our trend for water level change suggested a -2 mm yr^{-1} *drop* in water level for the time period between June 2015 and December 2016 (Figure 20). Shortening the NOAA Sewells Point dataset to the same temporal scale resulted in a trend of -130.94 mm yr^{-1}.

Discussion

These SETs represent the first to be installed in tidal forests in the lower James River watershed, and these data represent the beginning of a long-term study examining sediment accretion and elevation change within the lower James River watershed. Using SETs coupled with marker horizons is widely considered to be the most precise method for measuring elevation change, accretion, and subsidence (Boumans & Day, 1993; Webb et al., 2013). By measuring our SETs and marker horizons at 2-month intervals, we have established a high-resolution dataset that has revealed seasonal patterns in variability in accretion, elevation change, and subsidence. These patterns provide insight into the driving factors in elevation change dynamics in the lower James River watershed, and uncover new questions and further research opportunities to enhance our understanding of how tidal forests in this area will respond to sea-level rise.

Overall, rates of change in elevation (negative trend) varied significantly from accretion rates (positive trend) throughout our sites. Most sampling stations exhibited positive accretion throughout the study period, coupled with a simultaneous loss in elevation. The deficit between elevation change and accretion observed during this study period suggest that shallow subsidence is occurring beneath the marker horizons.

The general pattern of elevation loss observed at most sites is likely due to a combination of several factors. Winter, 2017 in Virginia was relatively mild, with warmest February temperatures on record for the state (NOAA, 2017). High rates of elevation loss during the winter sampling period may be due in part to increased rates of microbial decomposition of organic matter resulting from a mild winter (Burdick & Peter, 2015). This may have been compounded by decreased winter accretion rates, possibly a consequence of diminished vegetation densities during the winter months. Vegetation density, with the exception of Kimages Creek sites, was lower in winter, and may have contributed to decreased accretion rates and would have a reduced effect in offsetting elevation loss. The observed drop in vegetation density may also contribute to higher rates of erosion during the winter, further reducing net accretion rates and possibility impacting elevation loss as well.

Seasonal Variability Driven by Hydrology

Hydrology has been shown to be a major driver of wetland elevation change, and seasonal variability in hydrology may have influenced accretion and elevation trends during our study (Cahoon et al. 2011). Shrink-swell factors, such as water infiltration during the tidal regime causing soil expansion, and water loss during dryer periods due to plant usage and evapotranspiration causing soil shrinkage, can control wetland elevations (Cahoon et al. 2011). Additionally, flooding and storm runoff can deliver surface sediment, bolstering surface accretion, and deliver nutrients, accelerating root growth. Increased flooding can decrease organic matter decomposition rates, limiting subsidence. Alternatively, dryer periods will likely decrease delivery of sediment and nutrients, lowering surface accretion rates and limiting rates of belowground root growth. Dryer sediments can allow for accelerated rates of decomposition of organic matter, which could increase subsidence rates and lower overall elevation.

Measured elevations peaked between our first and second SET measurements, following high water events around September 3, 2016, caused by Tropical Storm Hermine, and October 10, 2016, caused by Hurricane Matthew, as highlighted by the blue ovals in Figure 21. The peak in elevations was followed by a subsequent drop in elevation as measured between the second and third SET measurements, following patterns in recent literature showing decreases in elevation following hurricane-induced sedimentation due to compaction under the weight of the newly deposited sediment (Cahoon, 2006; Baustian & Mendelssohn, 2015). The high water events that occurred in between the first and second SET measurements likely increased both surface accretion and total elevation in the short-term. Accretion was likely bolstered through increased rates of sediment deposition. Total elevation was likley increased due to increases in belowground elevation through soil swelling, enhanced root growth via nutrient addition due to storm runoff, and simultaneous decreases in subsurface root decay rates.

Following Hurricane Matthew, we see an overall decreasing trend in water levels in our study area, with three especially low periods, as highlighted by the beige-colored ovals in Figure 21. This general decrease in water levels, with three especially low periods occurring between October 16, 2016 and December 16, 2016, could have lead to soil shrinkage and increasing rates of root-zone decomposition, decreasing overall elevation via subsurface processes. Additionally, with relatively few storm events, there was likely a reduction in nutrient delivery from the watersheds, reducing rates of belowground root growth.

While total elevation gains peaked at the second SET measurement before steadily decreasing, surface accretion typically continued to trend upward from the second SET measurement, peaking around mid January before leveling off, with some marker horizons experiencing subsequent erosion. High rates of surface accretion during this period may have been initially driven by the aforementioned high water events (Figure 21), and maintained subsequently by steady tidal sediment delivery at many of the SET locations, while total elevation was lost during the same period due to hydrology-driven subsurface processes.

The high water periods highlighted by the blue ovals in Figure 21 may have driven both high rates of surface accretion that helped to maintain total elevations at the study sites during the short-term, as well as driven subsurface soil swelling due to water infiltration. High levels of flooding may have also introduced nutrients to bolster belowground root growth and slowed decomposition rates during this period. Lack of subsequent high water events and an overall downward trend in water levels during the period between October 16, 2016 and December 16, 2016 may have led to increased rates of subsurface subsidence due to higher rates of root decomposition and soil shrinkage during the relatively dry period, explaining the elevation loss we see after our second SET measurements.

Kimages Creek

Of all the wetland study sites, Kimages Creek experienced the greatest rates of elevation loss, despite positive accretion rates. This was especially true at SETs 6 and 11, which were placed near the stream channel on the middle and southern peninsulas, respectively (Figures 9 and 10). The high rates of elevation loss at SETs 6 and 11 may be the result of lateral migration of the stream channel following the removal of the earthen dam in 2010. Recent meta-analyses of marsh response to sea-level rise state that high rates of subsidence in marsh ecosystems are often the result of anthropogenic stressors, such as prior land use (Kirwan et al., 2016). Since the dam was breached, the Kimages Creek wetland ecosystem has been a state of flux as it adjusts and restores to its pre-dam state. Herbaceous vegetation is reestablishing in the current marsh ecosystem, but the areas near the stream channel are mudflat. The channel banks do not have the established trees and root systems present in our mature tidal forest ecosystem sites to prevent erosion. As such, the Kimages Creek channel is migrating laterally as it attempts to achieve an equilibrium. Visual, as well as empirical, evidence of channel migration and high levels of TSS recorded at the mouth of Kimages Creek support the possibility of high rates of erosion and exiting sediment due to lateral migration of the channel. It is possible that this migration and consequent erosion may be contributing to sediment loss from the stream bank near SETs 6 and 11, pulling material away and causing a rapid drop in elevation despite the concurrent sediment deposition with incoming tides. Consequently, the relatively high rates of elevation loss observed at SETs 6 and 11 may not be representative of the entire Kimages Creek wetland ecosystem.

Harris Creek

Harris Creek exhibited the only positive elevation trajectories during this study. Despite similar rates of accretion, Harris Creek and Kimages Creek had the most disparate elevation change rates in our study (Table 2, Figure8). Stagg et al. (2016) concluded that, while surface accretion was the most significant driver of elevation change in marsh ecosystems, subsurface root-zone processes were the most important factor in tidal forested freshwater wetlands. With nearly equal rates of accretion between Harris Creek and Kimages Creek, subsurface root growth and expansion may be a dominant factor in the disparity between elevation change rates in the two wetlands. While the watersheds of both Harris Creek and Kimages Creek drain agricultural and developing properties that may introduce sediment and nutrients into the wetlands, the Harris Creek tidal forest may be experiencing enhanced subsurface root growth that drives positive elevation gain. The Kimages Creek marsh, being dominated by herbaceous vegetation, lacks an abundance of woody vegetation, inhibiting a similar response. Additionally, higher lignin content in OM within Harris Creek would likely lower decomposition rates relative to those in Kimages Creek. Combined with higher rates of subsidence due to low lignin concentrations, the lack of subsurface root growth and expansion may be determinant in limiting elevation gain in Kimages Creek relative to Harris Creek.

While Harris Creek, Presquile NWR, and James River NWR all had similar TSS concentrations in their respective creek systems, elevation drops observed at Presquile NWR and James River NWR may be due to less nutrients being received to boost subsurface root growth. The Presquile NWR watershed drains only the relatively small area of the island, limiting sediment and nutrient availability. It is possible that the watershed drainages of Powells Creek and James River NWR are not supplying a sufficient load of sediment and nutrients to sustain positive elevation growth. More data and analyses focused on the contributing factors to subsurface processes will be needed to further develop our understanding of elevation change dynamics in these ecosystems.

Regression of Predictor Variables

We ran two multiple regression analyses with accretion rate and elevation change rate as our response variables. Our predictor variables were vegetation density, TSS, tidal inundation height, and distance to sediment source. In our model for accretion, only 14 percent of variability in our response was explained by our predictor variables, offering little insight into accretion drivers. Elevation change also was not well explained by any predictor variables.

As elevation change, particularly in tidal forests, is influenced in large part by sub-surface processes, it is possible that our chosen environmental drivers, which primarily govern surface accretion, are not particularly influential on rates of elevation change. Previous studies have shown surface accretion to be dependent, to varying degrees, on each of our potential predictor variables. As such, collection of more data over a longer temporal scale may be necessary to reveal their explanatory power (Gleason et al., 1979;Chmura & Hung, 2004; Mudd et al., 2010; Baustian et a., 2012; Kirwan et al., 2016).

Wetland Elevation Change compared with RSLR

When compared to the NOAA Sewells Point estimate of SLR based on their 120-year dataset (4.61 +/- .23 mm yr^{-1}), accretion rates at our sites suggest resilience to SLR, however, total elevation change rates suggests otherwise (Figure 22). Our trend for water level change at the VCU Rice Rivers Center pier suggests a drop in water level between June 2015 and December 2016 of -2 mm yr^{-1} (Figure 20). However, due to the short temporal length of this dataset, this may possibly be a short-term downward oscillation of local sea level, and not necessarily indicative of long-term trends. It has been suggested that, in studies examining SLR trends, a long-term dataset is necessary to avoid short-term fluctuations skewing generated trends (Turner, 1991; Callaway et al., 1997).

When we minimize the NOAA Sewells Point tide gauge data to match the time period of the Rice Rivers Center pier sonde, the trend line exhibited a dramatic drop in RSLR of -130.94 mm yr^{-1} (Figure 23). While this potentially supports the downward trend in RSLR obtained from the Rice Rivers Center sonde during the same time period, this is a short-term downward flux in water levels skewing our trend. Comparing the -130 mm yr^{-1} rate to the 4.61 mm yr^{-1} trend generated by the full 120-year Sewells Point dataset clearly demonstrates the benefits of a long dataset in correcting for short-term variability.

Comparisons to other Studies in Mature Tidal Forests

Craft (2012), in his study of accretion in tidal freshwater forests along the Ogeechee, Atlamaha, and Satilla Rivers along the Georgia Coast, USA, found accretion rates of 1.3 mm yr^{-1} (Cs^{137}) and 2.2 mm yr^{-1} (Pb^{216}). Based solely on accretion rates, and not taking into account subsidence that may have occurred beneath the sample cores (60 cm depth), Craft concluded that accretion rates were not keeping pace with the current rate of local RSLR (3.0 mm yr^{-1})(Craft, 2012).

Ensign et al. (2014) used marker horizons to measure surface accretion in tidal forests along the Waccamaw (South Carolina, USA) and Savannah Rivers (Georgia, USA), finding average rates ranging from 6 to 7.4 mm yr^{-1}. These rates do not take into account subsidence beneath the marker horizons, and as the authors note, with low bulk density and high OM content in the soil, subsidence at their sites was likely relatively high (Ensign et al., 2014). When the authors factored in modest amounts of subsidence (50% of accretion), mean accretion rates for the study sites would be only approximately 0.56 mm yr^{-1} faster then local rates of RSLR (3.15 mm yr^{-1}), allowing them to only barely keep pace, if at all (Ensign et al., 2014).

Contrasting with previous studies, a recent study in tidal forested freshwater wetlands using SETs coupled with marker horizons along the Savannah and Waccamaw Rivers found elevation change rates of 2.4, 2.9, and 23.5 mm yr^{-1} for the Savannah River sites (upper, middle, and lower forest, respectively), and 4.5, 4.3, and -3 mm yr^{-1} for the Waccamaw River sites (upper, middle, and lower forest, respectively)(Stagg et al., 2016). The authors concluded that, when compared with long-term rates of RSLR (3.1 mm yr^{-1}), the Savannah upper forest sites were marginally resilient to SLR, and the Waccamaw upper forest sites easily kept pace. When compared to short-term rates of RSLR 5.6 mm yr^{-1}, Savannah upper forest sites did not keep pace, but the Savannah lower forest and Waccamaw upper forests sites did keep pace.

Our mean accretion rates for our mature tidal forest sites of 11.53 mm yr^{-1} are indeed faster than local RSLR (4.61 mm yr^{-1}), and by a greater margin than observed in the Craft (2012) and Esign et al. (2014) studies. However, being able to account for subsidence using the SET-marker horizon method, we see that our mature forest sites, based on short-term trends of total mean elevation change rates (-15.18 mm yr^{-1}), do not keep pace with RSLR. When compared to the results from Stagg et al. (2016), which also used the SET-marker horizon method, we see that their sites generally exhibited positive elevation gains, and kept pace with RSLR, if only marginally. However, it should be noted that the Stagg et al. (2016) study occurred over a 5-year study period versus our 1-year current temporal scale, and the authors noted the importance of a longer study period to correct for short-term oscillations in variability (Stagg et al., 2016).

In our restored site at Kimages Creek, mean elevation change was -35.32 mm yr^{-1} for all six SETs. Removing our outliers (SETs 6 and 11), mean elevation change became -12.40 mm yr^{-1}. Our rates vary greatly from mean elevation rates found in the Kirwan et al. (2016) meta-analysis of marsh vulnerability to SLR, 3.0 mm yr^{-1} and 6.9 mm yr^{-1} for high marsh and low marsh, respectively. Marsh survivability is dependent on at least 1 meter of tidal flux and at least 30 mg/L TSS (Kirwan et al., 2016). The area of the James River on which Kimages Creek is located experiences a tidal flux of around 1 meter, and our sampling at Kimages Creek found mean TSS to be approximately 28.44 mg/L, very nearly the minimum for marsh resilience (Kirwan et al., 2016). Based on our data compared with findings from the Kirwan et al. (2016) meta-analysis, Kimages Creek should be marginally resilient to SLR. However, marsh resilience to sea-level rise may be impacted by previous anthropogenic stressors, such as the prior impoundment of Kimages Creek (Kirwan et al., 2016). Erosion caused by lateral migration of the restored creek channel may be impacting elevation change dynamics and contributing to high rates of elevation loss in Kimages Creek, as supported by the especially high rates of subsidence observed at our stream bank sites, SETs 6 and 11.

Ideally, these types of data are collected over a time span of 10+ years, with a minimum of 3 years used for analyses. Our abbreviated study period consisted of 4 collection dates taken at 2-month intervals over a span of 8 months. With high levels of variability in measured elevation during the relatively short study period, we will need to collect more data before we are able to establish long-term trends (Turner 1991; Callaway et al., 1997).

While it is too early to make statements about the resilience of our sites with high levels of confidence, it would be reasonable to assume that, based on current data, our mature and restored sites are likely not keeping pace with RSLR, and if they are, resilience is likely marginal. It will be essential to continue monitoring our SETs, and to contribute to the growing field of SET-marker horizon studies in tidal freshwater forested wetlands. Comparing our long-term trends with those of other studies in tidal forests will help us to determine whether or not these wetlands types are keeping pace with RSLR, and what factors dictate tidal forest resilience.

Conclusions

Our most substantial local estimate of SLR from Sewells Point suggests a RSLR rate of 4.61 +/- .23 mm yr^{-1}. Short-term trends in sediment accretion rates would suggest that wetlands in our study are keeping pace with RSLR, however, the overall trend of elevation loss would suggest otherwise. While Harris Creek appears to maintain both positive elevation change rates and accretion rates, elevation change rates do not appear to keep pace with RSLR. In Presquile NWR and James River NWR, high rates of subsidence appear to offset the sediment deposition. In Kimages Creek, the migrating channel may be impacting elevation loss near the creek banks, but all of the sites in the restored marsh appear to be losing elevation faster then sediment is accreting.

Subsurface processes may be dominant factors dictating total elevation change rates in mature tidal forests in the lower James River watershed. Hydrologic conditions seem to be a primary driver of both surface accretion and belowground processes influencing total elevation gain and loss. Mechanical and biological processes influenced by storm events, nutrient availability, and hydrologic conditions are likely interacting to affect variability we have observed between our mature tidal forests sites. These mechanical and biological processes, compounded by prior anthropogenic disturbances, may be influencing the variability observed in total elevation change rates between Kimages Creek and Harris Creek.

Due to high levels of variability, we cannot confidently forecast long-term trends for surface elevation this early in the study. Natural processes that dictate variation in elevation change and accretion, coupled with seasonal variability and externalities such as storm events, have resulted in an oscillating variability in surface elevation typical of these studies. We may be observing short-term fluctuations that do not necessarily reveal long-term trends. Consequently, a longer dataset is necessary to identify long-term trends in elevation change.

Continuing to study accretion and elevation change dynamics in tidal freshwater forests will give us insight into which wetlands are most at risk in the face of RSLR. Based on this understanding, we can begin to identify where paths of retreat may exist for specific wetlands and assess the best management of those paths of retreat. If tidal forests in the lower James River watershed are unable to keep pace with sea-level rise, it may be imperative to maintain migratory paths, absent of fracturing or interference, in order to perpetuate the existence of the critically important tidal freshwater forest wetland type and associated ecosystem services (Torio & Chmura, 2013).

The installation of the first SETs at VCU Rice River Center and within our partnered wildlife refuges has afforded us a novel opportunity to help to develop a more complete understanding of elevation change dynamics in mature and restored tidal freshwater forested wetlands in the lower James River watershed. By assimilating our SETs into the NOAA Chesapeake Bay Sentinel Site Cooperative and forging partnerships with James River NWR and Presquile NWR, we are contributing a larger network of coastal wetland elevation change data, and facilitating cooperative adaption to climate change and sea-level rise.

Future of the Study

For the abbreviated inception period of this study, we sampled each SET 4 times over an 8-month period, giving us a relatively high-resolution dataset. Moving forward in our long-term study, we will measure the SETs on a quarterly basis. This dataset will be incorporated into the NOAA Chesapeake Bay Sentinel Sites Cooperative dataset, filling in a data gap regarding tidal freshwater forested wetlands in the lower Chesapeake Bay and James River watersheds.

The tidal forests that we are studying have high levels of surface organic matter based on field observation, the decay of which will impact subsidence rates. Parsing out organic and mineral sediment contributions to accretion would be useful in the future of the study. Also, studying rates of subsurface root growth and decomposition would improve our understanding of elevation change in our forested wetlands sites. In addition to comparing linear trends of elevation change between study locations, comparing incremental changes in elevation over time over a longer study period will be useful in understanding effects of seasonal variability, particularly when paired with climatic data such as precipitation and temperature.

Real Time Kinematic (RTK) base stations will be used to survey our SET benchmarks and tie them into a known datum. RTK base stations are capable of sub-centimeter accuracy in the z-coordinate. By surveying our SETs and tying into a known datum, we will not only be able to directly compare our SET elevations to one-another, but also compare them to current local sea levels using data tied into a known datum. This will give us a better understanding of how these wetlands are responding to sea-level rise by letting us compare SET elevations directly to water elevations, in addition to rates of elevation change.

In order to gain insight into long-term accretion rates in these wetland sites that can be compared to contemporary accretion rates, we will take sediment cores and test for trace cesium^{137} activity. Testing for cesium^{137} will give us decadal scale accretion rates that are not as sensitive to seasonal variability and short-term fluctuations as our contemporary rates. Comparing these long-term accretion rates with our short-term, contemporary accretion rates will bolster our understanding of how these wetlands have historically accreted sediment and allow us to better predict future trends.

In order to develop cesium^{137} core profiles, cores taken to a depth of 1 meter using 10.2 cm diameter PVC tubes will be taken at each site ($n = 4$) to measure mean decadal-scale accretion rates. The cores will be segmented into 2 cm discs to be tested for trace activity of cesium^{137} using a high purity germanium detector (Callaway et al., 1997; Neubauer et al., 2002; Craft et al., 2003; Craft, 2012). The detected levels of cesium^{137} will allow us to develop a cesium^{137} profile for the core wherein we can establish the peak levels of cesium^{137} (1963) and its initial appearance, or the cesium^{137} horizon (1954). Thus, we will be able to determine an approximate average rate of accretion over the prior 50 years. Additionally, this method will allow us to measure carbon accumulation, specifically, which is of interest in understanding how much carbon is being sequestered in the sediment of our wetland study sites. This data will allow us to not only measure and compare average long-term accretion and carbon accumulation rates in between the wetland sites, but also compare average long-term accretion rates against our measurements of contemporary short-term accretion rates at each site. Furthermore, quantifying carbon accumulation in our wetland sites will serve to enhance the usefulness of our study in influencing policy decisions concerning wetland management.

Benchmark Stability

Based on recent studies suggesting the monitoring of benchmark elevations and considering the softness of the substrates in which our benchmarks are installed, we feel that a GPS or RTK monitoring regime for our benchmark elevations will be a necessary component as this study moves forward (Swales et al., 2016).

Tables and Figures

Table 1 Accretion and elevation change rates (mm $month^{-1}$)(calculated by linear regression of grand means for sampling stations). Accretion Rate – Elevation Rate = Subsidence Rate. Positive values under Subsidence indicate soil expansion rather than subsidence, and are highlighted in yellow.

SET	Location	Accretion Rate	Elevation Rate	Subsidence
SET_1	Harris Creek	0.4479	-0.3611	0.809
SET_2	Harris Creek	0.2604	-0.64655	0.90695
SET_3	Harris Creek	1.97915	0.59095	1.3882
SET_4	Kimages Creek	0.97915	-0.4603	1.43945
SET_5	Kimages Creek	1.5	-1.70415	3.20415
SET_6	Kimages Creek	1.57815	-6.691	8.26915
SET_7	Kimages Creek	-0.125	-2.25905	2.13405
SET_8	Presquile NWR	1.28125	-2.4278	3.70905
SET_9	Presquile NWR	-0.8021	-1.9779	1.1758
SET_10	Presquile NWR	0.4427	-2.3597	2.8024
SET_11	Kimages Creek	1.54165	-5.2235	6.76515
SET_12	Kimages Creek	0.58335	-1.32085	1.9042
SET_13	Harris Creek	0.26565	0.99235	(+) 0.7267
SET_14	Harris Creek	2.3854	0.59375	1.79165
SET_15	Harris Creek	0.96355	0.02905	0.9345
SET_16	James River NWR	2.1979	-2.1861	4.384
SET_17	James River NWR	1.875	-1.0597	2.9347
SET_18	James River NWR	0.5	-1.9757	2.4757

Table 2 Mean accretion, elevation change, and subsidence rates for each wetland (mm yr^{-1} +/- standard error), and total mean accretion, elevation change, and subsidence rates for all wetlands combined (mm yr^{-1} +/- standard error).

Wetland Location	*Accretion (mm yr^{-1})*	*Elevation Change (mm yr^{-1})*	*Subsidence (mm yr^{-1})*
Harris Creek	*12.60 (+/-) 4.52*	*2.40 (+/-) 3.10*	*10.21 (+/-) 4.20*
Kimages Creek	*12.11 (+/-) 3.34*	*-35.32 (+/-) 12.01*	*47.43 (+/-) 14.02*
Presquile NWR	*3.69 (+/-) 7.26*	*-27.06 (+/-) 1.68*	*30.75 (+/-) 8.89*
James River NWR	*18.29 (+/-) 6.25*	*-20.89 (+/-) 4.15*	*39.18 (+/-) 6.90*
Total Means	*11.67 (+/-) 3.01*	*-20.22 (+/-) 8.1*	*31.89 (+/-) 7.99*

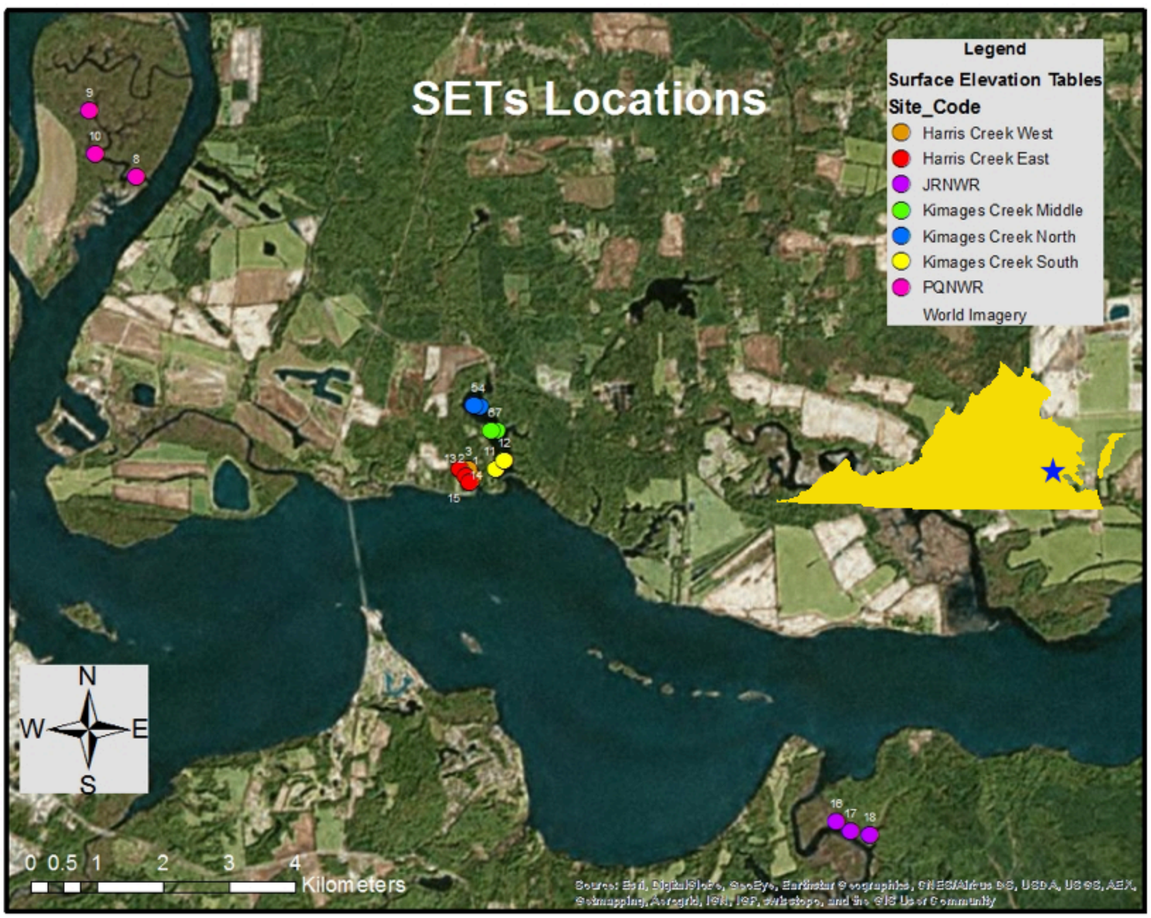

Figure 1 SET locations for Kimages Creek, Harris Creek, James River National Wildlife Refuge (JRNWR), and Presquile National Wildlife Refuge (PQNWR), in the lower James River watershed. Numbers identify individual SET locations.

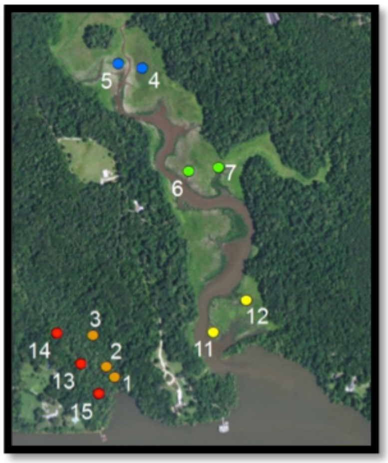

Figure 2 Close-up view of SET locations in Harris Creek (left) and Kimages Creek (right). Numbers identify individual SET locations.

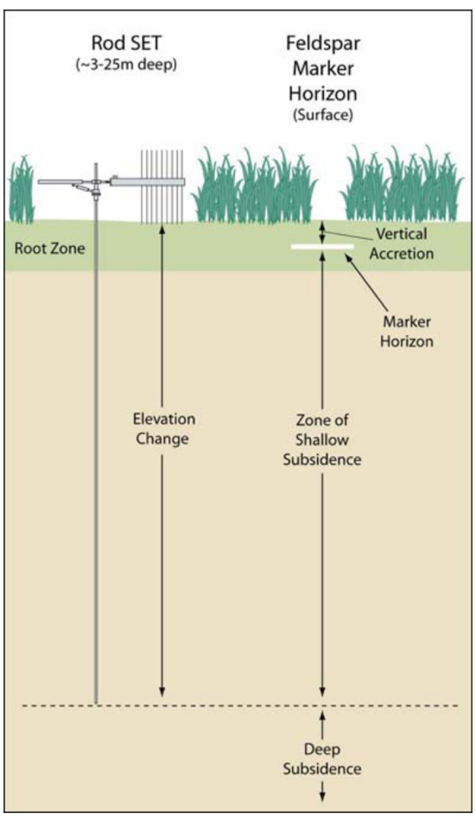

Figure 3 Diagram of SET and marker horizon sampling station. Figure obtained from https://www.pwrc.usgs.gov/set/theory.html.

Figure 4 Measuring the SET instrument in James River NWR. The pins have been locked into place at sediment surface, and we are measuring from the top of the pins to the top of the arm.

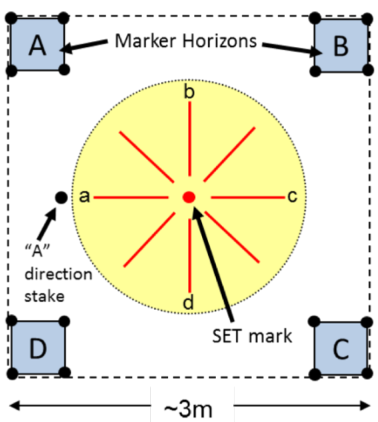

Figure 5 Diagram of a typical SET-Marker Horizon sampling station.

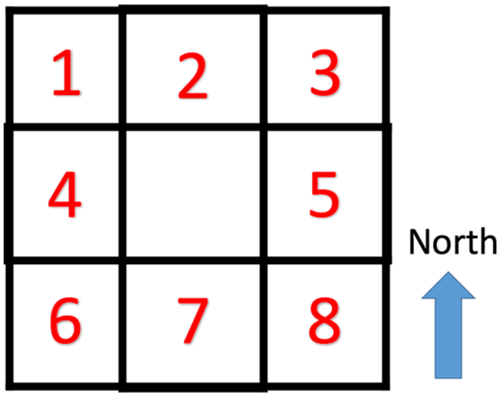

Figure 6 Diagram of the subplot layout for vegetation surveys in a sampling station.

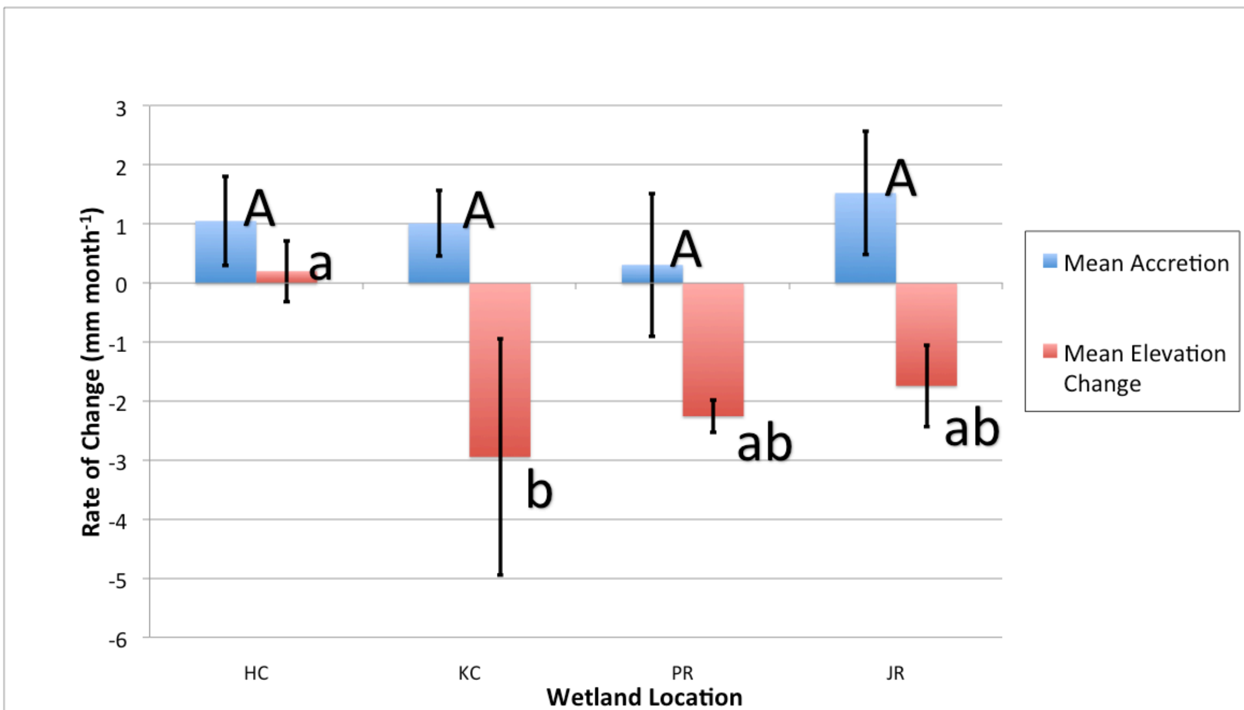

Figure 7 Mean rates of elevation change and accretion for each wetland location (mm month^{-1} +/- standard error). Rates of change calculated via least squares regression. Uppercase letters indicate significant differences among group means of accretion rates (p=0.41). Lowercase letters indicate significant differences among group means of elevation change rates (p=0.01).

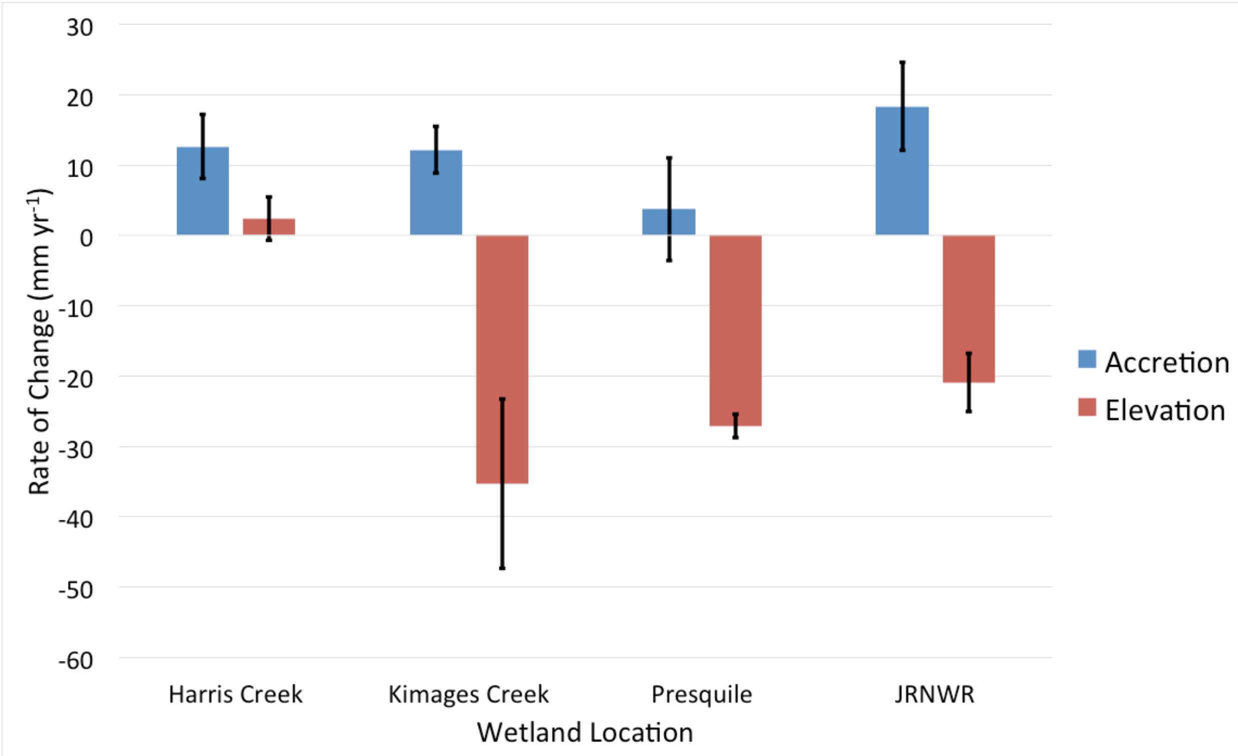

Figure 8 Extrapolated trends for mean elevation change and accretion rates within each wetland location (mm yr^{-1} +/- standard error).

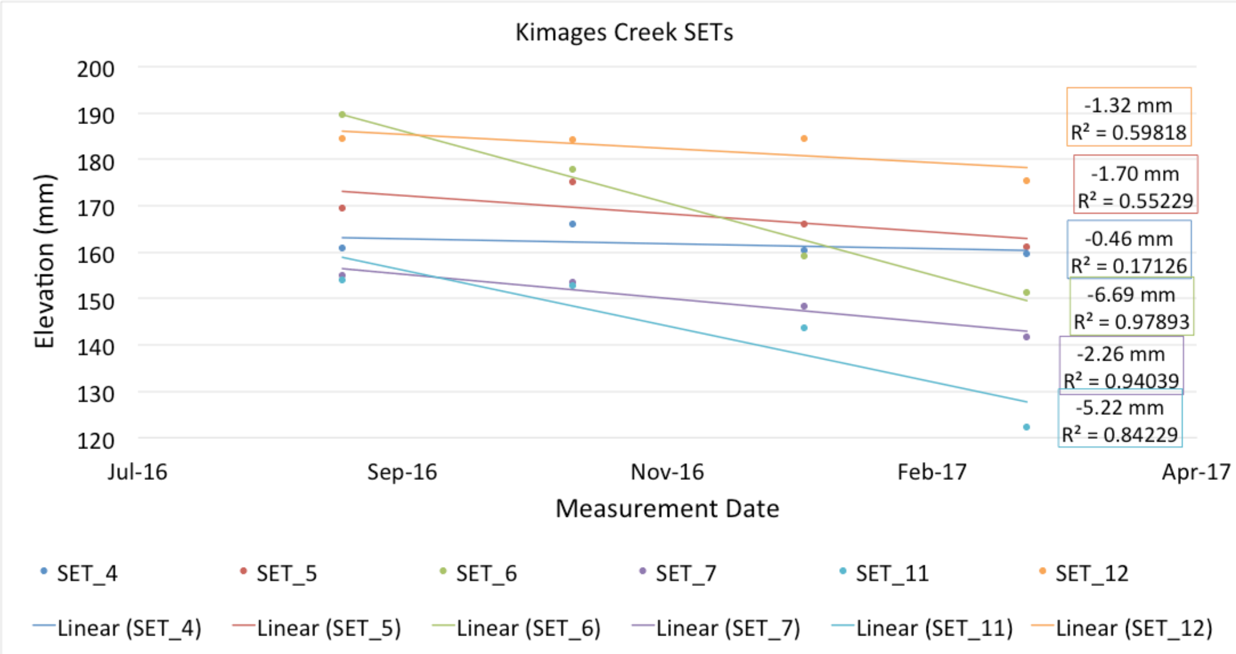

Figure 9 Rates of elevation change (mm month^{-1}) at Kimages Creek SETs, calculated via least squares regression.

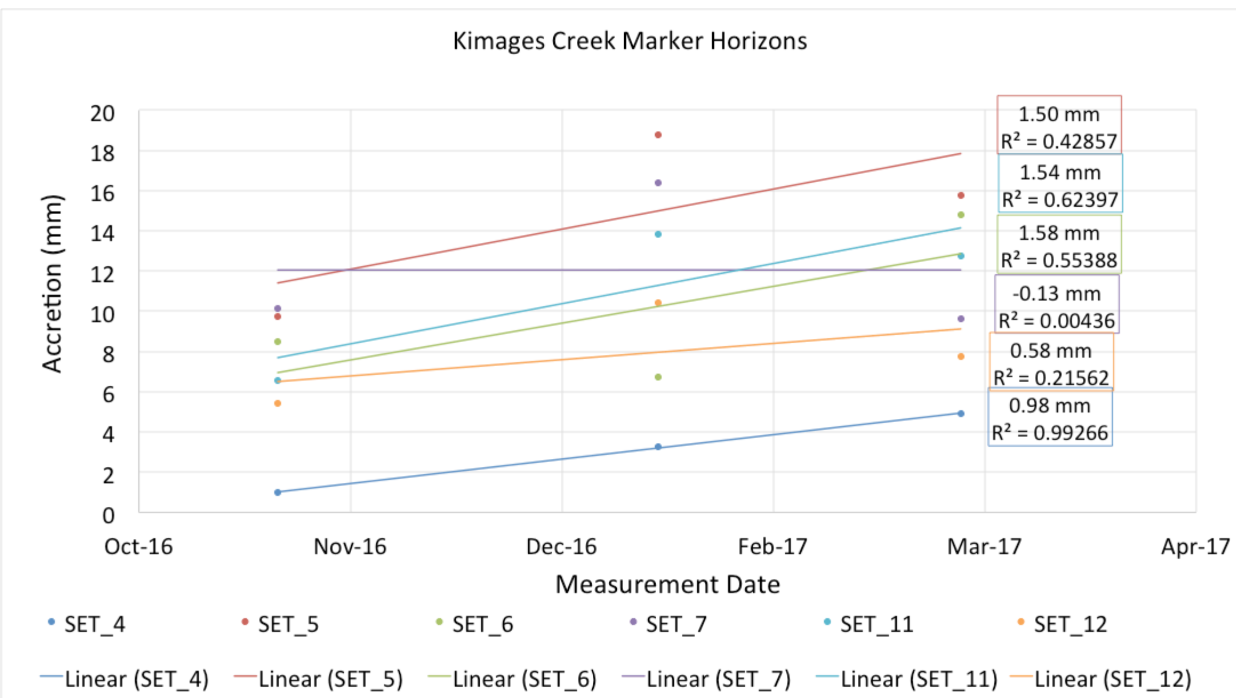

Figure 10 Rates of accretion (mm month^{-1}) at Kimages Creek MHs, calculated via least squares regression.

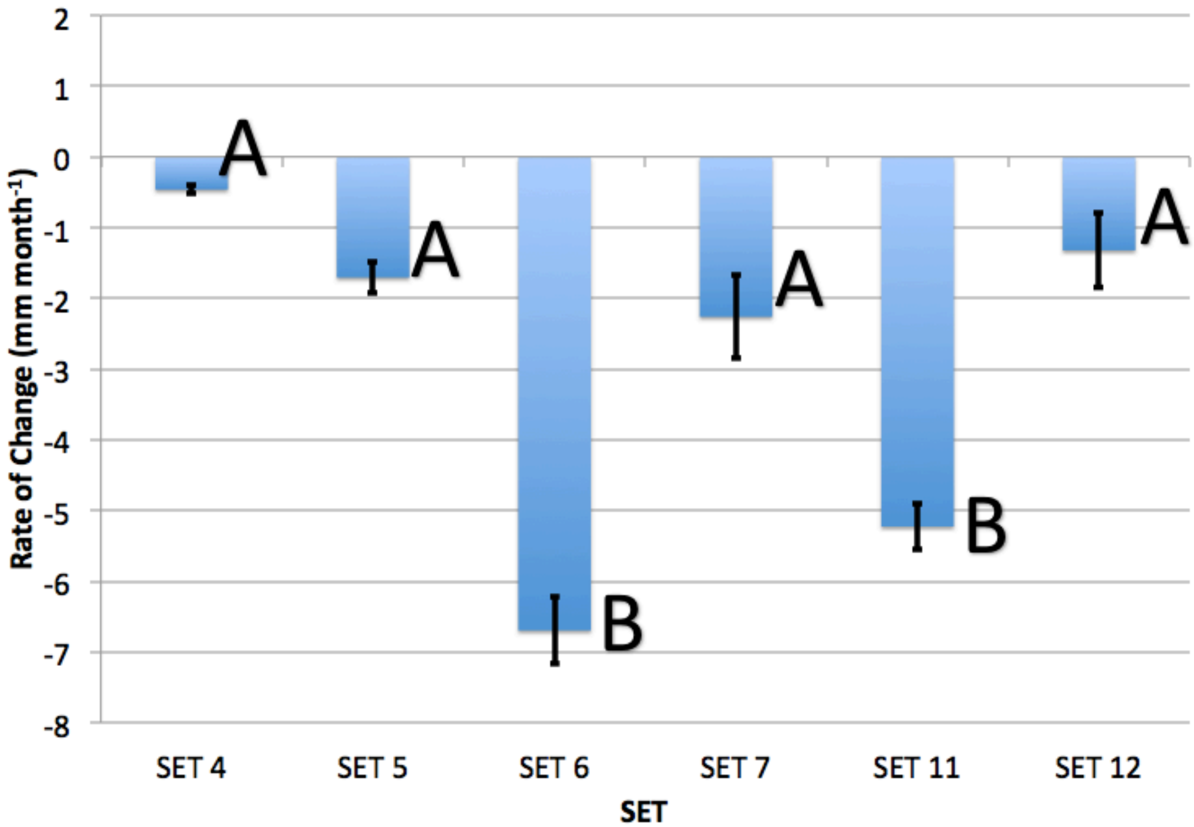

Figure 11 Group means for elevation change trends at SETs within Kimages Creek (mm $month^{-1}$ +/- standard error). Letters indicate significant variation among group means ($p<0.0001$).

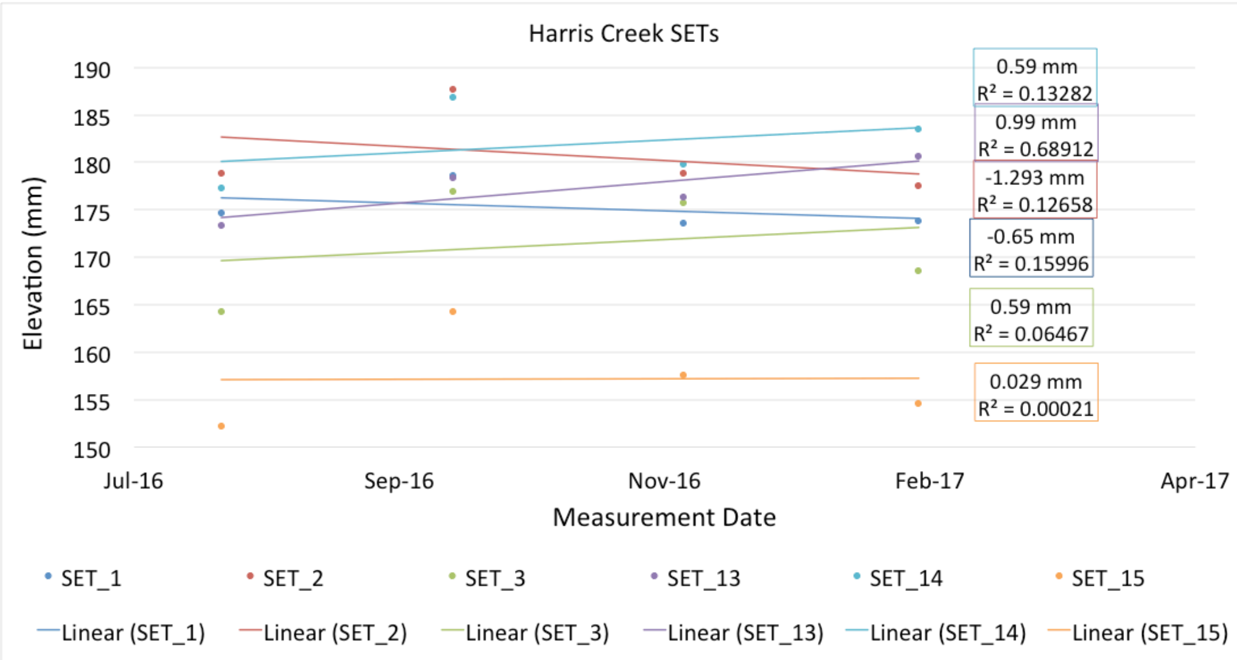

Figure 12 Rates of elevation change (mm month^{-1}) at Harris Creek SETs, calculated via least squares regression.

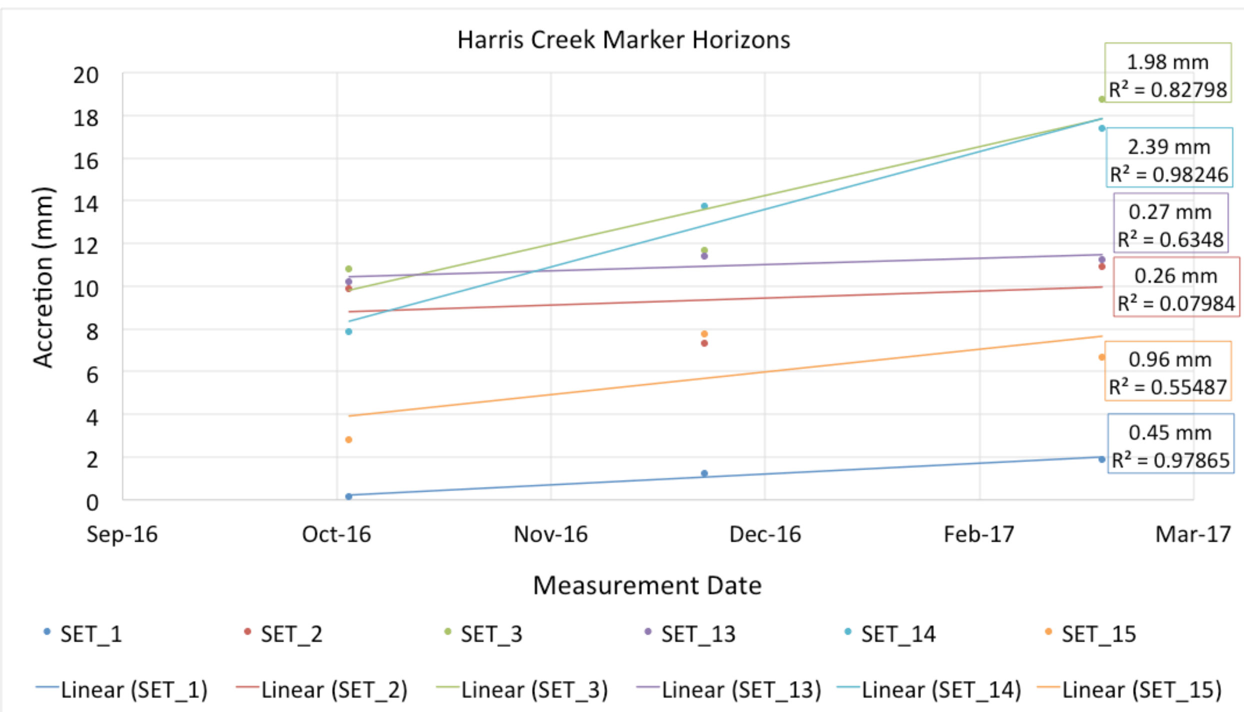

Figure 13 Rates of accretion (mm month^{-1}) at Harris Creek MHs, calculated via least squares regression.

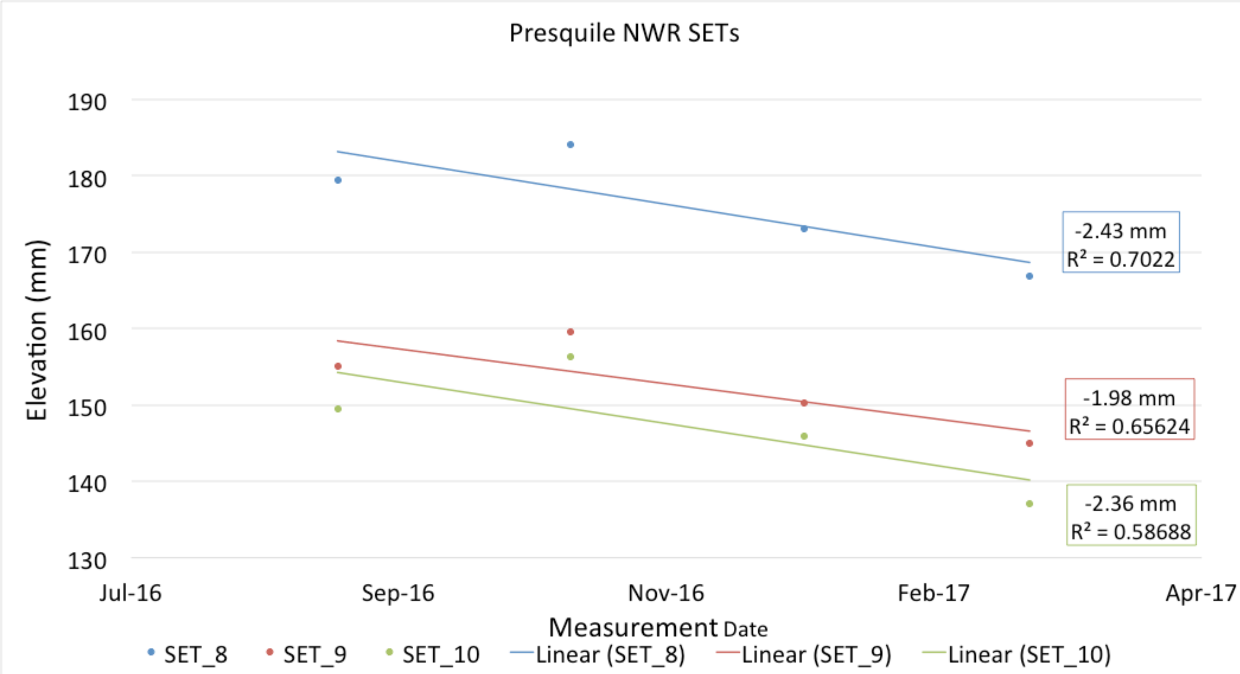

Figure 14 Rates of elevation change (mm month^{-1}) at Presquile NWR SETs, calculated via least squares regression.

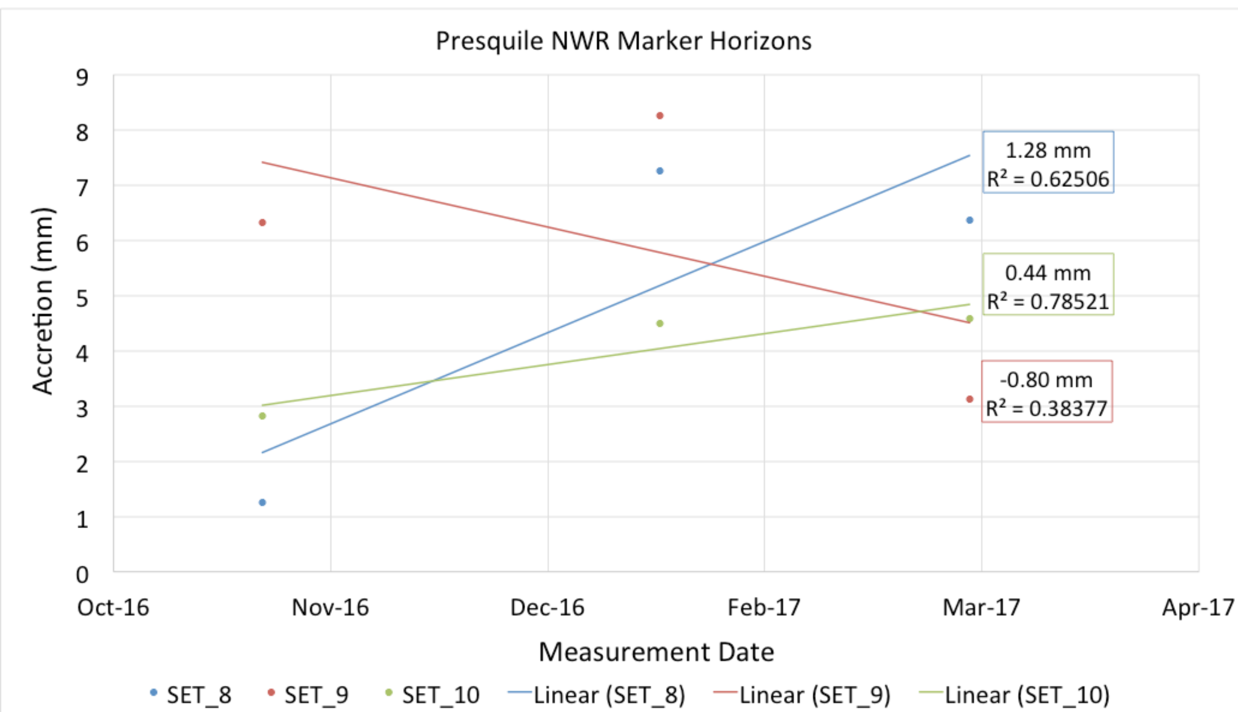

Figure 15 Rates of accretion (mm month^{-1}) at Presquile NWR MHs, calculated via least squares regression.

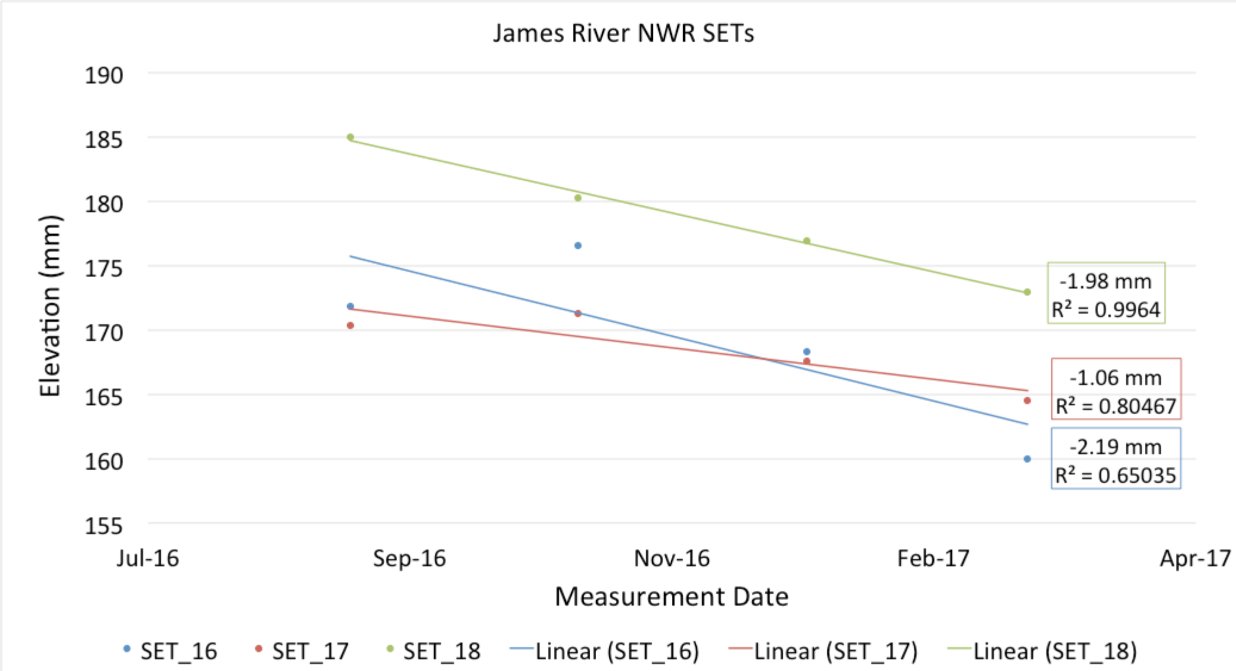

Figure 16 Rates of elevation change (mm month^{-1}) at James River NWR SETs, calculated via least squares regression.

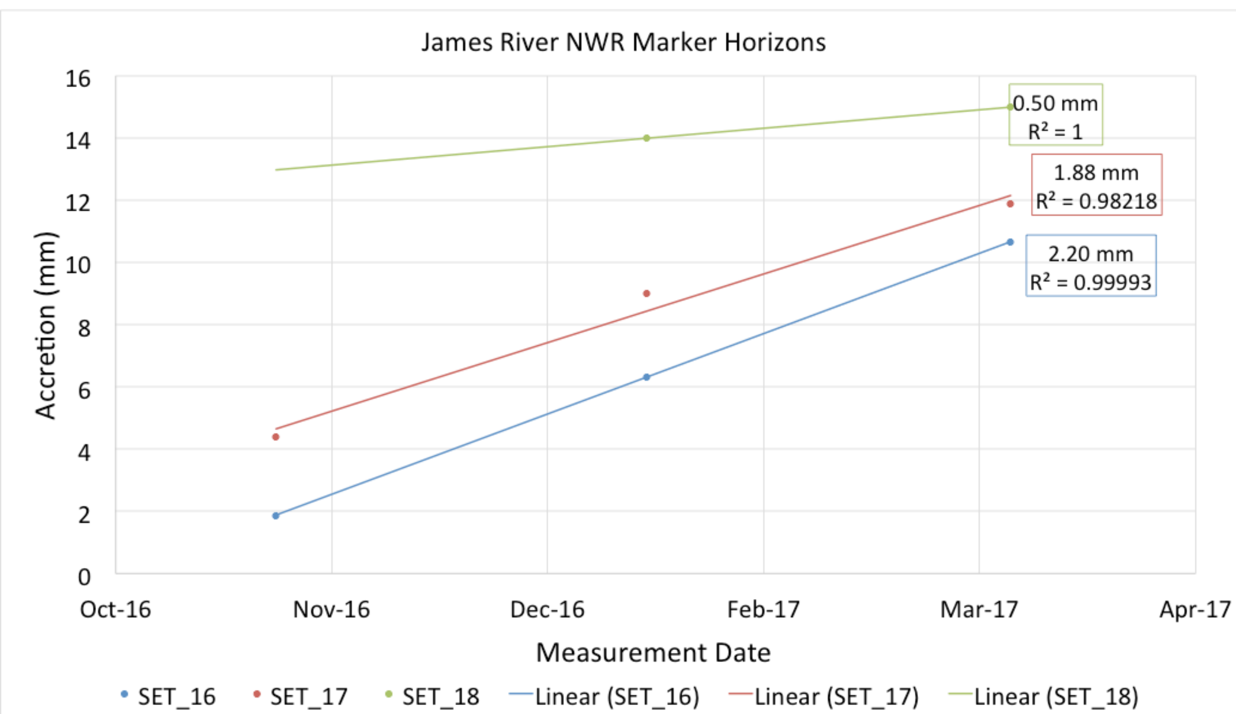

Figure 17 Rates of accretion (mm month^{-1}) at James River NWR MHs, calculated via least squares regression.

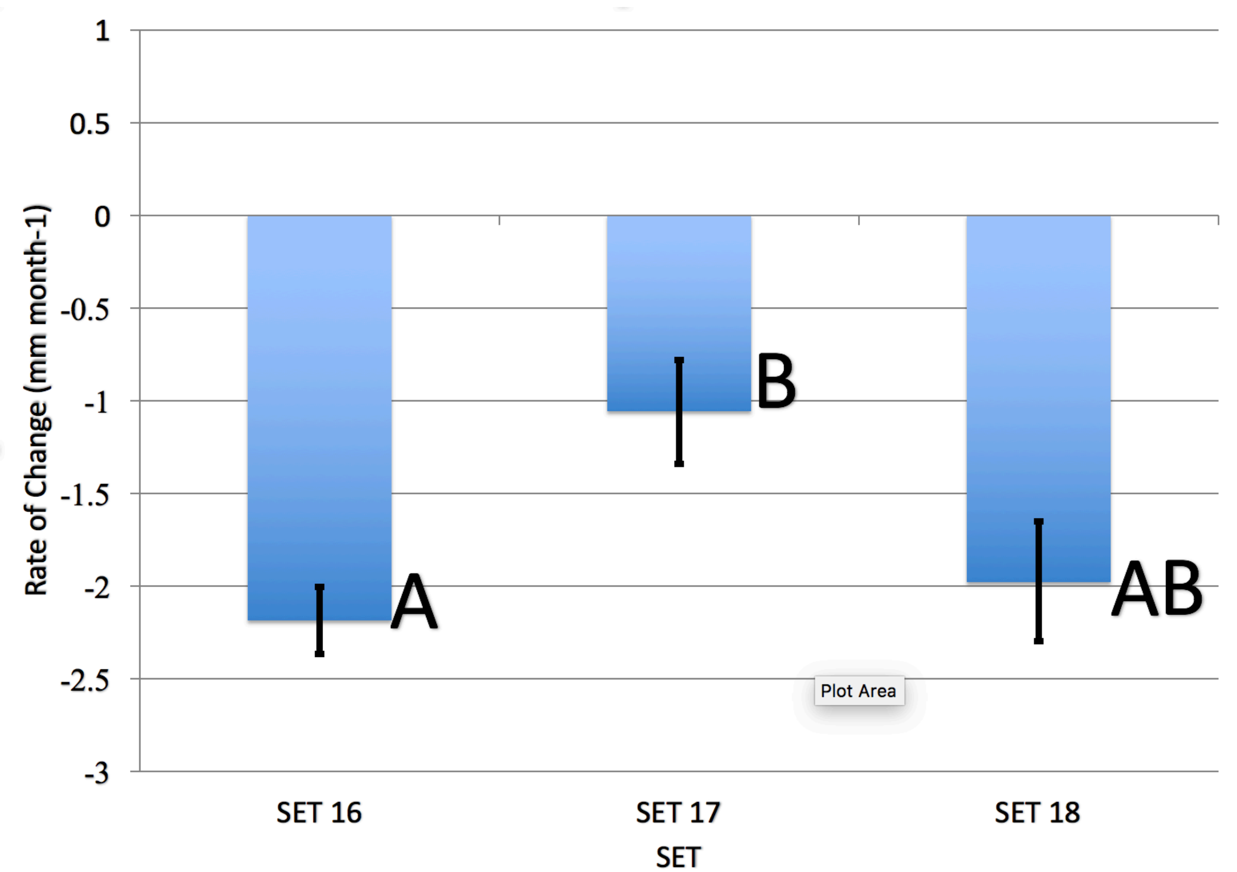

Figure 18 Group means for elevation change trends at SETs within James River NWR (mm month^{-1} +/- standard error). Letters indicate significant variation among group means (p=0.04).

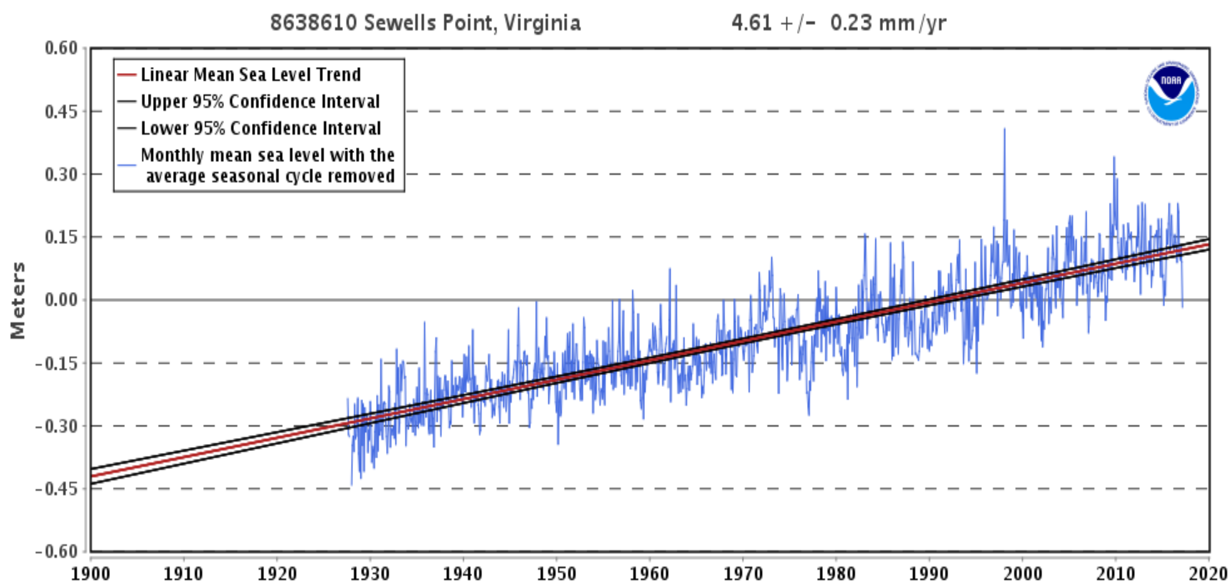

Figure 19 SLR trend (mm yr^{-1}) from the NOAA tide gauge at Sewells Point, Virginia. Figure obtained from https://tidesandcurrents.noaa.gov/sltrends/sltrends_station.shtml?stnid=8638610

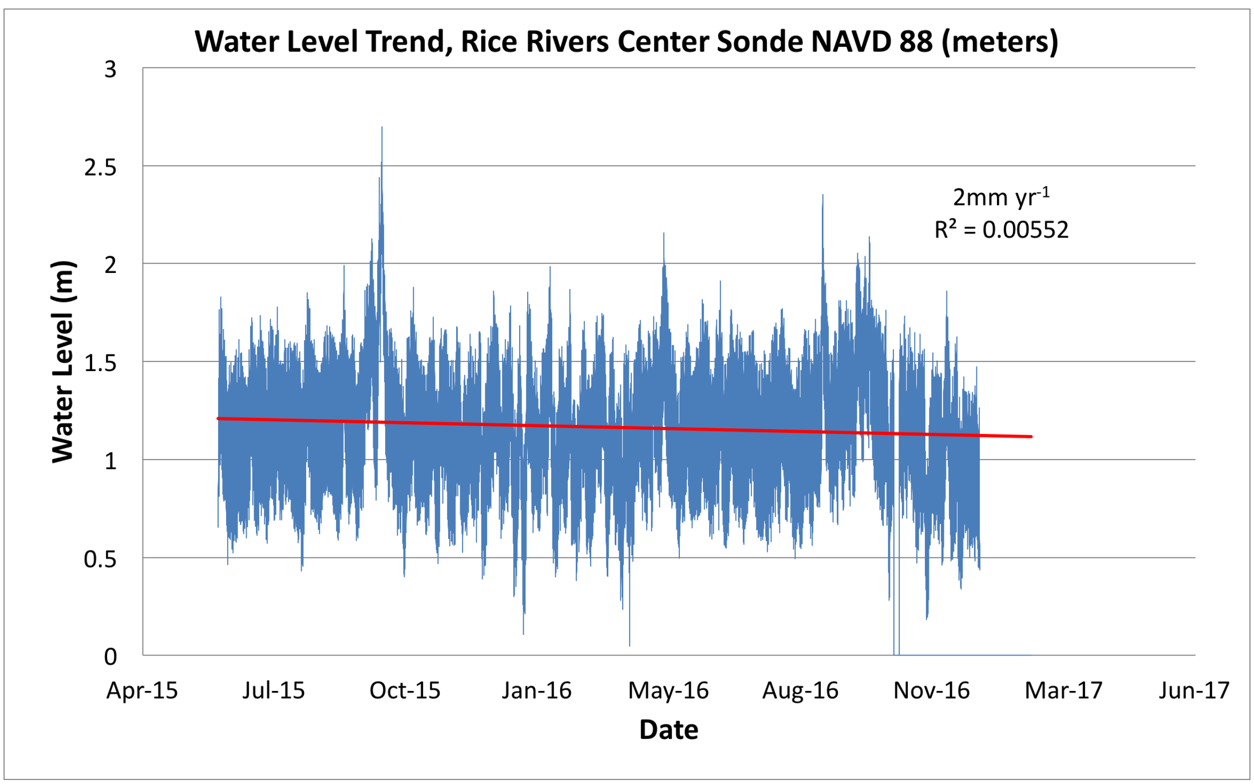

Figure 20 Water level data (m) from the sonde at the Rice Rivers Center pier, NAVD 88 datum. Water level trend (mm yr^{-1}) generated by least squares regression.

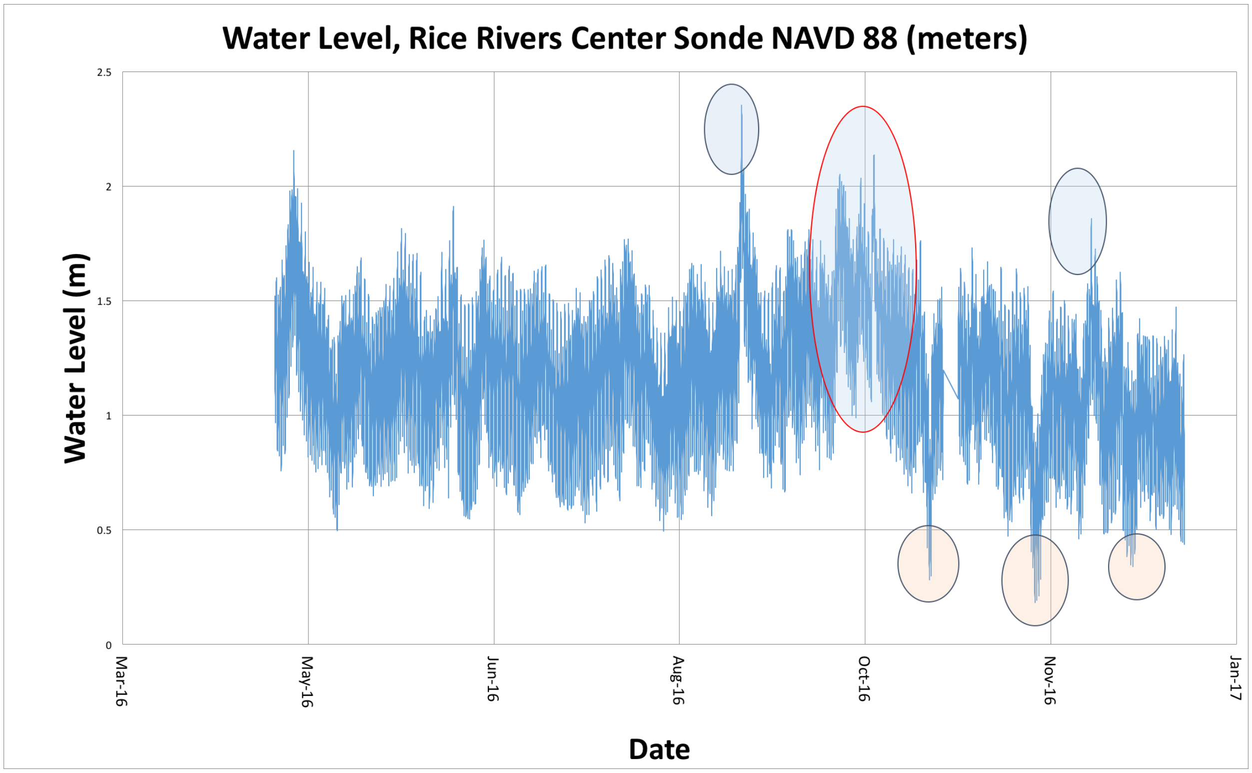

Figure 21 Water level data (m) from the sonde at the Rice Rivers Center pier minimized to the temporal scale of our sampling period (NAVD 88). Blue ovals represent high-water events. The large blue oval with the red outline represents increased water levels due to the Hurricane Matthew tidal surge. The beige ovals represent low water-events.

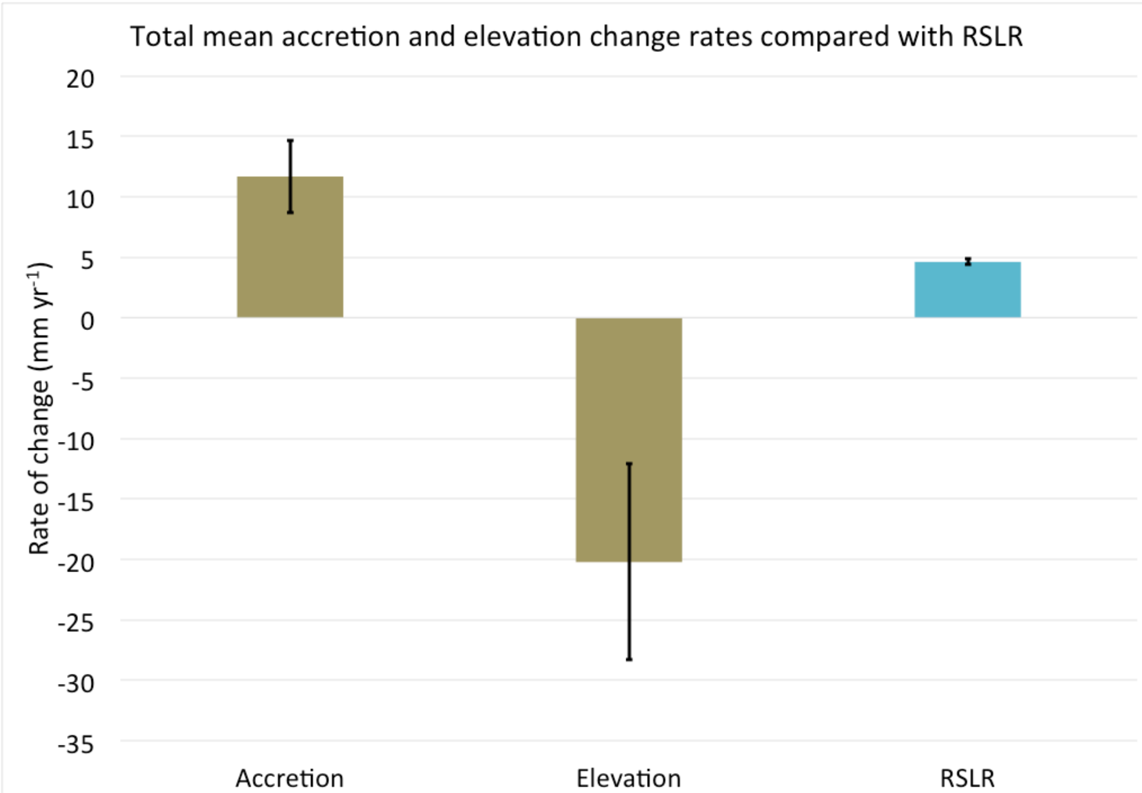

Figure 22 Total mean accretion and elevation change rates for our study locations in the lower James River watershed (mm yr^{-1} +/- standard error), compared with RSLR generated from the NOAA tide gauge at Sewells Point, Virginia (mm yr^{-1} +/- standard error).

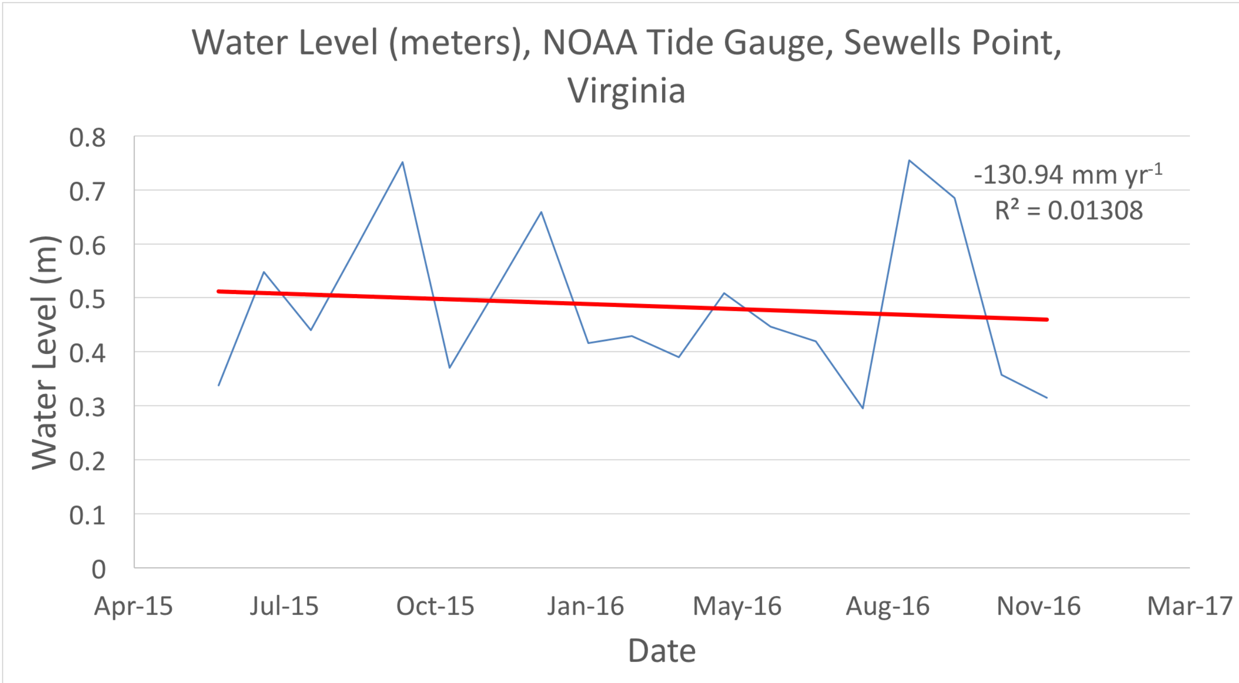

Figure 23 Water level data (m) from the NOAA tide gauge at Sewells Point, Virginia, at the same temporal scale as the Rice Rivers Center Sonde data. Water level trend (mm yr^{-1}) generated by least squares regression.

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Appendix A

Table A SET installation dates and measurement dates. Asterisks indicate incidents where marker horizons were measured on separate dates from the respective SET. In these instances, marker horizons were measured within a week of the SET sampling date.

			Measurement Dates			
Wetland Location	SET ID	Installation Date	1st	2nd	3rd	4th
Kimages Creek	SET 4	6/10/16	8/25/2016*	11/2/16	1/4/17	3/2/17
Kimages Creek	SET 5	6/10/16	8/25/16	11/2/16	1/4/17	3/2/17
Kimages Creek	SET 6	6/10/16	9/20/16	11/28/16	1/30/17	3/10/17
Kimages Creek	SET 7	6/10/16	9/20/16	11/28/16	1/30/17	3/10/17
Kimages Creek	SET 11	6/21/16	9/7/16	11/1/16	1/5/17	3/4/17
Kimages Creek	SET 12	6/21/16	9/8/2016*	11/2/16	1/5/17	3/4/17
Harris Creek	SET 1	5/25/16	8/22/16	10/19/16	12/20/16	2/23/17
Harris Creek	SET 2	5/25/16	8/22/16	10/19/16	12/20/16	2/23/17
Harris Creek	SET 3	6/9/16	8/19/2016*	10/31/16	12/20/16	2/23/17
Harris Creek	SET 13	6/24/16	8/22/2016*	10/24/16	12/21/16	2/24/17
Harris Creek	SET 14	6/24/16	8/22/2016*	10/24/16	12/21/16	2/24/17
Harris Creek	SET 15	6/24/16	8/22/2016*	10/24/16	12/21/16	2/24/17
Presquile	SET 8	6/16/16	9/18/16	11/15/16	1/18/17	3/9/17
Presquile	SET 9	6/16/16	9/18/16	11/15/16	1/18/17	3/9/17
Presquile	SET 10	6/16/16	9/18/16	11/15/16	1/18/17	3/9/17
JRNWR	SET 16	7/12/16	9/13/16	11/17/16	1/16/17	3/16/17
JRNWR	SET 17	7/12/16	9/13/16	11/17/16	1/16/17	3/16/17
JRNWR	SET 18	7/12/16	9/13/16	11/17/16	1/16/17	3/16/17

Installation Dates

These are the first SETs to have been installed in tidal forested freshwater wetlands in the lower Chesapeake Bay and James River watersheds. Locations were selected with the intentions of best elucidating accretion and elevation change rates in forested tidal freshwater wetlands along the James River east of the fall line, as well as within a restored tidal forest currently existing as a tidal marsh. All SET installation dates as well as sampling periods are displayed in

Table A. All SET locations are displayed in Figure 1, with a close-up of Harris Creek and Kimages Creek SETs in Figure 2.

On 25 May 2016, the first two SETs (SET 1 and SET 2) were installed along the eastern bank of Harris Creek under the supervision of Alex Demeo and Claudia Deeg from VIMS. A third (SET 3) was installed on the east bank of Harris Creek on 9 June 2016. Three SETs (SET 13 - 15) were installed on the western bank of Harris Creek on 24 June 2016.

At Kimages Creek, four SETs (SETs 4 – 7) were installed on 10 June 2016. SETs 4 and 5 were installed at the north of the wetland just south of the "tree island," on opposite sides of the creek. SETs 6 and 7 were installed on the middle peninsula, both on the eastern side of the creek. On 21 June 2016, SETs 11 and 12 were installed on the southern peninsula of Kimages Creek, both on the eastern side of the creek.

With assistance from Cyrus Brame of the U.S. Fish and Wildlife Service and his intern, Robert Gabay, three SETs (SETs 8-10) were installed along the southeastern creek on Presquile Island on 16 June 2016. All of these SETs were installed on the southeastern bank of the creek.

On Powells Creek at James River National Wildlife Refuge, three SETs (SETs 16-18) were installed on 12 July 2016. All of these SETs were installed on the eastern bank of the creek. The southernmost (SET 18) was installed along a small tributary, about 100 meters away from the main channel of Powells Creek. Cyrus Brame and Robert Gabay were also present to help with site selection and installation in JRNWR.

All SETs in all wetlands were read on a 2-month cycle, and from installation we allowed each SET two weeks to settle prior to the zero measurement.

Installation Procedures

Installation of the 18 SET benchmarks was executed by groups of at least 3 persons beginning 10 June 2016 with the final SETs being installed 12 July 2016. The first SET was installed near the mouth of Harris Creek on the eastern bank under the supervision of Alex Demeo and Claudia Deegs from Virginia Institute of Marine Sciences. Moving forward, SETs were installed without VIMS supervision but following the protocol that was laid out during our training with VIMS and adapted for the unique needs of our habitats and substrates.

Equipment and material was carried to each site by hand or carried by boat, depending on location specifics. Site selection was based first on general location as we moved up the estuarine gradient in the selected channels. After we reached a general area, we chose specific locations for the benchmarks in areas that represented neither a hummock nor a hollow; we selected benchmark locations to represent accretion and elevation change as they occur at average wetland elevations.

After benchmark locations were selected, we followed standard protocols and procedures for SET installations, with slight modifications to accommodate the unique substrates and hydrology of the habitats to be studied. We prepared the site by placing two 3.05 m length, 24.40 cm X 5.08 cm lumber planks on top of plastic Rubbermaid step stools in a parallel orientation, about 0.60 meters apart. The planks were placed to span the site with the future benchmark location at center. The step stools had small sealant-treated planks attached to the feet to keep them from sinking into the substrate. This setup allowed us to work from the platforms while minimizing impact to substrate within the sampling stations. Equipment and materials sensitive to water were placed on a large tarp to keep dry in the saturated environments.

At the point selected for the benchmark, a hole was dug approximately 0.61 m deep and 15.24 cm in diameter using posthole diggers. Placed immediately in the hole was a pre-cut, 0.61 m segment of 15.24 cm diameter PVC. This is where our protocol differed from standard, as the water content of the saturated substrate would immediately back fill the hole dug if we waited until after rod installation to install the PVC collar. By immediately installing the collar, we prevented backfill of the hole during the remainder of benchmark installation and establishment.

With PVC installed, we connected two 1.23 m steel rod segments and a driving tip, with joints secured using electric tape. We began driving rods by hand, attaching more rods as needed, until reaching the "by-hand" refusal. Attaching a driving head to the top rod, we began driving the rods with a 21 lb. Bosch demolition hammer. We continued attaching rods, moving the driving head to the top rod, and driving with the demolition hammer until we reached refusal. Refusal was identified when rod movement was less than one foot per minute using the demolition hammer. If the rods were at refusal, but sitting above sediment surface, and we were unable to drive further, we cut them to the appropriate height with an angle grinder; approximately 2 cm below the top of the PVC collar.

At this point, a stainless steal receiver was attached to the benchmark pipe using four bolts adjusted to best level the top of the receiver as measured with a torpedo level. Quick-Set concrete was mixed on site using locally procured water, and poured to the brim of the PVC collar with the top of receiver protruding. The cement was crowned at the top using a trough to promote water shedding and a pre-stamped monument marker was placed in the cement as it dried.

We built a 9 m^2 scaffolding around each benchmark pipe using 3 meter segments of 1.27 cm PVC placed in a square orientation attached to 5 foot PVC leg segments that were driven about 1 m into the ground. This scaffolding sat approximately 0.46 – 0.61 meters above the ground and both helped to identify the plots and prevent hunters and other researchers from unknowingly trampling the grounds within the sampling station.

In the corners of the sampling stations we placed four .25 m^2 feldspar marker horizons. In certain circumstances where a hummock, log, or other natural obstruction prohibited marker horizon placement in the corner of a sampling station, the horizon was offset 50 cm from the corner. Marker horizons were placed about 1 cm in depth in accordance with pre-existing USGS protocols. We established the marker horizons at the zero measurement of their respective SETs and first measured them at the second SET measurement.

Vita

Ronaldo Lopez was born on July 25, 1983, in Queens, New York, and is an American citizen. He graduated from Rockbridge County High School, Lexington, Virginia in 2001. He received his Bachelor of Arts in Economics from Hampden-Sydney College, Farmville, Virginia in 2005. He graduated with a Master of Science in Environmental Studies from Virginia Commonwealth University in 2017.