# Sentinels of Change — Are Salt Marshes in LIS Keeping Pace with Sea Level Rise?

## Final Report

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# Appendix Tables

| Acronym/ Abbreviation | Definition | Acronym/ Abbreviation | Definition |
|---|---|---|---|
| s | Accretion Rate (cm yr-1) | $^{210}Pb$ | Lead-210 |
| t | Age (yr) | $^{210}Pb_{xs}$ | Lead-210 Excess |
| ANOVA | Analysis of Variance | A | Lead-210 Activity at Depth |
| BI | Barn Island | $A_o$ | Lead-210 Initial Activity |
| CIC | Constant Initial Concentration Model | $Q_x$ | Lead-210 Inventory Below Depth x |
| C | Carbon | $Q_o$ | Lead-210 Inventory Total |
| $^{14}C$ | Carbon-14 | LIS | Long Island Sound |
| $CO_2$ | Carbon Dioxide | LOI | Loss on Ignition |
| C:N | Carbon:Nitrogen | m | Meter |
| CP | Caumsett Park | uM | Micromole |
| °C | Celcius | mg | Milligram |
| $^{137}Cs$ | Cesium-137 | mm | Millimeter |
| cm | Centimeter | mmol | Millimole |
| chl *a* | Chlorophyll a | NEIWPCC | New England Interstate Water Pollution Control Commission |
| CRS | Constant Rate of Supply Model | NY | New York |
| CT | Connecticut | NYC | New York City |
| x | Depth (cm) | N | Nitrogen |
| DIN | Dissolved Inorganic Nitrogen | $\lambda$ | Radioactive Decay Constant |
| DIP | Dissolved Inorganic Phosphorous | $^{226}Ra$ | Radium-226 |
| DSi | Dissolved Inorganic Silica | RSLR | Relative Sea Level Rise |
| DO | Dissolved Oxygen | $S_i$ | Sediment Accretion Rate Inorganic |
| g | Gram | $S_o$ | Sediment Accretion Rate Organic |
| GC | Gas Chromatograph | $S_t$ | Sediment Accumulation Total |
| H | Helium | $D_i$ | Sediment Density |
| hr | Hour | SET | Sedimentation Erosion Table |
| HI | Hunter Island | r | Slope of Regression |
| FID | Flame Ionization Detector | Temp | Temperature |
| JC | Jarvis Creek | UC | Udalls Cove |
| keV | Kiloelectron Volt | WR | Wading River |
| km | Kilometer | yr | Year |

# Executive Summary

Salt marshes provide a range of key ecosystem services including mitigating storm surge, filtering nutrients, and providing essential habitat for fish and birds. Yet as a borderland between terrestrial and marine ecosystems they are exposed to numerous anthropogenic impacts that degrade their ability to perform these activities. Rapid increases in relative sea level rise (RSLR) are one of the primary ways human activities have altered salt marshes. While historically salt marshes have been capable of maintaining their elevation in relation to sea level rise, the acceleration of sea level in recent years has put them at a greater risk of drowning (Valiela et al. 2015). The goal of this research was twofold. The first goal was to quantify recent rates of salt marsh accretion at sites throughout Long Island Sound (LIS) and compare them to recent (>2000) sea level rise in the area. The second goal was to better understand how fast the salt marshes in LIS were respiring organic matter to understand whether organic matter loss is influencing accretion rates. To do this, seasonal rates of salt marsh respiration (and photosynthesis for vegetated sites via $CO_2$ fluxes) as well as rates of litterbag decomposition were quantified. Six sites were chosen for this study, three on the north and three on the south side of LIS with two in Connecticut (CT) and four in New York (NY) (Figure 2). Importantly, accretion rates at these sites had previously been measured, which allowed for the comparison of recent rates with historic rates. These six sites spanned a nutrient gradient, west (high) to east (low), allowing for the indirect examination of the role of increased nutrients in salt marsh decomposition.

Five gauges in LIS were used to determine sea level rise rates since 2000. Sea level was found to be rising 0.69 to 0.75 cm $yr^{-1}$ in this region, which is 3 to 4X higher than regional historic rates and the current global average, but consistent with recent studies in the area. Historic LIS salt marsh accretion rates ranged between 0.08 and 0.68 cm $yr^{-1}$ with a mean of 0.2 cm $yr^{-1}$. The accretion rates (range: 0.21 to 0.48 cm $yr^{-1}$) measured in this study with radionuclides ($^{210}Pb$ and $^{137}Cs$) were 2 to 3X slower than rates of current RSLR These results indicate that at the time of these measurements, these salt marshes were not increasing fast enough to keep pace with RSLR. Barn Island followed by Hunter Island appear to have the lowest accretion rates with Wading River and Udalls Cove the highest rates. There was no significant different between current and past rates collected and in the literature, though accretion rates increased in CRS model data of individual accretion rates over time. The general trends between the two measurements were also consistent.

Decomposition rates indicate how quickly marshes lose organic matter that otherwise would contribute to accretion rates. Rates in the six study sites ranged from 0.08 to 0.36 $month^{-1}$ with an average rate of 0.16 $month^{-1}$. Barn Island and Jarvis Creek had the highest and lowest rates respectively and the remaining marshes all had relatively similar rates.

Mean $CO_2$ fluxes between sites were only different at Udalls Cove, where significantly higher emission of $CO_2$ were measured in unvegetated areas. As expected they were also different between vegetated or unvegetated areas and by season. The range of $CO_2$ fluxes in vegetated areas was from -25.66 to 22.34 mmol $m^{-2}$ $hr^{-1}$ and from -8.29 to 22.29 mmol $m^{-2}$ $hr^{-1}$ in unvegetated areas with averages of -5.76 and 2.54 mmol $m^{-2}$ $hr^{-1}$ respectively. The overlap in ranges was caused by a higher emission of $CO_2$ measured in Udalls Cove from a vegetated site in the summer. Seasonal trends of the largest emission in winter and largest uptake in summer were found. Udalls Cove proved an exception to this trend as it had particularly high rates of $CO_2$ emission throughout the year. $CO_2$ uptake in vegetated stands significantly increased with air and soil temperature as well as soil moisture, which is not surprising since plants grow fastest during warmer seasons such as summer and wetter seasons such as spring. In contrast, $CO_2$ emissions from un-vegetated plots significantly increased with air and soil temperature and had no relationship with soil moisture. Litterbag decomposition rates and $CO_2$ fluxes from vegetated plots were significantly related in spring and fall although the relationship is most linear in fall.

Overall, this study shows that salt marshes do not appear to be keeping pace with current rates of sea level rise in LIS although recent rates of accretion may be increasing. Additionally, $CO_2$ fluxes varied seasonally by site and Udalls Cove had significantly higher emission than the rest of the sites, which could indicate an enhanced loss of organic matter. No patterns were found between accretion rates, decomposition, $CO_2$ fluxes and distance from the east river – the proxy for nutrient concentrations. However, site specific nutrient concentrations are likely a more important measurement and such additional information may add clarity to the patterns observed.

# Introduction

Coastal systems are strongly impacted by anthropogenic forcings across local (e.g., land use change, nutrient pollution) and global (e.g., warming water temperatures, ocean acidification) scales. Determining how these forcings interact to alter the environment is complicated and challenging. One way to tease apart these changes is to examine long-term datasets. In doing so unexpected ecosystem responses are often observed such as large changes in nutrient fluxes, algal blooms, and other environmental processes (Fulweiler et al. 2007; Fulweiler and Nixon 2009; Nixon et al. 2009a).

Salt marsh ecosystems are particularly vulnerable to human activities as they lie at the border between terrestrial and marine environments. Thus, they experience human impacts from both sides. However, these systems are also remarkably important because they provide a range of ecosystem services. For example, they are critical nursery habitat for commercially valuable fish and birds (Brawley et al. 1998; Warren et al. 2002; Kimball and Able 2007), they mitigate storm surge (Williams 2012), filter nutrients (Sousa et al. 2012), and sequester carbon (Sousa et al. 2010).

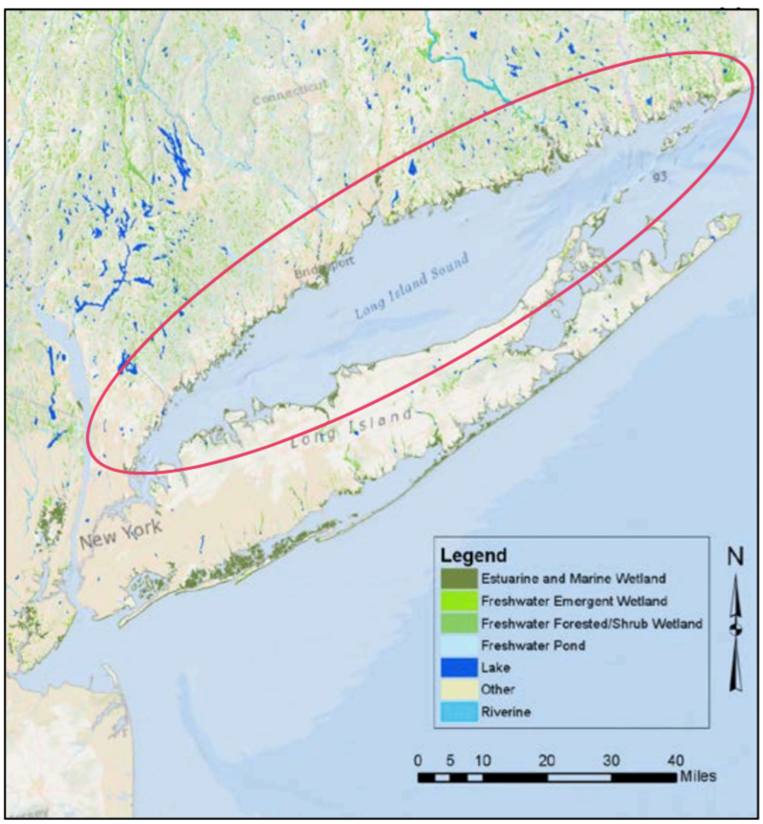


**Figure 1.** Map showing wetland environments in Long Island Sound (LIS). Dark green represents estuarine and marine wetland areas and are the focus areas of this proposal. LIS borders are roughly outlined with the pink oval.

While historically North American salt marshes have mostly kept pace with sea level rise recent estimates predict rates of salt marsh loss of >65% along the Northeast coast of the US (New Jersey to Maine) (Nicholls et al. 1999). These projections align with predictions by global climate models that this region will experience a faster and larger degree of relative sea level rise (RSLR) compared to global averages (Yin et al. 2009). In fact, these impacts are already being observed, as the Northeast has already experienced rates of RSLR 3 to 4X the global average throughout the last three decades (Sallenger et al. 2012). Determining how salt marshes will respond to RSLR is difficult because their vertical accretion depends on a suite of factors, including sediment supply, organic matter production, and decomposition rates (Mccaffrey and Thomson 1980b; Nixon 1982; Kirwan and Mudd 2012). Experimental work (Langley et al. 2009), model simulations (Kirwan and Mudd 2012), and some field observations (Kolker et al. 2010) indicate that salt marshes may keep pace with RSLR due to increased primary production accompanying elevated atmospheric $CO_2$ concentrations, and higher sediment availability following increased tidal inundation. On the other hand, wetter marsh platforms alter vegetation patterns (Warren and Niering 1993; Donnelly and Bertness 2001) and suggest that marshes in the Northeast (defined here as New Jersey to Maine; Roman et al. 2000; Hartig et al. 2002) may not be keeping up with RSLR. In particular, marshes appear to be vulnerable when the rate of RSLR is rapid (0.5 cm $yr^{-1}$ of RSLR) and/or where sediment loads are low (<20 mg $L^{-1}$) (Kirwan et al. 2010). In these conditions the marsh platform deepens and vegetation dies off as a result of inundation stress (Nyman 1993; Morris et al. 2005; Nelson and Zavaleta 2012).

Recently, RSLR and salt marsh accretion rates were examined in Narragansett Bay, RI (Carey et al. 2015). To do this, sites measured over thirty years ago by Bricker-Urso were revisited (Bricker-Urso et al. 1989) and radionuclide tracers (lead-210 ($^{210}$Pb) and cesium-137 ($^{137}$Cs)) were used to reassess vertical accretion rates. Since the time of that study, RSLR in Narragansett Bay has significantly increased, from 0.26±0.02 cm $yr^{-1}$ (1931-1983) to 0.41±0.07 cm $yr^{-1}$ (1984 - 2011) (NOAA 2012), similar to the trends observed throughout the Northeast (Sallenger et al. 2012). Moreover, the last decade has seen dramatic increases in RSLR, rising to 0.91±0.2 cm $yr^{-1}$ (2000-2011) (Noaa 2012), likely driven by both climate and meteorological factors (Kolker et al. 2009a). We hypothesized that the combination of an increased rate of RSLR, limited sediment supply (Nixon 1982; Roman et al. 2000), and warmer winter temperatures (Nixon et al. 2009b) could restrict the ability of these organic-rich marshes of the Northeast to keep pace with accelerated RSLR.

This work has highlighted at least two critical points relevant to LIS. First, Northeast salt marshes respond quickly to RSLR and thus LIS salt marsh accretion rates may also not be keeping pace with RSLR. Second, salt marshes are harbingers of climate change. Careful monitoring of accretion rates and other important site variables should help track, understand, and hopefully mitigate some of these changes. Importantly, this work exposes an interesting question – are increased temperatures causing enhanced decomposition of salt marsh organic matter, leading to decreases in organic accretion rates? Additionally, how wide-spread is this phenomenon?

Long Island Sound (LIS) provides a unique opportunity to study these questions while simultaneously developing a program that will allow for the assessment of how climate change is impacting this area. LIS is fringed with salt marshes (Fig. 1) and has a west to east nutrient gradient from heavy nitrogen loading closest to New York City (NYC) to much lower concentrations at the east end of LIS. It has been well studied for a variety of important environmental characteristics including dissolved oxygen, nutrients, air and water temperature, wind, precipitation, etc. The salt marshes around LIS have been studied for a variety of ecosystem services and numerous accretion rates have been estimated using radionuclide measurements. This report contains a large synthesis of existing data and newly collected data for marsh accretion and decomposition rates. This work will provide a solid foundation on which to monitor future climate change impacts on LIS.

# Objectives

**The purpose of this research was to determine if accretion rates in LIS salt marshes are keeping pace with sea level rise.** The **specific objectives** of this proposal were to:

1. Quantify salt marsh accretion rates and other site characteristics at a select set of sites in LIS using two dating methods ($^{137}$Cs and 'excess' $^{210}$Pb).

2. Measure rates of salt marsh decomposition (through litterbags % weight loss and $CO_2$ effluxes) at sites throughout LIS.

3. Collect and synthesize data throughout LIS for environmental factors indicative of local human impacts and climate change including: nutrient loading, air and water temperatures, sea level rise, wind, and precipitation; additionally all previously measured salt marsh accretion rates in LIS were gathered for comparisons to the data collected in this study. *Please note that another proposal was funded that focused on this objective and recently a robust review of environmental conditions and the current state of LIS was recently published titled, "LIS – Prospects for the Urban Sea" (Latimer et al. 2013). Thus, the research and this report focus on historic accretion rates and objectives one and two. Some long-term data was collected that related this research and those results can be found in Appendix 1.*

# Methodology and Data Collection

## Site Description

LIS is over 160 km long and over 32 km wide with the East River on its western end and the Atlantic Ocean on its eastern end (Figure 1). This creates a decreasing salinity gradient from the ocean to the river. Due to its size, six salt marsh sites were selected, with three on the southern New York/Connecticut shore and three on the northern Long Island, New York shore (Figure 2). The sites were also chosen to enable examination of any potential differences along the west to east nutrient gradient. The gradient is highest to the west and decreases exponentially going east into LIS (Figure 3). As much as possible, sites that had been studied in the past were chosen in order to build on previous work.

### *Hunter Island (HI), Pelham Bay State Park, Bronx, NY*

Hunter Island is a long peninsula extending into the Western end of LIS just north of the mouth of the East River and next to the Hutchinson River. This study site was a small, well-preserved marsh on the north side of the base of the peninsula. Most of the marsh platform was covered with *Spartina patens* with a band of tall-form *Spartina alterniflora* along the creeks and a small amount of short-form *S. alterniflora* in between. There were many mussels in the tall form *S. alterniflora*, but little evidence of crab boroughs. Salt marsh accretion rates had been measured previously by Cochran et al. (1998).

### *Udalls Cove Park Preserve (UC), NYC Park, Queens, NY*

Udalls cove is a large marsh in a residential area of Queens close to the western end of LIS in Little Neck Bay. One part of the marsh has a sewage pipe running through it from Nassau County. There were some bare, sediment only areas across the marsh and a gradient of tall- to short-form *S. alterniflora* extending a few meters into the marsh with a small amount of *Typha* scattered along the edge and *S. patens* upland of this low marsh area. This marsh was not previously studied, but the marsh directly adjacent to this marsh, Alley Pond Park, Queens, NYC, was studied by Cochran et al. (1998) and these rates were used for historical comparisons. At the time of this study, Alley Pond Park was in the process of being restored and could not be accessed.

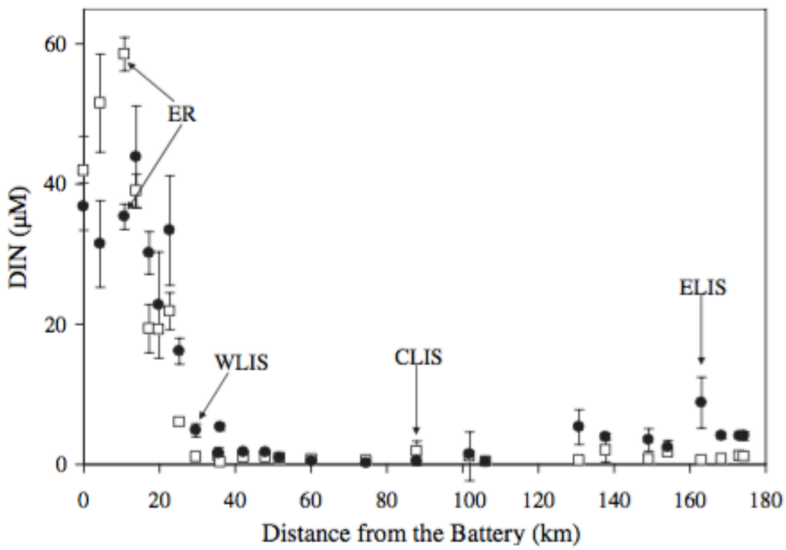


**Figure 3.** Dissolved inorganic nitrogen in LIS. This figure is from Gobler et al. 2006 showing the concentration of dissolved inorganic nitrogen starting in the western end of LIS and going to the eastern end. Concentrations are highest at very close proximities to the city and decrease at a fast pace going east into the LIS.

### *Jarvis Creek (JC), Branford Land Trust, Branford, CT*

Jarvis Creek marsh is located along the northern short of CT in the heart of one of the Brandford Land Trust Marsh and is relatively isolated. It is along the northern shore towards the middle of the sound. Until 1979, the upper portion of this marsh had significant flow restriction caused by a tide gate (Anisfeld 2008), but the sampling site was seaward of this structure. Vegetation in this marsh was primarily short-form *S. alterniflora* with some tall-form *S. alterniflora* along portions of the tidal channel. The vegetation was relatively thin and there was considerable evidence of crab boroughs. Measurements were previously taken in this marsh by Anisfeld (2008).

### *Caumsett State Historic Park Preserve (CP), State Park, Lloyd Neck, NY*

Caumsett State Historic Park Preserve salt marsh is located along the northern end of a large peninsula extending into LIS. It was a relatively untouched area with a considerable amount of wildlife and the marsh itself was just on the landward side of a barrier beach. The vegetation was a mix of short and tall-form *S. alterniflora* with *S. patens* meters upland of the creek. There were numerous mussels and some crabs in this marsh. Previous measurements had been taken here by Cochran et al. 1998.

### *Barn Island Wildlife Management Area (BI), State Park, Stonington, CT*

Barn Island salt marsh is located on the very eastern end of LIS. This was a very peaty marsh with *S. alterniflora* tall and short-form along the creeks and mostly *S. patens* on the platform with some patches of *Salicornia europaea*. Many studies have been conducted at Barn Island (Bloom 1964; Harrison and Bloom 1977; Young et al. 1985; Warren and Niering 1993; Orson et al. 1998; Carey et al. 2015) and the $^{210}$Pb rates from Orson et al.

(1998) were used for comparison.

### *Wading River Marsh (WR), Nature Conservancy, Wading River, NY*

Wading River Marsh is a salt marsh located on the eastern edge of Long Island. The small Wading River flows north through the marsh and discharges into LIS. It is inland of a barrier beach lined with houses separating it from the ocean, but the river itself is not restricted. It is next to the site where the Shoreham Nuclear Power Plant was sited, which was never brought into operation. The platform is primarily short-form *S. alterniflora* with long-form *S. alterniflora* along the creek and *S. patens* further upland. There is evidence of both mussels and crab boroughs along the creek. A study was done by Young et al. (1985) at this location.

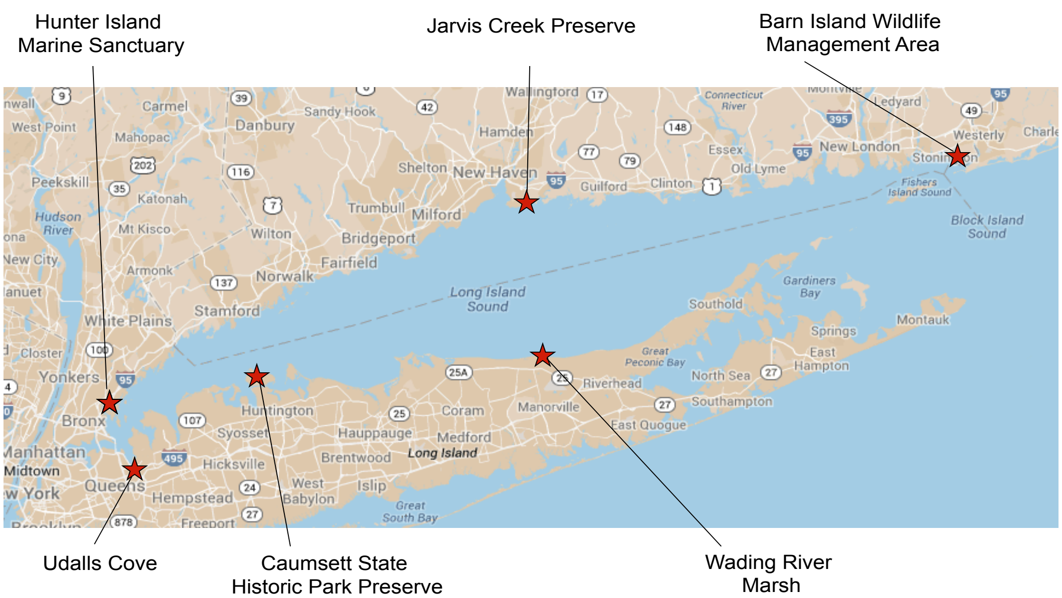


**Figure 2.** Map of Long Island Sound showing the study sites for this project. Red stars indicate the site location.

# Salt Marsh Core Collection

For each site, sediment cores were collected using a half cylinder gouge auger corer (Figure 4). One core was used for sediment characteristic analysis and the other was used to measure salt marsh accretion via radionuclide techniques (Table 1). The two sediment cores were taken at each marsh in a relatively solid, dry area of the low marsh platform with careful consideration to minimize compression of the core while also retrieving a core of approximately 50 to 75 cm. During extraction, cores were monitored for compaction by visual inspection of the top of the core and if compaction was noticed, a new core was taken. After each core collection the auger corer was rinsed with deionized water and dried with a clean cloth. The cores were immediately frozen and taken back to the lab for processing where they were cut into 1 cm increments down to a depth of up to 50 cm (Bricker-Urso et al. 1989).

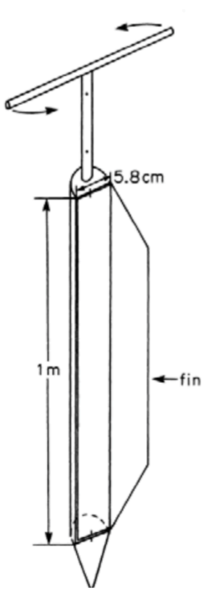


**Figure 4.** Example of salt marsh corer. A 6 cm diameter gouge auger corer was used to collect all sediment cores at all sites.

# Relative Sea Level Rise (RSLR)

Sea level data from 5 tidal gauges in Connecticut (CT) and New York (NY) in LIS was examined. Only gauges with more than 15 years of data were used (Table 2). The closest gauges to each site were determined by calculating the Euclidean distances between all of the sites and all of the gauges and for each site choosing the gauge at the shortest distance. Only these sites were used going forward. Monthly averages were attained and yearly averages calculated from the data. Current RSLR was also calculated using sea level data from after 2000, which is possibly a turning point in RSLR acceleration and salt marsh response (Valiela 2015) and has been used by other studies (Raposa et al. 2015; Carey et al. 2015b). The gauges and data are maintained by the National Ocean and Atmospheric Administration (http://tidesandcurrents.noaa.gov/tide_predictions.html).

# Historic and Current Salt Marsh Accretion Rates

## Historic Accretion Rates

A thorough literature review was conducted and seventy-four salt marsh accretion rates from twenty-one studies in CT and NY were compiled (Table A2.3). They were found using references of related papers, personal communications, and web-based bibliographic software with keywords such as Long Island Sound, Connecticut, New York, accretion, salt marsh, accretion rates, sedimentation. Certain environmental parameters, such as loss on ignition (LOI), tidal range, marsh type, and marsh area (high, low, and middle marsh), were also extracted from the published literature and governmental monitoring programs.

The data review was restricted to fluvial or back barrier marshes (Wood et al. 1990, Goodman et al. 2007), which were identified by the study descriptions or by visual identification of rivers leading into marshes using satellite images. Rates from low, mid-, or high salt marsh locations were also included. Any rates from marshes that were described as restricted or recovering from restriction were not included(Anisfeld et al. 1999).

The majority of the rates were determined with radionuclide methods such as $^{210}$Pb and $^{137}$Cs. Other methods included marker horizons, $^{14}$C, sedimentation erosion tables (SET), historic information, storm deposits, opaque spherules and pollen. In order to compare salt marsh accretion rates across these studies the impact of method on accretion rates was examined first.

## Current Accretion Rates

One sediment core for salt marsh accretion rate analysis was collected from each of the six study sites. At one site (Barn Island) duplicate cores were collected to check consistency. The same subset of cm sections was used to analyze accretion rates as was used for LOI and half of the bulk density samples. Sediment chronology was calculated using radionuclide profiles of $^{210}$Pb, $^{226}$Ra, and $^{137}$Cs. A pure germanium gamma spectrometer (Canberra, GCW3023) was used to measure gamma emissions of $^{210}$Pb (46.4 keV), $^{214}$Pb (352 keV, used to measure $^{226}$Ra), and $^{137}$Cs (661 keV) in the Moran Laboratory at the Graduate School of Oceanography at the University of Rhode Island (Carey et al. 2015). Then the $^{137}$Cs-derived accretion rates were calculated by dividing the depth of the $^{137}$Cs activity peak by the number of years between core collection and 1963 as the year of maximum production (Turner et al. 2002). $^{210}$Pb accretion rates were calculated using two models – the constant initial concentration model (CIC) (Krishnaswamy et al. 1971) and the constant rate of supply model (CRS) (Appleby and Oldfield 1978; Mccaffrey and Thomson 1980a). Both models are based on ‘excess’ $^{210}$Pb ($^{210}$PbXS) activities. The CIC model assumes a constant rate of sedimentation over time and is represented by the following equation:

$$A = A_0 e^{(-\lambda/s)x}$$

Where A is the $^{210}Pb_{XS}$ activity at a given depth, $A_0$ is the initial activity of $^{210}Pb_{XS}$ at the marsh surface, $\lambda$ is the radioactive decay constant for $^{210}$Pb (0.03101 yr$^{-1}$), $s$ is the accretion rate (cm yr$^{-1}$), and $_x$ is the depth of the core (cm). Assuming minimal compaction, a single accretion rate calculated from plotting the natural log of the excess activity vs. the depth of the core. Slope of the line is given by:

$$r = -\lambda / s$$

where $r$ is the slope of the regression and accretion rate is $s$.

The CRS model allows for variable sedimentation rates (Appleby and Oldfield 1978; Mccaffrey and Thomson 1980; Kolker et al. 2009) and is represented by the following equation:

$$Q_X = Q_0 e^{-\lambda t}$$

$Q_x$ is the inventory of $^{210}Pb_{xs}$ below depth $x$ (dpm $cm^{-2}$), $Q_0$ is the total inventory (dpm $cm^{-2}$), $t$ is the age (yr) of depth $x$ (cm).

The percent of accretion from organic or inorganic material was also estimated (Bricker-Urso et al. 1989). To do so the following equation was used,

$$S_i = ((St)(LOI)) \times D_i^{-1},$$

where $S_t$ is the total average sediment accumulation (g $cm^{-2}$ $year^{-1}$), LOI is the ratio of loss on ignition for organic (%LOI/100) and inorganic (1-%LOI/100), $D_i$ is sediment density (1.1 g $cm^{-3}$ and 2.6 g $cm^{-3}$ for organic and inorganic, respectively), and $S_i$ is the inorganic (or organic) sediment accretion rate (cm $year^{-1}$).

# Sediment Characteristics

Two cores were collected from each site besides BI where three cores were collected. The cores were cut in 1 cm increments to a depth of 50 cm. One core from each site was used to determine the chronology of sediment using radionuclide profiles $^{210}Pb$ and $^{137}Cs$. In BI, two cores were used to determine the sediment geochronology. The second core was used to assess sediment characteristics such as bulk density, water content, LOI, and sediment C:N. For bulk density and water content, every cm section of the core at every depth was analyzed in JC, BI, and WR. In HI, UC, CP the top 10 cm, every other cm to a depth of 20, and every fifth cm to a depth of 50 were analyzed. This same subset was used for LOI analysis. Only the top 10 cm were analyzed for CHN (Table 1). All samples analyzed were averaged to get the final site value.

**Table 1.** Sediment Core Information. Subsets of the approximately 50 samples were analyzed for each sediment characteristic. The size of the subsets varied by characteristic and ranged from the whole core for some bulk density measurements to the top 10 cm for CHN. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

**Sediment Cores**

| Site | Core no. | Core Vegetation | Core depth (cm) | Analysis | Previous Study |
|---|---|---|---|---|---|
| HI | 1 | *S. alterniflora* | 50 | Accretion Rate | Cochran et al., 1998 |
| | 2 | *S. alterniflora* | 50 | Sediment Characteristics | |
| UC | 3 | *S. alterniflora* | 50 | Accretion Rate | Cochran et al., 1998* |
| | 4 | *S. alterniflora* | 50 | Sediment Characteristics | |
| JC | 5 | *S. alterniflora* | 50 | Accretion Rate | Anisfeld et al., 1999 |
| | 6 | *S. alterniflora* | 49 | Sediment Characteristics | |
| CP | 7 | *S. alterniflora* | 50 | Accretion Rate | Cochran et al., 1998 |
| | 8 | *S. alterniflora* | 50 | Sediment Characteristics | |

| | | | | | |
|---|---|---|---|---|---|
| BI | 9 | *S. alterniflora* | 50 | Accretion Rate | Orson et al., 1998 |
| | 10 | *S. alterniflora* | 50 | Sediment Characteristics | |
| | 11 | *S. alterniflora* | 50 | Accretion Rate | |
| WR | 12 | *S. alterniflora* | 50 | Accretion Rate | Young et al., 1985 |
| | 13 | *S. alterniflora* | 50 | Sediment Characteristics | |

*This study was done in Alley Pond Park, which is an adjacent salt marsh to Udalls Cove. At the time of this study, Alley Pond Park was being restored, so Udalls Cove was used, but rates from Alley Pond Park were used for comparison.

## Bulk Density

Subsection samples (1 cm increments) were weighed for wet weight, dried in an oven at 60 °C for at least 48 hours, weighed, and then remained in the drying oven until a constant dry weight was obtained (Bricker-Urso et al. 1989; Carey et al. 2015). The bulk density for each sample was calculated by dividing the dry weight by the volume to get g $cm^{-3}$. The final bulk density values that were used for analysis for each site were the average of the measurements for all depths analyzed.

## Loss on Ignition (LOI)

After being dried, samples were ground using a KLECO 4 canister ball mill. Between 3 and 4 g of sample was weighed out and burned in a muffle furnace for 6 hours at 500 °C to remove all organic matter (Bricker-Urso et al. 1989; Carey et al. 2015). Sediment organic matter was determined by subtracting the post-burn weight from the pre-burn weight and converting it into a percentage. The final LOI values that were used for analysis for each site were the average of the measurements for all depths analyzed.

## Water Content

The water content was calculated by subtracting the dry weight from the wet weight and converting it into a percentage. The final water content values that were used for analysis for each site were the average of the measurements for all depths analyzed.

## Sediment C:N

Replicate samples from each 1 cm increment of salt marsh core were weighed out to 5 ± 0.5 mg, packed into tin capsules, and sent to the BU Stable Isotope Lab for analysis. Samples were combusted in a Fisons NA1500 elemental analyzer and measured using Eager200 software (Swider et al. 1989; Anisfeld et al. 1999). Internal standards were run every 6 samples. The final C:N values that were used for analysis for each site were the average of the measurements for all depths analyzed.

# Decomposition

## Litterbags

Known amounts of dead vegetation were placed in each marsh for an extended period of time and the amount of biomass lost over that time was determined. This provided a measure of how long it takes for organic matter to decompose at each site. Senesced salt marsh vegetation (i.e., *S. alterniflora*) was collected from each of the marsh sites and air dried in the laboratory at room temperature (~25 °C). Six sets of triplicate litterbags were constructed for each site. Each litterbag, containing 1 g of site-specific dried plant material, was sealed into a nylon mesh bag and weighed. In July 2014, litterbags were distributed at each site in the low marsh zone and anchored in place with small wire staples (Figure 5). Three litterbags were collected from each site six times with the last bags being retrieved in March 2015. The litterbags were to be collected by January, but snow and ice covering the salt marsh platform blocked access. The first five collections were made approximately one month apart though logistics only allowed for their collection less than a week after a month had passed. The five collections were made over a span of approximately six months ending at the end of December 2014. After being collected, all litterbags were brought back to the lab, dried at 60 °C for 3 days or until a constant weight was reached (Valiela et al. 1985; Foote and Reynolds 1997; White et al. 1978; Warren et al. 2001; Anisfeld 2008). For all litterbags a decline in material was seen except for the last litterbags collected in March that had over-wintered and had noticeable new growth on them when collected. Thus, the percentage of litter lost was calculated from only the July to December measurements. The decomposition rates were taken from the slopes of the exponential regressions of the percentage of material remaining and days since placed. The rate per day was multiplied by 30 days to get the rate per month. Decomposition rates from previous studies in LIS were also collected (Warren et al. 2001; Anisfeld 2008) and all of the data from all studies were converted to a percentage lost per day in order to aid comparisons.


**Figure 5.** Example of litterbag used to measure decomposition. This is an example of a finished litterbag deployed and stapled into one of the study sites, UC.

## Litterbag C:N

Duplicate aliquots of 0.2 g of litter from each litterbag were weighed for CHN analysis. Samples were processed in the same manner described above for the sediment C:N samples (Swider et al. 1989; Anisfeld et al. 1999). The final C:N values that were used for analysis for each site were the average of the measurements from all the bags collected at each site.

# Salt Marsh $CO_2$ Fluxes

## $CO_2$ Fluxes

Carbon dioxide fluxes were measured using closed static chambers once each season to capture the annual range of $CO_2$ fluxes (Moseman-Valtiera et al. 2011, Jorgensen et al. 2012; Emery and Fulweiler 2015). At each site triplicate $CO_2$ fluxes in representative vegetated and non-vegetated locations were measured close to noon during peak photosynthesis on days with no rain and steady PAR readings. Approximately 1 week before the $CO_2$ measurements were made, stainless steel collars (20 cm diameter, 5 cm deep) were driven into the sediment until the top rubber flange was flush with the ground. For each $CO_2$ flux measurement a Plexiglas cylinder (20 cm diameter, 44-46 cm

**Figure 6.** Example of gas chamber used in this study to measure $CO_2$ fluxes. Six gas samples were collected over a half hour and samples brought back to the lab for analysis on a gas chromatograph. Chamber air was mixed with a fan and temperature inside the chamber was recorded.

height) was placed on the collar flange. The top of the chamber was sealed with a pyrex cap and seals between the collar, cylinder, and cap were made airtight with closed cell foam gaskets (Figure 6). Small, battery powered fans mixed the air inside the chambers and temperature, which was used for flux calculations, and humidity were measured with digital sensors attached to the inside of the chamber (Emery and Fulweiler 2015). Gas samples (25 mL) were collected 6 times over the course of the incubation. The first gas sample was collected immediately after the chamber was closed and additional samples were collected approximately every 5 minutes thereafter. Gas samples were collected by inserting a 60 mL plastic syringe with a stopcock into a rubber septum installed on the side of the chamber. This sample was then immediately injected into an ultra-high purity He flushed and evacuated 12 mL Exetainer vial (Labco, UK).

Gas samples were stored in the refrigerator until analysis on a Shimadzu GC-2014 gas chromatograph (GC) equipped with a flame ionization detector (FID) for $CO_2$. The GC was fitted with Hayesep Q and N columns with $N_2$ as the carrier gas at a flow rate of 25 mL per minute. The $CO_2$ concentration was determined using a standard curve and all curves had $R^2 \geq 0.995$ on 5 or 6 points. Flux rates were calculated as the linear change in concentration over time, divided by the chamber footprint area. Only significant ($p<0.05$) fluxes were reported. All other fluxes were assigned a value of 0. Samples were analyzed on the GC within 60 days with the exception of half of the spring samples from Caumsett State Park, which were analyzed 3.5 months later because the GC was undergoing repair. These values were found to be comparable to the fluxes calculated from the first half of the samples from that site, which were run within the 60-day period.

## Environmental Conditions

Data on environmental conditions was collected in order to determine potential driving forces for $CO_2$ fluxes. At each gas headspace sampling point, atmospheric air pressure and temperature were also recorded using a Hach HQd dissolved oxygen probe HACH probe (Hach Instruments, Loveland CO) or a digital weather station. PAR measurements were taken using a Li-Cor LI-190 Quantum Sensor (Li-Cor Biosciences, Lincoln, NE) or a Li-Cor LP-80 PAR/LAI Ceptometer (Li-Cor Biosciences, Lincoln, NE). They were taken within a few meters of the chambers, but above the plant canopy, to represent the maximum conditions experienced by the photosynthesizing leaves. Sediment temperature was measured at 10 cm depth in the sediment in the flux footprint immediately after flux measurement using a digital thermometer and moisture probe.

# Statistics

Accretion and sea level rise rates were converted to cm $yr^{-1}$. When a range was given for one location in a study, the average of the range was taken and used in the analysis. ANOVA was used to test for differences between sites in sediment characteristics, decomposition measurements, gas fluxes, and environmental parameters. ANOVAs were also used to test for differences between marsh areas, marsh type, and current and historic LOI and accretion rates. For sediment characteristics, litter bag C:N, and $CO_2$ fluxes, a one-way ANOVA was run with six levels of sites. For analysis of $CO_2$ fluxes a three-way ANOVA was performed with two levels of vegetation, six levels of sites, and four levels of seasons. For marsh area, marsh type, and current and historic LOI and accretion rates, one-way ANOVAs were run with three levels of marsh area, two levels of marsh types, and two levels of time period for current and historic LOI and accretion rates. When significant differences were found between groups, a post hoc means comparison with Tukey's test for Honestly Significant Differences (HSD) was performed.

For all other analyses, a linear regression analysis was used. A flux was only calculated when the linear regression was significant ($p< 0.05$). All statistical analyses were done in R statistical software Version 0.98.1049.

## Sea Level Rise

Linear regressions were run to compare the water level over time for each tidal gauge. The slopes of the equations for the line of best fit associated with the regression were used as the rates of RSLR. Breakpoint analyses were also performed on the sea level records for each gauge using the segmented package named "Regression Models with Breakpoints/Changepoints Estimation" Version: 0.5-1.1, which was written and is maintained by Vito M. R. Muggeo. This was able to detect whether there is a significant change in slope within a data set (Muggeo 2003) and this was used to determine the start point to be used for all of the records to calculate current RSLR. The year 2000 was chosen as the starting year and the previous rates of RSLR were calculated using the data from these records taken before the year 2000.

# Results and Discussion

## Relative Sea Level Rise (RSLR)

Global sea level has been rising at a rate of 2.1 mm $yr^{-1}$ and this trend has been accelerating (Church and White 2006). Consistent with global trends, sea level in LIS has been rising over the past century (Figure 7). Five gauges were found within LIS and used in this analysis: 3 in CT and 2 in NY. Trends calculated using the entire length of each record ranged from 0.26 to 0.82 cm $yr^{-1}$ (Table 2). Because the length of the sea level rise record impacts the calculated sea level rise rate, rates from the year 2000 forward were also determined. Calculating sea level rise rates in this way provided a range of 0.69 to 0.75 cm $yr^{-1}$ (Table 2). These rates are 3 to 4X higher than the global average (Church and White 2006). This is consistent with the findings of other studies (Yin et al. 2009; Sallenger et al. 2012; Andres et al. 2013), which found a hotspot of RSLR in this area due to the weakening of the Gulf Stream, which normally pulls water away from the northeast coast into the Atlantic Ocean. It is also consistent with RSLR rates calculated in other recent studies using this breakpoint (Carey et al. 2015b; Raposa et al. 2015).

**Table 2.** Sea Level Gauge Information. Data was collected from the closest tidal gauges to our study sites that had sea level records longer than 15 years. The current SLR was calculated using only sea level data from 2000 to 2014. The Historic SLR was calculated using all data from before the year 2000. Because the tidal gauges with shorter records (Kings Point and New Haven) did not have records longer than 15 years before the year 2000, they were excluded from calculations of historic SLR. Instead, the next closest tidal gauge with a record longer than 15 years was used, which was Bridgeport for all sites which had previously been closest to Kings Point or New Haven. The range of current SLR, 0.69 to 0.75 was 3 to 4X higher than historic SLR. All data was collected from NOAA. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

**Sea Level Gauges**

| Site | NOAA Station | Length of Record (yr) | Tidal Range (m) | Current SLR (cm $yr^{-1}$) | Historic SLR Before 2000 (cm $yr^{-1}$) | NOAA Station Before 2000 |
|---|---|---|---|---|---|---|
| HI | Kings Point | 17 | 2.18 | 0.73 | 0.25 | Bridgeport |
| UC | Kings Point | 17 | 2.18 | 0.73 | 0.25 | Bridgeport |
| JC | New Haven | 16 | 1.87 | 0.75 | 0.25 | Bridgeport |
| CP | Kings Point | 17 | 2.18 | 0.73 | 0.25 | Bridgeport |
| BI | New London | 76 | 0.78 | 0.69 | 0.21 | New London |
| WR | New Haven | 16 | 1.87 | 0.75 | 0.25 | Bridgeport |

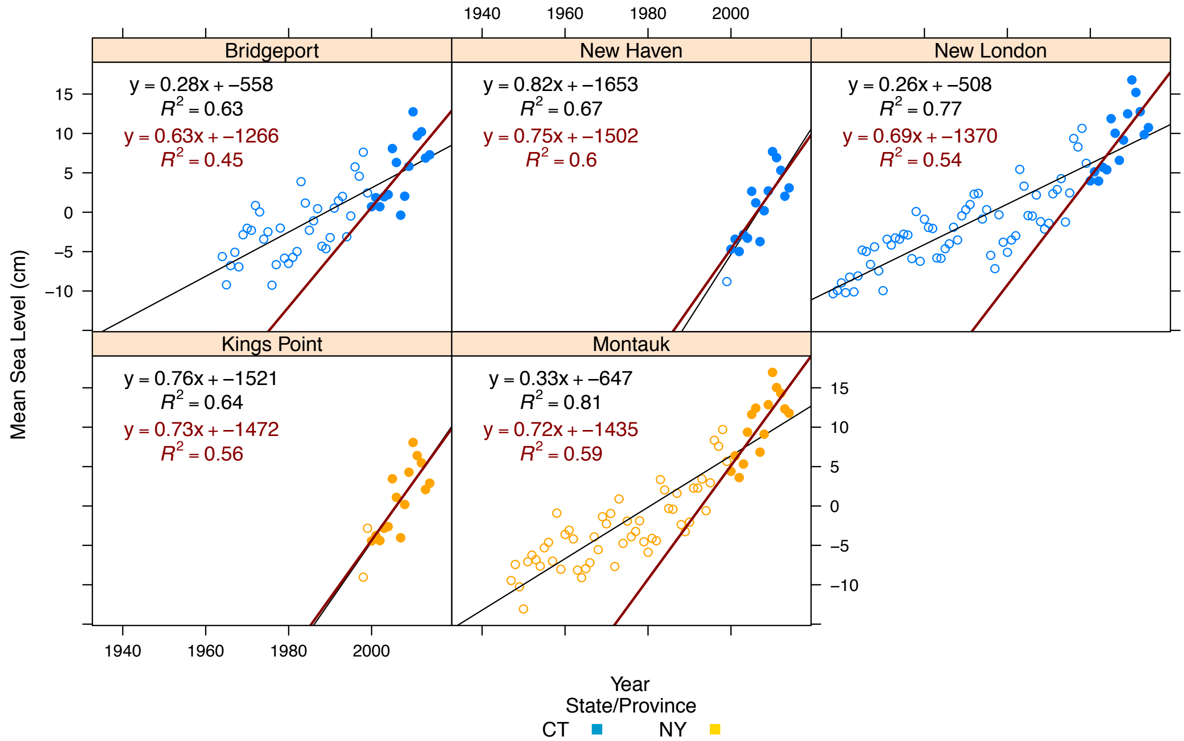


**Figure 7.** Relative sea level data in all five tidal gauges in CT and NY. The filled in circles and red lines are the data from 2000 to 2014 and the regression line for those data for each tidal gauge. The equation and $R^2$ for the regression using the entirety of the data are in black and those for the regression using the subset are in red. The year 2000 was used as the beginning of the records used to calculate current rates of sea level rise because of evidence this was a turning point in sea level rise acceleration and marsh response (Valiela, 2015). All gauges and data are maintained by the National Ocean and Atmospheric Administration (NOAA). Data were obtained here: http://tidesandcurrents.noaa.gov/tide_predictions.html.

# Historic and Current Salt Marsh Accretion Rates

## Historic Accretion Rates

From the literature search on accretion rates in LIS, rates spanning a forty-year period (1972 to 2013) were found. It was found that only the accretion rate determined by $^{14}C$ appeared to be different than the rest and because this was only one data point it was excluded this study from further analysis. No statistical difference between accretion rates within NY or CT were observed nor were differences in accretion rates in different marsh areas found (i.e., low, mid, high marsh) (Figure 8). Accretion rates also do not appear to have changed over time. There were no significant relationships between rates and the percentage of organic matter, which has been found to be an important control on salt marsh accretion rates in other studies (Carey et al. 2015). No statistical difference in accretion rates of different marsh types was observed. Specifically, fluvial marshes had higher rates than back barrier marshes (Figure 9; $p = 0.00021$). This relationship could be driven by increased sediment availability in marshes close to rivers.  To compare historic accretion with sea level rise, RSLR rates between 1938 and 1999, which ranged from 0.21 to 0.25 cm $yr^{-1}$ (Table 2, Figure 10) were calculated. Accretion rates measured throughout LIS salt marshes over this time period ranged from

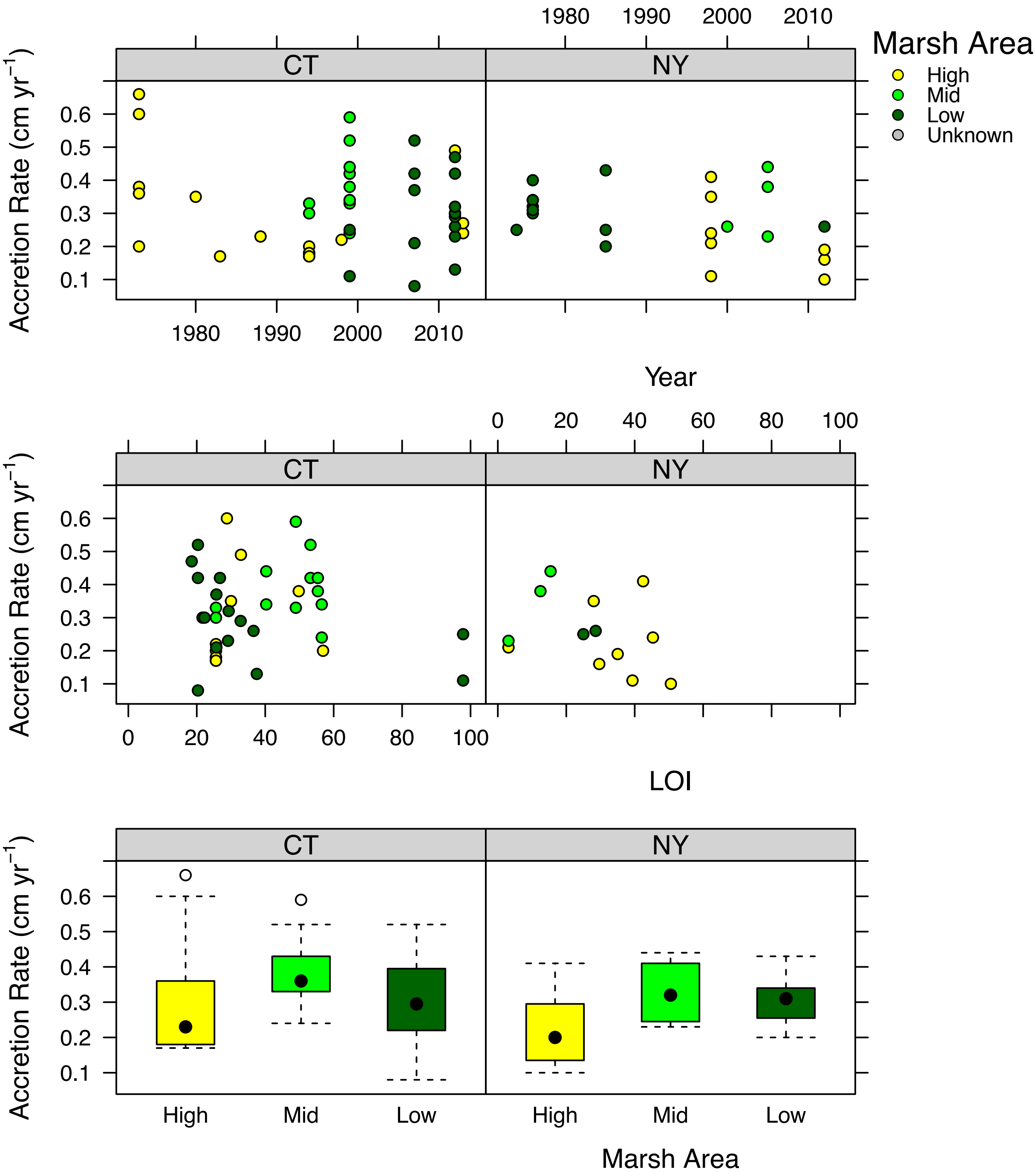


**Figure 8.** Historic accretion rates by different environmental parameters. Accretion rates, as well as other environmental parameters, measured in marshes in LIS were gathered from 21 different studies in the literature. The colors represent marsh area with dark green being low marsh, light green being mid marsh, and yellow being high marsh. Relationships between accretion and a) time, b) LOI, and c) marsh area were all analyzed. No significant differences were found.

0.08 – 0.68 cm $yr^{-1}$ with a mean rate of 0.32 cm $yr^{-1}$. In the past, 77% of marshes appeared to be keeping pace with past sea level rise.

# Current Accretion Rates

$^{210}Pb$ activities ranged from 0.73 to 30.80 dpm $g^{-1}$ (Table 3) and decreased exponentially in all cores (Figure 11). $^{137}Cs$ activities ranged from 0 to 2.83 dpm $g^{-1}$. Peaks were found at depths between 13 and 25 cm in all of the cores taken. The year 1963 was used as the peak year of maximum production of $^{137}Cs$ by which vertical accretion rates were calculated (Delaune et al. 1978; Turner et al. 2000).

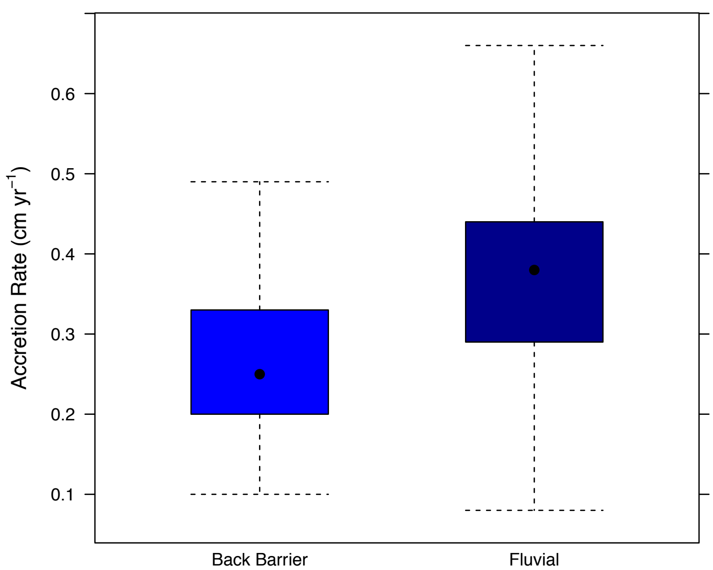


**Figure 9.** Accretion rates from marshes of different types (back barrier=light blue and fluvial=dark blue). Shared letters indicate no significant difference. Accretion rates in fluvial marshes were significantly higher than accretion rates in back barrier marshes ($p=0.0021$).

One assumption of the $^{210}Pb$ model is that salt marshes are receiving $^{210}Pb$ at a constant rate and only from atmospheric deposition (McCaffrey and Thompson 1980; Cochran et al. 1998), which has previously been found to be the case in this region (Benninger 1987). To check this it is possible to compare the total inventories of $^{210}Pb$ to a known range values expected if atmospheric deposition is the only source of $^{210}Pb$, 22 to 38 dpm $cm^{-2}$ (Graustein and Turekian 1986; Cochran et al. 1998; Kolker et al. 2009; Hill et al. 2015; Carey et al. 2015). If the inventory is above this range, the marsh is most likely receiving $^{210}Pb$ from an external source besides atmospheric deposition (Cochran et al. 1998; Hill et al. 2015) and if it is below this, the marsh could be becoming depleted in $^{210}Pb$ through processes such as erosion (McCaffrey and Thompson 1980; Bricker-Urso et al. 1989; Roman et al. 1997). At the sites in this study, inventories had an average of 42.1 dpm $cm^{-2}$ and values ranged from 29.2 to 67.0 dpm $cm^{-2}$. The average was a little above the normal range and excess $^{210}Pb$ was found at Hunter Island, Udalls Cove, and Barn Island with Udalls Cove having the highest value. It is possible that there are other sources of $^{210}Pb$, particularly to Udalls Cove, which also has a sewage pipe running through it. This is consistent with the findings of Cochran et al. (1998) who did a previous study in the adjacent marsh Alley Pond Park. He found a larger than expected inventory of $^{210}$Pb in this marsh and attributed it to its location at the intersection of multiple major roadways (Northern Boulevard, Long

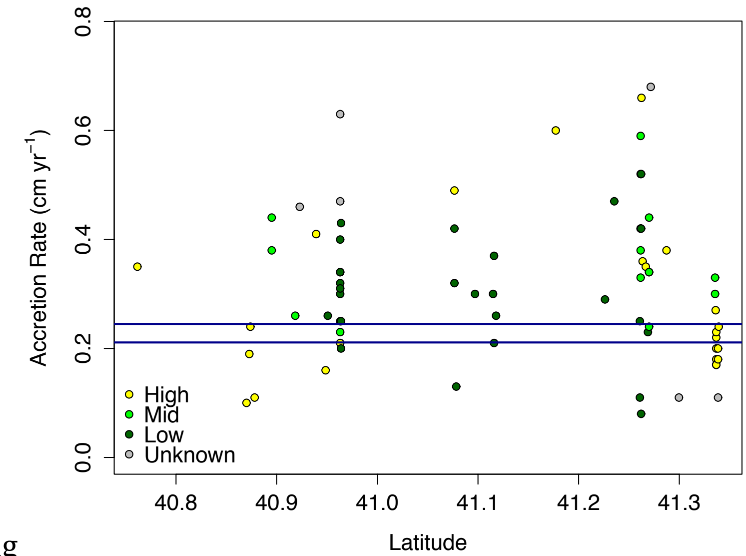


**Figure 10.** Historic accretion rates across different latitudes in LIS. LIS accretion rates were collected from 21 different studies in the literature. The colors represent marsh area (low marsh=dark green, mid marsh=light green, and high marsh=yellow). The blue box represents the range of historic sea level calculated from sea level data before 2000. Most accretion rates are above historic RSLR.

Island Expressway, and the Cross Island Parkway). Similar inventories have been found in other marshes in this area and other areas (Hill et al. 2015; Hewitt and Rashed 1991). In these studies, higher levels of $^{210}$Pb were also attributed to external input of metals from urban environments. This would make the accretion rates calculated using $^{210}$Pb less reliable, though potentially still useable (Cochran et al. 1998; Hill et al. 2015). The validity of rates from Udalls Cove is supported by the fact that the date calculated by the $^{210}$Pb models for the depth of the $^{137}$Cs peaks 1953 (Table A2.2), was relatively close to 1963, so the external input of $^{210}$Pb has most likely been constant over time. It would therefore be accounted for in the background $^{210}$Pb and not influence calculation of the chronologies from excess $^{210}$Pb (Hill et al. 2015). None of the inventories were below the expected range implying that none of the marshes appear to be experiencing erosion.

**Table 3.** Accretion rates calculated from radionuclide profiles of $^{210}Pb$, $^{226}Ra$, and $^{137}Cs$ in sediment cores from six marshes along LIS. Profiles were measured using a pure germanium gamma spectrometer and calculated using the constant rate of supply model (CRS), constant initial concentration model (CIC), and the $^{137}Cs$ peak method. The $^{210}Pb$ methods are measuring accretion rates over a 100-year span and the $^{137}Cs$ method is measuring rates over a 30-year span. The $^{137}Cs$ rates are generally higher than the rates from the other two methods. For BI, two cores were collected and their results have been averaged. BI has the lowest accretion rates across the methods with HI having the second lowest. WR and UC have the highest rates across most methods and JC and CP have lower rates in the CRS model and higher rates in the CIC model and $^{137}Cs$ method. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| Site | | **$^{210}Pb$ CRS model** | | **$^{210}Pb$ CIC model** | | **$^{137}Cs$** | |
|---|---|---|---|---|---|---|---|
| | **LOI (%)** | **Accretion rate (cm yr$^{-1}$)** | **Inventory (dpm cm$^{-2}$)** | **Accretion rate (cm yr$^{-1}$)** | **$R^2$** | **Accretion rate (cm yr$^{-1}$)** | **Inventory (dpm cm$^{-2}$)** |
| HI | 31 | 0.26 | 43.6 | 0.21 | 0.92 | 0.34 | 9.42 |
| UC | 24 | 0.44 | 67.0 | 0.40 | 0.94 | 0.48 | 10.77 |
| JC | 21 | 0.21 | 29.2 | 0.39 | 0.91 | 0.48 | 4.08 |
| CP | 28 | 0.22 | 34.0 | 0.44 | 0.87 | 0.48 | 3.08 |
| BI | 48 | 0.25 | 43.7 | 0.26 | 0.85 | 0.30 | 2.00 |
| WR | 23 | 0.39 | 35.1 | 0.46 | 0.70 | 0.38 | 6.45 |

The $^{137}Cs$ inventory can also be used to determine whether or not atmospheric deposition is the primary source of $^{137}Cs$ and whether erosion is occurring (Graustein and Turekian 1986; Cochran et al. 1998; Carey et al. 2015). Based on the 30-year half-life and previously measured inventories for Hunter Island, Udalls Cove, and Caumsett Park of 6.0, 16.0, and 5.0 respectively (Cochran et al. 1998), the total inventory of $^{137}Cs$ was calculated, which we would currently expect to find at the study sites considering the natural decay of the radionuclide since the previous measurements were taken. The current $^{137}Cs$ inventories should have been 4.15, 11.08, and 3.46 dpm cm$^{-2}$. The inventories for those sites were 9.42, 10.77, and 3.08 respectively and the range of all of the inventories ranged from 2.00 to 10.77 dpm cm$^{-2}$. These values and ranges agree very well, with the exception of at Hunter Island where the overall inventory increased rather than decrease from decay. It is possible that an external source of $^{137}Cs$ could exist at this site. There was a very distinct peak in the

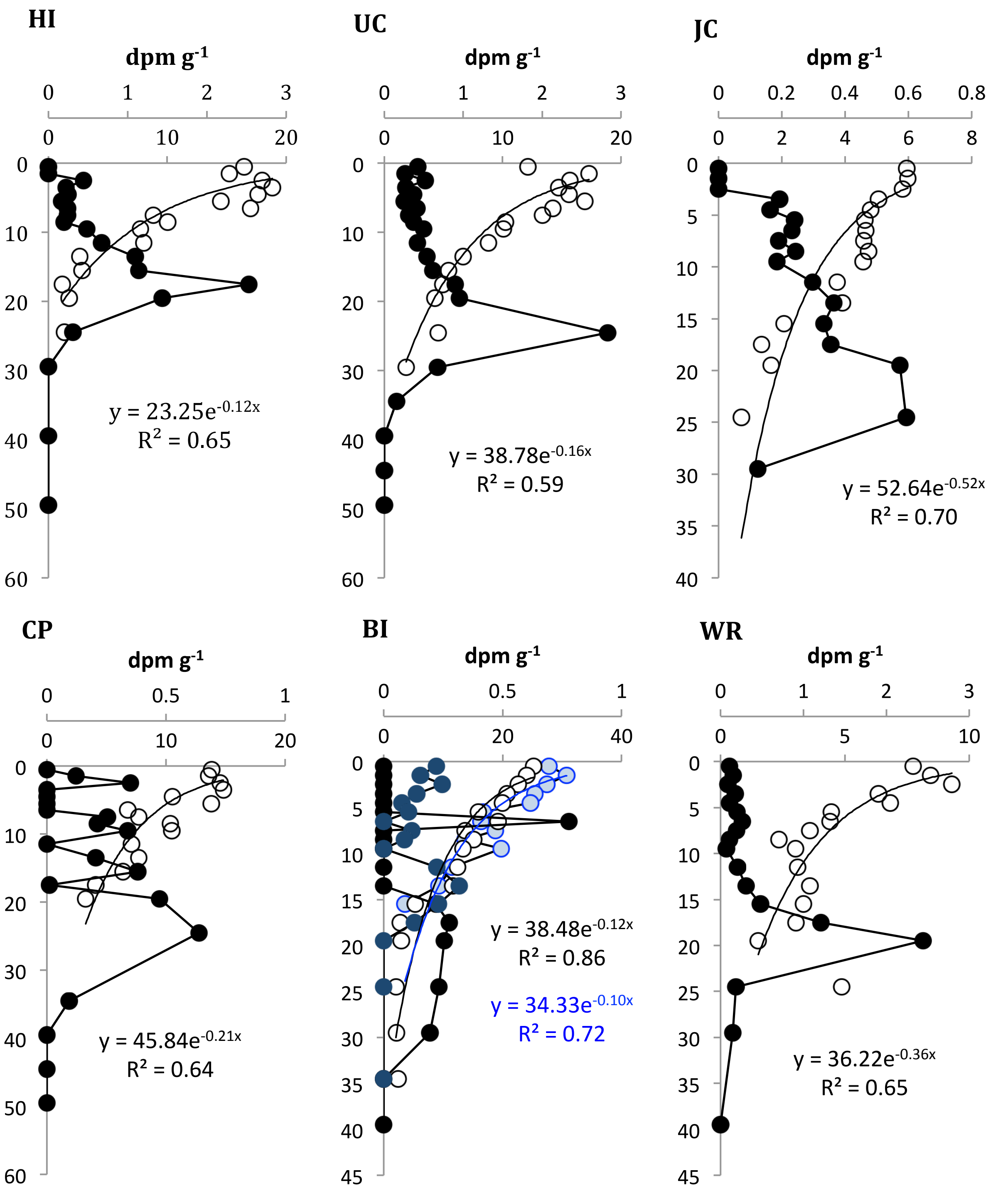


**Figure 11.** Sediment geochemical profile of $^{210}Pb_{xs}$ and $^{137}Cs$ for all six sites throughout LIS. Sediment cores were taken in 2014 and sub-sectioned into 1 cm increments. These were sent to the Moran Laboratory at the University of Rhode Island where they were analyzed for radionuclides. The white circles are $^{210}Pb_{xs}$ and the black circles are $^{137}Cs$. Both of these are on difference axes. For BI, two cores were taken and the light blue is $^{210}Pb_{xs}$ and the dark blue is $^{137}Cs$. $^{210}Pb$ decayed exponentially in all sites and $^{137}Cs$ peaks were identified for all cores. In BI 1, there was a second peak in the top portion of the sediment that could be a signal from the Chernobyl accident since that sediment layer was dated 1989, which is close to when the accident occurred. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

**Table 4.** Regression statistics from the individual CRS chronologies. The CRS model is capable of estimating accretion rates throughout the depth of the core, which is a proxy for time. The p-value, $R^2$, and slope are provided from the fits from each core. Three of the cores show significant increases over time in accretion rates. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| Site | CRS Model Core Results | | |
|---|---|---|---|
| | **P-Value** | **$R^2$** | **Slope** |
| HI | 0.121 | 0.10 | 0.0014 |
| UC | 0.178 | 0.18 | 0.0025 |
| **JC** | **0.00236** | **0.46** | **0.0017** |
| **CP** | **0.0169** | **0.32** | **0.0019** |
| BI1 | 0.174 | 0.57 | 0.0015 |
| **BI2** | **0.00473** | **0.49** | **0.0030** |
| WR | 0.233 | 0.36 | 0.0027 |

Hunter Island core, so this addition most likely did not influence calculated accretion rates. No external input or depletion of $^{137}Cs$ has occurred at the other sites.

Accretion rates from the $^{210}Pb$ Constant Initial Concentration (CIC) model ranged from 0.21 to 0.46 cm $yr^{-1}$. A very similar range of 0.21 to 0.44 cm $yr^{-1}$ was measured with the $^{210}Pb$ Constant Rates of Supply (CRS) model. While their ranges were similar, the rates diverged between the two models for Caumsett Park and Jarvis Creek with the CIC model calculating rates that were twice as high as those calculated by the CRS model (Table 3). One possible explanation for this is that the accretion rates at these sites could have changed over time. The CIC model assumes that accretion rates are constant over time. The accretion rate is calculated using the regression of the natural log of $^{210}Pb$ and depth by dividing the decay constant by the slope. The CRS model is capable of taking into consideration any changes in accretion rates over time. Differences between results from the CIC and CRS models in other studies have been found to mainly result from this difference between the models (Appleby and Oldfield 1978). In the $^{210}Pb_{xs}$ data, the exponential declines with depth imply that the assumptions of the CIC model have not been met and the CRS model results could be more reliable (Hill et al. 2015; Carey et al. 2015). Looking at the CRS model results alone, accretion was found to be increasing in Jarvis Creek and Caumsett Park as well as in Barn Island core 1 (Table 4). The slope for Barn Island is a little higher than the slopes for Jarvis Creek and Caumsett Park. This supports the conclusion that the accretion rates calculated using the CRS model might be more accurate.

**Table 5.** Historical comparison of accretion rates and RSLR in LIS. The rates of sea level rise were calculated from tidal gauges and data maintained by NOAA. Historic rates were calculated from sea level data before 2000 and current rates were calculated from sea level data from 2000 to 2014. For each of the study sites, the closest tidal gauge was determined and accretion rates were compared to current SLR rates associated with these gauges. Some of the tidal gauges did not have records longer than 15 years before 2000, so rates from the next closest tidal gauge were used instead. The current accretion rates from BI1 and BI2 have been averaged. In general accretion rates remained within the same range, though BI, HI, and UC increased overall, while JC decreased. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| **Site** | **Historic Sea Level Rise (cm yr$^{-1}$)** | **Current Sea Level Rise (cm yr$^{-1}$)** | **Historic Accretion Rate (cm yr$^{-1}$)** | **Current Accretion Rate (cm yr$^{-1}$)** | | |
|---|---|---|---|---|---|---|
| | | | | **$^{210}$Pb CRS Model** | **$^{210}$Pb CIC Model** | **$^{137}$Cs** |
| HI | 0.25 | 0.73 | 0.11[a] | 0.26 | 0.21 | 0.48 |
| UC | 0.25 | 0.73 | 0.35[a] | 0.44 | 0.40 | 0.34 |
| JC | 0.25 | 0.75 | 0.5[b] | 0.21 | 0.39 | 0.34 |
| CP | 0.25 | 0.73 | 0.41[a] | 0.22 | 0.44 | 0.48 |
| BI | 0.21 | 0.69 | 0.18[c] | 0.245 | 0.26 | 0.48 |
| WR | 0.25 | 0.75 | 0.46[d] | 0.39 | 0.46 | 0.38 |

[a]Accretion rate determined with $^{210}$Pb CRS; Cochran et al. 1998
[b]Accretion rate determined with $^{210}$Pb CIC; Anisfeld et al. 1999
[c]Accretion rate determined with $^{210}$Pb; Orson et al. 1998
[d]Accretion rate determined with Marker Horizons; Young 1985

Accretion rates from the $^{137}$Cs peaks ranged from 0.30 to 0.48 cm yr$^{-1}$. This range was close to though higher than the range from the other two $^{210}$Pb based methods. This was driven by the fact that all of the individual $^{137}$Cs rates were higher than both the CRS and CIC rates for every site besides Wading River where the $^{137}$Cs model was similar to the CRS rate and below the CIC rate. This trend has been found in other studies as well (Turner et al. 2006; Mudd et al. 2009). These higher rates could be due to the high mobility of $^{137}$Cs in marine sediments (Appleby and Oldfield 1978; Carey et al. 2015). This is caused by the abundance of cations that displace the $^{137}$Cs molecules and cause them to migrate in the sediment column (Foster et al. 2006). It could also be because the $^{137}$Cs accretion rates are representing rates from a shorter period of time, 30 years instead of 100 years for the $^{210}$Pb rates (Anisfeld et al. 1999; Chmura et al. 2001; Carey et al. 2015). In a literature review comparing accretion rates taken by $^{137}$Cs and $^{210}$Pb, Turner et al. (2006) also found accretion rates from $^{137}$Cs were generally higher than accretion rates from $^{210}$Pb. The pattern within the $^{137}$Cs rates was more consistent with the CIC model values with higher rates for Jarvis Creek and Caumsett Park. The $^{137}$Cs peaks at these two sites were also the least distinct peaks with the lowest dpm readings (Figure 11), so it is possible the peaks lay in between the 20 and 25 cm samples for each, which would lead to a slightly lower accretion rate. This would be more consistent with the CRS model rates and implies the CRS rates might also be more accurate.

# Accretion Rates Summary

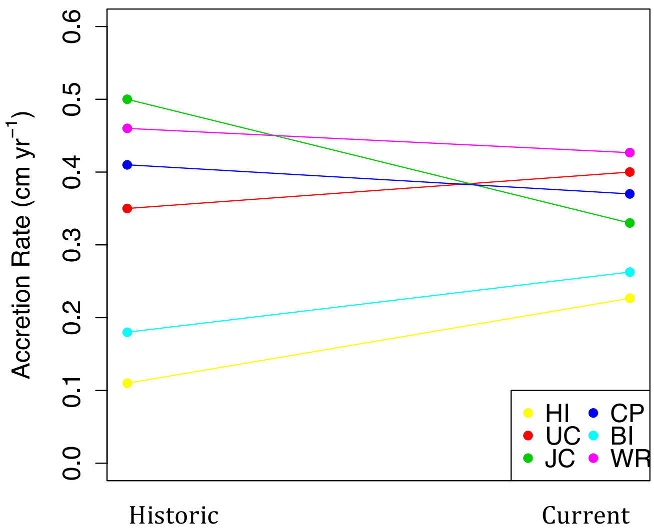


**Figure 12.** Historic accretion rates from each of the six study sites were compared to current rates from this study. The historic rates were taken before 2000 and were gathered from the literature (Cochran et al. 1998: HI, UC, CP; Anisfeld et al. 1999: JC; Orson et al. 1998: WR; Young et al. 1985: WR). The rate for UC was taken in an adjacent marsh Alley Pond Park. The averages of the accretion rates calculated from the $^{210}$Pb CRS model, $^{210}$Pb CIC model, and $^{137}$Cs peak method were used for current rates. The two cores from BI have been averaged. Accretion rates have not changed overall, but some, like BI, HI, and UC have increased while others have decreased, like JC. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

### *Past and Present Accretion Rate Comparison*

Salt marsh accretion rates at the six sites studied here were also measured in the past (Young 1985; Orson et al. 1989; Cochran et al. 1989; Anisfeld et al. 1999), allowing us to assess if accretion rates have changed overtime. All of the rates used for comparisons from previous studies were taken in the 1980s and 1990s using the same $^{210}$Pb method, with the exception of Wading River, which was taken using a marker horizon method (Table 5). Three of these used the CRS model, one used the CIC model, and one did not state which model was used. Overall, the mean historic rate, 0.33 cm yr$^{-1}$, was not statistically different from the mean current CRS, CIC, and $^{137}$Cs rates, which were 0.29, 0.36, and 0.41 cm yr$^{-1}$, respectively (Figure A2.1). There was also no significant difference found between the historic rates and only the rates found using the CRS model, which was determined to be the most reliable (Figure A2.2). The current accretion rates from this study are also similar to other values found in recent studies in this area (Hill et al. 2015; Carey et al. 2015; Kolker et al. 2009).

There were some trends within the rates for each site. Rates in Barn Island, Hunter Island, and Udalls Cove have increased according to all models though Udalls Cove has higher overall rates (Figure 12). In Caumsett Park, rates have increased in comparison to the previous CIC and $^{137}$Cs rates, but decreased in comparison to the CRS rates.

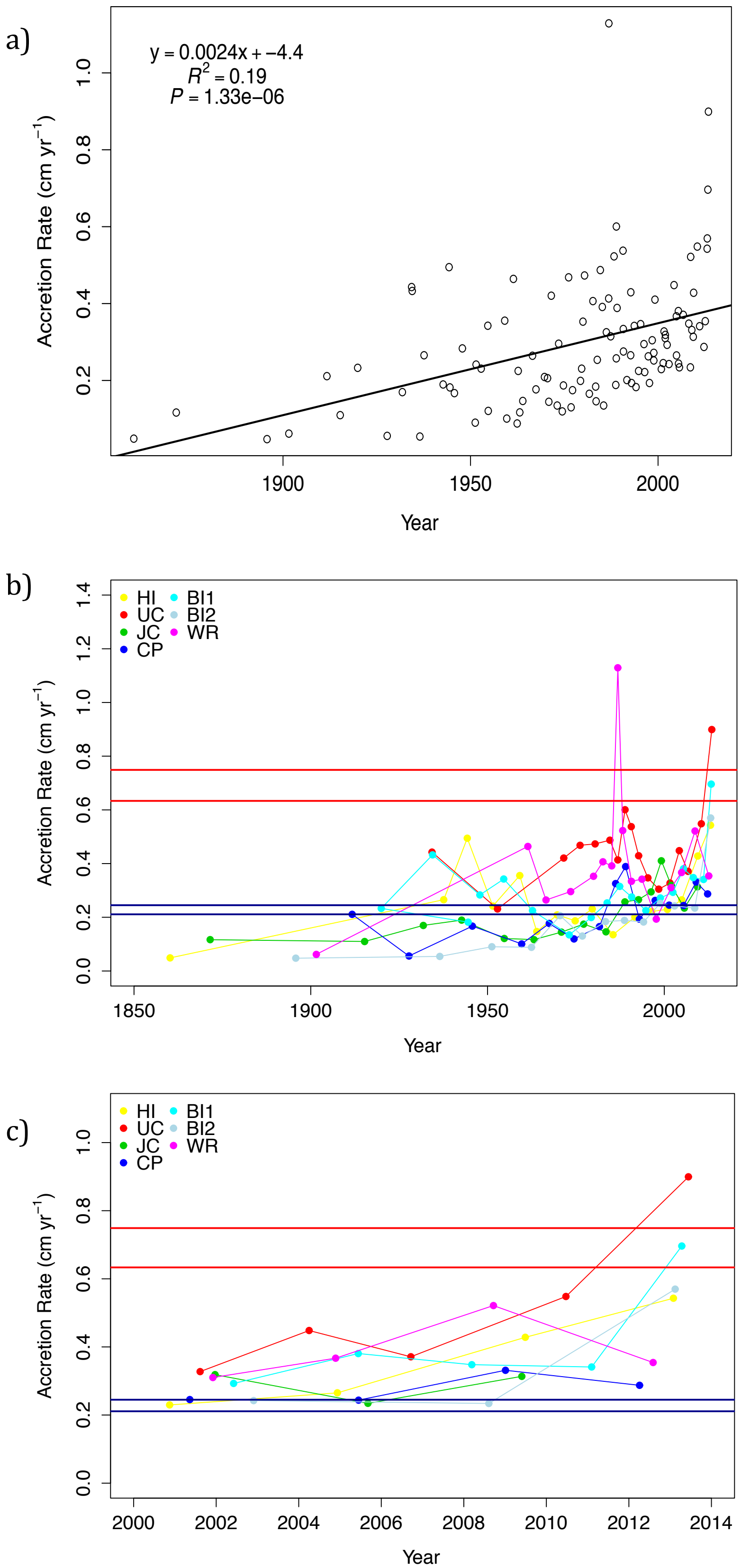
a)
y = 0.0024x + −4.4
R² = 0.19
P = 1.33e−06
Accretion Rate (cm yr⁻¹)
Year
b)
HI
UC
JC
CP
BI1
BI2
WR
Accretion Rate (cm yr⁻¹)
Year
c)
HI
UC
JC
CP
BI1
BI2
WR
Accretion Rate (cm yr⁻¹)
Year

**Figure 13.** Accretion rates calculated using the CRS model for each 1 cm interval of each core collected from the six sites in this study. By calculating rates at intervals, the CRS model is capable of producing accretion rates over time. a) All of these individual accretion rates from all of the cores show a significant increase in accretion rates over this time period. Panel b) shows each of the individual chronologies, which makes it possible to see what is occurring in each core. Panel c) is also looking at the individual chronologies and accretion rates, but only looking at the last 15 years, which is the time over which the current RSLR rates were calculated. The ranges for historic and current RSLR are also plotted on these two graphs in blue and red respectively. There is a lot of variability in the WR and UC cores. The final rates of a few cores are above the current range of RSLR, while the rest remain below. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

In Wading River they have decreased slightly. In Jarvis Creek accretion rates have decreased according to all models, though only slightly with the $^{137}Cs$ model. Another way to examine trends over time is to use the data from the CRS model. This model provides separate rates for individual depths, which are associated with different years. These chronologies show that, overall, accretion rates have been increasing over time (Figure 13a; $R^2 = 0.19$, $p < 0.0001$). Another study by Hill et al. (2015) found similar results using the same method. It is also possible to look at rates within each core individually across the whole time period (Figure 13b) although the individual rates within the CRS models should be taken with caution, as there is no replication for each measurement. All of the individual relationships were not significant (Table 4) though they all had similar upward trends. In the last 15 years of each record (Figure 13c), the trends follow a similar pattern to historic rates. Accretion rates increased in Barn Island, Hunter Island, and Udalls Cove though rates are highest in Udalls Cove. Rates over this time period are low and have remained the same in Caumsett Park and in Jarvis Creek and they are high, but decreasing in Wading River.

*Relative Sea Level Rise*

The main goal of this study was to determine whether or not salt marshes are accreting fast enough to keep pace with RSLR. Previous studies in this region have shown that the rates of accretion in the low marsh, which is where all of the cores for this study were taken, are typically 1.5 times the rate of RSLR (Bricker-Urso et al. 1989; Cochran et al. 1998, Orson et al. 1998). The accretion rates (range: 0.21 to 0.48 cm $yr^{-1}$) found in this study are 2 to 3X slower than rates of current RSLR (range: 0.69 to 0.75 cm $yr^{-1}$). This indicates that while many of these marshes appear to be increasing their rate of accretion, they are not increasing fast enough to keep pace with RSLR. Individual dates within the CRS chronologies confirm that while rates are rising they are slower than current RSLR for the majority of the record (Figure 13c). They also suggest that some sites might be closer to reaching RSLR rates than others. The last accretion rates in Barn Island, Hunter Island, and Udalls Cove are close to, within, or above the range of current RSLR. Final accretion rates for Jarvis Creek, Caumsett Park, and Wading River are well below the range and rates in Caumsett Park and Wading River appear to be decreasing. Even considering the replication limitations of our assessment, these patterns are likely still consistent with the comparison of past and present accretion rates.

**Table 6.** Summary of organic and inorganic matter accretion rate data. This was done with the $^{210}Pb$ CRS model data. The results from BI1 and BI2 have been averaged. Organic accretion was a higher percentage of non-pore space accretion in BI. Inorganic accretion was a majority of non-pore space accretion in most other sites besides HI, and was a particularly large percentage of non-pore space accretion in JC and WR. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| Site | Average Accretion Rate (cm yr$^{-1}$) | | Percent (%) of Accretion Due to Organic vs. Inorganic Matter | | Percent (%) of Accretion Due to Pore Space | |
|---|---|---|---|---|---|---|
| | Organic | Inorganic | Organic | Inorganic | Non-Pore Space | Pore Space |
| HI | 0.017 | 0.017 | 7.3 | 7.3 | 14.6 | 85.4 |
| UC | 0.025 | 0.033 | 6.2 | 8.4 | 14.7 | 85.3 |
| JC | 0.015 | 0.024 | 7.2 | 11.4 | 18.6 | 81.4 |
| CP | 0.013 | 0.013 | 6.0 | 6.4 | 12.4 | 87.6 |
| BI | 0.016 | 0.008 | 7.6 | 3.6 | 11.2 | 88.8 |
| WR | 0.025 | 0.037 | 6.4 | 10.3 | 16.7 | 83.3 |

### *Organic vs. Inorganic Matter Content*

An important factor in understanding accretion and marshes responses to RSLR is the relative amounts of organic and inorganic matter accretion in each marsh (Table 6). It is possible to calculate the amount of accretion driven by each type of matter by multiplying the total accretion by LOI and bulk density and dividing by the sediment or organic matter density. With this rate, a percentage of the total accretion rate can be calculated as well as the percent due to pore space. It was not possible to look at this over time, but there were clear differences between sites. Barn Island and Hunter Island were on the lower end of both organic and inorganic accretion since they had lower total rates, but Barn Island was the only site with a higher percentage of accretion due to organic accretion and Hunter Island had equal percentages of organic and inorganic accretion. Jarvis Creek and Caumsett Park had lower organic accretion and Caumsett Park had low inorganic accretion, but Jarvis Creek had a higher rate of inorganic accretion and the highest percentage of inorganic accretion. Wading River and Udalls Cove had the highest amount of organic and inorganic accretion, but a higher percentage of accretion from inorganic accretion. A trend was seen where the marshes with the highest dependence on organic matter accretion have the lowest accretion rates. The marshes with more inorganic matter accretion have the highest rates of accretion in these data.

### *Nutrient Gradient*

The differences between sites were also examined. This allowed for the assessment of whether or not accretion rates changed based on proximity to NYC, which is a source of nutrients. The range of rates for Hunter Island and Udalls Cove, the two sites on the western end of the nutrient gradient, were 0.21 to 0.34 cm yr$^{-1}$ and 0.40 and 0.48 cm yr $^{-1}$.

For Jarvis Creek and Caumsett Park, the sites in the middle of the gradient, the rates ranged from 0.21 to 0.48 cm $yr^{-1}$ and 0.22 to 0.48 cm $yr^{-1}$. For Barn Island and Wading River on the eastern end of the gradient, the ranges were 0.25 to 0.30 cm $yr^{-1}$ and 0.38 to 0.46 cm $yr^{-1}$. Looking at just these accretion rates, there is no relationship between accretion rates and distance from NYC since the highest and lowest rates were found on either end of the gradient. This means that nutrients might not be driving accretion in this region. One consideration is that the distance from NYC was used as a proxy for nutrient concentrations. It is possible that there are local sources of nutrients that are having a greater impact on marsh dynamics than the nutrients from NYC treated wastewater. Also, based on other studies (Figure 3; Gobler et al. 2006) and results found here (Figure A1.3 thru A1.4), the concentration of nutrients decreases exponentially 20 km from NYC. It is possible that the influence of this nutrient gradient might be smaller than the range of this study. It could also be influenced by patterns of currents within LIS, which would determine where the nutrients were transported (Latimer et al. 2013).

# Sediment Characteristics

The remainder of the data was collected from the six sites used in this study. They are presented in this order: Hunter Island (HI), Udalls Cove (UC), Jarvis Creek (JC), Caumsett Park (CP), Barn Island (BI), and Wading River (WR). From left to right, these sites are aligned from west to east across LIS with the sites on the north shore of LIS listed first (Figure 2).

**Table 7.** Salt marsh sediment characteristics determined from one core at each site across LIS. Cores were divided into 1 cm sections down to 50 cm. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). For bulk density and water content, every cm section of the core at every depth was analyzed in JC, BI, and WR. In HI, UC, CP the top 10 cm, every other cm to a depth of 20, and every fifth cm to a depth of 50 were analyzed. This same subset was used for LOI analysis. Only the top 10 cm were analyzed for CHN.

| Site | Bulk Density (g $cm^{-3}$) | Water Content (%) | LOI (%) | C:N |
|---|---|---|---|---|
| HI | 0.25 | 73.8 | 29.6 | 18.1 |
| UC | 0.29 | 68.0 | 23.7 | 16.5 |
| JC | 0.27 | 73.4 | 21.6 | 21.3 |
| CP | 0.26 | 73.7 | 26.5 | 15.6 |
| BI | 0.14 | 83.7 | 51.6 | 14.7 |
| WR | 0.33 | 66.9 | 21.6 | 16.1 |

## Bulk Density

The bulk density of the salt marsh sediments at these sites ranged from 0.14 to 0.33 g $cm^{-3}$ (Table 7). These values are within the range of bulk densities found in other sites in the region (Anisfeld et al. 1999; Kolker et al. 2009; Carey et al. 2015). Barn Island had a significantly lower bulk density ($p < 0.0001$) than all of the sites and Wading River had a significantly higher bulk density ($p = 0.0303$) than all of the sites besides Udalls Cove (Figure 14). Hunter Island, Udalls Cove, Jarvis Creek, and Caumsett Park all had very similar bulk densities. The bulk density is an indication of the amount of sediment that is available for salt marsh accretion. Many of the marshes in this area have relatively low sediment loads compared to other marshes in other regions such as the Gulf of Mexico and the Bay of Fundy (Turner et al. 2006; Chmura et al. 2001). Higher sediment loads can by caused by extra input from rivers and the type of bedrock or other geologic features (Amos 1987). This could explain the historically higher accretion rates in fluvial marshes. Bulk density remained fairly constant at depth with a slight decrease with depth in Jarvis Creek and Barn Island and increase in Udalls Cove and Caumsett Park (Figure A3.1), which could imply some compaction.

## Loss on Ignition

The organic matter content, which was the average LOI throughout each core, from the sites in this study ranged from 22 to 52 % (Table 7) and was within the range of previous measurements collected from the literature review associated with this study and from other recent studies (Cochran et al. 1998; Anisfeld 2008; Hill et al. 2015; Carey et al. 2015). Consistent with the bulk density results, Barn Island has a significantly higher percent organic matter than all of the other sites ($p < 0.0001$). Hunter Island also had a significantly higher percent organic matter than Jarvis Creek and Wading River (Figure 15; $p = 0.0215$). Most salt marshes in the northeast

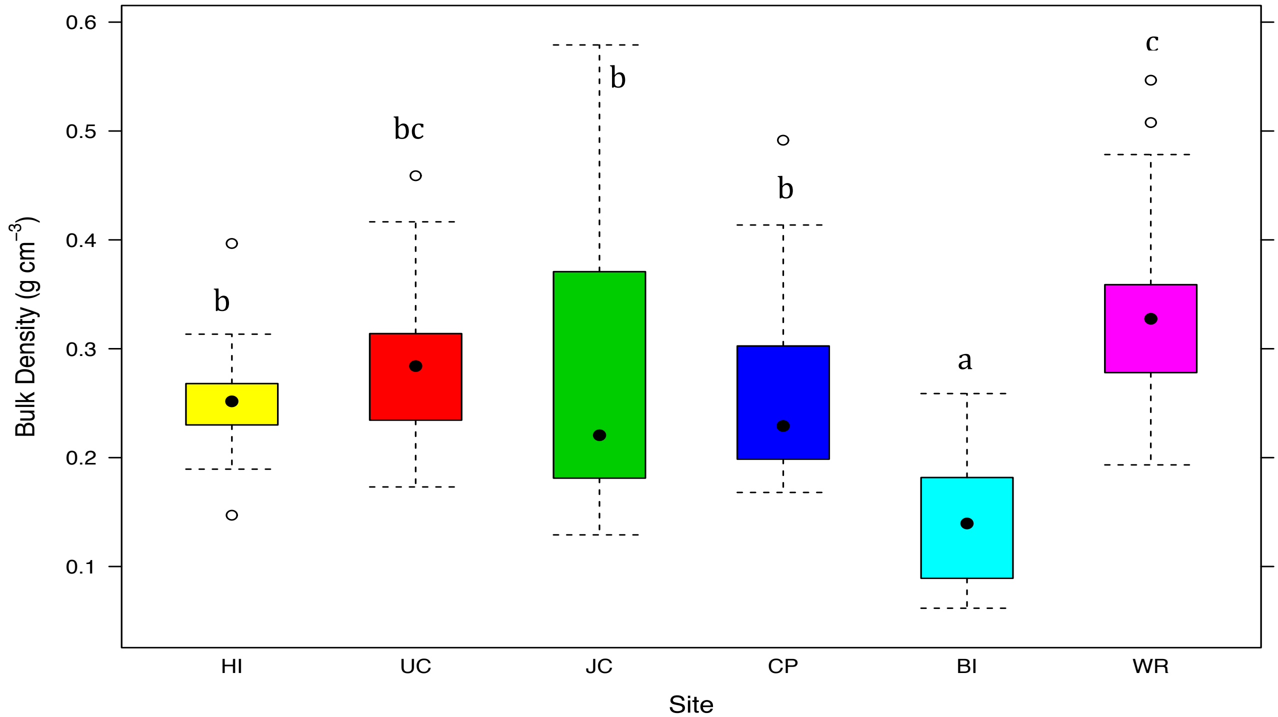


**Figure 14.** Mean salt marsh bulk density based on 6 cm diameter core, sub-sectioned at 1 cm increments. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. Bulk density was significantly lower in BI and higher in WR (ANOVA,$p<0.0001$;$p=0.0303$).

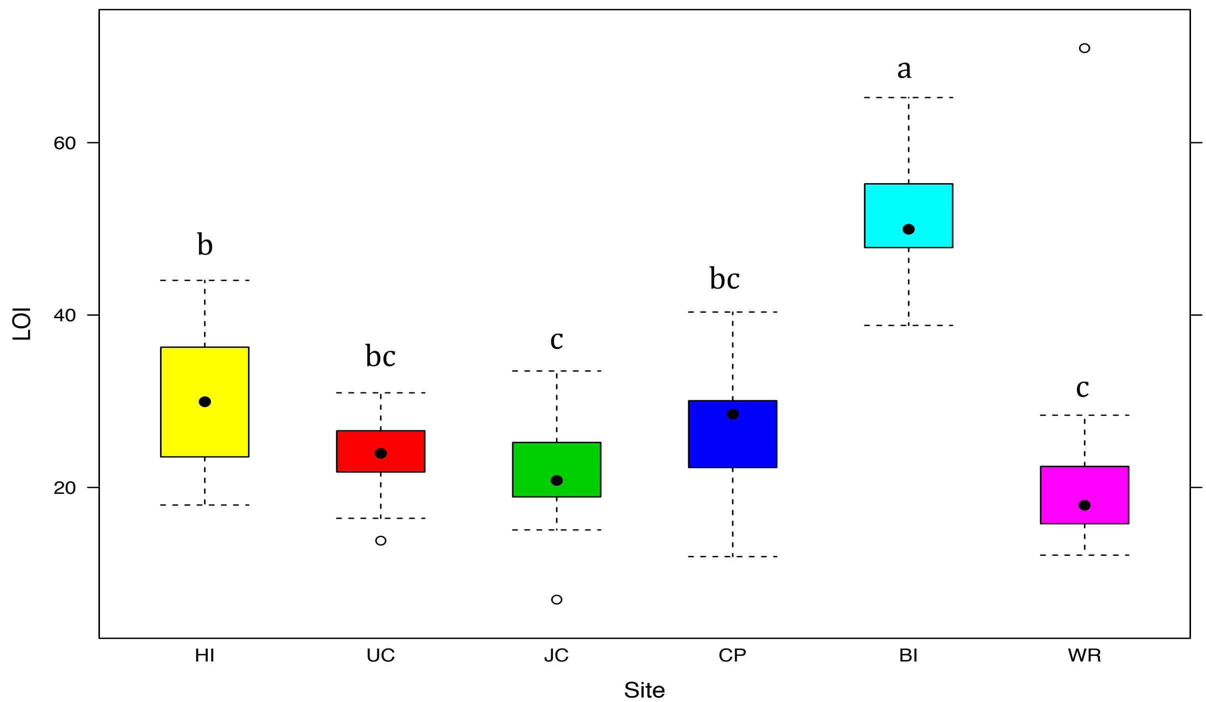


**Figure 15.** Mean whole salt marsh core (n=1) LOI from each of the six sites in this study. Cores were sub-sectioned at 1 cm increments down to 50 cm. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. LOI was significantly higher in BI and lower in WR and JC (ANOVA,$p<0.0001$;$p=0.0215$,$p=0.0175$).

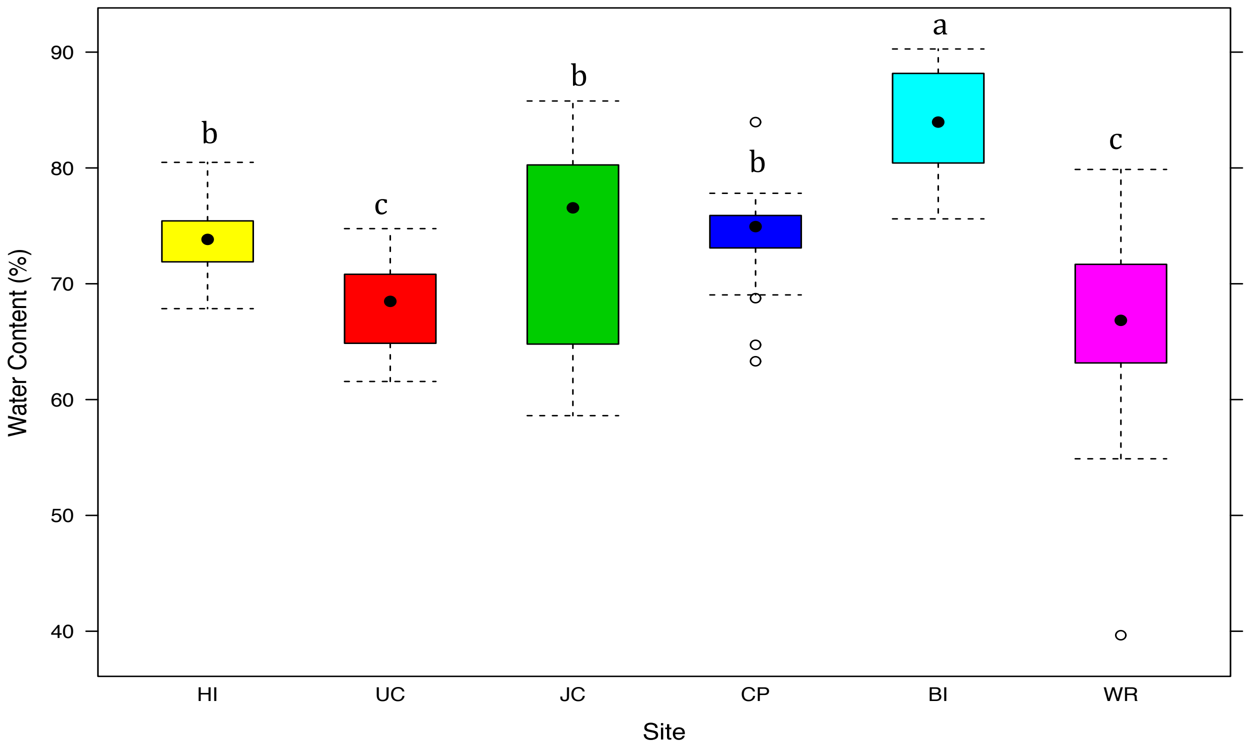


**Figure 16.** Mean whole salt marsh core (n=1) water content from each of six sites in this study. Cores were sub-sectioned into 1 cm down to 50 cm. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. Water content was significantly higher in BI and lower in WR and UC (ANOVA,p<0.0001;p=0.0005,p=0.0034).

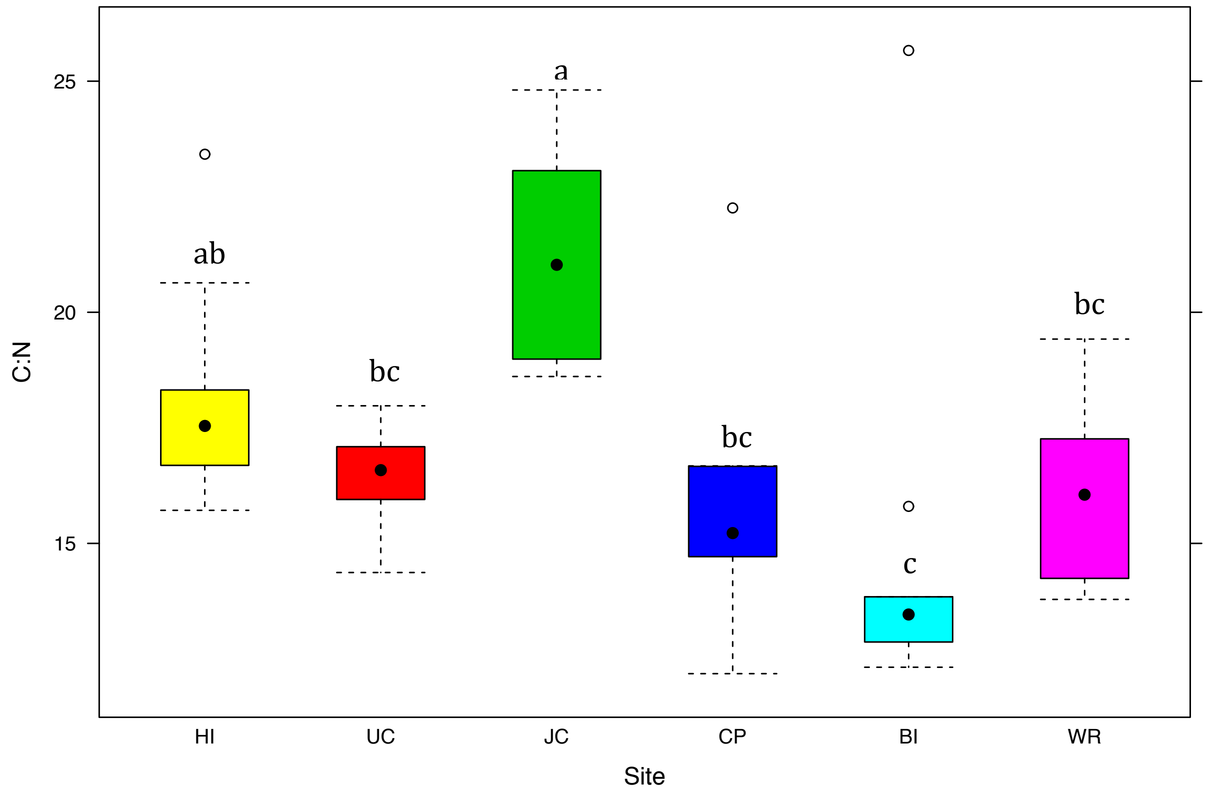


**Figure 17.** Mean whole salt marsh core (n=1) C:N from each of the six sites in this study. Cores were sectioned into 1 cm increments down to 50 cm. Only the top 10 cm were used for C:N analysis. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. C:N was significantly lower in BI and higher in JC (ANOVA,p=0.0424;p=0.0010).

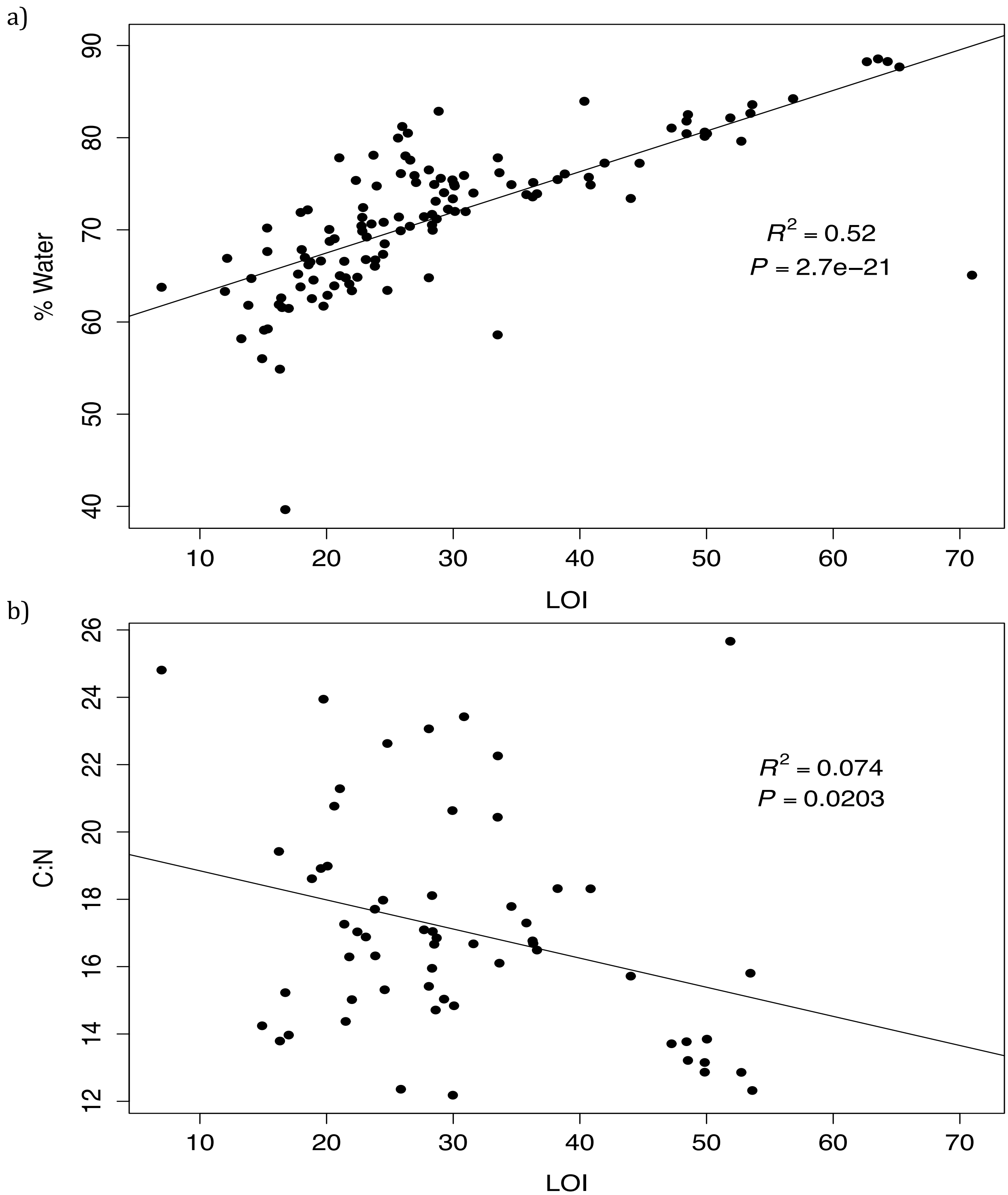


**Figure 18.** Relationships of a) water content and b) C:N with LOI in sediment samples from all six of the salt marshes in this study. There is a strong increase in water content with increasing LOI and a slight decrease in C:N with increasing LOI.

are known to have relatively high organic matter content and be sediment starved (Nixon 1982; Kirwan and Temmerman 2009; Kolker et al. 2009). The build up of organic matter, which is determined by the balance between the primary productivity and decomposition of vegetation, can be crucial for accretion rates in sediment starved salt marshes like most of those found in Long Island Sound. It can be harder for these marshes to maintain high accretion rates (Redfield 1972; Allen 2000). Here, Barn Island is an example of a marsh with high organic matter content and a low accretion rate. In the Udalls Cove, Caumsett Park and Hunter Island cores the LOI decreased with depth and in the Barn Island core, it increased with depth (Figure A3.2). The increase in LOI with depth in this core also implies that the marsh platform at Barn Island has contained a high percentage of organic matter over a longer period of time. It has built up organic matter despite decomposition and compaction. The currently lower LOI at surface sediment could indicate a change in this trend. The amount of organic matter could be decreasing from forces such as decomposition or other environmental changes.

## Water Content

For all of the marshes, the majority of the sediment profile was pore water with water content ranging from 70 to 84 %, which are within the range of water content found at previous studies (Table 7) (Bricker-Urso et al. 1989; Cochran et al. 1998). Similar to LOI, Barn Island has the highest water content (Figure 16;$p < 0.0001$) and Wading River the lowest ($p = 0.0005$), but Udalls Cove also has a lower water content than all sites besides Wading River ($p = 0.0334$). Hunter Island, Udalls Cove, Jarvis Creek, and Caumsett Park all had similar water content. In Jarvis Creek and Barn Island the water content increased with depth (Figure A3.3). There was a positive relationship between pore water and LOI (Figure 18a; $R^2 = 0.52$, $p < 0.0001$). This could be explained by the theory that the quality of organic matter in marshes can lead to the retention of pore water within the marsh sediment. This can be important for accretion since the inclusion of pore water into the structure of the marsh can help maintaining the marshes elevation in relation to sea level rise (Bricker-Urso et al. 1989; Carey et al. 2015). The ability of marshes to maintain this dynamic could be important for accretion.

## Sediment C:N

The carbon to nitrogen ratio within the sediment for each site ranged from 14.7 to 21.3 (Table 7), which is similar to C:N ratios found in sediments in other marshes (Anisfeld et al. 1999). This is most likely due to the relatively large percentage of organic matter in marshes. Vascular plants, normally have a C:N ratio above 20 and most marine vegetation, phytoplankton, have C:N ratios below 10, so higher C:N ratios indicate the presence of more vascular plants in the sediment (Prahl et al. 1994; Wegner et al. 2003). The C:N ratio in Jarvis Creek was significantly higher than all of the other sites besides Hunter Island (Figure 17; $p = 0.0010$). Barn Island was also significantly lower than Hunter Island ($p = 0.0424$). There was also a weak negative relationship between C:N and LOI (Figure 18b; $R^2 = 0.07$, $p = 0.0203$). Lower C:N ratios could lead to higher LOI through the growth of more vegetation when more nitrogen is available in the sediment.

## Sediment Characteristics Summary

As expected, bulk density, organic matter content, and water content are all related. Bulk density is negatively related to organic matter content due to the inherent balance between the presence and accumulation of organic matter and sediment, which are the primary components of salt marsh sediments (Figure A3.5a). Organic matter and water content are positively correlated to each other possibly due to the incorporation of pore water into peaty organic matter (Bricker-Urso et al. 1989; Carey et al. 2015) (Figure 18a). Based on these results, Barn Island sediment contained more organic matter and Wading River contained more inorganic material. Udalls Cove and Jarvis Creek appeared to contain relatively more inorganic and Hunter Island and Caumsett more organic matter. Barn Island is on the higher end of the range of values for organic matter content typically found in this region (Anisfeld et al. 1999; Hill et al. 2015; Carey et al. 2015). All of the marshes are primarily made of pore water, possibly indicating the overall importance of organic matter (Bricker-Urso et al. 1989). The balance between these three inputs is an important factor in salt marsh accretion rates. C:N followed a different pattern with the highest values in Jarvis Creek and lowest values in Barn Island and can affect processes such as decomposition.

*Accretion Rates*

These results are consistent with the organic and inorganic matter accretion rates. The marshes with the highest organic matter contents have the lowest accretion rates and the marshes with the highest bulk densities have the highest rates of accretion. However, the trend between accretion rates and organic matter content were not significant regardless of what accretion model was used (Figure A3.6). The reverse relationship also did not exist with bulk density. Looking at the organic matter content values for Barn Island and Wading River, which were on the extremes of the LOI range in this study, in comparison with Carey et al.'s values, Barn Island and Wading River appear to be closest to marshes with the second highest and second lowest LOI. The range of this study's samples might not cover a range of LOI necessary to see this relationship.

# Decomposition

Increased temperatures, which have been found to increase decomposition rates (White et al. 1978), are hypothesized to be impacting salt marsh accretion by increasing decomposition to the point where more carbon is being respired than is sequestered by the growth of vegetation (Kirwan and Blum 2011). This would be particularly important in salt marshes that have a higher percentage of organic matter (Carey et al. 2015).

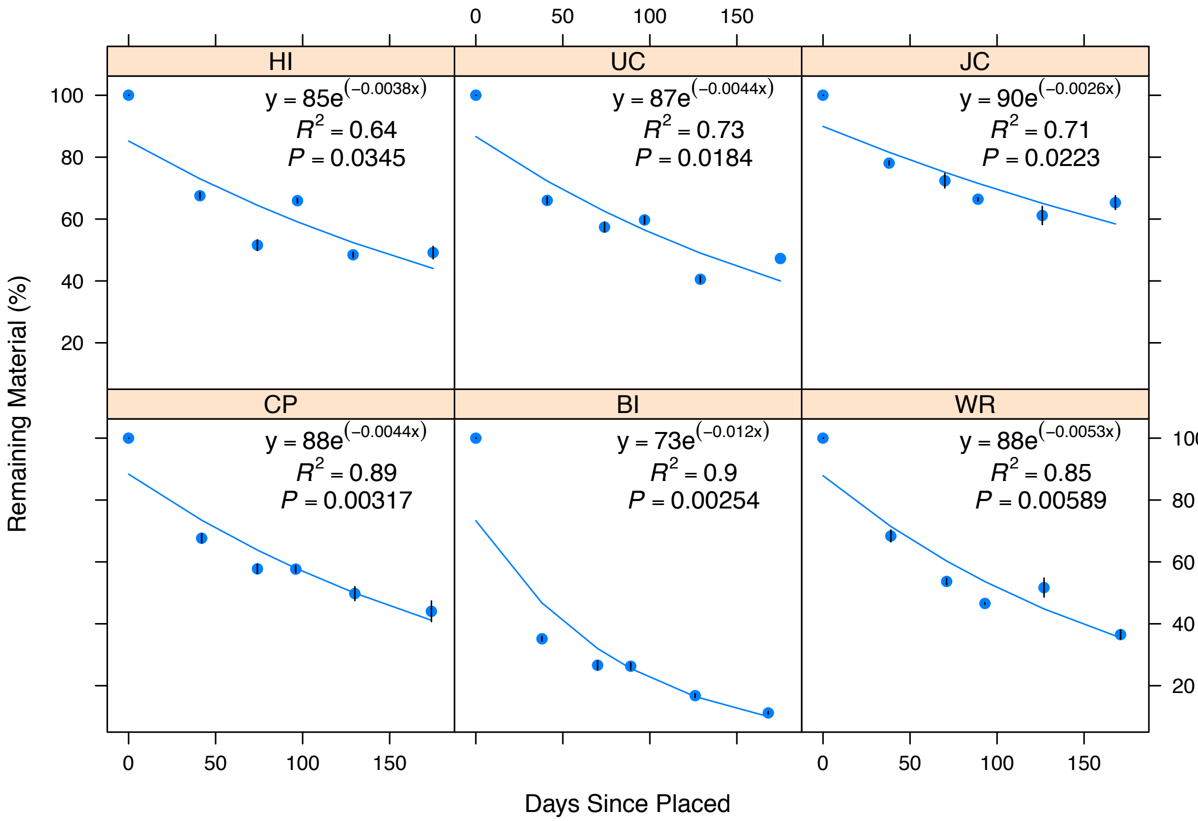


**Figure 19.** Percent of material remaining in litterbags after removal from the six low marsh salt marsh sites in this study. Litterbags were created with dried material from their respective marshes and then left attached to the marsh surface for up to 6 months. Exponential decay was experienced in all sites. The fastest rates were seen in BI and the lowest rates in JC. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

**Table 8.** Litter bag decomposition rates. The decay rate per day for each site was taken from the exponential fit of the percent of remaining material. To calculate the monthly decomposition rates, the daily rates were multiplied by 30. The percent remaining after 6 months are the percentage lost from the litter bags collected at the end of December since they were placed in the marsh in July. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| Site | Rate month$^{-1}$ | Percent Remaining After 6 Months |
|---|---|---|
| HI | 0.11 | 49 |
| UC | 0.13 | 47 |
| JC | 0.08 | 65 |
| CP | 0.13 | 44 |
| BI | 0.36 | 11 |
| WR | 0.16 | 37 |

## Litterbags

The material within the litterbags decayed at an exponential rate at every site (Figure 19). An average of 42% of the original material was lost over the course of 6 months from summer to winter with an average decay rate of 0.16 month$^{-1}$ (Table 8). Barn Island had the highest rates of decomposition at 0.36 month$^{-1}$, retaining only 11% of its original material by the end of 6 months. Jarvis Creek decomposed slowest with a rate of 0.08 month$^{-1}$ and retained 65% of the original material. Wading River, Caumsett Park, Udalls Cove, and Hunter Island were all close to the average rate, with Wading River being slightly higher than the other rates.

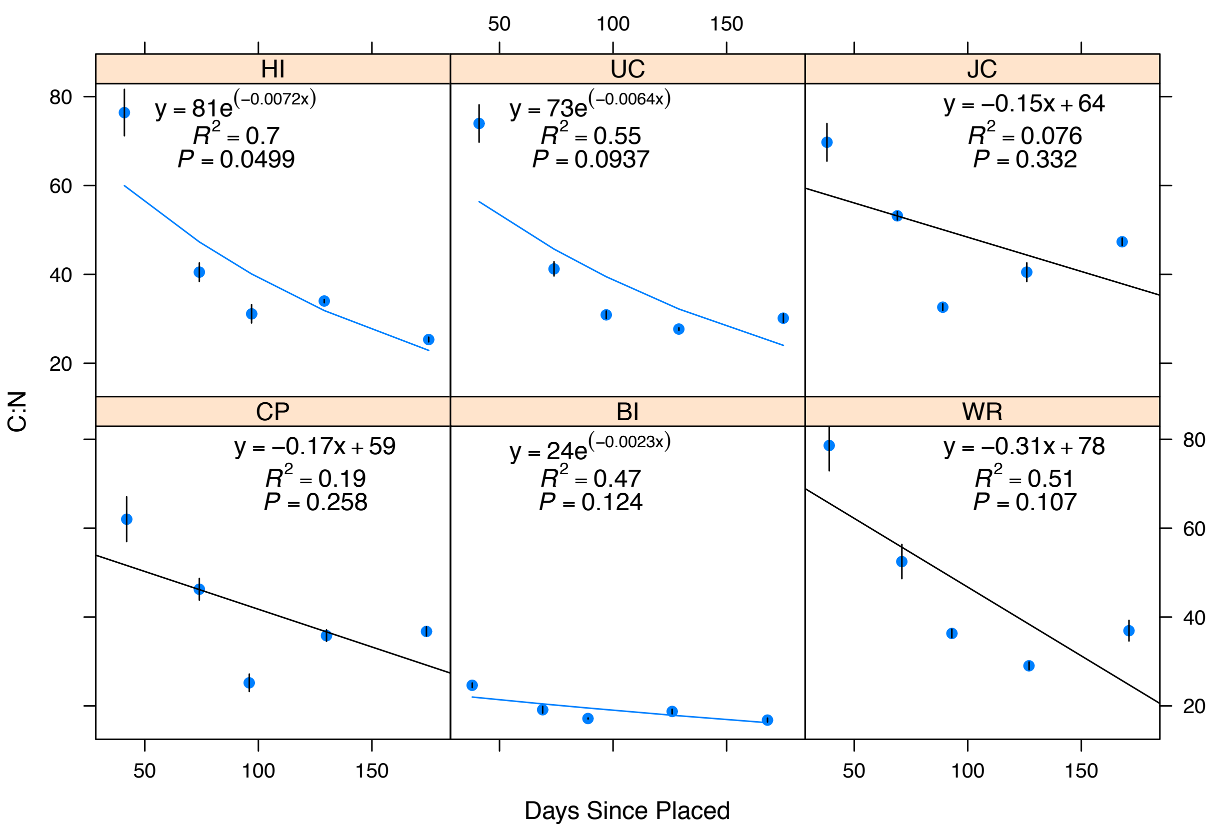


**Figure 20.** C:N ratios in litterbags used to measure decomposition in each of the six salt marshes in this study. Exponential and linear fits were run through the data for all sites. Linear fits were also attempted and when they provided a better fit, they were plotted along with the exponential fit and the equation, $R^2$, and p value were plotted to the right of the exponential equation, $R^2$, and p value. Half of the sites fit an exponential relationship and half fit a linear relationship best. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

## Litterbag C:N

The C:N ratio within the vegetation in the litterbags in this study ranged from 110 to 15. Anisfeld (2008) found the majority of C:N ratios in decomposing salt marsh organic matter in this region to be between 10 and 30 with the highest values at 40, but his values were from within the sediment. Much of the data from this study were within this range, but since the samples from this study were vegetation decomposing at the marsh surface in litterbags, it is not surprising that the values are higher. In the litter bags, there was also a decreasing trend in C:N ratios over time. However, this trend was only significant in Hunter Island (Figure 20). By site, the same pattern was found in litterbag C:N as in sediment C:N. Barn Island had the lowest C:N ratio and Jarvis Creek had the highest average ratio (Figure 21), but the difference between Barn Island and Jarvis Creek was not statistically significant (p = 0.0799). The C:N ratio were also consistent with and confirmed the patterns seen with decomposition. In these marshes, the C:N ratio decreased over time, which means the marshes are decomposing quickly and losing nitrogen in the biomass. In the sediment of these marshes, the C:N ratios were high, but followed the same trend as decomposition. Barn Island had the lowest C:N ratio and highest rates of decomposition and Jarvis Creek had the highest C:N ratio and lowest rates of decomposition.

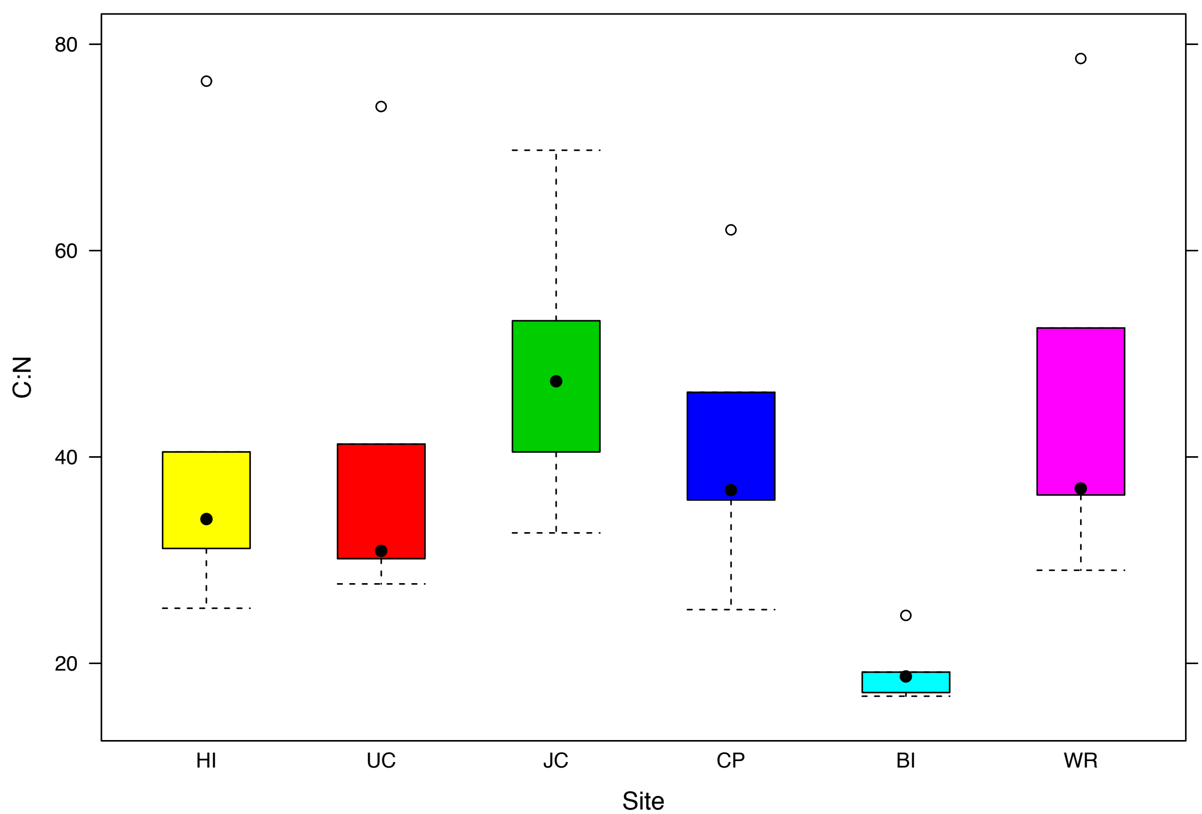


**Figure 21.** Mean salt marsh litter bag C:N from each of the six sites in this study. Litter bags were collected every month for 6 months. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). No significant differences were found.

## Historic Decomposition Rates

The data from this study and other studies were recalculated as the percent of litter lost per day (Table 9). One study by Warren et al. (2001) measured decomposition of *S. alterniflora* in Great Island after one and two months. A study by Anisfeld (2008) measured decomposition in the low marsh after three months. After one and two months, decomposition was faster in Barn Island than in Great Island and slower in the rest of the sites. After 3 months, decomposition rates at all of the sites in this study were higher than the rates found in the three sites studied by Anisfeld et al. in CT. In this context, Barn Island still appears to have particularly high decomposition rates.

**Table 9.** Litterbag decomposition rate comparison. The decay rate per day for each site was calculated by dividing the percent of material lost by the days passed since the litterbags were placed in the field. The three sets of numbers from this study are from the first, second, and third months with results from the first month on top, the second in the middle, and the third at the end. The "Days" columns contain the number of days that passed between when the litterbags were placed in the field and when they were collected. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

| **This Study** | | | **Literature** | | |
|---|---|---|---|---|---|
| **Site** | **Days** | **Rate (% $d^{-1}$)** | **Site** | **Days** | **Rate (% $d^{-1}$)** |
| HI | 41 | 0.79 | Great Island[a] | 27 | 1.20 |
| UC | 41 | 0.83 | | | |
| JC | 38 | 0.58 | | | |
| CP | 42 | 0.77 | | | |
| WR | 38 | 1.71 | | | |
| BI | 39 | 0.81 | | | |
| HI | 74 | 0.65 | Great Island[a] | 58 | 0.93 |
| UC | 74 | 0.58 | | | |
| JC | 70 | 0.39 | | | |
| CP | 74 | 0.57 | | | |
| WR | 70 | 1.05 | | | |
| BI | 71 | 0.65 | | | |
| HI | 97 | 0.35 | Hoadley Creek[b] | 98 | 0.15 |
| UC | 97 | 0.42 | | | |
| JC | 89 | 0.38 | Jarvis Creek[b] | 95 | 0.27 |
| CP | 96 | 0.44 | | | |
| WR | 89 | 0.83 | Sherwood Island[b] | 96 | 0.28 |
| BI | 93 | 0.57 | | | |

[a] Warren et al. 2001 – only the values of *Spartina alterniflora* decomposition
[b] Anisfeld 2008

## Decomposition Summary

The higher rates of decomposition found at Barn Island could be caused by temperature, organic matter content, or nutrients.

*Temperature*

Rates of decomposition are sensitive to temperature (Valiela et al. 1985; Morris and Whiting 1986) and higher temperatures would lead to increased decomposition rates (Kirwan and Blum 2011). The scope of this study only allowed for the comparison of decomposition rates in different sites and no significant relationships existed between soil temperatures in this study and different sites or distance from the East River. Water temperature would also be relevant for decomposition rates in salt marsh sediments and water temperatures in this area have been found to be rising over time on a national (Karl et al. 2009), regional (Nixon et al. 2009b; Fulweiler et al. 2015), and local scale (Rice and Stewart 2013). This could play a role in decomposition.

*Organic Matter*

A higher percentage of organic matter can also provide more material to be decomposed, which could inherently allow for faster rates. Barn Island, which had the highest rates, does have significantly higher organic matter content than all the sites besides Hunter Island (Figure 15), so it is possible that the increased supply of substrate, organic matter, could be causing the enhanced decomposition rates at this site. At Jarvis Creek, there was less organic matter than Barn Island and Hunter Island, but it was not significantly different from the other sites, so this might or might not explain the low decomposition rates here.

*Nutrients*

If there are higher concentrations of nutrients, it makes it easier for decomposers to break down organic matter (Turner 2011; Deegan et al. 2012;Wigand et al. 2014). Other studies have found increased decomposition rates after the addition of nutrients (Haines and Hanson 1979; Marinucci et al. 1983; Valiela et al. 1985). In the sites in this study, Barn Island is at the eastern end of LIS and farthest from the source of nutrients, NYC, creating our gradient. This would mean it should have the lowest amount of nutrients, which should lead to decreased decomposition rates. It is possible that local sources of nutrients could be causing increased decomposition. Barn Island is not a fluvial marsh as it is not fed from a river directly upland of it, but the mouth of the Pawcatuck River is directly south east of this marsh and empties into Little Narraganset Bay. It is possible that the discharge from this river could increase the nutrient concentration of this bay, which could influence decomposition rates in Barn Island since this water would be brought onto the marsh by the tide. One study on on the nutrient budget of the Pawcatuck River found that the total dissolved inorganic nitrogen and phosphorus exports are high in relation to the exports to other nearby estuaries (Fulweiler and Nixon 2005). At certain times of the year, these concentrations do reach as high as 40 to 60 μM DIN, which is similar to the highest concentrations found in the western end of LIS (Figure 3; Gobler et al. 2006; Figure A1.3). The high decomposition rates at BI could be influenced by this local source of nutrients. Based on the increase of LOI with depth in Barn Island, it is also possible that decomposition has been increasing over time and allowing less organic matter to build up, which would therefore decrease accretion rates. (Kirwan and Blum 2011).

The cause of the high decomposition rates in Barn Island is not clear, but it could be tied to the low accretion rates found in Barn Island. If this is true, it does not appear to be a wide spread phenomenon. In this study, Barn Island had the highest decomposition rates. The only other site with relatively high rates was Great Island. This site is within the mouth of the Connecticut River, which has a much larger watershed than the Pawcatuck River. It is therefore possible that this river also has high concentrations of nutrients, which could be causing higher rates of decomposition in Great Island. There was no evidence of trends in decomposition rates being related to the nutrient gradient in LIS.

# Salt Marsh Respiration

$CO_2$ fluxes differed significantly between vegetated and unvegetated sediment and by season (Figure A4.1; $p < 0.0001$; $p = 0.0011$, respectively). The majority of fluxes in vegetated sediment, with the exception of winter fluxes, were negative where $CO_2$ is being drawn down by photosynthesis, as would be expected during the growing season (Emery and Fulweiler 2014). The unvegetated sediment $CO_2$ fluxes were overall positive, with the exception of Caumsett Park, which had a small overall draw down of $CO_2$.

## Vegetated $CO_2$ Fluxes

The range of fluxes from vegetated areas was -25.66 to 22.34 mmol $m^{-2}$ $hr^{-1}$ with a mean rate across the sites of -5.76 mmol $m^{-2}$ $hr^{-1}$. There was no difference in $CO_2$ fluxes in vegetated areas in different sites (Figure 22a). Respiration varied more by season with greater emission of $CO_2$ in the winter than in all other seasons (Figure 22b; $p < 0.0001$) since the vegetation is not actively growing and photosynthesizing. There was also a trend of less negative fluxes at vegetated sites in the spring than in the summer ($p = 0.0648$). The normal seasonal trend of mostly negative fluxes in the summer and to a lesser degree in the fall and spring and positive fluxes in the winter was the most significant trend in $CO_2$ fluxes.

There were some deviations from this pattern. In Udalls Cove, a large $CO_2$ emission was observed during the summer. The largest emission of $CO_2$ at Barn Island was during the spring and it was the only site to have overall emission during the spring.

## Unvegetated $CO_2$ Fluxes

The range of fluxes from unvegetated areas was -8.29 to 22.29 mmol $m^{-2}$ $hr^{-1}$ with a mean rate of 2.54 mmol $m^{-2}$ $hr^{-1}$ across seasons. In unvegetated areas, $CO_2$ fluxes in Udalls Cove were significantly higher than fluxes in Caumsett Park (Figure 23a; $p = 0.0031$) and almost higher than fluxes in Jarvis Creek ($p = 0.0585$). Seasonally, positive fluxes were highest in the summer with very low fluxes the rest of the year with a few exceptions (Figure 23b). However, Udalls Cove was the only site where these trends were significant with summer being significantly higher than all other fluxes besides spring. The only deviations from this trend were higher positive fluxes in Hunter Island and Udalls Cove in the spring and in Udalls Cove in the winter. Hunter Island had the highest flux found in all of the unvegetated

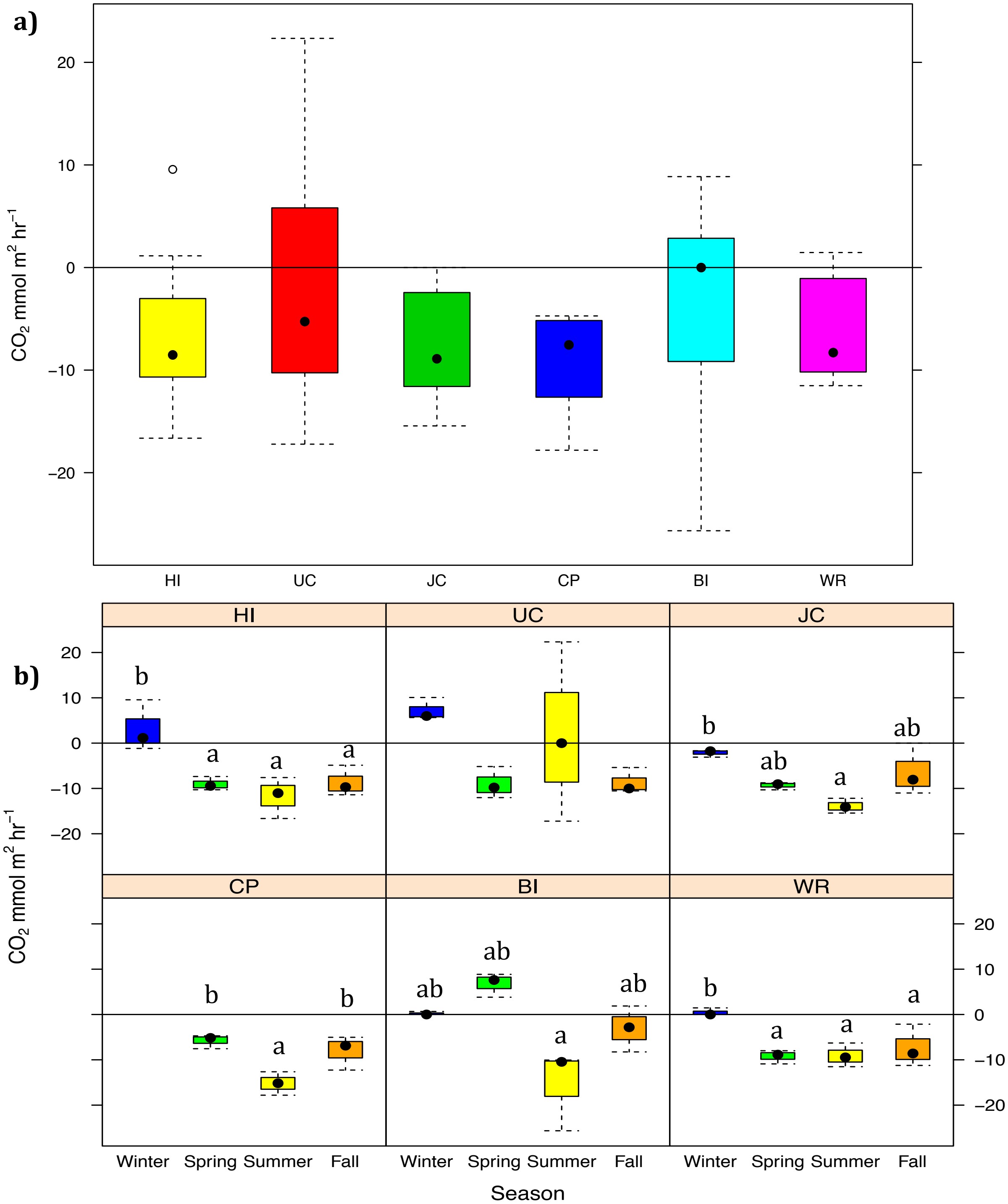


**Figure 22.** Respiration rates in vegetated areas. Respiration rates are grouped by a) site and b) both site and season. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. $CO_2$ fluxes in vegetated areas had a consistent pattern of larger negative fluxes in the summer and emission to zero fluxes in the summer. The only exception was UC where high emission in the summer prevented this relationship from being significant.

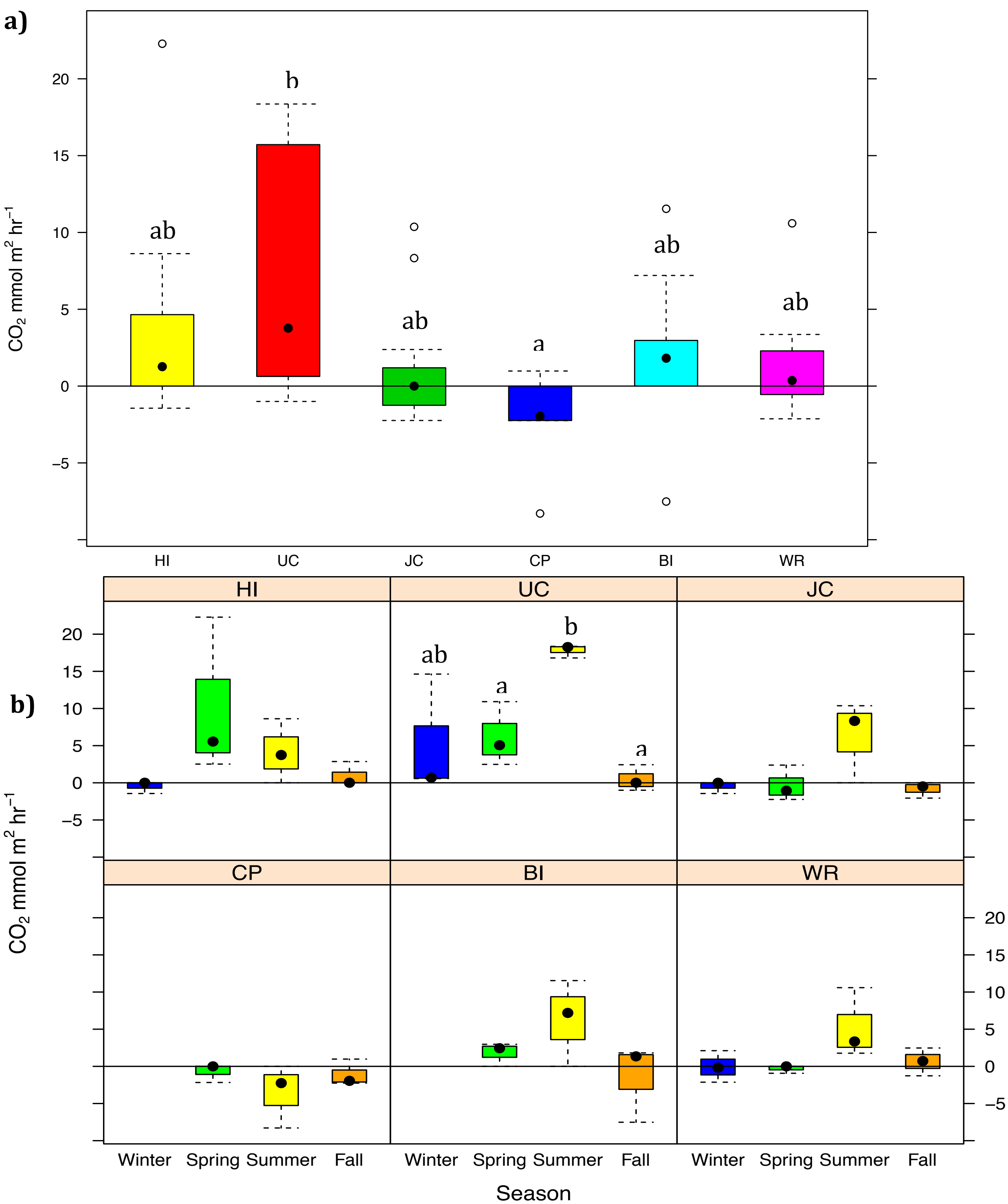


**Figure 23.** Respiration rates in unvegetated areas. Respiration rates are grouped by a) site and by b) both site and season. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River). Shared letters indicate no significant difference. $CO_2$ fluxes in unvegetated areas were very low with the largest rates of emission in the summer. The only significant patterns were seen in UC where significantly higher emission occurred in the summer (ANOVA,p=0.0031).

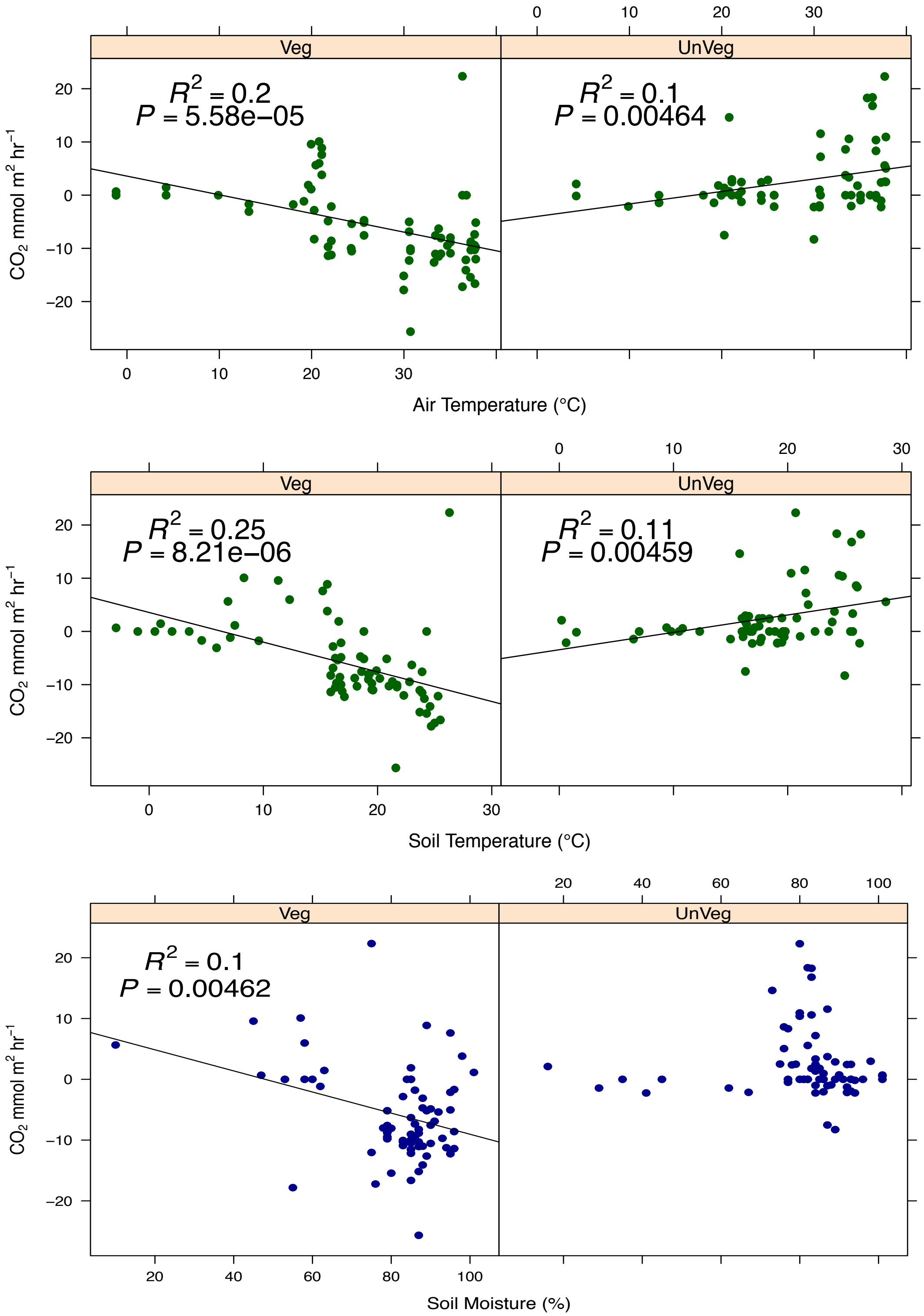


**Figure 24.** Regression of $CO_2$ and environmental parameters. This was done using all sites during all seasons. $CO_2$ was tested against a) air temperature (°C), b) soil temperature (°C), and c) soil moisture (%). For vegetation, negative relationships were found for all three variables ($R^2$=0.205,p<0.0001; $R^2$=0.247,p<0.0001; $R^2$=0.10,p=0.00462). For unvegetated areas positive relationships were found for air temperature and soil temperature ($R^2$=0.105,p=0.0464; $R^2$=105,p=0.0459).

areas and most of the vegetated areas, with the exception of the highest fluxes at Udalls Cove. Caumsett Park was the only site where an average negative flux was detected during every season.

## Environmental Parameters

Air temperature, soil temperature, and soil moisture were also measured at the sites. PAR was measured, but this remained consistent across sites since samples were taken in similar conditions. $CO_2$ fluxes were positively related to air temperature in vegetated and unvegetated sites (Figure 24a; $R^2 = 0.21$, $p < 0.0001$; $R^2 = 0.11$, $p = 0.0046$, respectively). As expected, this same relationship was found with soil temperature (Figure 24a; $R^2 = 0.25$, $p < 0.0001$; $R^2 = 0.11$, $p = 0.0046$). Photosynthesis was inhibited in vegetated areas by colder temperatures and respiration was enhanced in unvegetated areas by warmer temperatures. There was a negative relationship between $CO_2$ fluxes and soil moisture in vegetated areas (Figure 24c; $R^2 = 0.1$, $p = 0.0046$), but this was most likely driven by low soil moisture readings in the winter when the soil was still frozen.

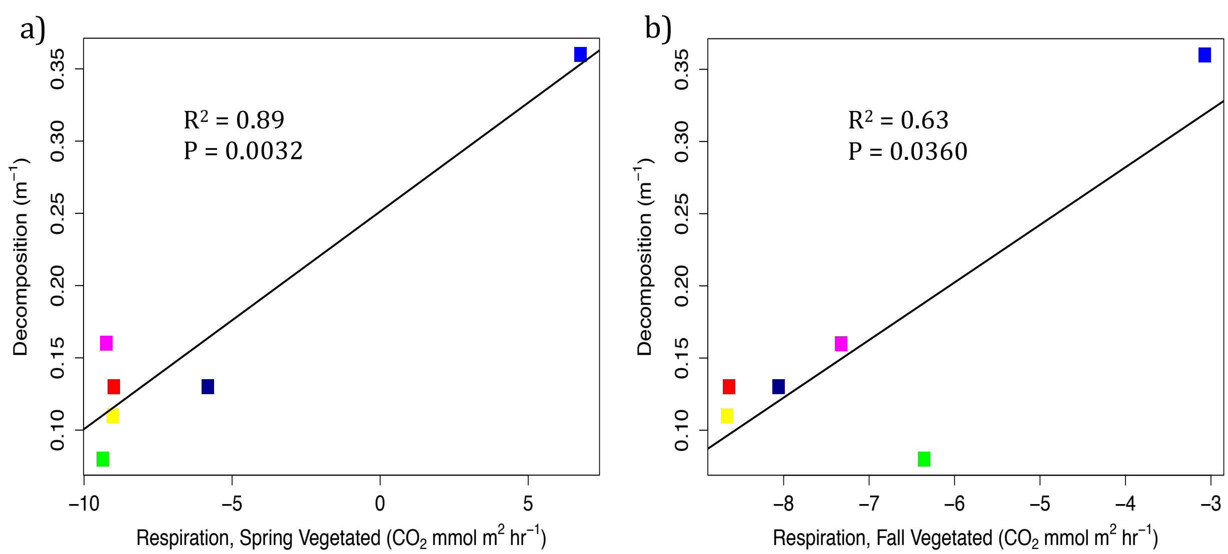


**Figure 25.** Regression of decomposition rates and the percent remaining material by respiration rates. Sites are represented by different colors (Hunter Island=yellow, Udalls Cove=red, Jarvis Creek=green, Caumsett Park=dark blue, Barn Island=light blue, Wading River=magenta). A positive relationship was found between decomposition rates and respiration in vegetated areas in the a) spring and b) fall ($R^2$=0.89, p=0.0032; $R^2$=0.63, p=0.0360).

## Respiration Summary

There was a large overlap in the range of $CO_2$ emissions from vegetated and unvegetated sites driven by the high emission of $CO_2$ measured from the vegetated site in Udalls Cove in the summer. These high emissions were the strangest deviation from seasonal trends with high rates during multiple seasons especially in relation to other sites. The only other sites that ever displayed higher than normal $CO_2$ emissions were Barn Island and Hunter Island in the winter and spring in vegetated and unvegetated areas respectively.

Since respiration rates are a measure of the amount of carbon dioxide that is released from the break down of organic matter and decomposition is a measure of how fast organic matter breaks down, a correlation between both measurements was expected. The higher respiration rates in the summer and the positive relationship between respiration and temperature are evidence of their correlation. There was also a direct relationship between respiration and decomposition rates in vegetated areas in the spring and fall (Figure 25a and b; $R^2$ = 0.89, p = 0.0032; $R^2$ = 0.63, p = 0.0360). However, this relationship did not exist in all areas and seasons. Besides these relationships, the overall trends by site were sometimes similar. Barn Island had the highest rates of decomposition and a high amount of respiration in vegetated areas in the spring. However, with Udalls Cove, which had the highest respiration rates, no high decomposition rates were detected. It is possible that other factors might separate the two processes.

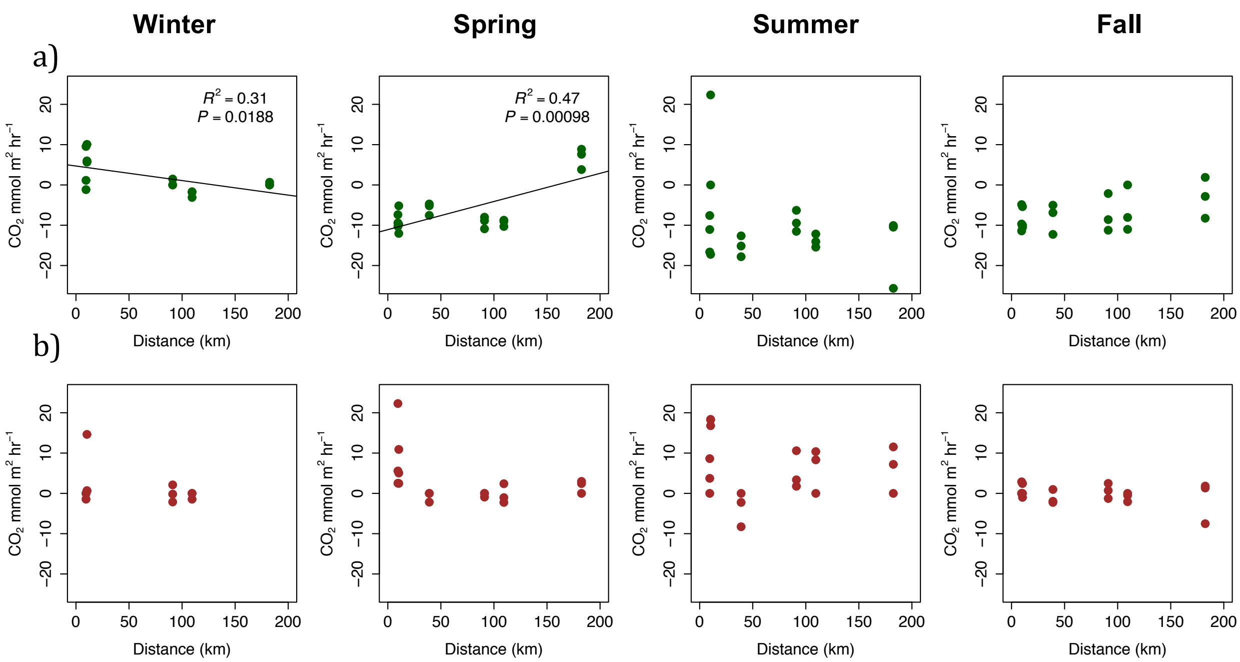


**Figure 26.** Regression of $CO_2$ and Distance from the East River in km. The columns contain measurements from the different season (winter=first, spring=second, summer=third, fall=fourth). Row a) has all vegetated areas (green) and row b) has all unvegetated areas (brown). The relationship with distance was used as a proxy for a nutrient gradient starting from the west/high going to the east/low with treated wastewater coming from NYC in the west. Positive relationships were found for vegetated areas in fall and spring and negative relationships were found in vegetated areas in the winter and unvegetated areas in the spring.

### *Nutrients*

Another question that could be addressed by this study is determining the influence of nutrients on marsh processes along the nutrient gradient. To determine whether or not this would have an effect, changes in respiration according to the sites' distance from the East River across the difference season and vegetation were considered. In vegetated sites, fluxes increased with distance in spring and decreased in the winter (Figure 26; $R^2$ = 0.47, p = 0.0010; $R^2$ = 0.31, p = 0.0188). In vegetated areas in the summer, there was mostly uptake across the gradient with the exception of some large emissions of $CO_2$ closer to the city, though no significant relationship was found. In spring, the trends were driven by greater uptake closer to the city and in the winter it was driven by greater emissions closer to the city. It is possible that on the higher end of the nutrient gradient, towards the western end of LIS, the plants are experiencing enhanced growth patterns. They are taking up more $CO_2$ than other sites in the fall and spring and releasing more than other sites in the winter. In the summer, the overall large negative fluxes could be overwhelming this trend or increased temperatures and nutrients are increasing respiration and overwhelming the increased negative fluxes close to the city. There were no significant trends in the unvegetated areas with distance.

# Conclusion

Salt marshes are important ecosystems that have historically been capable of maintaining their elevation in relation to sea level rise (Redfield 1972; Morris et al. 2002) but given the rapid acceleration of RSLR particularly in the northeast (Yin et al. 2009; Sallenger et al. 2012; Andres et al. 2013), they are at a greater risk of drowning (Turner et al. 2000; Bromberg and Bertness 2005; Morris et al. 2013; Valiela et al. 2015). The primary focus of this study was to determine whether or not these marshes are keeping pace with sea level rise and understand the driving forces behind which marshes are and which are not. Three forces considered here are dynamics between organic or inorganic matter accretion; temperature, decomposition and respiration; and nutrient concentrations.

### *Organic and Inorganic Matter*

The balance between organic and inorganic matter does vary by site and there is some indication that it could be correlated with accretion. The lowest accretion rates were correlated with the site with highest organic matter and highest water content, Barn Island, and the highest accretion rates were correlated with the sites with highest bulk density, Wading River and Udalls Cove. This relationship is consistent with current theories and previous studies (Redfield 1972; Allen 2000; Carey et al. 2015). In between these two extremes, a clear relationship does not exist, so this is at least not the only factor determining accretion rates. The amount of pore water in a marsh, which could build up in peaty marshes to help maintain elevation in relation to sea level rise, makes up three quarters or more of the sediment profile of all marshes. This indicates the importance of organic matter and pore water in maintaining marsh accretion (Bricker-Urso et al. 1989; Carey et al. 2015).

### *Temperature, Decomposition, and Respiration*

Increased temperature has been found to increase decomposition (White et al. 1978) and this could lead to decreased accretion rates (Kirwan and Blum 2011). Temperature does appear to be one factor driving respiration rates, though there was no direct correlation with decomposition rates. Trends in respiration rates and decomposition rates were not always consistent, though there was some evidence for a relationship during certain seasons. In terms of accretion rates, there was no relationship between any of these variables and accretion, though the same relationship existed for decomposition and accretion as for organic matter accretion. The one site with the highest decomposition was the same site that had the lowest accretion rates. However, the trend did not hold true for most sites as the site with the second highest decomposition rates also had the highest accretion rates. Higher rates of decomposition were also not wide spread as only one site appeared to be decomposing organic matter at a particularly fast rate, so it is possible high decomposition rates is an important factor in determining accretion, but its effects are not wide spread.

### *Nutrients*

Decomposition rates have been found to increase with nutrient addition (Haines and Hanson 1979; Marinucci et al. 1983; Valiela et al. 1985) and decomposition can decrease rates of organic accretion. There was no direct correlation between accretion rates and the nutrient gradient incorporated into the design of this experiment. However, it is possible that local sources of nutrients could play an important role in increasing decomposition rates in salt marshes like Barn Island and Great Island, which both had high decomposition rates and were close to or in the mouth of rivers with large watersheds and potentially high nutrient loads. Harrison and Bloom made the last measurement of accretion rates at this marsh in 1973, which is too outdated for comparisons. This indicates the importance of measuring local nutrient concentrations in salt marshes.

The effect of the nutrient gradient was only seen on respiration rates in the spring and winter in vegetated areas. It is possible that vegetation patterns in salt marshes closer to the city are experiencing enhanced growth cycles in comparison to salt marsh vegetation elsewhere. The uptake of $CO_2$ closer to the city was enhanced in the spring and the emission of $CO_2$ was enhanced during the winter. This is consistent with nutrients both having been found to cause greater growth of aboveground biomass in salt marshes (Morris et al. 2013) as well as increased respiration.

### *Accretion and Relative Sea Level Rise*

Accretion rates in salt marshes in LIS do still appear to be increasing overall, though there is some variation. Barn Island and Hunter Island had the lowest rates, though Jarvis Creek and Caumsett might also have similarly low rates based on the $^{210}Pb$ CRS model, and Udalls Cove and Wading River had the highest rates. There were positive trends for all marshes using the CRS data and significant relationships for Barn Island, Jarvis Creek, and Caumsett Park. Barn Island, Hunter Island and Udalls cove also increased in comparison to historic rates as well as based on trends towards the end of the CRS model data. Despite this, none of the rates measured in this study were close to the current rates of RSLR. Most of them were 2 to 3X lower. While accretion rates are not the only measurement necessary to understand the elevation of the marsh platform above sea level, they are an important indication of this relationship.

Determining how salt marshes will respond to RSLR is difficult because of the many important factors involved such as sediment supply, organic matter production, temperature trends, decomposition and respiration rates, and nutrient concentrations. Given the results in this study, while accretion rates in marshes appear to be increasing, it is possible that further salt marsh losses will occur due to high RSLR. Another study using wetland delineation from aerial photography found evidence of salt marsh loss in this region, which supports the assessment that more salt marsh loss could be expected in the future (Cameron et al. 2015). Careful monitoring of accretion rates will continue to be important in the future to help track, understand, and hopefully mitigate some of these stressors.

# Recommendations

This was one of the first studies combining measurement of salt marsh $CO_2$ fluxes, decomposition rates, and accretion rates in LIS marshes. The results highlights the inherent dynamics and variability of salt marsh systems. Currently, marshes are rarely revisited to repeat previous accretion rates measurements with the exception of Barn Island, CT (Harrison and Bloom, 1977; Young 1985; Warren and Niering 1993; Orson et al. 1998; Carey et al. 2015b) and Flax Pond, NY (Siccama and Porter 1972; Armentano and Woodwell 1975; Muzyka 1976; Richard 1978; Cochran et al. 1998; Kolker et al. 2009). The previous studies at the sites in this assessment were conducted approximately 15 to 30 years ago (Young et al. 1985; Orson et al. 1998; Cochran et al. 1998; Anisfeld et al. 1999). It will be important to determine trends more frequently over the approaching decades given the current exponential increase in sea level. We recommend that a few sentinel sites are chosen and studied annually for a period of 3 to 5 years. The number of sites would necessarily depend on funding given the logistic difficulties associated with monitoring sites along the entire perimeter of Long Island Sound, but should be no less than three. As funding allowed we suggest adding additional sites. After the end of this 3 to 5 year study we recommend re-evaluating to determine if the study should continue or if resources should be focused elsewhere.

In these studies, it will be important to attain good spatial coverage. Aerial photography can accomplish this and these efforts should be continued (Cameron, 2015), but a suite of ground-based methods is necessary to understand the mechanisms associated with salt marsh loss. This understanding can help decide optimal management strategies. At the chosen sites we recommend at least seasonal sampling for the following parameters: net $CO_2$ fluxes, above and belowground biomass, sediment characteristics (e.g., LOI, bulk density, etc.), inorganic nutrient concentrations, and greenhouse gas emissions. We recommend that accretion rates be measured more frequently. When measuring accretion rates it is important capture erosion and marsh platform elevation changes using methods such as sedimentation erosion tables (SET) or sedimentation. SET can provide a more accurate measurement (Boumans and Day 2008) and sedimentation pins can be deployed inexpensively over a large area (Carey et al. 2015b). Decomposition should also be measured and litterbags should be collected monthly throughout the growing season of each year. Decomposition and net $CO_2$ flux measurements will reveal biomass losses, which is crucial to understand in the organic-rich marshes of this region. Above and belowground biomass will indicate specifically where biomass is being lost if it is and sediment characteristics will illuminate the constituents of the marsh platform. Inorganic nutrient concentrations will give insight into the degree of nutrient loading and how this is influencing processes such as decomposition. Greenhouse gas emissions should also be monitored to measure the degree to which our salt marshes are contributing to the greenhouse gas effect and to better understand rates of anaerobic decomposition.

In the future, taking a volumetric approach to salt marsh dynamics could be a useful framework for answering the question of whether or not these marshes are keeping pace with RSLR. Sediment budgets are commonly implemented tools (Trimble et al. 1981; Meade 1982; Fitzgerald and Heteren 1999), but this approach was only recently applied to a salt marsh in the Bay of Fundy (van Proosdij et al. 2006). Proosdij et al. (2006) increased spatial coverage of accretion measurements and measured marginal erosion to determine the volumetric change in a marsh. This was sufficient for marshes in the Bay of Fundy where sediment supply is much higher (Amos and Tee 1989) and marshes are less dependent on organic accretion (Johnson 1967). In LIS salt marshes, it will be important to include measurements of decomposition, net $CO_2$ fluxes, pore water and the other biomass related measurements to fully understand volumetric changes. This approach could lend insight into how much of the marsh is being lost, where it is being lost from, and why it is being lost.

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# Appendix 1: Long Island Sound Environmental Data

## Methods and Data Analysis

A robust review of environmental conditions and the current state of LIS was recently published titled, "LIS – Prospects for the Urban Sea" (Latimer et al. 2013). Here we report a synthesis of research conducted after this effort. Specifically we gathered available data for atmospheric temperature and wind conditions, inorganic nutrients, and water column chlorophyll a (chl *a*) and dissolved oxygen (DO).

## Results and Discussion

Wind and water temperature did not vary across the length of LIS, but between 2004 and 2010 wind significantly increased over time (Figure A1.1; $R^2 = 0.16$, $p = 0.0003$; O'donnell et al. 2013; Gobler et al. 2006; Codiga and Nehra 2012; Baumann et al. 2015; and George et al. 2015). This remained consistent when looking at records at three separate stations at different locations along the east, west gradient (Figure A1.2). This could have lead to increased sediment re-suspension in LIS and therefore higher rates of deposition and salt marsh accretion (Ward et al. 1984; Reed 1989; Kolker et al. 2009). We observed no significant trends in temperature, however we only examined a 12 year period and it is likely that the time series is too short to detect a temperature increase. Temperature increases have been observed locally and regionally (Nixon et al. 2009b; Karl et al. 2009; Rise and Stewart 2013).

We also examined changes in dissolved inorganic nitrogen (DIN), phosphorus (DIP), and silica (DSi) from 2000 to 2011 at 5 stations in LIS (Figure A1.3 thru A1.4; Gobler et al. 2006; George et al. 2015). We observed no trends in inorganic nutrient concentrations over this time period. As expected, however, inorganic nitrogen and phosphorus did significantly ($p < 0.0001$) decrease with distance from NYC, the major nutrient loading area of LIS. Dissolved silica did not decrease along this gradient (Figure A1.5). These data help put the six marsh sites into a larger environmental context. Specifically, we had selected the salt marsh sites because they had been previously studied and because they fell along the west/high to east/low nutrient gradient of LIS.

Surface water dissolved oxygen across LIS, at least for the stations examined here, is significantly increasing over the last decade (Figure A1.7a; $R^2 = 0.59$, $p < 0.0001$; Cuomo and Valenta 2003; Latimer et al. 2013; Collins et al. 2013; Wallace et al. 2014; and Baumann et al. 2015). Additionally, surface water column chl *a* appears to be decreasing over this same time period (Figure A1.7b; $R^2 = 0.43$, $p = 0.0002$; Gobler et al. 2006; Codiga and Nehra 2012; and George et al. 2015). While this is the case, samples were taken at different locations over time for both of these parameters (Figure A1.8), which could have influenced the results since there is a known difference in conditions along LIS. Continued monitoring will be essential in determining whether or not efforts to decrease eutrophication in LIS have been effective (Latimer et al. 2013).

# Conclusion

Our literature review of environmental data revealed an increase in wind speed over time, but no trends in temperature over time or distance for water or soil temperature. The temperature record found in this study is most likely too short to detect the increasing trend found by other studies (Nixon et al. 2009b; Karl et al. 2009; Rise and Stewart 2013)). We did find more evidence of the nutrient gradient with higher levels of dissolved inorganic nitrogen and phosphorous at the western end of Long Island Sound, which supported the use of this location as a nutrient gradient. There was also an increase in dissolved oxygen and a decrease in chl *a* over this time period, but measurements were taken at different locations over time and these spatial differences could be driving trends, so further measurements will be necessary to determine trends in these parameters.

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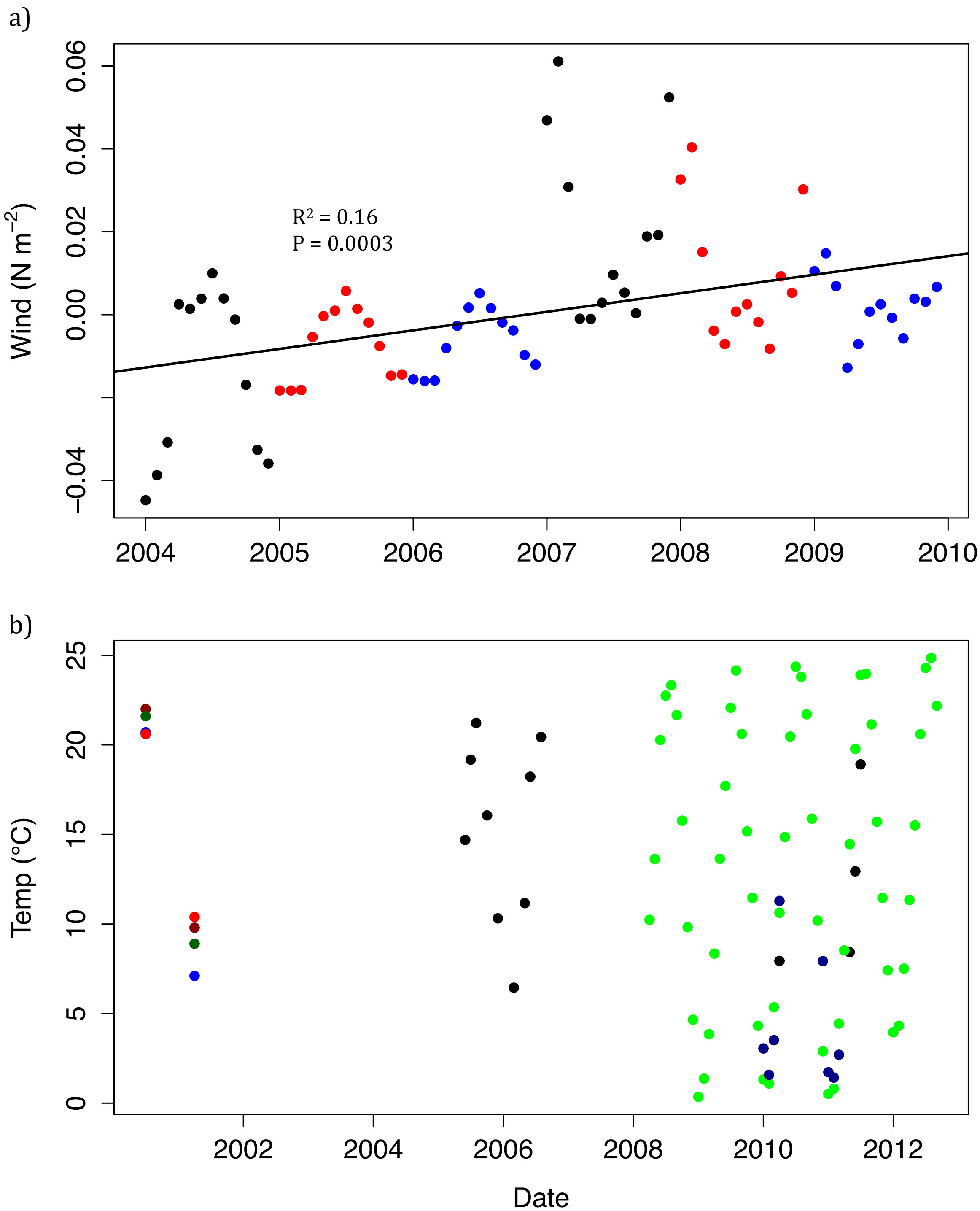



**Figure A1.1.** Mean monthly wind speed (a; 2004 – 2010) and mean monthly water temperature (b; 2000 – 2012) for 3 and 7 sites respectively across LIS. Wind data is from O'donnell et al. (2013) and temperature data is from Gobler et al. (2006); Codiga and Nehra (2012); Baumann et al. (2015); and George et al. (2015). The points are colored by different locations. Wind speed appears to be increasing over time ($R^2$=0.16, p=0.0003), but there is no difference in temperature over this time period.

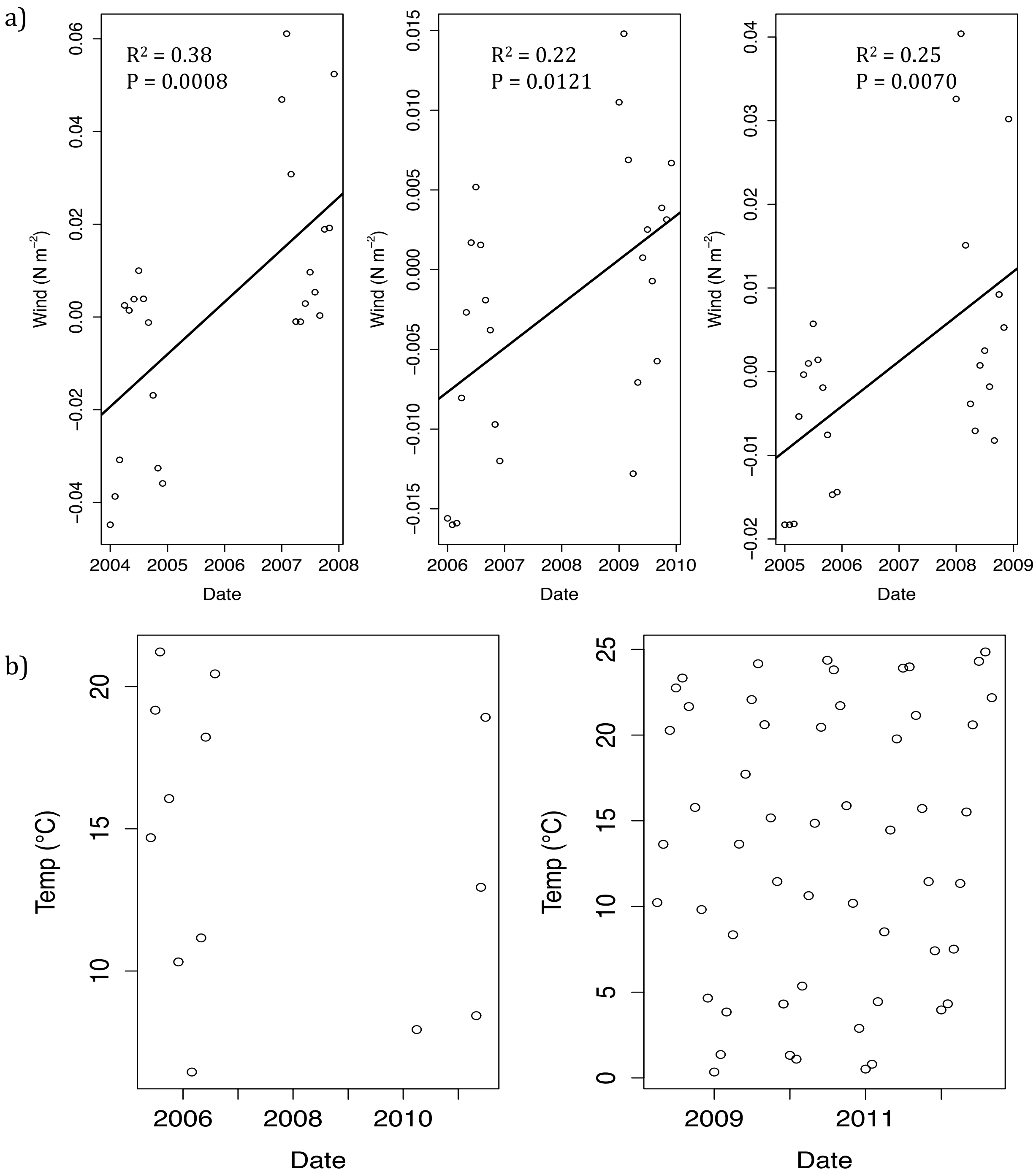



**Figure A1.2.** Mean monthly wind speed (a; 2004 – 2010) and mean monthly temperature (b; 2000 – 2012) for 3 (longitude from left to right: 72.62, 73.181, 73.57) and 2 (longitude from left to right: 72.159 73.137) sites across LIS, which had more than 10 data points at each. Wind data is from O'donnell et al. (2013) and Temperature data is from Gobler et al. (2006); Codiga and Nehra (2012); Baumann et al. (2015); and George et al. (2015). Wind speed appears to be increasing over time for all three locations ($R^2$=0.38, p=0.0008; $R^2$=0.22, p=0.0121; $R^2$=0.25, p=0.0070), but there is no difference in temperature at any site over this time period.

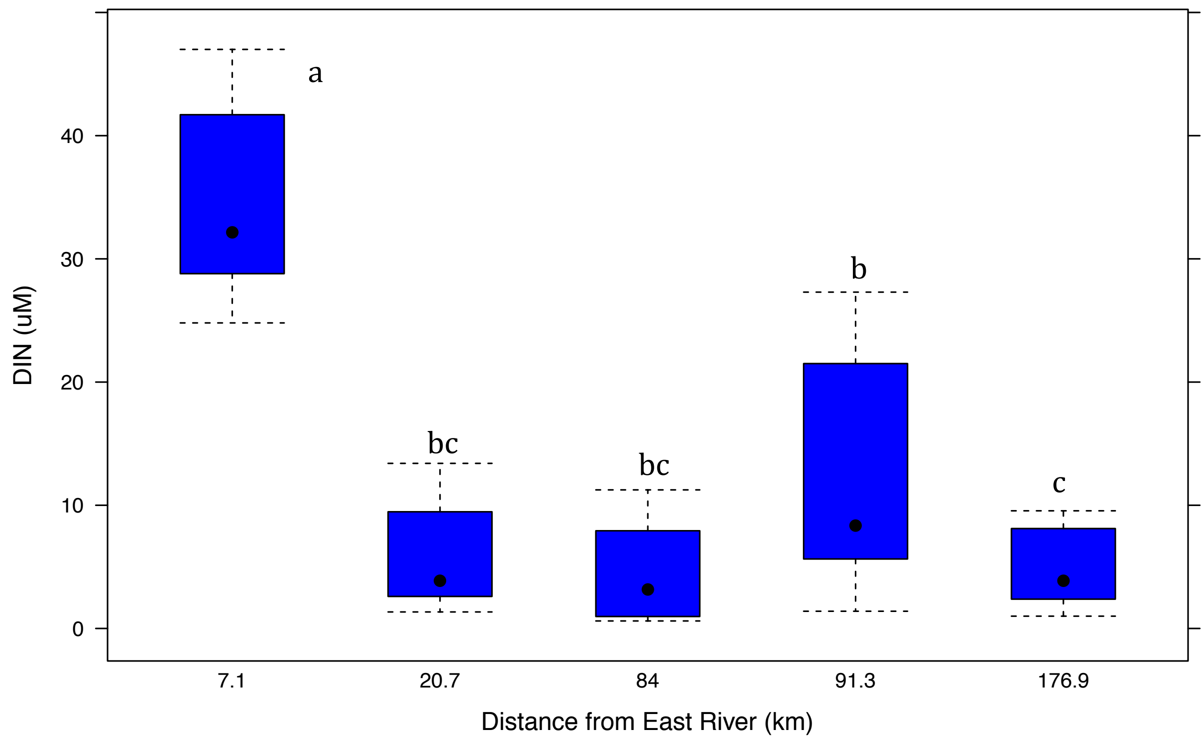



**Figure A1.3.** Mean dissolved inorganic nitrogen (2000 – 2011) at five locations at different distances (7.1 – 176.9 km) from the East River in NYC. These data are from Gobler et al. (2006) and George et al. (2015). Shared letters indicate no significant difference. There were higher concentrations closer to the river (ANOVA, $p<0.0001$) which provides support for the use of LIS as a nutrient gradient in this study.

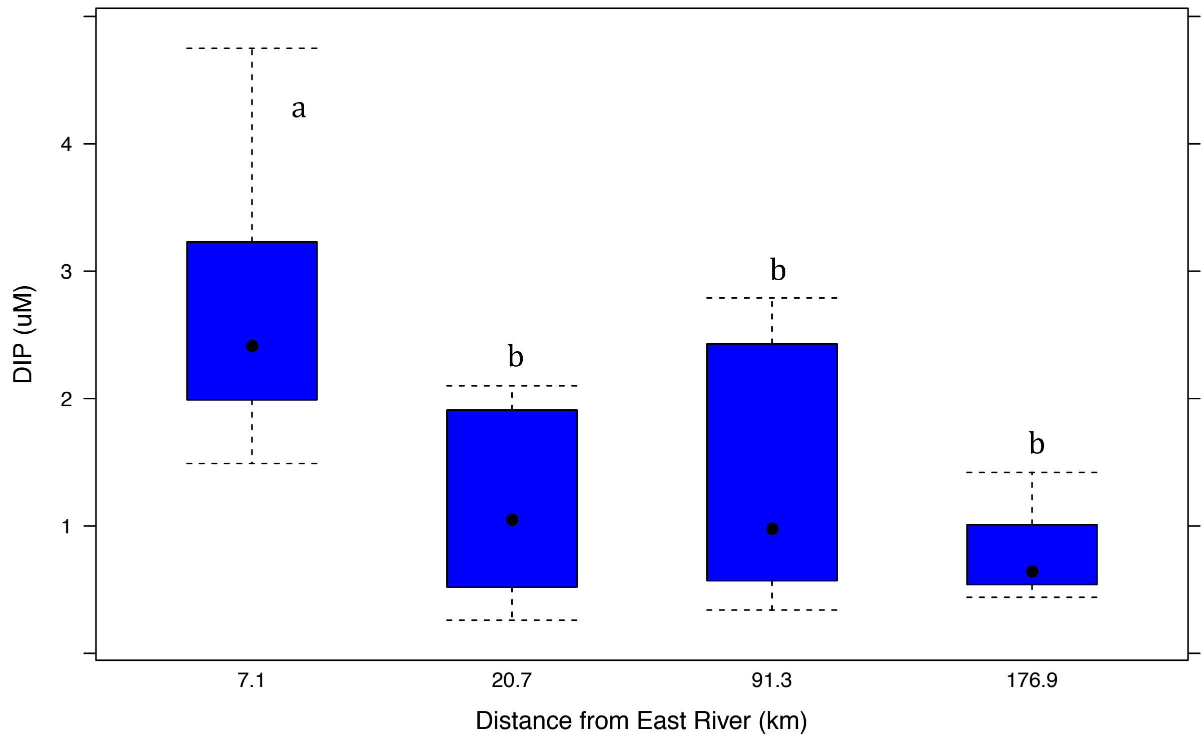



**Figure A1.4.** Mean dissolved inorganic phosphorus (2000 – 2004) at four locations at different distances (7.1 – 176.9 km) from the East River in NYC. These data are from Gobler et al. (2006). Shared letters indicate no significant difference. There were higher concentrations closer to the river (ANOVA,p<0.0001) which provides support for the use of LIS as a nutrient gradient in this study.

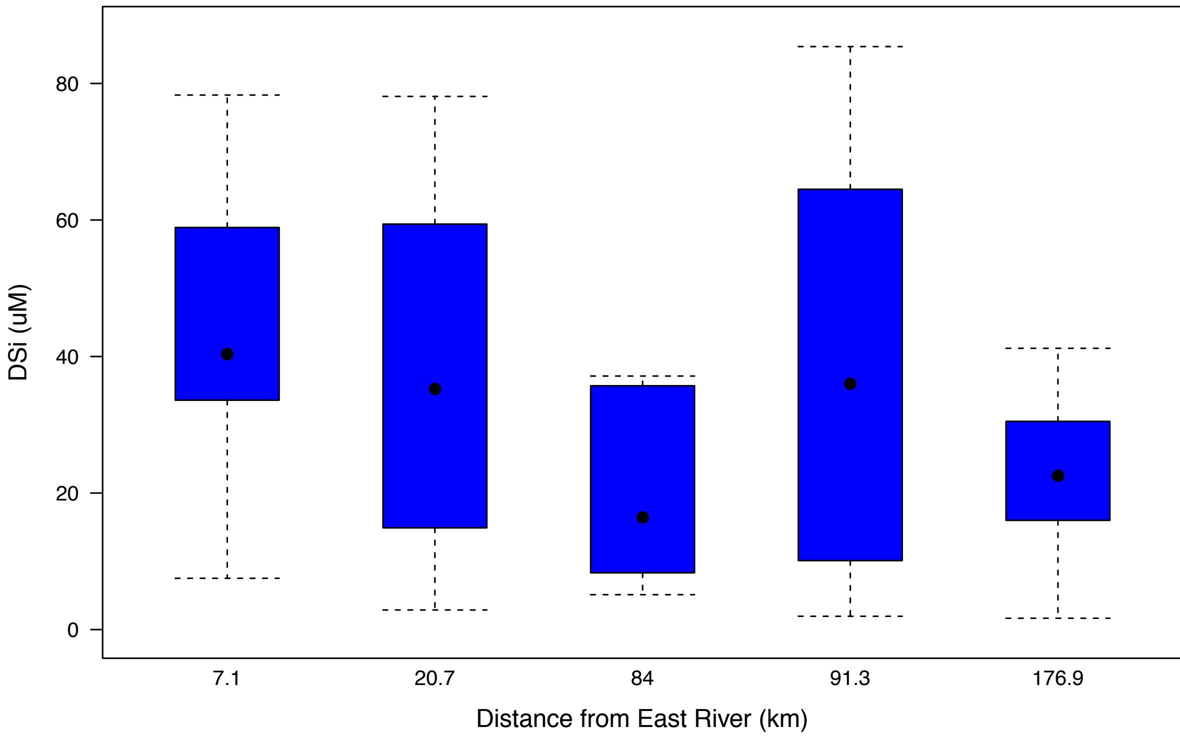



**Figure A1.5.** Mean dissolved inorganic nitrogen (2000 – 2011) at five locations at different distances (7.1 – 176.9 km) from the East River in NYC. These data are from Gobler et al. (2006) and George et al. (2015). There were slightly higher concentrations closer to the river though there were no significant differences between the locations.

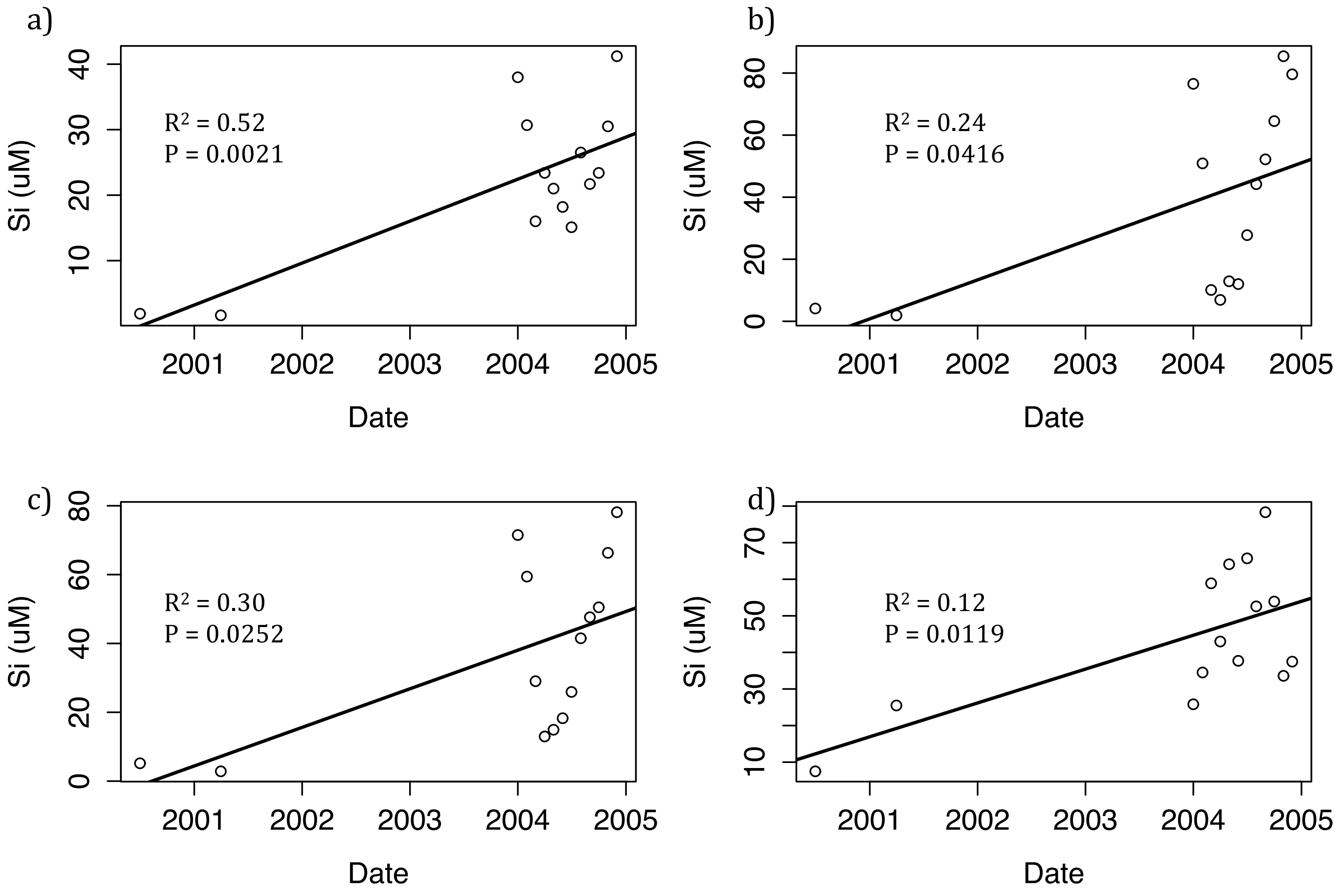



**Figure A1.6.** Mean dissolved inorganic nitrogen (2000 – 2005) at five locations (a) 72.197 b) 73.078 c) 73.717 d) 73.935). These data are from Gobler et al. (2006) and George et al. (2015). DSi significantly increased over time at all locations ($R^2$=0.52,p=0.0021; $R^2$=0.24,p=0.0416; $R^2$=0.30,p=0.0252; $R^2$=0.12,p=0.0119).

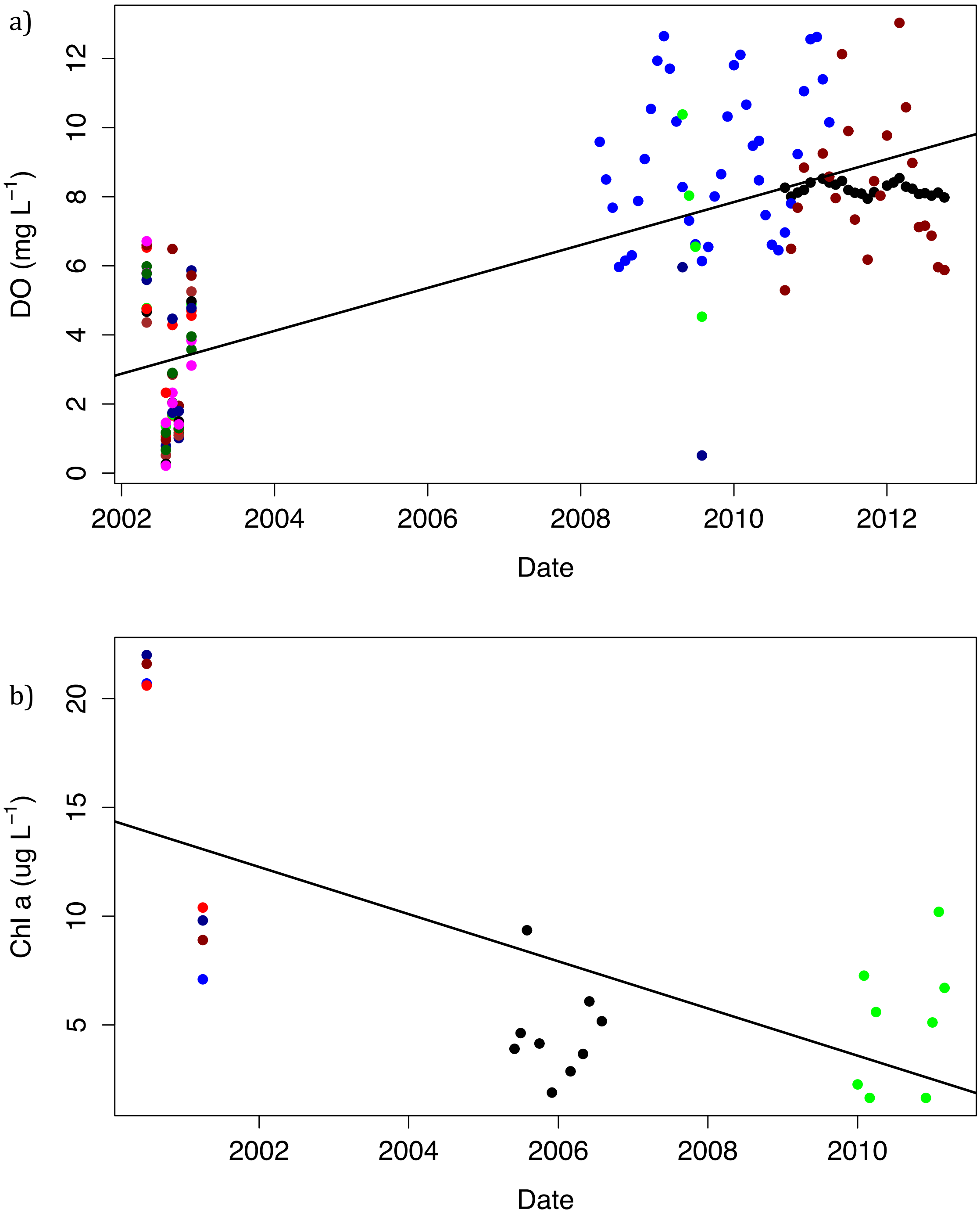



**Figure A1.7.** Mean monthly dissolved oxygen (a; 2002 – 2012) and mean monthly chlorophyll a (b; 2000 – 2010) for 18 and 6 sites respectively across LIS. DO is from Cuomo and Valenta (2003); Latimer et al. (2013); Collins et al. (2013); Wallace et al. (2014); and Baumann et al. (2015) and chl a data is from Gobler et al. (2006); Codiga and Nehra (2012); and George et al. (2015). The points are colored by different locations. DO appears to be increasing over time ($R^2$=0.59,p<0.0001) and chl a is decreasing over this time period ($R^2$=0.43, p=0.0002).

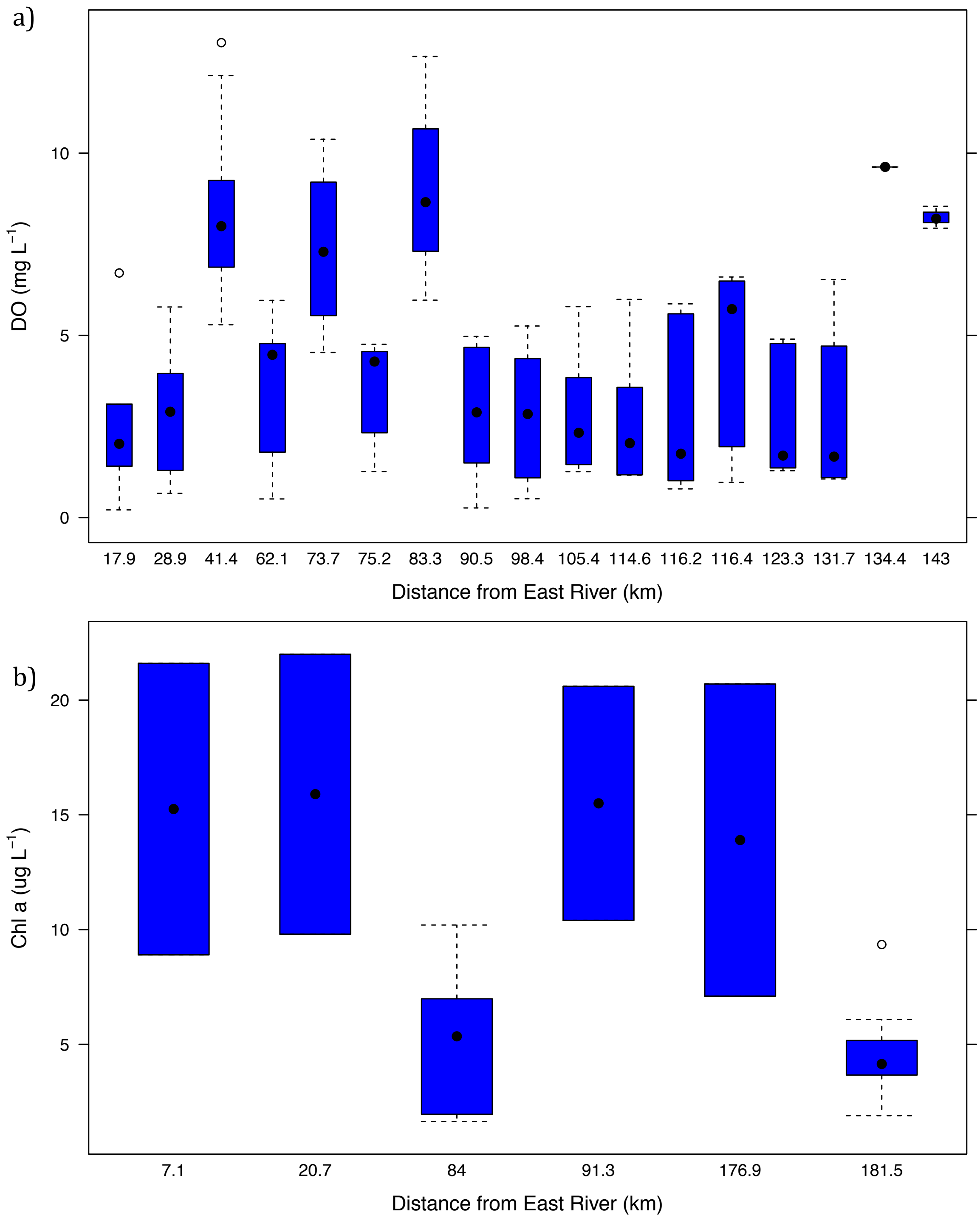



**Figure A1.8.** Average dissolved oxygen (a; 2002 – 2012) and chlorophyll a (b; 2000 – 2010) for 6 and 18 sites respectively across LIS. DO is from Cuomo and Valenta (2003); Cuomo et al. (2011); Collins et al. (2013); Wallace et al. (2014); and Baumann et al. (2015) and chl a data is from Gobler et al. (2006); Codiga and Nehra (2012); and George et al. (2015). There were no clear trends in DO along the nutrient gradient or chl a, though there was a weak decreasing trend going towards the eastern/lower end of the gradient.

# Appendix 2: Accretion Rates

**Table A2.1.** Site information for the study sites for this project. The distances calculated are the Euclidian distances. If marsh type was not found in the literature, satellite images were used to determine marsh type.

| **Marsh Site** | **State** | **Latitude** | **Longitude** | **Distance from NYC (km)** | **Marsh Type** | **Primary vegetation** |
|---|---|---|---|---|---|---|
| HI | NY* | 40.8712 | 73.8092 | 9.49 | Back Barrier | *S. alterniflora* along creek, *S. patens* on marsh platform |
| UC | NY | 40.7787 | 73.7442 | 10.33 | Back Barrier | *S. alterniflora* along creek, *S. patens* on marsh platform |
| JC | CT | 41.2639 | 72.7399 | 109.38 | Fluvial | short-form *S. alterniflora* |
| CP | NY | 40.9415 | 73.4798 | 39.04 | Back Barrier | *S. alterniflora* |
| BI | CT | 41.3394 | 71.8729 | 182.47 | Back Barrier | *S. alterniflora* along creek, *S. patens* and *Salicornia* on marsh platform |
| WR | NY | 40.9628 | 72.8627 | 91.12 | Fluvial | short-form *S. alterniflora* and *Salicornia* |

*This site was in NY, but on the north shore of the LIS.

**Table A2.2.** Dates from $^{210}$Pb CIC model at the $^{137}$Cs peak depth. All of the depths are within 12 years of 1963 with the exception of HI, which is 33 years earlier.

| **Site** | **$^{210}$Pb date at $^{137}$Cs peak** |
|---|---|
| HI | 1930 |
| UC | 1953 |
| JC | 1951 |
| CP | 1959 |
| BI | 1953 |
| WR | 1969 |

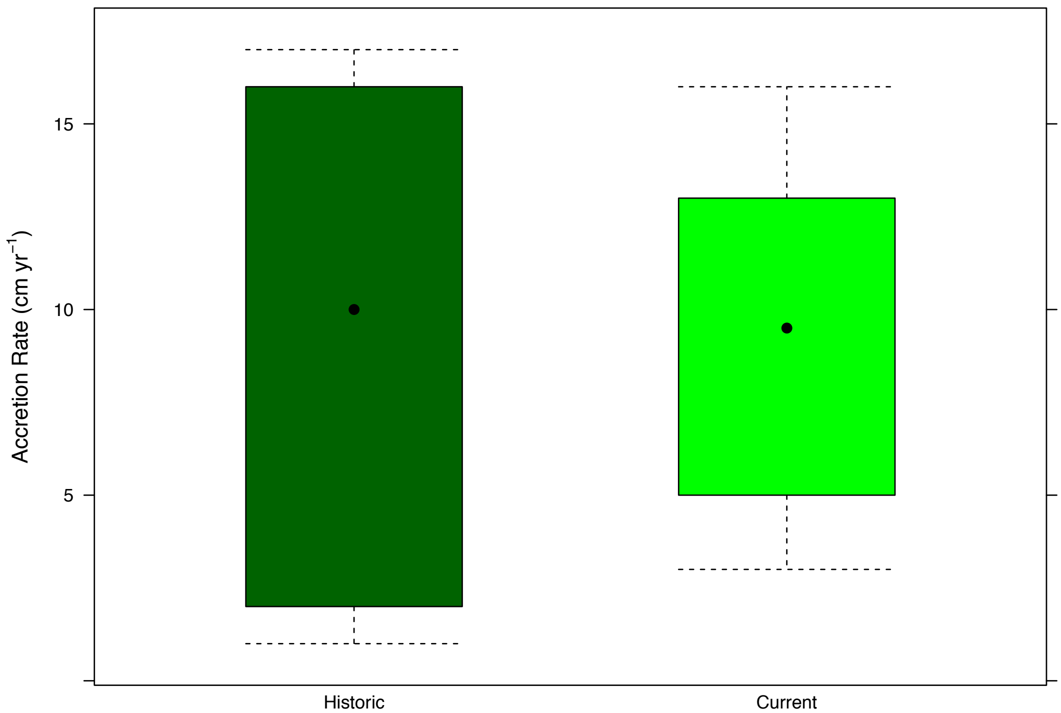



Figure A2.1. Accretion rates from this study and previous studies. Current accretion rates from the $^{210}$Pb CRS model, $^{210}$Pb CIC model, and $^{137}$Cs peaks were all used. Shared letters indicate no significant difference. No significant difference was found.

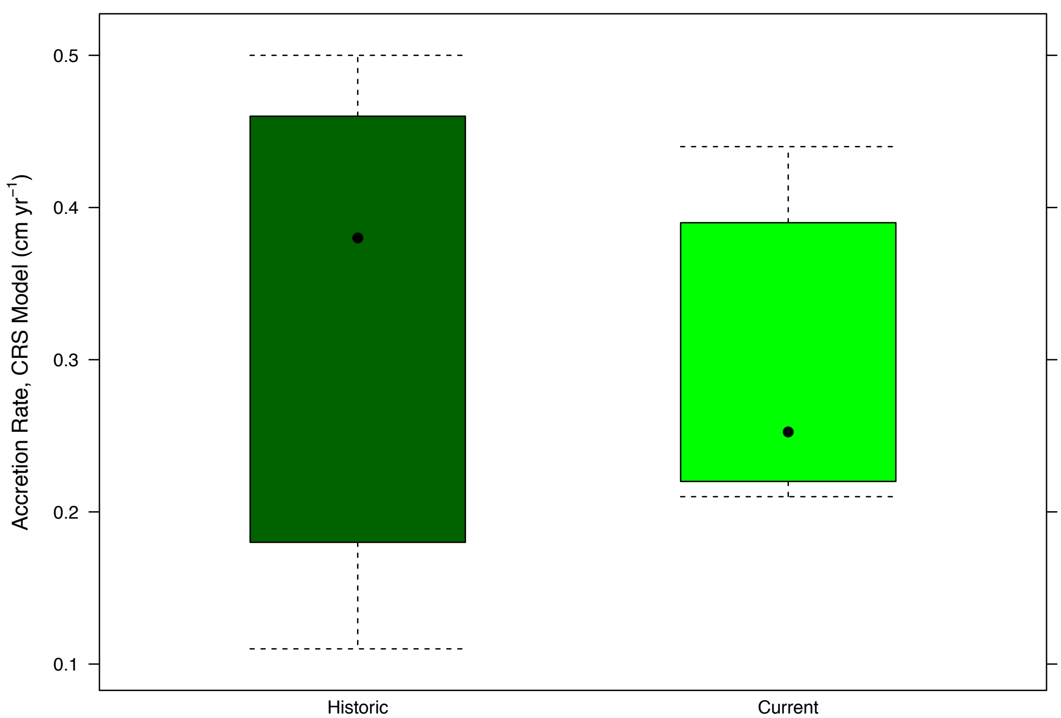



**Figure A2.2.** Accretion rates from this study and previous studies. Only current accretion rates from the $^{210}$Pb CRS model were used. Shared letters indicate no significant difference. No significant difference was found.

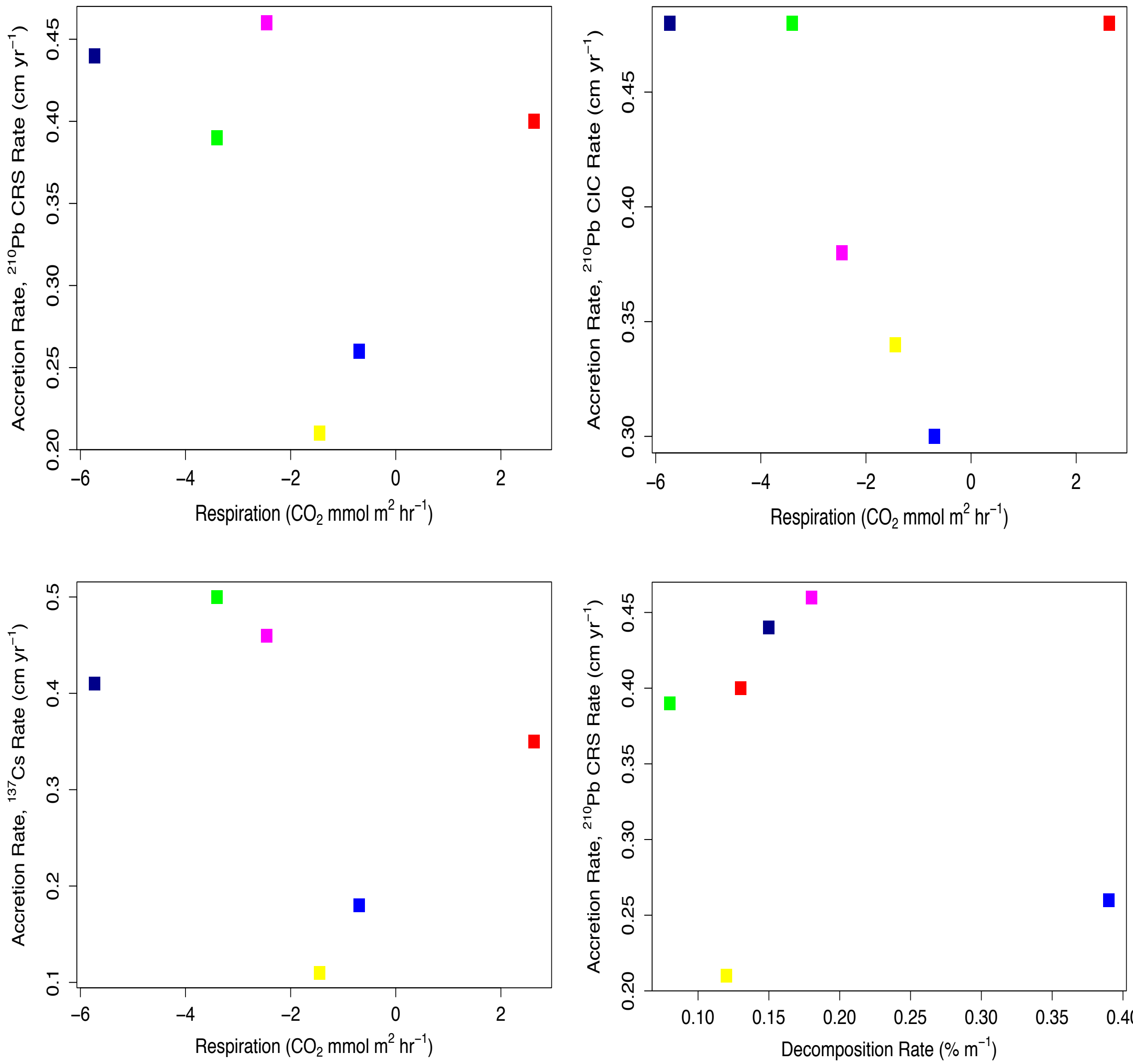



**Figure A2.3.** Plots of Accretion Rates vs. Decomposition Rates and Respiration Rates. The a) $^{210}$Pb CRS rate, b) $^{210}$Pb CIC rate, and c) $^{137}$Cs peak rate were all run against the average respiration for each site. They were also run against all of the subsets of respiration by season and vegetation (not shown here). No relationships were found. For Decomposition, the $^{210}$Pb CRS rate was run against the decomposition rate month$^{-1}$. This was also run against the different methods for accretion rate and the percent of material remaining as another measure of decomposition, but no relationships were found.

**Table A2.3.** Site information and data from literature review of all previous studies in Long Island Sound.

| State | Site | Method | Type | Lat | Long | Year | Accretion Rate (cm $yr^{-1}$) | Marsh Area | LOI |
|---|---|---|---|---|---|---|---|---|---|
| CT | Branford and Guilford, Sybil-1[a] | $^{210}Pb$-CIC | Fluvial | 41.2608 | 72.8094 | 1999 | 0.11 | Low | 97.8 |
| CT | Branford and Guilford, Hoadley-1[a] | $^{210}Pb$-CIC | Fluvial | 41.2617 | 72.7336 | 1999 | 0.52 | Mid | 53.2 |
| CT | Branford and Guilford, Hoadley-2[a] | $^{210}Pb$-CIC | Fluvial | 41.2617 | 72.7336 | 1999 | 0.38 | Mid | 55.4 |
| CT | Branford and Guilford, Hoadley-3[a] | $^{210}Pb$-CIC | Fluvial | 41.2617 | 72.7336 | 1999 | 0.59 | Mid | 48.9 |
| CT | Branford and Guilford, East-1[a] | $^{210}Pb$-CIC | Fluvial | 41.27 | 72.6567 | 1999 | 0.34 | Mid | 40.3 |
| CT | Branford and Guilford, East-2[a] | $^{210}Pb$-CIC | Fluvial | 41.27 | 72.6567 | 1999 | 0.24 | Mid | 56.5 |
| CT | Branford and Guilford, Sybil-1[a] | $^{137}Cs$-1963 | Fluvial | 41.2608 | 72.8094 | 1999 | 0.25 | Low | 97.8 |
| CT | Branford and Guilford, Hoadley-1[a] | $^{137}Cs$-1963 | Fluvial | 41.2617 | 72.7336 | 1999 | 0.42 | Mid | 53.2 |
| CT | Branford and Guilford, Hoadley-2[a] | $^{137}Cs$-1963 | Fluvial | 41.2617 | 72.7336 | 1999 | 0.42 | Mid | 55.4 |
| CT | Branford and Guilford, Hoadley-3[a] | $^{137}Cs$-1963 | Fluvial | 41.2617 | 72.7336 | 1999 | 0.33 | Mid | 48.9 |
| CT | Branford and Guilford, East-1[a] | $^{137}Cs$-1963 | Fluvial | 41.27 | 72.6567 | 1999 | 0.44 | Mid | 40.3 |
| CT | Branford and Guilford, East-2[a] | $^{137}Cs$-1963 | Fluvial | 41.27 | 72.6567 | 1999 | 0.34 | Mid | 56.5 |
| CT | Barn Island[b] | Marker Horizon-Glitter Addition | Back Barrier | 41.3385 | 71.8719 | 1973 | 0.20 | High | 56.9 |
| CT | Great Island[b] | Marker Horizon-Glitter Addition | Fluvial | 41.2872 | 72.3266 | 1973 | 0.38 | High | 49.8 |
| CT | Hammock River marsh[b] | Marker Horizon-Glitter Addition | Back Barrier | 41.2635 | 72.5075 | 1973 | 0.36 | High | |
| CT | Stony Creek marshb | Marker Horizon-Glitter Addition | Fluvial | 41.2624 | 72.7369 | 1973 | 0.66 | High | |
| CT | Nells Island[b] | Marker Horizon-Glitter Addition | Fluvial | 41.1772 | 73.1107 | 1973 | 0.60 | High | 28.8 |
| CT | South Shore CT, 7[c] | $^{210}Pb$-CRS | Back Barrier | 41.0785 | 73.4231 | 2012 | 0.13 | Low | 37.5 |
| CT | South Shore CT, 14[c] | $^{210}Pb$-CRS | Back Barrier | 41.2689 | 72.7631 | 2012 | 0.23 | Low | 29.1 |
| CT | South Shore CT, 11[c] | $^{210}Pb$-CRS | Back Barrier | 41.1180 | 73.3250 | 2012 | 0.26 | Low | 36.6 |
| CT | South Shore CT, 12[c] | $^{210}Pb$-CRS | Fluvial | 41.2262 | 73.0354 | 2012 | 0.29 | Low | 32.8 |
| CT | South Shore CT, 9[c] | $^{210}Pb$-CRS | Back Barrier | 41.0970 | 73.3875 | 2012 | 0.30 | Low | 21.7 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | Barrier | | | | | | |
| CT | South Shore CT, 8B[c] | $^{210}Pb$-CRS | Back Barrier | 41.0765 | 73.4153 | 2012 | 0.42 | Low | 26.8 |
| CT | South Shore CT, 13[c] | $^{210}Pb$-CRS | Fluvial | 41.2353 | 72.9914 | 2012 | 0.47 | Low | 18.5 |
| CT | South Shore CT, 8A[c] | $^{210}Pb$-CRS | Back Barrier | 41.0765 | 73.4153 | 2012 | 0.49 | High | 32.9 |
| CT | Clinton, Farm River, core FR11B[d] | $^{210}Pb$ | Fluvial | 41.2667 | 72.8517 | 1980 | 0.35 | High | 30.0 |
| CT | Pataguanset tidal wetlands, Watts Island[e] | $^{14}C$ | Fluvial | 41.2996 | 72.2199 | 1981 | 0.11 | | |
| CT | Barn Island, Headquarters Marsh[f] | $^{210}Pb$ | Back Barrier | 41.3385 | 71.8719 | 1994 | 0.18 | High | 25.6 |
| CT | Barn Island, Bloom's Point In[f] | $^{210}Pb$ | Back Barrier | 41.3367 | 71.8709 | 1994 | 0.20 | High | 25.6 |
| CT | Barn Island, Bloom's Point In[f] | $^{137}Cs$ | Back Barrier | 41.3367 | 71.8709 | 1994 | 0.17 | High | 25.6 |
| CT | Barn Island, Bloom's Point In[f] | Marker Horizon-Glitter Addition | Back Barrier | 41.3367 | 71.8709 | 1994 | 0.18 | High | 25.6 |
| CT | Barn Island, Bloom's Point Out[f] | $^{137}Cs$ | Back Barrier | 41.3367 | 71.8709 | 1994 | 0.17 | High | 25.6 |
| CT | Barn Island, Davis Marsh[f] | $^{210}Pb$ | Back Barrier | 41.3355 | 71.8558 | 1994 | 0.33 | Mid | 25.6 |
| CT | Barn Island, Davis Marsh[f] | $^{137}Cs$ | Back Barrier | 41.3355 | 71.8558 | 1994 | 0.30 | Mid | 25.6 |
| CT | Hoadley Creek[g] | Marker Horizon-Clay Feldspar | Fluvial | 41.262 | 72.734 | 2007 | 0.42 | Low | 20.3 |
| CT | Hoadley Creek[g] | Marker Horizon-Clay Feldspar | Fluvial | 41.262 | 72.734 | 2007 | 0.52 | Low | 20.3 |
| CT | Sherwood Island[g] | Marker Horizon-Clay Feldspar | Back Barrier | 41.116 | 73.325 | 2007 | 0.21 | Low | 25.7 |
| CT | Hoadley Creek[g] | SET | Fluvial | 41.262 | 72.734 | 2007 | 0.08 | Low | 20.3 |
| CT | Sherwood Island[g] | SET | Back Barrier | 41.116 | 73.325 | 2007 | 0.37 | Low | 25.7 |
| CT | East Haven Salt Marsh[h] | Historic information | Fluvial | 41.2716 | 72.8652 | 1972 | 0.68 | | 29.6 |
| CT | Barn Island, Headquarters Marsh[i] | Storm Deposits | Back Barrier | 41.3385 | 71.8719 | 1988 | 0.11 | | |
| CT | Barn Island, Wequetequock Cove[i] | Storm Deposits | Back Barrier | 41.3368 | 71.8764 | 1988 | 0.23 | High | |
| CT | Barn Island, Bloom's Point In[j] | Marker Horizon- | Back | 41.3367 | 71.8709 | 1983 | 0.17 | High | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Glitter Addition | Barrier | | | | | | |
| CT | Barn Island, Bloom's Point In[f] | Storm Deposits | Back Barrier | 41.3367 | 71.8709 | 1998 | 0.22 | High | 25.6 |
| CT | Barn Island, Headquarters Marsh[k] | SET | Back Barrier | 41.3392 | 71.8719 | 2013 | 0.24 | High | |
| CT | Barn Island, Headquarters Marsh[k] | SET | Back Barrier | 41.3361 | 71.8714 | 2013 | 0.27 | High | |
| NY | Flax Pond, Site 2[l] | $^{210}$Pb | Back Barrier | 40.9631 | 73.1442 | 1973 | 0.63 | | 25.6 |
| NY | Flax Pond, Site 3[l] | $^{210}$Pb | Back Barrier | 40.9631 | 73.1442 | 1973 | 0.47 | | 16.0 |
| NY | Stony Brook Harbor[m] | $^{210}$Pb | Back Barrier | 40.9185 | 73.1588 | 2000 | 0.26 | Mid | |
| NY | Youngs Island, Stony Brook Harbor[m] | $^{210}$Pb-CRS | Back Barrier | 40.9229 | 73.1512 | 2000 | 0.46 | | |
| NY | Fresh Pond[n] | $^{210}$Pb | Back Barrier | 40.9640 | 72.7728 | 1985 | 0.43 | Low | |
| NY | Fresh Pond[n] | Opaque Spherules | Back Barrier | 40.9640 | 72.7728 | 1985 | 0.25 | Low | |
| NY | Fresh Pond[n] | Pollen | Back Barrier | 40.9640 | 72.7728 | 1985 | 0.20 | Low | |
| NY | Alley Pond[o] | $^{210}$Pb-CRS | Back Barrier | 40.7617 | 73.7511 | 1998 | 0.35 | High | 28.0 |
| NY | Hunters Island[o] | $^{210}$Pb-CRS | Back Barrier | 40.8781 | 73.7881 | 1998 | 0.11 | High | 39.3 |
| NY | Caumsett Park[o] | $^{210}$Pb-CRS | Back Barrier | 40.9392 | 73.4839 | 1998 | 0.41 | High | 42.5 |
| NY | Flax Pond[o] | $^{210}$Pb-CRS | Back Barrier | 40.9631 | 73.1442 | 1998 | 0.21 | High | 3.1 |
| NY | Goose Creek[o] | $^{210}$Pb-CRS | Fluvial | 40.8739 | 73.8183 | 1998 | 0.24 | High | 45.3 |
| NY | Flax Pond[p] | Historic information | Back Barrier | 40.9631 | 73.1442 | 1974 | 0.25 | Low | 25.0 |
| NY | South Shore NY, 4[c] | $^{210}$Pb-CRS | Back Barrier | 40.8700 | 73.8084 | 2012 | 0.10 | High | 50.6 |
| NY | South Shore NY, 5[c] | $^{210}$Pb-CRS | Back Barrier | 40.9486 | 73.7194 | 2012 | 0.16 | High | 29.6 |
| NY | South Shore NY, 3[c] | $^{210}$Pb-CRS | Back Barrier | 40.8728 | 73.8124 | 2012 | 0.19 | High | 35.0 |
| NY | South Shore NY, 6[c] | $^{210}$Pb-CRS | Fluvial | 40.9508 | 73.7000 | 2012 | 0.26 | Low | 28.6 |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| NY | Flax Pond[q] | $^{210}Pb$-CRS | Back Barrier | 40.9631 | 73.1442 | 2005 | 0.23 | Mid | 3.1 |
| NY | Nissequague, Core A[r] | $^{210}Pb$-CRS | Fluvial | 40.895 | 73.2067 | 2005 | 0.44 | Mid | 15.4 |
| NY | Nissequague, Core B[r] | $^{210}Pb$-CRS | Fluvial | 40.895 | 73.2067 | 2005 | 0.38 | Mid | 12.5 |
| NY | Flax Pond[s] | $^{210}Pb$ | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.40 | Low | |
| NY | Flax Pond[s] | $^{210}Pb$ | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.34 | Low | |
| NY | Flax Pond[s] | $^{210}Pb$ | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.30 | Low | |
| NY | Flax Pond[s] | $^{210}Pb$ | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.32 | Low | |
| NY | Flax Pond, bare mud flats[t] | Historic information | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.34 | Low | |
| NY | Flax Pond, high intertial S. alterniflora peat[t] | Marker Horizon-Brick Dust and Glitter Addition | Back Barrier | 40.9631 | 73.1442 | 1976 | 0.31 | Low | |

[a]Anisfeld et al. 1999 [b]Harrison and Bloom 1977 [c]Hill and Anisfeld 2015 [d]McCaffrey and Thompson 1980 [e]Orson et al. 1987 [f]Orson et al. 1998 [g]Anisfeld 2008 [h]Siccama and Porter 1972 [i]Warren and Niering 1993 [j]Young 1985 [k]Warren et al. 2015 [l]Armentano and Woodwell (1975) [m]Cademartori E.A. (2000) [n]Clark and Patterson 1985 [o]Cochran et al. 1998 [p]Flessa et al. (1977) [q]Kolker et al. 2005 [r]Kolker et al. 2009 [s]Muzyka (1976)[t] Richard 1978

# Appendix 3: Sediment Characteristics

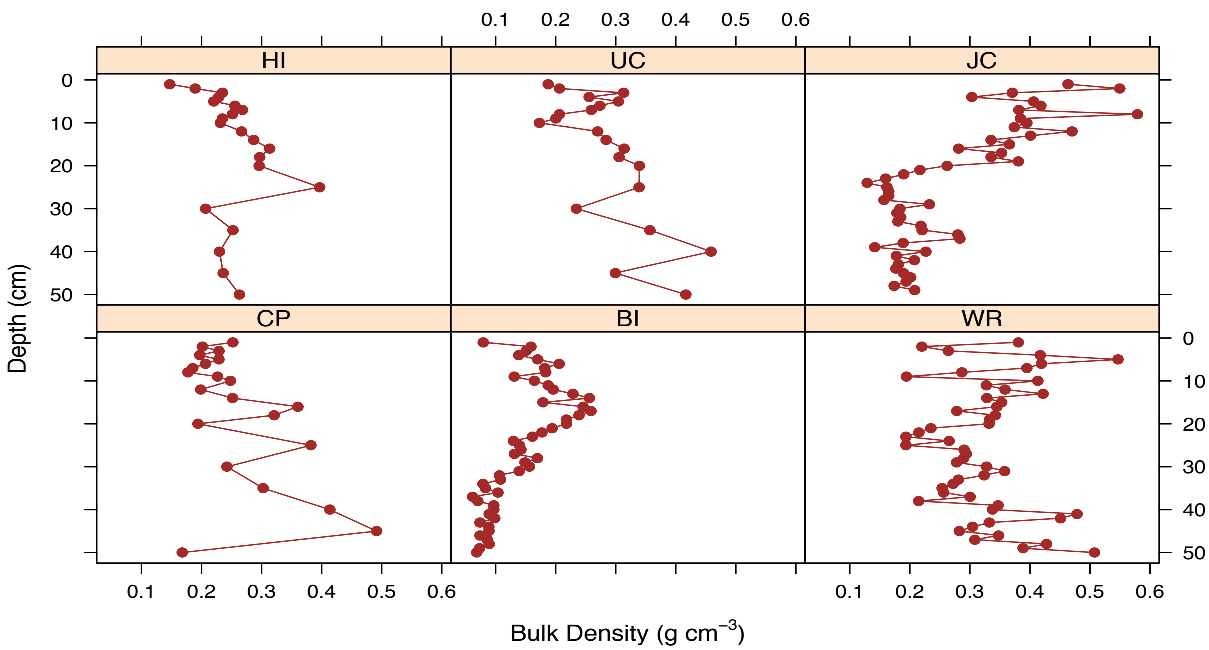



**Figure A3.1.** Depth profile of bulk density in one core from each of six sites in this study. Cores were sub-sectioned into 1 cm increments do a depth of 50 cm. Bulk density decreased in JC and BI and Increased in UC and CP. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

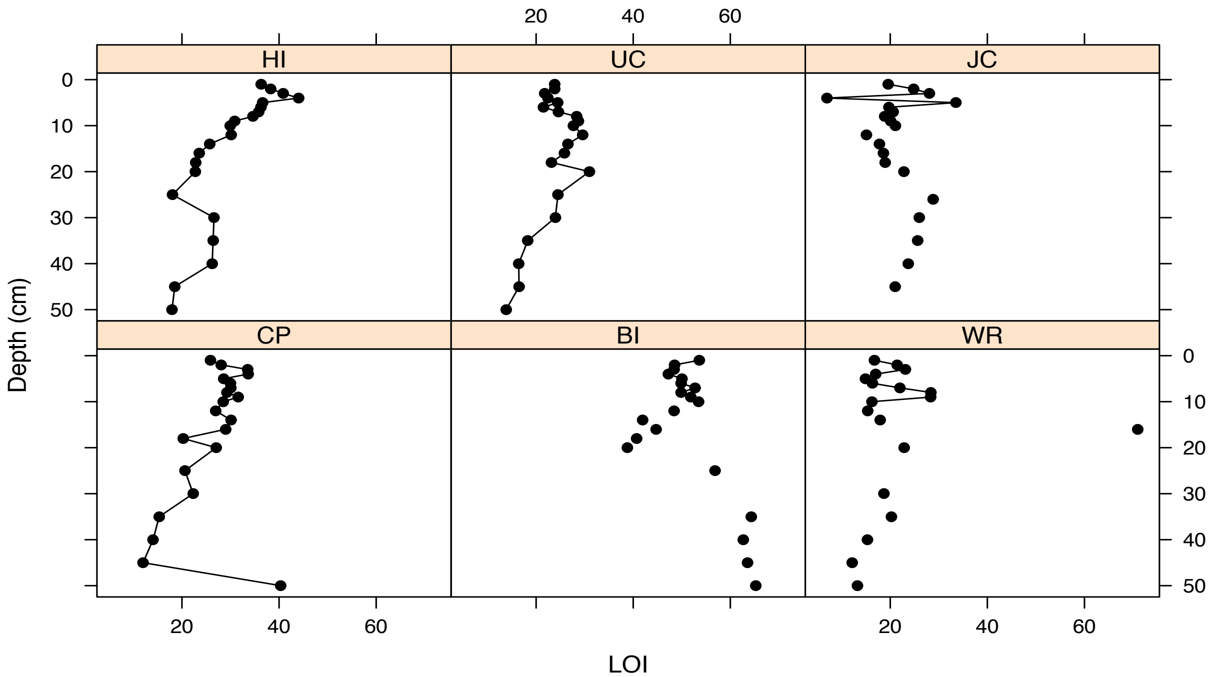



**Figure A3.2.** Depth profile of LOI in one core from each of six sites in this study. Cores were sub-sectioned at 1 cm increments to a depth of 50 cm. LOI increased with depth at UC, CP, and HI and decreased in BI. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

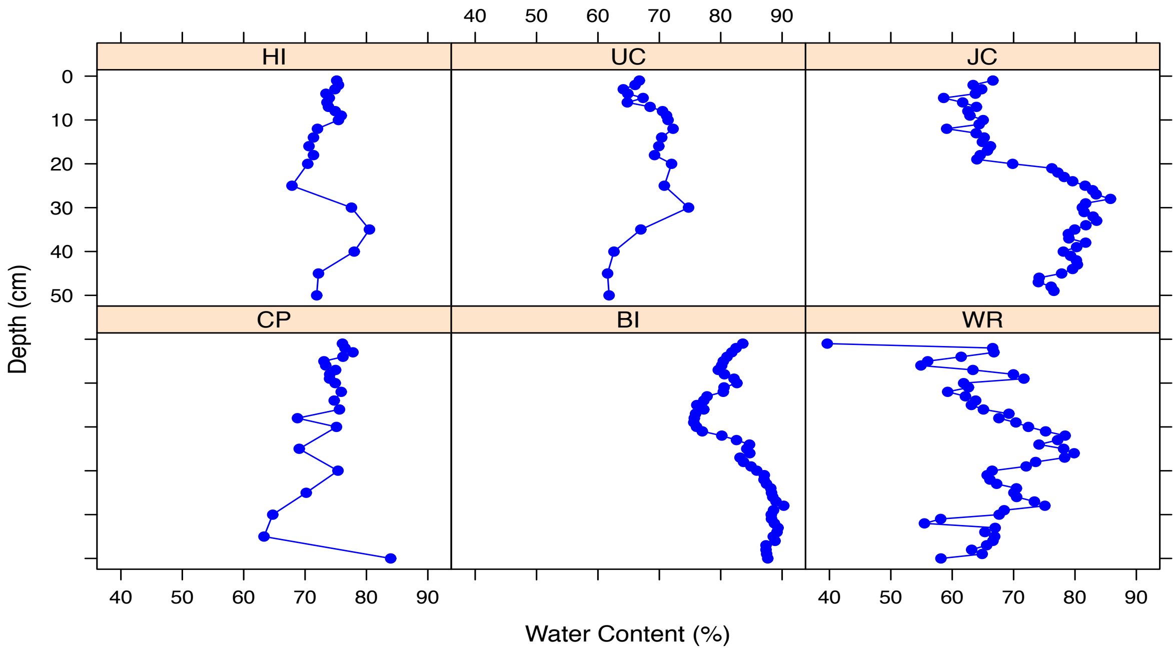



**Figure A3.3.** Depth profile of water content in one core from each of six sites in this study. Cores were sub-sectioned in 1 cm increments to a depth of 50 cm. Water content increased with depth at JC and BI. Sites are arranged from west to east across LIS (HI=Hunter Island, UC=Udalls Cove, JC=Jarvis Creek, CP=Caumsett Park, BI=Barn Island, WR=Wading River).

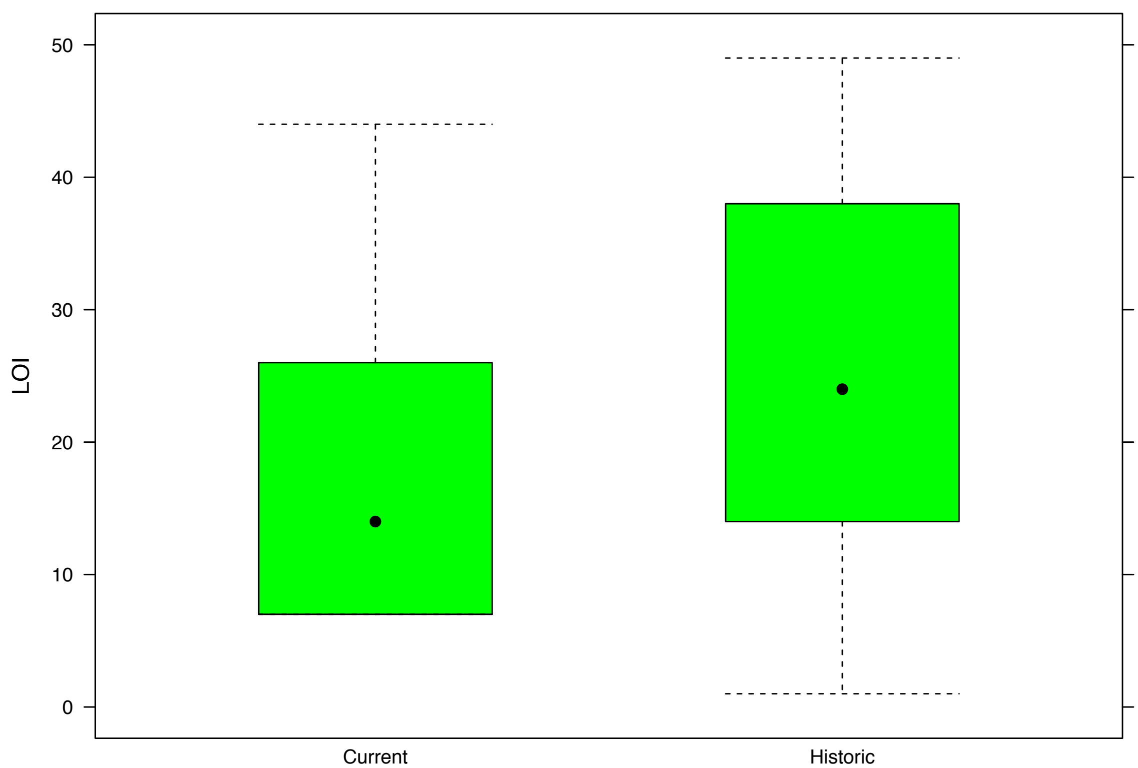
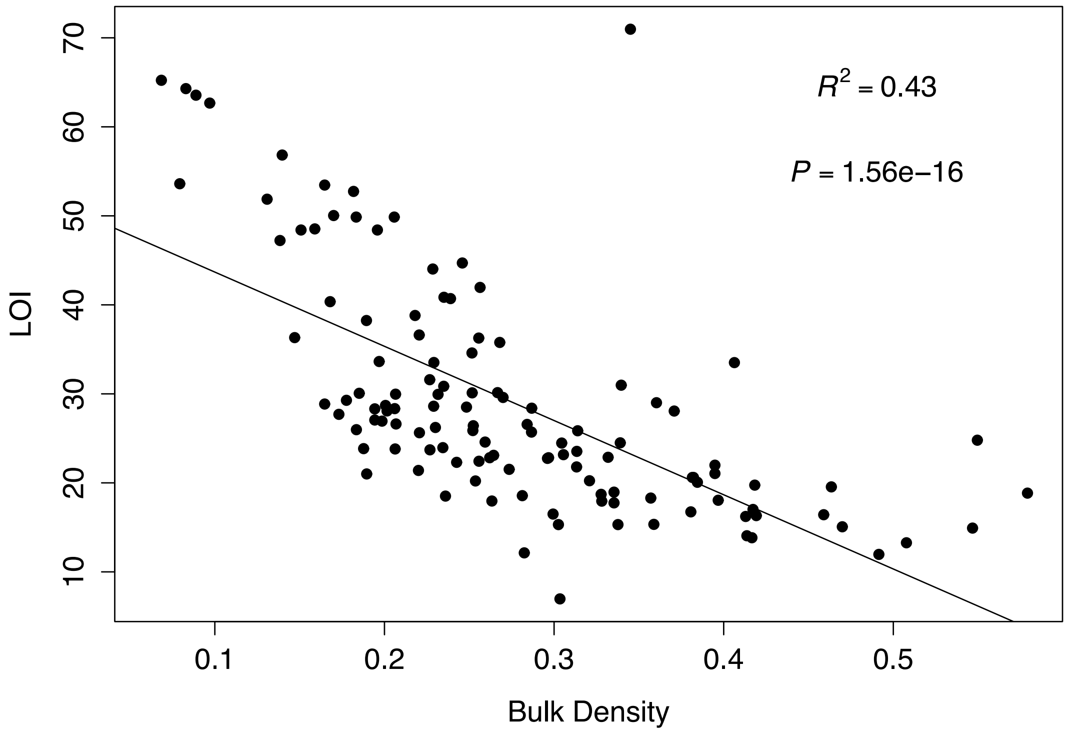



**Figure A3.4.** LOI from this study and previous studies. Shared letters indicate no significant difference. No significant difference was found.

a)




b)

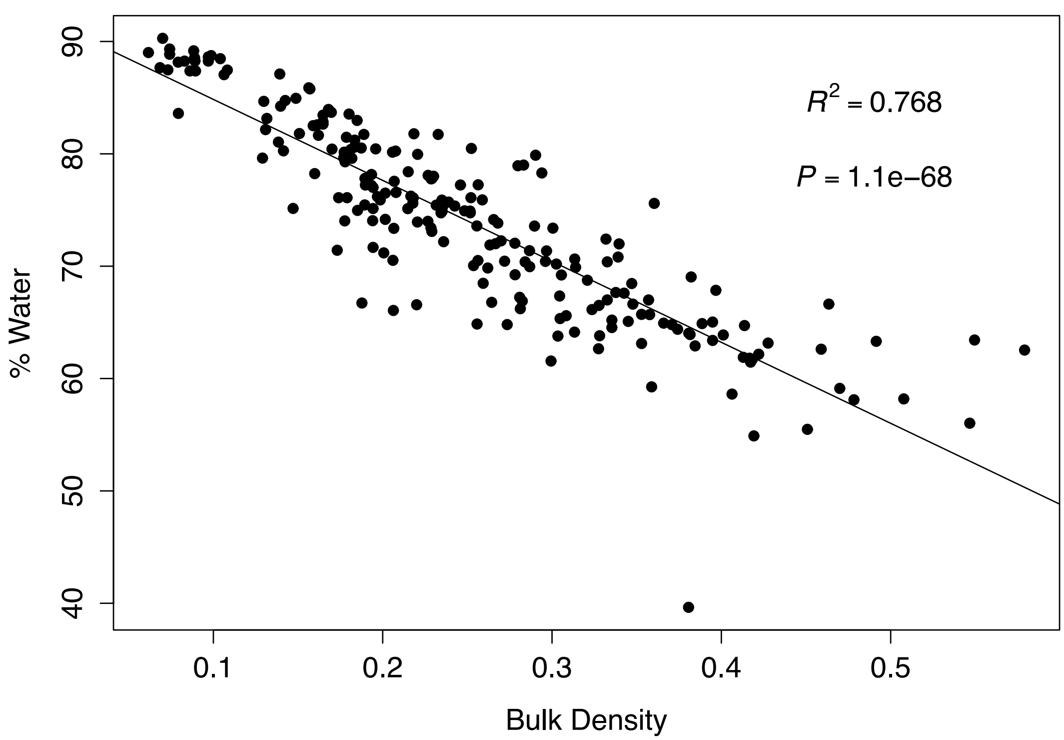



**Figure A3.5.** Relationships of a) LOI and b) water content with bulk density in sediment samples from all six of the salt marshes in this study. There is a strong decrease in LOI and water content with increasing bulk density ($R^2$=0.43, p<0.0001; $R^2$=0.77, p<0.0001).

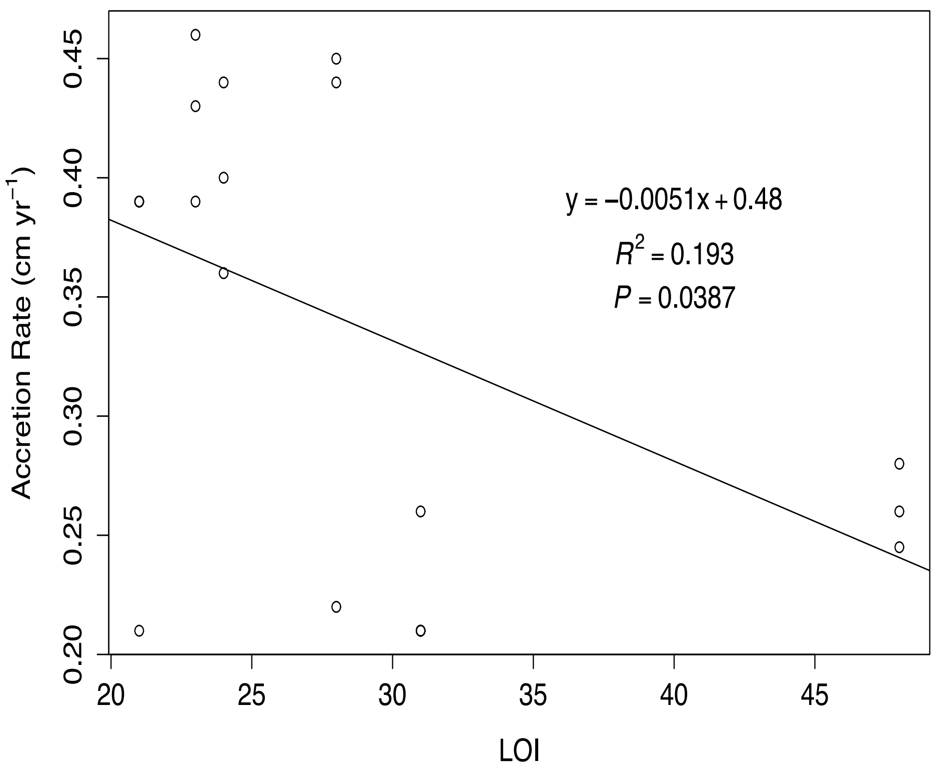



**Figure A3.6.** All Current Accretion rates and LOI. This relationship is between the $^{210}$Pb CRS, $^{210}$Pb CIC, and $^{137}$Cs accretion rates from this study and LOI from each site. There was a significant decrease in accretion rates with increased LOI ($R^2$=0.19,p=0.0387), but regressions with the accretion rates from each model individually and LOI were not significant.

# Appendix 4: Respiration Rate

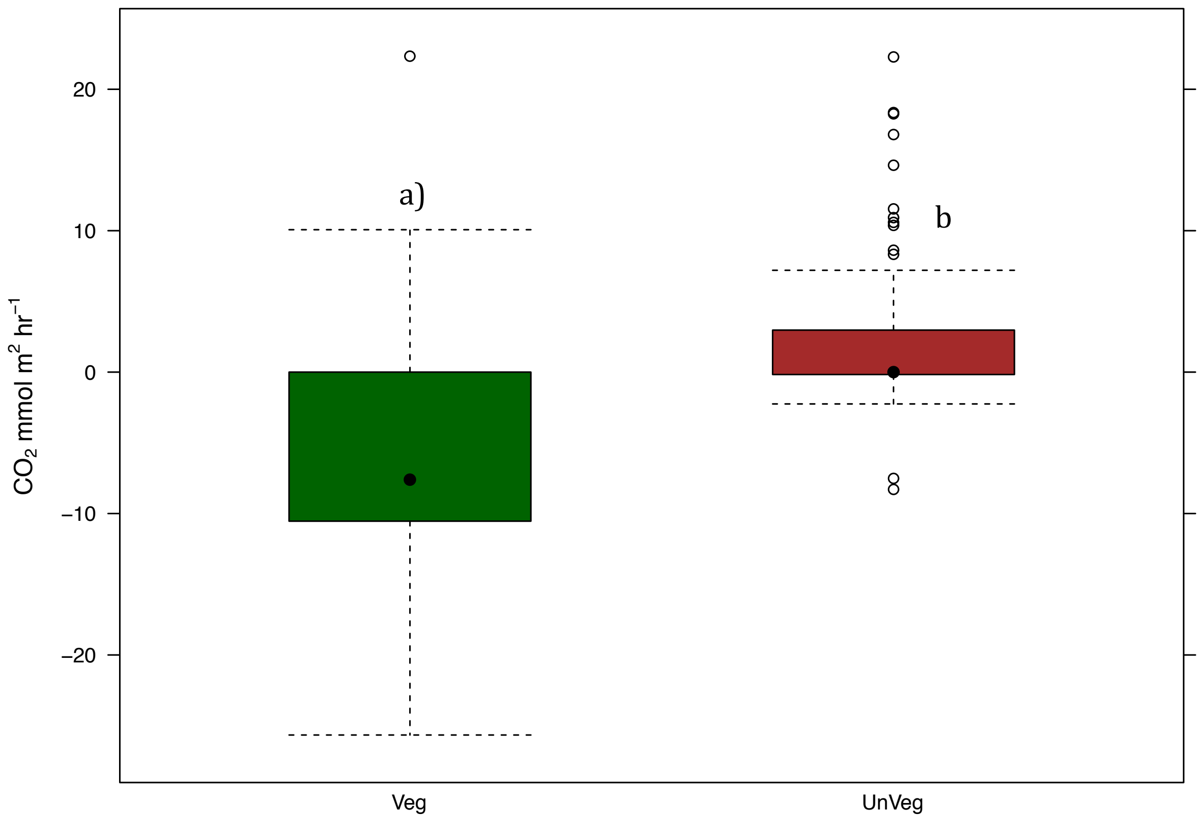



**Figure A4.1.** Respiration rates from this study by the presence of vegetation. Shared letters indicate no significant difference. Vegetated areas had significantly more negative fluxes than unvegetated areas ($p < 0.0001$; $p = 0.0011$).

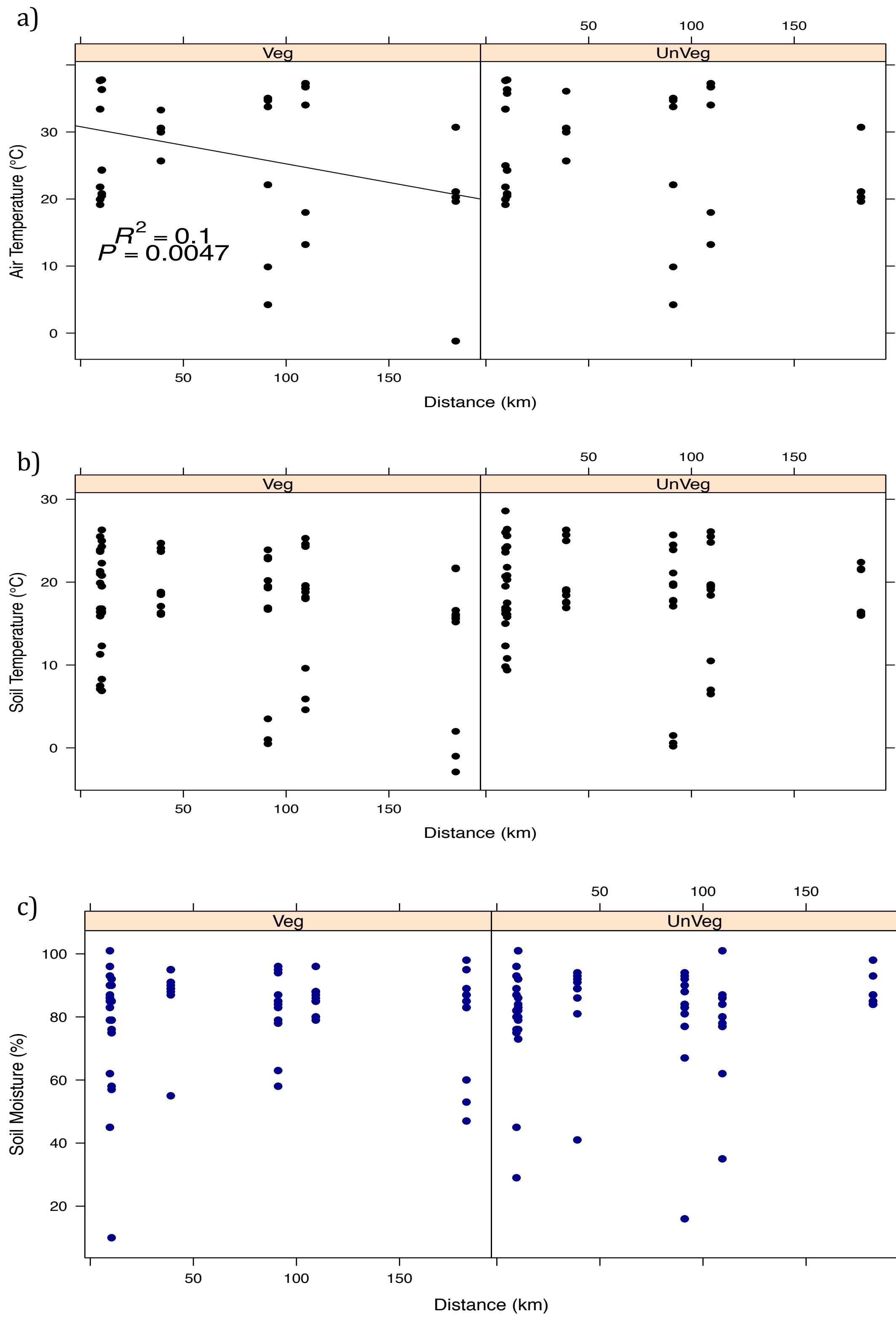



**Figure A4.2.** Air temperature, soil temperature and soil moisture by distance from the East River. A linear regression was run between distance and a) air temperature (°C) b) soil temperature (°C) and c) soil moisture (%). This was done using all sites during all seasons. There was a significant negative relationship between air temperature and distance in vegetated areas ($R^2$=0.10, p=0.0047).

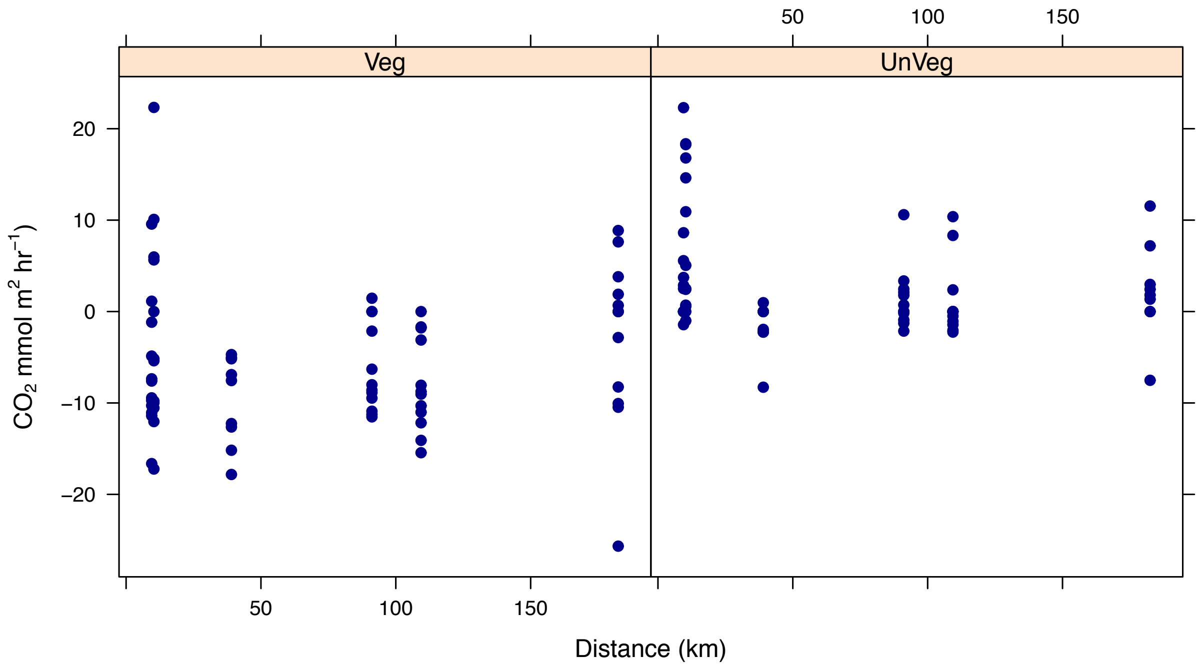



**Figure A4.3.** $CO_2$ fluxes by distance from the East River. This was done using all sites during all seasons. No significant relationships were found.