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Biogeochemical and dynamical constraints on salt marsh health on Long Island

A Thesis Presented

by

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Abstract of the Thesis

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Long Island salt marshes have been disappearing in the last several decades. This loss of marshes is likely multifactorial but several stressors have been identified as main contributors. This study focuses on two of those contributors: high pore water sulfide concentrations in salt marsh sediments and increasing rates of sea level rise. There is a clear correlation between high concentrations of sulfide and salt marsh loss. At the same time, these stressed marshes are being subjected to exponentially increasing rates of sea level rise, preventing the marshes from accreting fast enough.

Seven salt marshes have been studied in this work: four on the north shore of Long Island and three in the Peconic Bay area (east of Long Island). Pore water sulfide, ammonium, and phosphate concentrations have been measured. 210Pb data has been collected from sediment cores for the measurement of accretion chronologies and compared with known rates of average sea level rise since the beginning of the 20th century. Solid phase data has also been collected allowing the determination of the degree of pyritization of the marshes studied. The higher the degree of pyritization, the lower the capability of salt marshes to sequester sulfide and reduce its concentration in the pore water. Aerial data has also been included in this work as a comparison of salt marsh loss through the years.

Results show that most of the marshes studied exhibit very high levels of pore water sulfide in addition to barely keeping pace with rates of sea level rise or not keeping pace at all. These results are in agreement with the aerial data observed and call for action to stop the disappearance of these important ecosystems.

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List of Abbreviations

EC: East Creek

WP: West Pond

FC: Frost Creek

FP: Flax Pond

HC: Hubbard Creek

MS: Mashomack

AH: Accabonac Harbor

DOP: Degree of pyritization

Acknowledgments

Introduction

Salt marshes are essential ecosystems that greatly contribute to the biodiversity found in many coastal areas around the world. In the United States, they are found mainly along the Gulf of Mexico and the southeast coast, with South Carolina currently having the most acreage of salt marsh (Chabreck, 1988; Seabrook, 2013). The Pacific Coast of the United States has less salt marsh area due to stronger wave action and more mountainous terrain (Chabreck, 1988). Salt marshes represent a transitional zone between land and sea, and depending on their ratio of salt water to fresh water are categorized into the following groups: freshwater marshes, salt marshes, and brackish marshes. These last two classifications are also known as tidal marshes because they are present in the intertidal area between high and low tides (Butler and Weis, 2009). Salt marshes have been strongly affected by anthropogenic activities in the coastal zone, and it is the goal of this thesis to examine the possible impacts of such activities on selected geochemical processes in marshes from different areas of Long Island, New York.

The disappearance of salt marshes in Long Island and elsewhere can impact biodiversity and other ecosystem properties because marshes serve as nurseries for a multitude of species, provide food for many organisms, protect coasts from erosion, and filter nutrients and toxic substances from the water column (Joosten et al., 2002; Bromberg Gedan et al., 2009).

The area of salt marsh on Long Island has decreased since the arrival of the first colonizers. Throughout all these years, marshes have been damaged due to multiple reasons such as salt hay farming, filling for urban development (e.g. JFK airport), shoreline development, shoreline hardening, dikes and impoundments, and dredging or mosquito control (Butler and Weis, 2009).

Table 1: Marsh loss statistics

Marsh Loss- 1974/5 to 2003 (DEC data; F. Mushacke)

Marsh	% Loss (Acres) Intertidal Marsh	% Loss (Acres) High Marsh	% Gain (Acres) Intertidal Marsh	% Gain (Acres) High Marsh
Jamaica Bay	46 (734)	—	—	—
East Creek (Sands Point)	0	0	0	0
West Pond (Glen Cove)	61 (13.2)	—	—	—
Frost Creek (Lattingtown)	47 (29.3)	—	—	41 (6.4)
Flax Pond (Old Field)	25 (15.4)	10 (2.1)	—	—
Hubbard Creek	—	30 (85.7)	67 (81.6)	—
Accabonac Harbor (East Hampton)	47 (23.8)	—	0.6 (1.1)	—
Mashomack (Shelter Island)	2 (0.5)	35 (1.5)	—	—

Mosquito ditching has important negative consequences for the marshes, including lowering the marsh water table level; changing vegetation patterns, formation of marsh pools and pannes, loss of native waterfowl and wildlife habitat (Daiber, 1986). In the last decades, the rate of loss has increased. Some data showing this can be seen in Table 1 (F. Mushacke, NY DEC). There are likely different causes for this increase in salt marsh loss. One possibility is related to the increase of nutrients released into coastal waters over the same period.

Input of excess of nutrients comes from both domestic and agricultural sources. CSO's (Combined Sewer Overflow events) also play a significant role in eutrophication and can contribute to an increase of organic matter loading to coastal waters and marshes. The New York City metropolitan area (west of Long Island) releases into the waters high amounts of ammonia with concentrations ranging between 45 and 100 µM.

Moving away from the city, waters of Long Island Sound show lower ammonia concentrations, ranging between 0 and 5 µM. Something similar happens with nitrate values higher closer to the city (8-20 µM) and much lower in waters of the Sound further away from the city (0.5-8 µM) (Valiela, 1995) . An increase of organic matter in the water column leads to hypoxic or anoxic waters which reduce the exposure of sediments to dissolved oxygen. When oxygen is consumed, bacteria start to use other agents as oxidants as a means to obtain energy from the decomposition of organic matter. One of these agents is sulfate, one of the most common anions in seawater, which is reduced to sulfide. Even though sulfur is an essential macronutrient in the growth of marsh plants (Leustek et al., 1999), high concentrations of sulfide are harmful to them (Bagarinao, 1992). In effect, sulfide acts as a phytotoxin at high concentrations. There are several means of action for this phytotoxin. For example, sulfide blocks the production of energy by negatively affecting the action of cytochrome c oxidase in mitochondria (Martin and Maricle, 2015). In addition, it hinders the action of other enzymes containing metals (Koch et al., 1990; Bagarinao, 1992; Raven et al., 1997; Lamers et al., 2013). Previous research on Long Island salt marshes has implicated increased sulfidic conditions as a major factor in salt marsh decline and contraction (Hartig et al., 2002; Kolker, 2005).

In this thesis, I investigate the possible causes for salt marsh loss in seven locations on Long Island. For this reason, several parameters including pore water sulfide and ammonium concentrations, ^{210}Pb activities, and solid-phase iron and sulfur species concentrations have been measured. Pore water sulfide is a good measure of the level of stress related to sulfide suffered by salt marsh plants, and comparison of ^{210}Pb marsh accretion rates with average sea level rates is a good indicator of whether the marshes are keeping pace with sea level rise.

Background

      Marshes are exposed to the action of tides and virtually every marsh has a region of high marsh (closer to main land) and a region of low marsh. Any kind of marsh is influenced by tides, but only salt and brackish marshes are exposed to salt water from the tides. In these two cases, both high marsh and low marsh are flooded periodically, but areas of low marsh are flooded more frequently and are thus more exposed to salt water. Some areas of the low marsh are flooded most of the day. Salt marsh sediment is composed of peat, which is formed by decomposing plant matter. Peat can be defined as sedimentarily accumulated material consisting of at least 30 % (dry mass) of dead organic material (Joosten et al., 2002). Since these deposits contain a lot of decomposing organic material and often have restricted circulation, their oxygen content is low (Scott et al., 2014). Anoxia is common in the pore water of salt marsh sediments but not necessarily in the overlying water. Levels of oxygen in the water column (overlying water) depend on different factors such as temperature, eutrophication, salinity, and tidal frequency. Levels of oxygen may be naturally low in marshes not affected by anthropogenic activities during periods of low photosynthetic activity and during low tide (Vernberg et al., 2001).

      Salt and brackish marshes are characterized by salt-tolerant plants called halophytes (from the Greek, with hals meaning "salt" and phyton meaning "plant"). Some plants are more tolerant to salt than others so there is a clear division between the kinds of plants found in the high marsh and in the low marsh. High marsh areas are waterlogged only periodically, normally twice per month due to high tides related to new and full moon periods. For this reason, high marsh plants such as salt hay (Spartina patens) and spike grass (Distichlis spicata) are less tolerant to salt than plants growing in the lower marsh. The low marsh offers a smaller variety of plants, but they are able to cope with the high levels of salt; these plants are dominated mostly by smooth cordgrass (Spartina alterniflora). S. alterniflora appears in two forms, tall and short, and its decomposing leaves and roots are one of the main components of saltmarsh peat (Karleskint, 2009).

      Salt marshes are also potentially experiencing stress related to sea level rise (Nicholls et al., 1999; Craft, 2007; Craft et al., 2008). Global warming due to human activities is partly responsible for this increase. An increase in sea level causes migration of cordgrass (Spartina alterniflora) to the high marsh, and the high marsh moves inland. Many marshes have lost the capability of moving inland due to coastal development. A fast pace of increase in sea level, such as that observed over the last century, can reduce the capability of plants such as cordgrass and other plant species to adapt in areas where inland migration is still possible. This leads to the disappearance of the marshes and their conversion to mudflats if they are not able to accrete at least at the same pace as sea level rise. For this reason, this project also focuses on the comparison of accretion rates of the studied marshes with mean sea level rise. Sea level estimates have been obtained from the tide gauge records at the Battery (NYC) and Montauk (both long-term average and changes with time) over the period ~1900 to the present (Fig. 1).
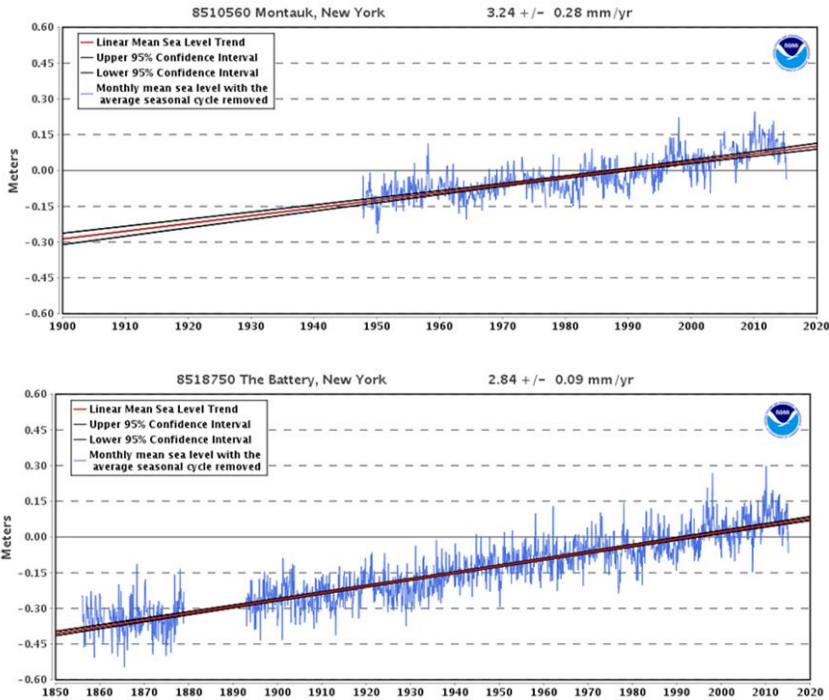
Figure 1: Sea level rise at Montauk and the Battery (NOAA)

Accretion rates can be calculated using the natural U-series radionuclide [210]Pb (half-life = 22.3 y). [210]Pb measurements can provide reliable sediment chronologies from about a hundred years ago (five half-lives) until present. The radioactive grandparent of ^{210}Pb, [226]Ra is naturally present in soils and rocks and decays to [222]Rn, which is a noble gas (half-life = 3.83 days). Due to its gaseous nature, a small percentage of [222]Rn is lost from the Earth's crust to the atmosphere. Once there, it quickly decays to several short-lived radioisotopes that eventually produce the longer-lived [210]Pb. The newly produced ^{210}Pb atoms attach to aerosol particles and are removed from the atmosphere through precipitation (rain, snow) and dry deposition. Atmospherically derived ^{210}Pb is retained in soils and marsh peat and, in the latter case, can be used as a chronometer to determine the accretion rate of the marsh surface. This atmospherically-derived ^{210}Pb is distinguished from that formed "in situ" by [226]Ra decay by subtracting the measured [226]Ra activity from the measured ^{210}Pb activity. This "unsupported" [210]Pb thus represents the ^{210}Pb that reached soils and sediments through atmospheric fallout and is called "excess" [210]Pb.

Different approaches can be taken when using ^{210}Pb as a natural clock with sediments. The model used in the present study is termed CRS (Constant-Rate- of-Supply), which assumes that the main contribution of excess ^{210}Pb to the sediment comes from atmospheric deposition and has remained constant with time. This model allows variation in accretion rates of the sediment to be determined and shows how the accretion rate of the marsh has varied over time. Details on the application of the CRS model to marshes are given in Appleby et al. (1978), Cochran et al. (1998), and Appleby (2008).

When determining ^{210}Pb chronologies in this fashion, several precautions must be taken. For example, the length of the core must be sufficient to span the entire record of ^{210}Pb decay (or about 5 half-lives). The core must also be taken carefully such that no material is lost, including the top-rooted layer of the sediment. Failure to take these measures may lead to wrong dates and accretion rates of the sediment studied (McKenzie et al., 2011).

One way of checking the validity of the data obtained from ^{210}Pb is by comparing the accretion rates with those derived from distributions of the anthropogenic radionuclide, ^{137}Cs. ^{137}Cs was produced in the atmosphere from testing of atomic (fission) weapons and as with ^{210}Pb, attached to aerosol particles and was scavenged from the atmosphere to the Earth's surface. The production and fallout of ^{137}Cs from the atmosphere was decidedly not "constant" with time; rather it peaked in 1963-64 with the signing of the Nuclear Test Ban Treaty. Thus rather than use the ^{137}Cs half-life to determine chronologies, it is the pattern of ^{137}Cs in a core that is used. Specifically, the depth of the maximum activity of ^{137}Cs is taken to represent 1963. This produces a single value for the accretion rate over the time interval 1963 to core collection. Agreement between ^{137}Cs and ^{210}Pb lends support to the validity of the ^{210}Pb chronologies, although migration of ^{137}Cs in a core is possible (Ritchie and McHenry, 1990).

Methods

a) Study sites

Seven marshes were sampled during the course of this study (Fig. 2), typically in the zone dominated by *Spartina alterniflora* with minor amounts of *Spartina patens*, thus, mid-marsh. These marsh sites may be divided into two distinct regions. The first one is located along the north shore of Long Island (Long Island Sound), from a marsh close to New York City (East Creek) to one near Stony Brook (Flax Pond). Four marshes were studied in this area; from west to east they are East Creek, West Pond, Frost Creek, and Flax Pond. The second region lies in the eastern end of Long Island in the Peconic Bay system. Three marshes were studied there: Hubbard Creek (located next to Great Peconic Bay), Mashomack, (on Shelter Island) and Accabonac Harbor, (in Gardiners Bay opposite Cartwright Island). Accabonac Harbor marshes were heavily ditched by hand during the Great Depression years with the goal of increasing mosquito control. We sampled this particular marsh in two areas: a "control" area where the ditches created more than 75 years ago have not been filled nor ditched again but left on their own, and a "treated" area where the ditches were blocked at their seaward end with the goal of trying to restore normal circulation and sedimentation in the marsh. No marshes from the south shore of Long Island have been included in this study.

Figure 2: Map of sampling sites (Map data ©2015 Google)

b) Field methods

All the samples were taken by Dr. J. Kirk Cochran and his team from 2008 to 2010. Sampling stations were established in each marsh to complement Surface Elevation Tables (SETs) installed by the NY DEC and the Nature Conservancy. Generally three SETs were installed in each marsh and pore water samples were collected near each SET emplacement. Pore water samples were taken using pore water "sippers" of different lengths (5, 10, 15, 25, and 50 cm). The sippers are hollow acrylic rods that end in a small opening. Each sipper is connected to

Tygon tubing which can be connected to a 50 ml plastic syringe. A valve connected to the Tygon tubing between the syringe and the sipper facilitates purging of the syringe. Pore water samples of ~50 ml are drawn into the syringe through the sipper. The samples are immediately filtered in the field through 0.45 μm filters. Aliquots for dissolved sulfide, nutrients and trace elements are collected. For all samples, the sulfide aliquots were fixed in the field by adding 0.5 ml of 0.05 M $Zn(C_2O_2H_3)2.2H_2O$ to each sample. Sampling was carried out in each marsh at least two times per year, spring and late summer/early fall, to attempt to capture the end-winter and end-summer seasonal temperature extremes.

c) Pore water geochemistry

Dissolved sulfide and ammonia were measured on the pore water samples using techniques described by Kolker (2005) and Cochran et al. (2013). Briefly, pore water sulfide was measured colorimetrically (Cline, 1969). Absorbances were measured at 670 nm using a spectrophotometer and sample absorbances were compared with those determined for standards of different sulfide concentrations. Standard curves had precisions of ± 1-5 %. Nitrate, nitrite (as $NO_3^- + NO_2^-$) and ammonia were measured using a Lachat Nutrient Autoanalyzer. Analytical precision was ± 5 %. Pore water pH and Eh were measured in the field with a YSI electrode. The model of YSI used is a Professional Plus 1020 handheld unit that measured pH on the total H^+ scale and used a Pt electrode for Eh. Salinity values were determined in the laboratory through measurement of chloride.

d) Solid phase geochemistry

Cores for solid phase geochemistry (sulfur and iron) and radiochemistry, ^{210}Pb, ^{226}Ra and ^{137}Cs) were taken in each marsh by carefully inserting an aluminum tube into the marsh peat.

After the tube was emplaced on the marsh surface, vertical cuts were made around the perimeter to minimize compaction as the core was inserted. Cores were taken in this manner at Accabonac Harbor (single cores in both treated and control areas), Hubbard Creek (single core), East Creek (three cores), Flax Pond (three cores), Frost Creek (single core), West Pond (single core) and Mashomack (single core). The number of a given core corresponds to the SET emplacement (and pore water sampling station) near which it was taken. Immediately after return to the laboratory, the cores were frozen. They were later defrosted only enough to permit the sediment to be extruded and then were sectioned into 1-2 cm intervals. Small aliquots of sediment were removed from each section for solid phase geochemistry and the remainder was weighed, dried, weighed again to determine water content and then ground to a powder. An aliquot of the dried sediment was ashed at 450°C to determine organic matter content through loss on ignition. The remainder was packed into small plastic containers for gamma spectrometry. Gamma peaks at 46 keV (^{210}Pb), 352 keV (^{214}Pb for 226Ra) and 991 keV (^{137}Cs) were recorded. Count rates were converted to activities (disintegrations per minute; dpm) using NIST Standard Reference Material 4350B for ^{226}Ra and $^{137}Cs.$ ^{210}Pb activities were corrected for sample self-absorption by counting a gamma source (^{241}Am, 60 keV) through each sample and

converted to dpm using a series of standards of known ^{210}Pb activity prepared at different densities. Errors on the radionuclide measurements are ± 1 standard deviation determined from the counting uncertainties on sample and background count rates.

Solid phase reactive iron was measured by leaching ~50 mg of dried sediment in 1N HCl for 24 hours at room temperature. Total Fe in the leach solution was measured colorimetrically using the ferrozine method (Stookey, 1970). Sulfur was measured in two pools -- acid volatile sulfides (AVS, generally iron monosulfides, FeS) and total sulfur, as described in Kolker (2005) and Cochran et al. (2013).

Results

a) Pore water data

Pore water sulfide (H_2S, HS^-) and ammonium (NH_4^+) profiles were measured at least twice in the marshes over a period of two years (2008 to 2010). Each marsh was sampled at three sampling stations, generally located near Surface Elevation Tables (SETs) established to track accretion and subsidence of the marsh surface over seasonal to annual time scales. Data for pore water sulfide and nitrogen species are presented in the appendix (Table A1). Figures 3-9 show depth profiles of pore water sulfide concentrations in each marsh. All the marshes, with very few exceptions, show an increase of sulfide concentrations and occasionally display a maximum at depth. At the marsh surface, there is much variability in sulfide concentrations with some marshes showing values close to zero and some other marshes showing values as high as ~2500 µM near the surface (i.e. Accabonac Harbor). In general, values of sulfide are higher at the end of the summer periods and are also higher in 2010 at the same time of the year in comparison with 2009.
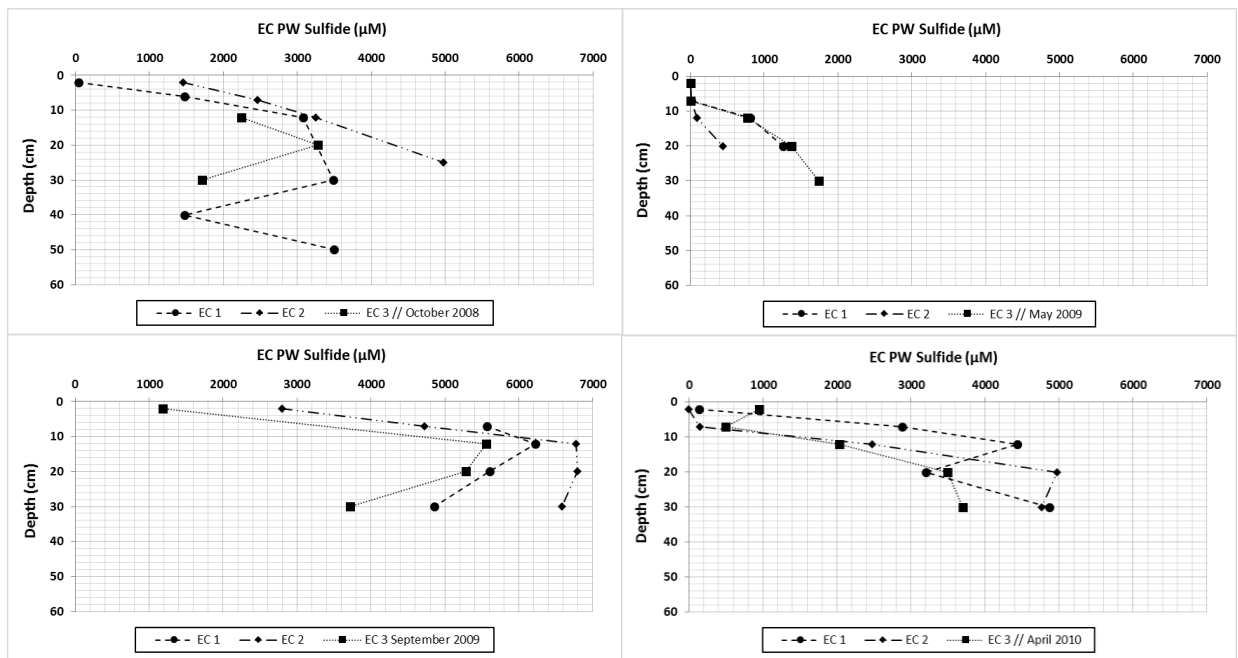
Figure 3: Pore water sulfide profiles for East Creek stations (2008 – 2010)
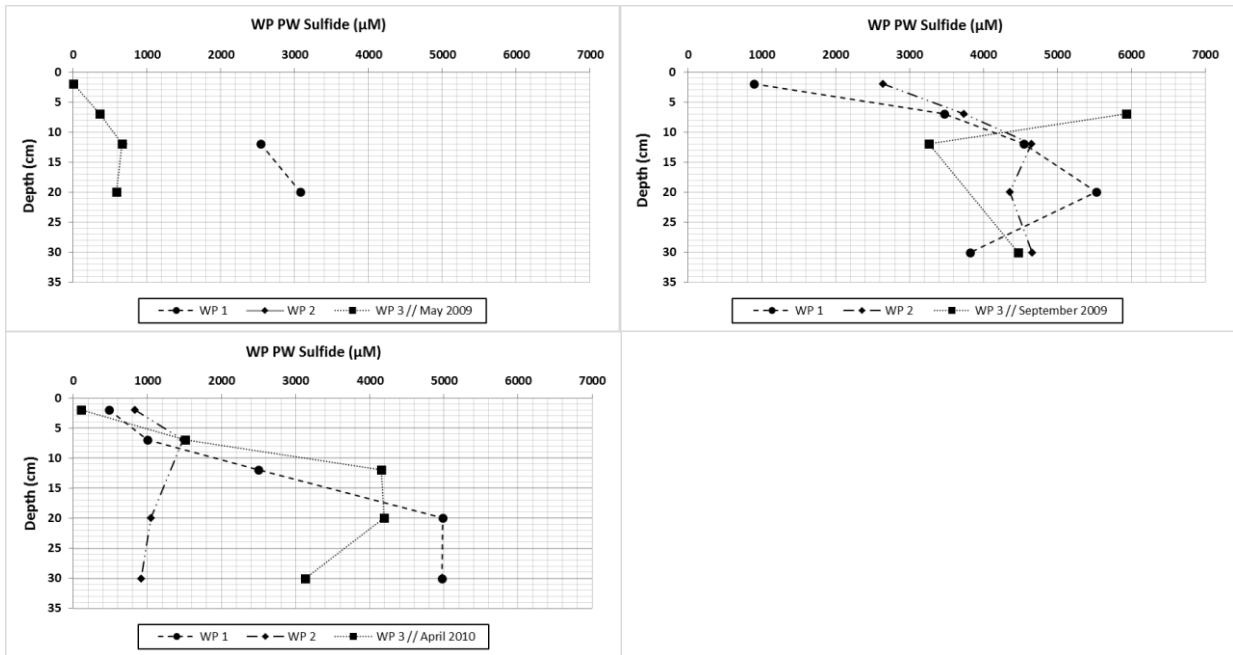
Figure 4: Pore water sulfide profiles for West Pond stations (2009 – 2010)
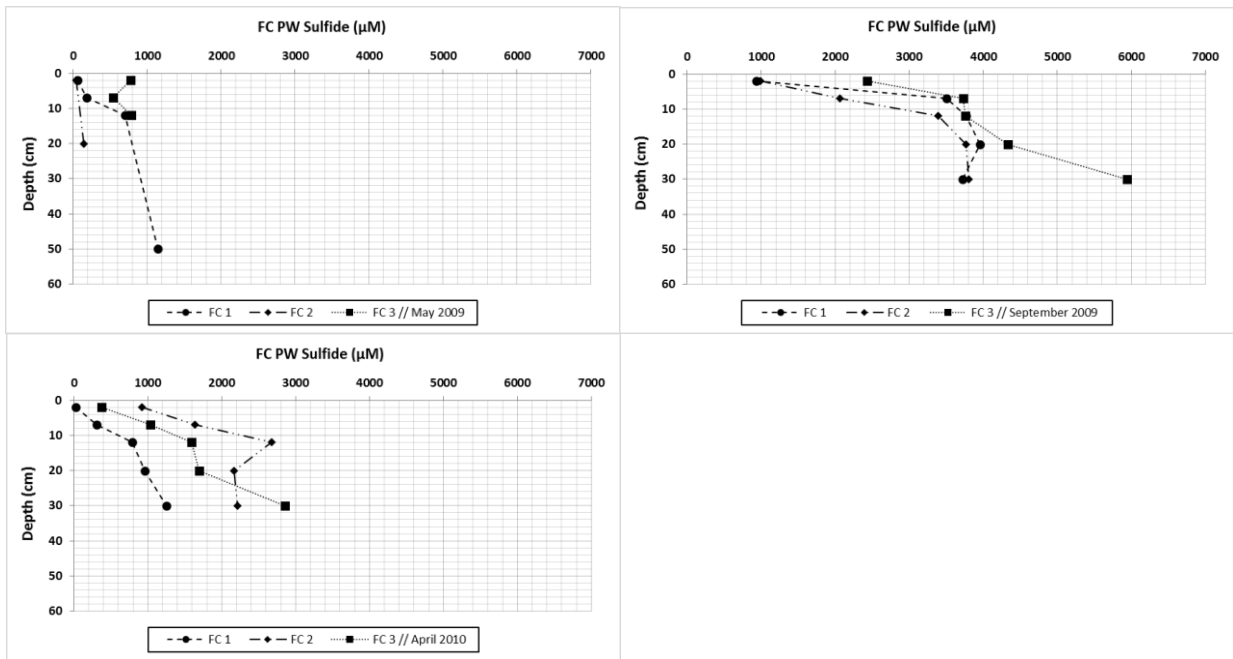
Figure 5: Pore water sulfide profiles for Frost Creek stations (2009 – 2010)
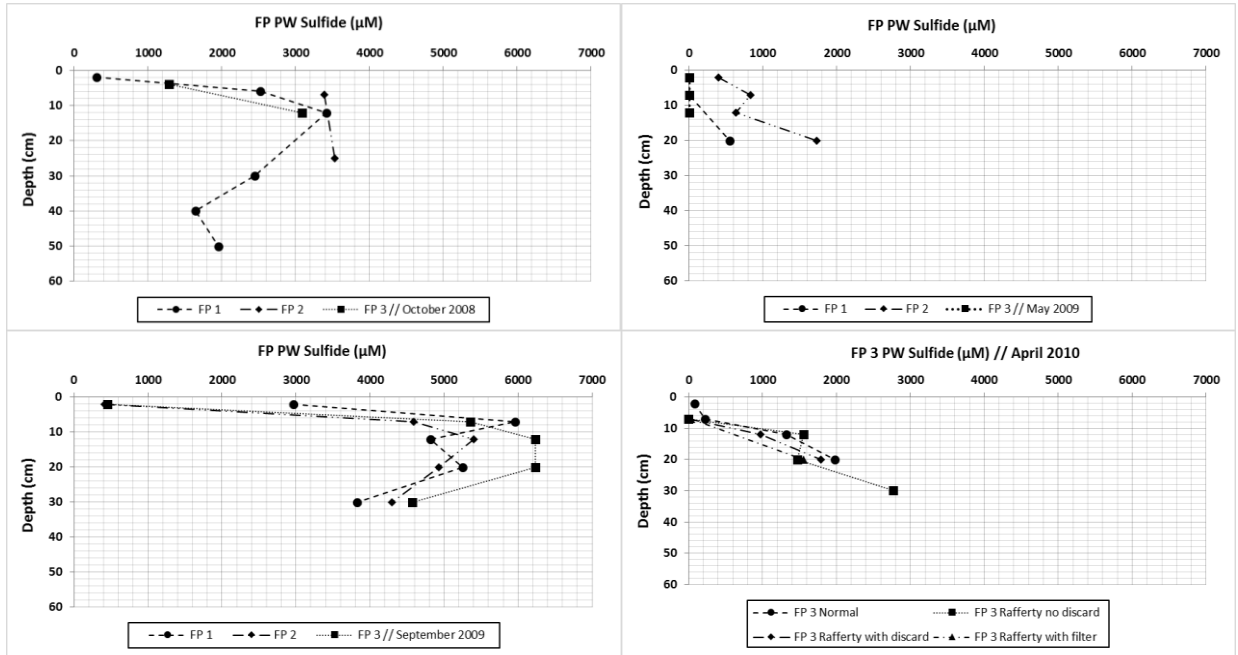
Figure 6: Pore water sulfide profiles for Flax Pond stations (2008 – 2010)
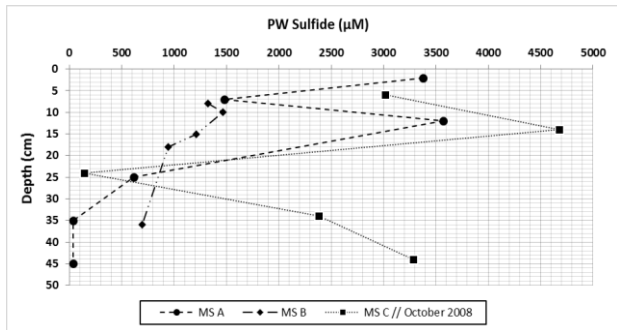
Figure 7: Pore water sulfide profile for Mashomack stations (2008)
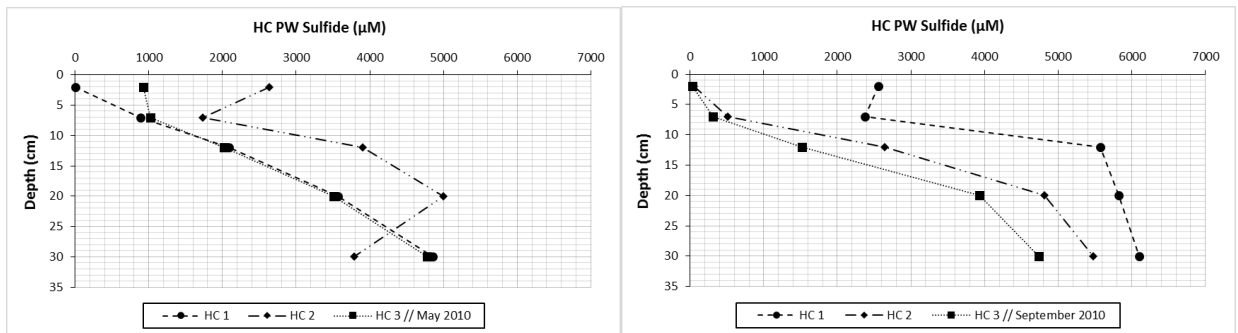
Figure 8: Pore water sulfide profiles for Hubbard Creek stations (2010)
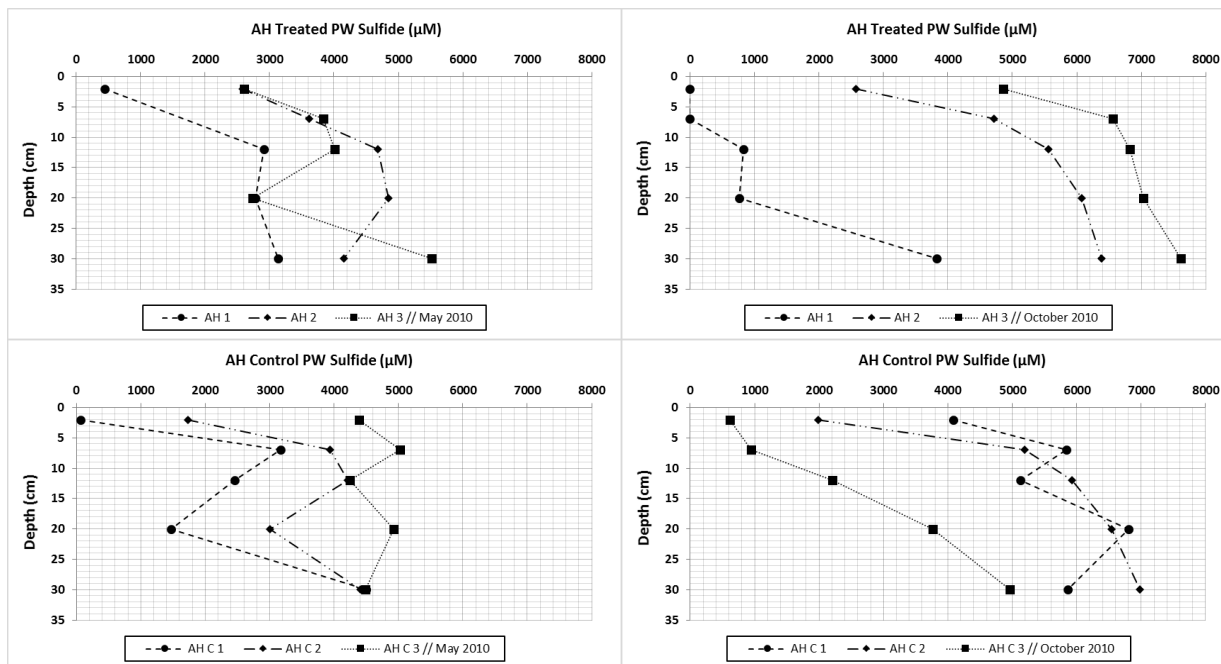
Figure 9: Pore water sulfide profiles for Accabonac Harbor stations (Treated and Control; 2010)

Concentration ranges of ammonium are relatively constant throughout the year for all the marshes and higher at all sites and times than NO_3^- concentrations (Appendix Table A1).

Chloride was measured in the marshes from the north shore of Long Island to evaluate pore water salinity. Pore water salinities ranged between 22.9 – 28 at East Creek, 21.5 – 26.3 at West Pond, 18.5 – 26.1 at Frost Creek and 23.4 – 30.0 at Flax Pond. Salinities for these marshes were measured during the months of May and September of 2009 and during April of 2010. No seasonal changes were observed. Of all the marshes, Flax Pond showed the highest salinity values (Appendix Table A1).

Pore water pH values usually ranged between 6 and 7 for all the marshes with the exception of Frost Creek and West Pond, which showed values ranging from 4.7 to 5.0 in May of 2009. E_h values were negative at all times with the exception of these two marshes when they showed low values of pH (Appendix Table A1).

Specifically, data from each marsh can be summarized as below:

East Creek

This marsh was sampled four times: in October 2008, May and September 2009, and April 2010. The highest values of pore water sulfide are found at the end of the summer period of 2009 and concentrations increase with depth to a maximum of 6800 μM at 12 cm and then decrease (Fig. 3).

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Ammonium concentrations offer little variation throughout time with values typically ranging from near zero at the surface to values as high as 400 µM at higher depths. The month of May of 2009 is the one that shows a higher range of values for ammonium concentrations (8 – 400 µM).

West Pond

Samples of this marsh were taken in May and September of 2009, and April of 2010. The general trend of pore water sulfide in this marsh is also one of increasing values with depth but some of the profiles show values that decrease slightly at depth (Fig. 4).

Ammonium concentrations for this marsh show little variation throughout the year.
One characteristic in this marsh is that in May of 2009 pH values are low (~4.75).

Frost Creek

The sampling periods for this marsh were comparable to those at West Pond (May and September 2009, and April 2010).   This marsh shows the typical pattern of increasing sulfide concentrations with depth, with higher values at the end of the summer and lower values during the spring (Fig. 5).

The range of pore water ammonium concentrations at this marsh does not change much throughout the year in comparison with the other marshes sampled.

Occasionally (e.g. Frost Creek 1 in May of 2009), the pore water in the upper few cm of sediment is characterized by low pH values, positive redox potentials and low sulfur pore water concentrations. These data suggest oxidation of solid phase sulfides.

Flax Pond

Flax Pond is a marsh closely situated to Stony Brook and is the easternmost site sampled on the Long Island north shore. Flax Pond was sampled in October 2008, May and September 2009, and April 2010. This marsh has pore water sulfide concentrations comparable to those observed at the other sites. Again, higher values are observed at the end of the summer period, reaching concentrations of almost 5500 µM for pore water sulfide (Fig. 6).

In contrast, the range of ammonium concentrations was very similar at all sampling times, with no inter-seasonal or interannual variations. However, unlike the other marsh sites, this range is narrower with values ~0 at the surface to no more than 195 µM with depth.

Mashomack

The only data available for this marsh is from October of 2008 (Fig. 7). Sulfide concentrations are quite high at the surface at each of the three sampling stations. The highest sulfide concentration observed is ~4700 µM at a depth of 14 cm. The lowest value of sulfide concentration is ~40 µM at a depth of 45 cm. With the exception of one of the sampling sites, concentrations of sulfide decrease with depth.

Ammonium concentrations range from almost zero to ~310 µM. In contrast to the general profiles of sulfide concentrations for this site, ammonium concentrations tend to decrease with depth.

Hubbard Creek

This marsh was sampled in May and September 2010, and little inter-seasonal variation is observed. Pore water sulfide values increase with depth from near-zero values at the surface to values ~6100 µM at the highest depth measured (30 cm: Fig. 8).

Ammonium concentrations also increase with depth with values near-zero at the surface to values as high as ~260 µM at higher depths. In these two periods, this marsh shows a pH range of 6 – 7. Redox values are always negative ranging between -284 and -151 in May and between -352 and -183 in September.

Accabonac Harbor

Accabonac Harbor was sampled in May and October 2010. There is not much difference in the pore water sulfide concentrations observed between the control and treated sites (Fig. 9). Sulfide concentrations in May range between ~1880 and 4300 µM and between ~2500 and 5900 µM in October.

Similarly, no inter-seasonal changes are observed at the treated site with respect to ammonium concentrations, with values ranging between 0 at the surface and ~590 µM at higher depths. The control site shows lower values in May (15 – 467 µM) in comparison with October (49 – 721 µM).

b) *Solid phase data*

Solid phase C-N-S and Fe data are shown in Table 2 for all the marshes with the exception of Mashomack. The greatest %C was found at West Pond (~19 – 32 %) and the lowest at Flax Pond (~5.9 – 12.9 %). Water content was similar in all the marshes ranging in most cases from ~70 to ~86 % and showed little variation with depth. Solid phase sulfur values were also measured for all the marshes with the exception of Mashomack. Acid volatile sulfide (AVS, FeS) values were quite similar for all the marshes with the exceptions of two marshes: Accabonac Harbor (control), where some depths show higher values than average (i.e. 4.41 µmol/g in the depth interval of 6-7 cm and 6.97 µmol/g in the depth interval 12-14 cm) and East Creek, with values ranging from 0.22 µmol/g (12-14 cm interval) to values as high as 42.3 µmol/g (6-7 cm). Total sulfur values were systematically high for Accabonac Harbor, Hubbard Creek, and East Creek (273 – 574 µmol/g) while Flax Pond, West Pond, and Frost Creek showed lower values (187 – 471 µmol/g). Leachable iron was higher in the north shore marshes with values ranging between 68 and 249 µmol/g. Leachable iron values for Hubbard Creek and Accabonac Harbor were in the range 51 – 164 µmol/g.

Table 2: Solid phase geochemical data (Collection dates are given in the radiochemical data tables; Appendix A2)

Site	Interval (cm)	% Water	LOI	AVS (µmol/g)	Fe$_{leach}$ (µmol/g)	% S	Total S (µmol/g)	FeS$_2$ (µmol/g)	% DOP
Accabonac Harbor T2	1-2	79.9		0.55	162	1.05	326	163	50.1
	6-7	83.3		0.00	155	1.54	480	240	60.7
	10-12	79		0.00	51	1.46	456	228	81.8
	16-18	83.3		0.00	141	1.46	456	228	61.7
Accabonac Harbor C2	1-2	82.7		0.13	74	1.84	574	287	79.4
	6-7	79.5		4.41	164	1.09	340	168	50.6
	10-12					1.35	422		
	12-14	82.2		6.97	161				
	18-20	75.3		0.00	145	1.15	358	179	55.3
	26-28	79.06		0.00					
	>28					0.88	273		
Hubbard Creek 2	1-2	85.4	56	0.98	113	1.48	461	230	67.0
	6-7	85.6	58	3.95	99	1.48	460	228	69.8
	10-12	85.9	52	0.93	108	1.67	520	260	70.6
	18-20	82.5	40	1.31	150				
	18-20	82.5	40	0.00	150				
	deepest					1.67	521		

Site	Interval (cm)	% Water	LOI %	AVS (µmol/g)	Fe$_{leach}$ (µmol/g)	% S	Total S (µmol/g)	FeS$_2$ (µmol/g)	% DOP
Frost Creek 1	1-2	70.5	28.8	0.39	213	0.62	193	96	31.2
	6-7	81.2	47.3	0.18	175	1.32	412	206	54.0
	10-12					1.13			
	12-14	81.3	36.9	0.16	182				
	12-14	81.3	36.9	0.036	160				
	18-20	81.2	40.7	0.32	110	1.14	356	178	61.8
	deepest					1.27			
West Pond 1	1-2	77.5	49.7	0	205	0.66	206	103	33.4
	6-7	78.5	41.7	0	125	1.18	368	184	59.5
	10-12					1.29			
	12-14	83.2	52.8	0	106				
	18-20	76.4	30.8	0	68	1.51	471	235	77.6
	26-28	80.8	31	0	112				
	deepest					1.23			
East Creek 1	2-3	82.4	51.2	24.34	249	1.33			
	6-7	80.5	50.4	42.25	186	1.34	418	188	50.2
	10-12					1.23			
	12-14	77.3	29.7		188				
	18-20	72.8	26.8	15.23	288	1.78	555	270	48.4
Flax Pond 3	1-2	74	28.1	0.17	235	0.7	218	109	31.7
	6-7	59.8	33.3	0.64	184	0.6	187	93	33.6
	10-12					1			
	12-14	73.4	24.8	0.21	196				
	18-20	75.1	22.3	0.11	193	0.88	274	137	41.5
	26-28	67.8	18.1	0.054	225				
	deepest					1.22			

c) Radionuclide data

The ^{210}Pb data are given in detail in Appendix Table A2. Hubbard Creek shows the highest ^{210}Pb activities with a maximum of 48 dpm/g. Figures 10-15 show accretion chronologies of the cores, as calculated using a constant rate-of-supply or constant flux model. This model modififes the basic radioactive decay equation to utilize integrated values of excess ^{210}Pb in the core:

$$Q = Q_0 \exp(-\lambda t) \qquad (1)$$

where Q = the inventory of excess ^{210}Pb (dpm/cm^2) below depth x in a core, Q_0 = the total inventory of excess ^{210}Pb in the core (dpm/cm^2), t is the age of depth x (y), and λ is the radioactive decay constant for ^{210}Pb (0.0311 y^{-1}). ^{210}Pb activities were measured in each

sectioned interval of the cores collected, facilitating the calculation of ages for each section interval. Accretion rates were thus determined as:

$$S = (x_n - x_{n-1})/(t_n - t_{n-1}) \qquad (2)$$

where x_n, x_{n-1} and t_n, t_{n-1} represent adjacent depths in the core (cm) and their respective ages (y) calculated from eqn 1. Errors on S were calculated from the propagated errors on the respective values of t, as described in Cochran et al. (1998). Compaction in a sediment core can alter the depth-age relationship and affect the calculated values of S. In the present case, however, water content was high and relatively constant with depth in the cores, and thus no corrections for compaction were made to the down-core accretion chronologies shown in Figs. 10-15. , Bioturbation also can affect down-core distributions of ^{210}Pb. Because of the fibrous nature of the marsh peat, bioturbation is limited relative to muddy subtidal sediments. However, burrows (for example, of fiddler crabs) can occur and care was taken to avoid burrows in collecting the cores.

One test of the applicability of the CRS model is characterizing the source of ^{210}Pb to the marsh. Previous studies (e.g. Cochran et al., 1998) have shown that the atmosphere is a dominant source of ^{210}Pb to many marshes. Indeed, inventories of excess ^{210}Pb in the marshes studied (~18 – 28 dpm/cm^2 ; Table 3) are comparable to that of the atmospheric supply of this radionuclide, ~28 dpm/cm^2 (Table 3; Graustein and Turekian, 1986; Renfro et al., 2013), supporting the use of the constant rate of supply model.

Table 3: ^{210}Pb$_{xs}$ inventories and ^{210}Pb and ^{137}Cs accretion rate

Marsh	$\sum$ ^{210}Pb (dpm/cm^2)	^{210}Pb Accretion Rates (cm/y)		^{137}Cs Accretion Rate (cm/y)
		1973-2003	*2003-2009*	*1963-2009*
Accabonac Harbor C2	22.8 ± 0.3	0.20 ± 0.13	0.40	0.18
Accabonac Harbor T2	25.7 ± 0.3	0.13 ± 0.027	0.12	0.18
Hubbard Creek 2	27.5 ± 0.4	0.22 ± 0.035	0.25	0.23
East Creek 1	18.9 ± 0.3	0.22 ± 0.049	0.25	0.28
East Creek 2	26.3 ± 0.6	0.27 ± 0.061	0.36	0.37
East Creek 2B	22.6 ± 0.4	0.25 ± 0.080	0.19	0.33
Flax Pond 1	17.9 ± 0.3	0.26 ± 0.067	0.15	0.28
Flax Pond 2	22.4 ± 0.4	0.20 ± 0.046	0.21	0.28
Flax Pond 3	28.4 ± 0.5	0.36 ± 0.18	0.23	0.41
Frost Creek 1	22.7 ± 0.4	0.21 ± 0.088	0.27	0.24
West Pond 1	21.0 ± 0.5	0.26 ± 0.076	0.19	0.33
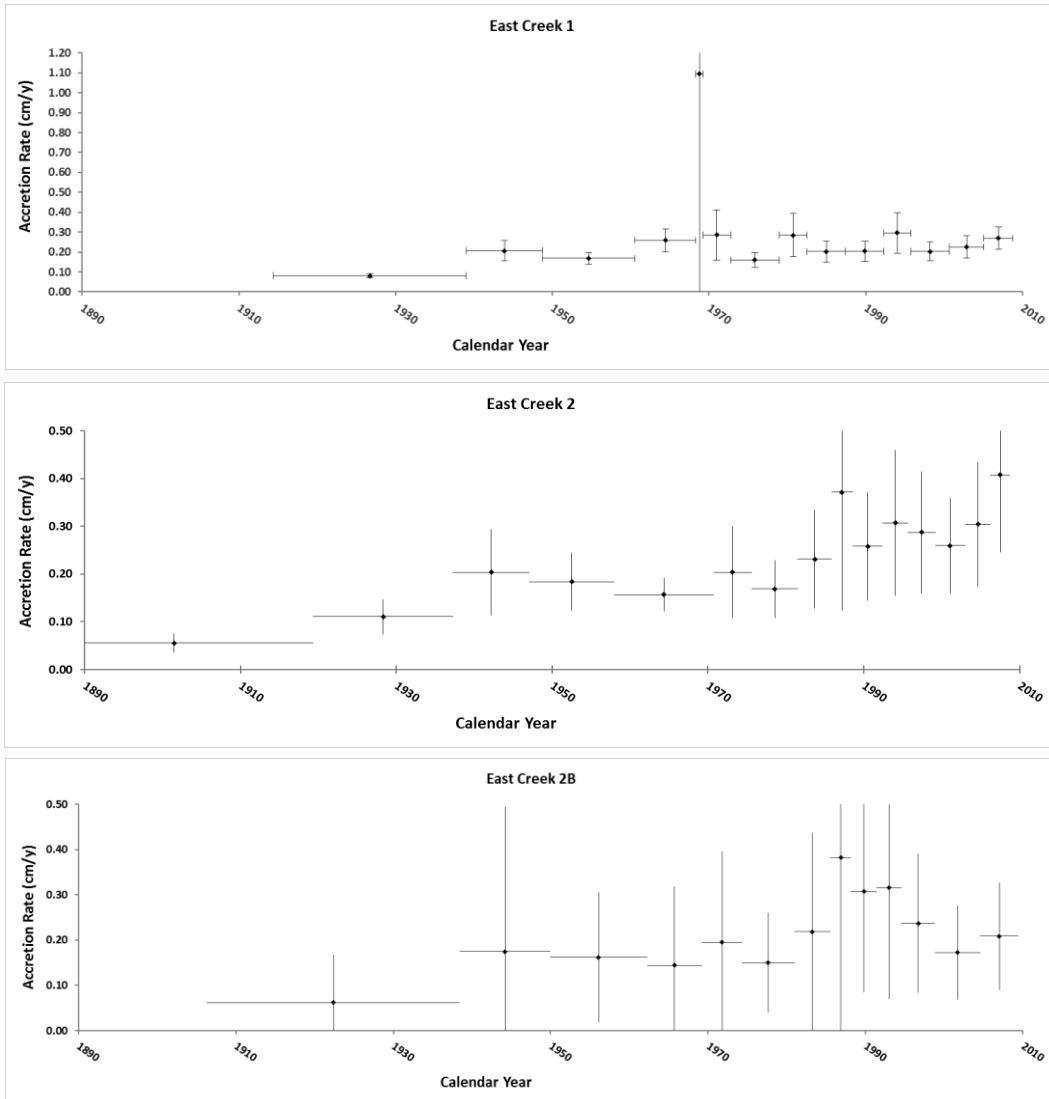
Figure 10: Marsh accretion chronologies for East Creek
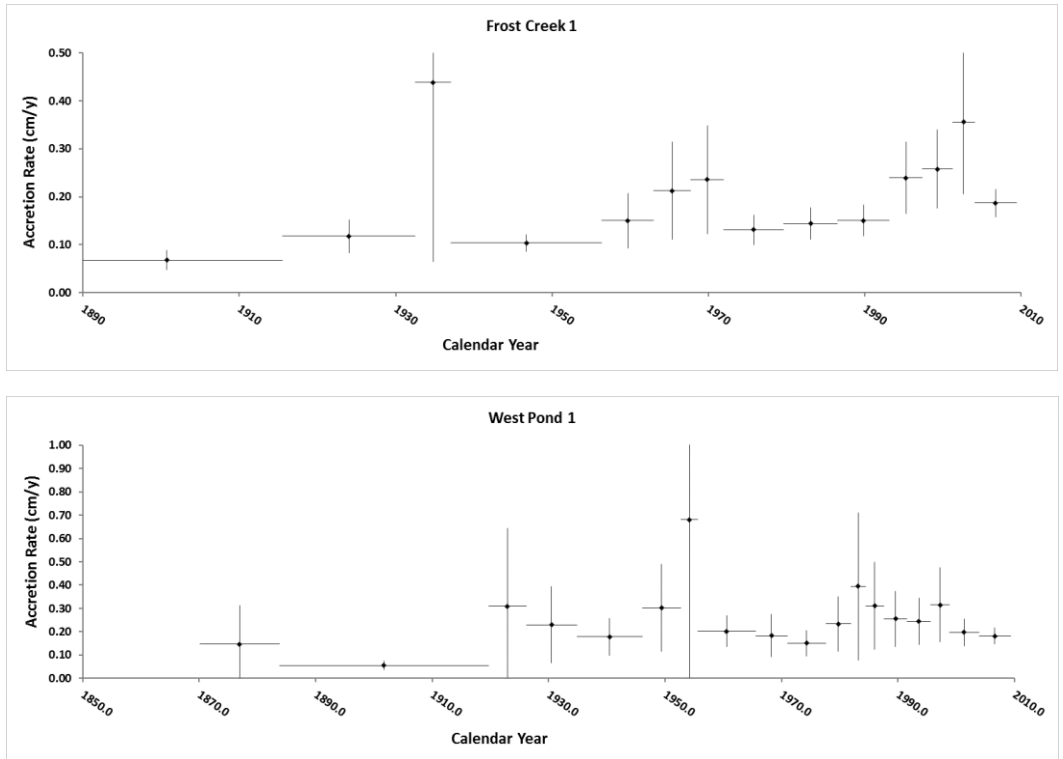
Figure 11: Marsh accretion chronologies for Frost Creek and West Pond
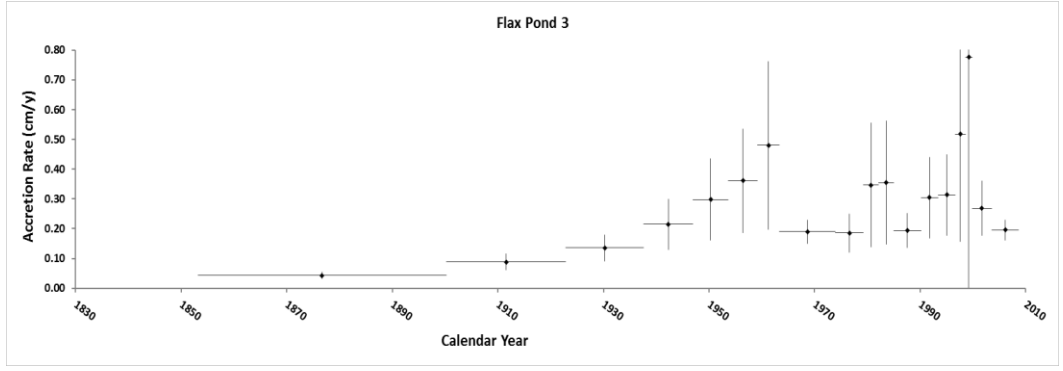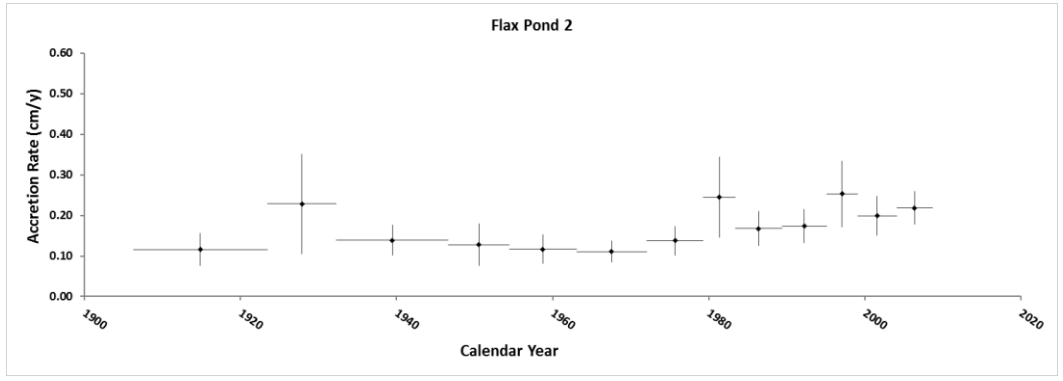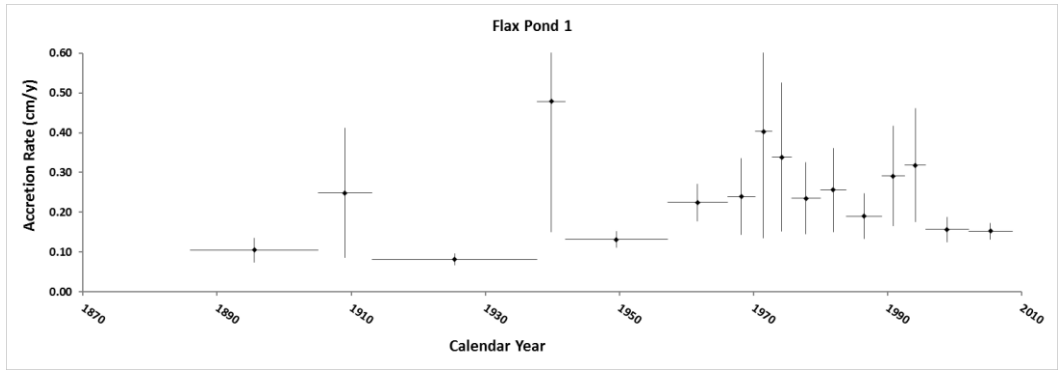
Figure 12: Marsh accretion chronologies for Flax Pond
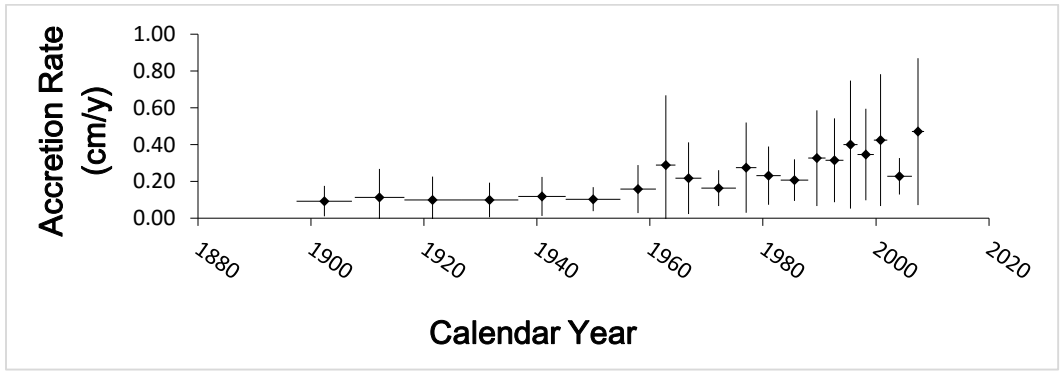
Figure 13: Marsh accretion chronology for Mashomack
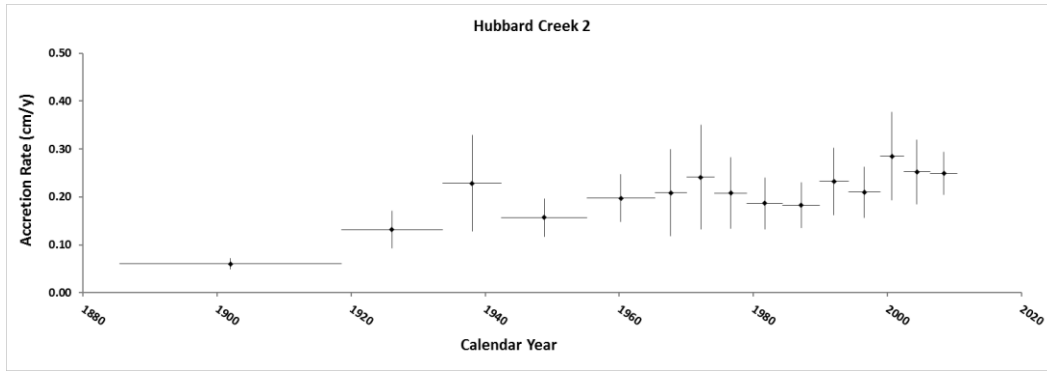
Figure 14: Marsh accretion chronologies for Hubbard Creek
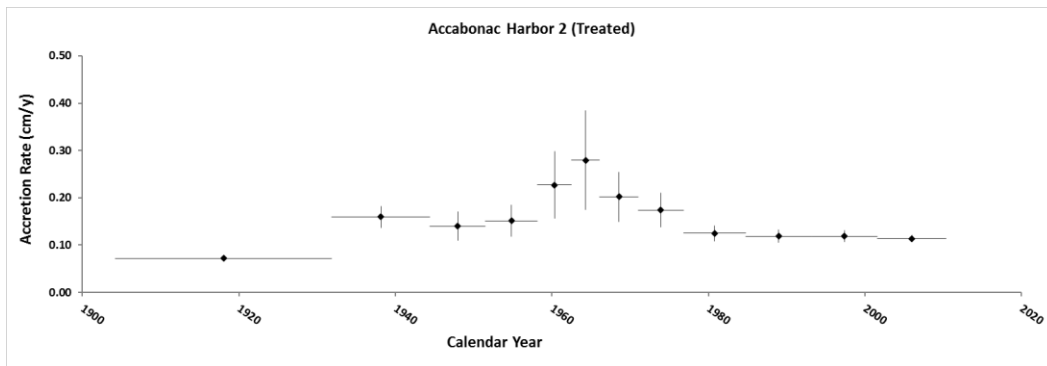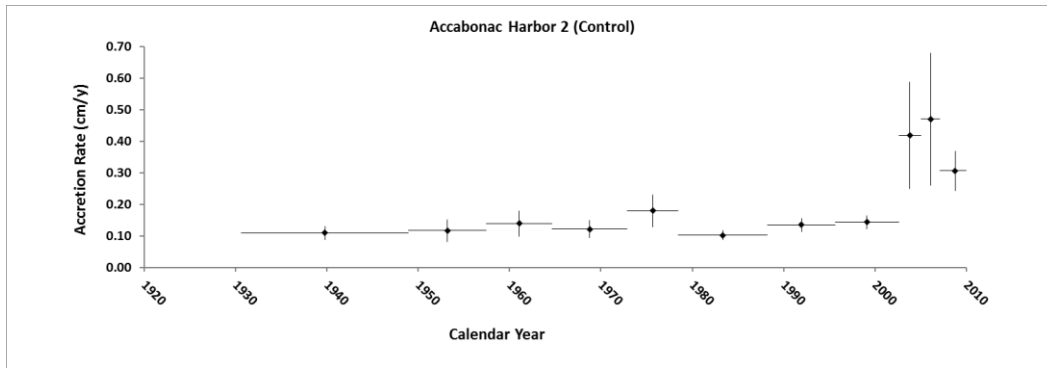
Figure 15: Marsh accretion chronologies for Accabonac Harbor

Discussion

a) *Marsh geochemistry*

a.1) *Pore water sulfide and ammonium depth profiles*

One of the goals of this study was to test Kolker's (2005) model for marsh loss in the context of increasing exposure of the root zone of the marsh to elevated concentrations of hydrogen sulfide. A variety of greenhouse studies shows that *S. alterniflora* is able to cope with low concentrations of hydrogen sulfide, but as levels rise above ~1000 µM, photosynthetic ability is affected and severe impacts including mortality can occur at concentrations ~3000 µM (Pezeshki et al., 1988; Koch and Mendelssohn, 1989; Koch et al., 1990; Seliskhar et al., 2004). We adopt the thresholds described by Kolker (2005) of stress on *S. alterniflora* at hydrogen sulfide levels from 2000 to 4000 µM and toxicity above 4000 µM. Seen against these levels, the studied marshes show levels of hydrogen sulfide typically exceeding 4000 µM in the root zone, especially in the late summer/early fall sampling.

One approach to synthesizing the pore water sulfide data is to examine the temporal variation in concentrations at depths corresponding to the root zone of the marsh plants. Measurement of pore water H_2S from low to high tide at Mashomack marsh showed no temporal change in H_2S at depths of 7 and 25 cm. (K. Cochran, pers. comm.). Davey et al. (2011) used computed tomography to examine the root distribution in marshes in Jamaica Bay and showed that living roots extended to ~20 cm. Figure 16 shows the average concentration of hydrogen sulfide in the depth 20 ± 5 cm for each marsh and plots the values as a function of sampling time. The data show a clear seasonal trend, with lower values in the spring vs late summer/early fall over the approximately 3 years of sampling (although all marshes were not sampled at all times). Concentrations frequently exceed 2000 µM and even 4000 µM in the late summer/early fall samplings, and such concentrations may cause stress and even mortality to *S. alterniflora*. The seasonal variation in hydrogen sulfide is consistent with the temperature-dependent process of bacterially mediated sulfate reduction (Howarth and Teal 1979; Hines et al. 1989; Neubauer et al. 2005).
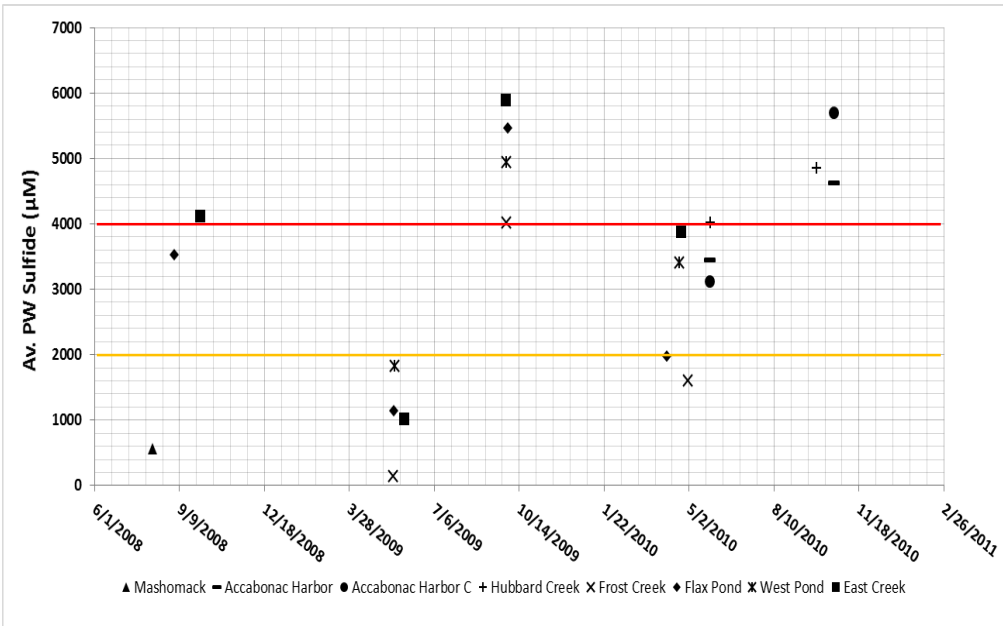
Figure 16: Pore water sulfide (20 ± 5 cm) vs. time- all sites

If hydrogen sulfide is interfering with the ability of *S. alterniflora* to take up nitrogen, one might expect a correlation between high pore water sulfide and ammonia (the dominant form of usable nitrogen in the marsh peat pore water). Figure 17 shows a weak correlation when the marshes are considered as a single population, but no clear trend within a marsh. Any correlation may be obscured by factors that affect NH_4^+ independently of sulfide, such as adsorption onto marsh sediments.
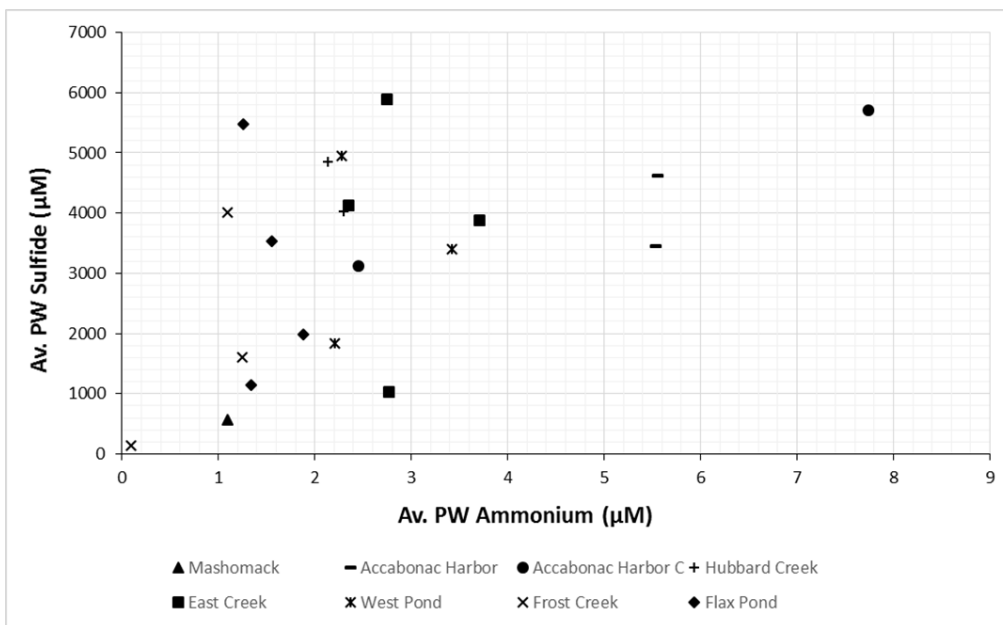
Figure 17: Average pore water sulfide vs. average pre water ammonium- all sites

Many processes are operating to control the pore water concentrations of H_2S and NH_4^+. These include: production as organic matter is remineralized, biological uptake into the marsh plants, diffusive and advective transport, and precipitation of iron sulfides and adsorption of NH_4^+ onto particle surfaces. It is beyond the goal of this study to determine the relative rates of each of these processes. Rather we use the measured pore water concentrations of H_2S as an indicator of the exposure of the marsh plants to this phytotoxin.

a.2) Solid phase iron and sulfur

Although the pore water data indicate that, at least at some times of the year, the marshes studied are exposed to concentrations of sulfide that are problematic for the plants, if such exposure is relatively brief (i.e. period of a season), the plants may be able to tolerate the exposure (Carlson and Forrest, 1982). One indicator of the extent to which high concentrations of sulfide persist in the pore water is to look at the solid phase sulfur, particularly the extent to which sulfide is present as iron sulfides. These minerals are generally iron monosulfides (FeS) and pyrite (FeS_2). Iron monosulfides are relatively reactive and are converted rapidly to pyrite in sulfidic environments (Berner, 1970; Howarth, 1979). Solid phase sulfur also may exist as elemental sulfur and associated with organic matter. We measured solid phase iron and sulfur on samples from the Hubbard Creek and Accabonac Harbor marshes and from all the marshes of the north shore of Long Island (East Creek, West Pond, Frost Creek, and Flax Pond; Table 2).

There are three main sulfur reservoirs in salt marsh sediment: acid volatile sulfide (AVS), organic sulfur, and pyrite (FeS_2). Experimentally, these reservoirs can be separated in several ways. Typically, AVS is separated by leaching sediment with 6 N HCl in an inert atmosphere and trapping the H_2S evolved in a solution of buffered zinc acetate. This procedure extracts the sulfur bound as iron monosulfides. A similar set-up involves a 12 N HCl leach of sediment in the presence of Cr (II). This procedure (termed Chromium Reducible Sulfur) extracts sulfur associated with both AVS and pyrite (Zhabina and Volkov, 1978; Canfield, Raiswell, Westrich, Reaves, and Berner, 1986; Fossing and Jørgensen, 1989). The cold CRS can also include a percentage of elemental sulfur. Finally, total sulfur can be analyzed via a CHNS analyzer. This analysis measures sulfur associated with iron sulfides as well as elemental sulfur and that associated with organic matter. It may also contain residual sulfate from evaporated pore water unless it is removed. In the present study we measured only AVS and total sulfur. Prior work with Long Island marshes (Kolker, 2005; Cochran et al., 2013) shows that the CRS pool is much larger than that of AVS, suggesting that pyrite is the dominant form of iron sulfide in the marsh peat. Although we cannot quantify the mount of organosulfur in the samples, we note that in the studies cited above, the total sulfur was commonly equal to (or even somewhat less than) the CRS. Kolker (2005) measured AVS, CRS, and total sulfur in two marshes (Nissequogue River, north shore of Long Island) and Big Egg (Jamaica Bay), and his results show that the organo-sulfur fraction ranges from 0 to 38 %, but is typically < 25 %. Here we assume that [total sulfur] – [AVS] is a measure of pyrite sulfur. Support for this assumption can be found in previous measurements of organic sulfur in marshes like the Great Sippewissett Salt Marsh in

Falmouth, Massachusetts (Howarth et al., 1983) or by looking at previous studies made in Long Island marshes such as Flax Pond. In these studies, organo-sulfur compounds were produced very slowly and the formation of other compounds was favored (Swider et al., 1989). However, this observation is clearly in contrast with organic sulfur contents found in other marshes of the East coast of the United States such as the Great Marsh of Delaware (Luther et al., 1986; Ferdelman et al., 1991), the Belle Baruch marsh system of Goat Island and Oyster Landing in South Carolina (King, 1988), or salt marshes from the Louisiana Barataria Basin where organic sulfur contents comprise between 64 and 87 % of the total sulfur (Krairapanond et al., 1991; Krairapanond et al., 1992).

AVS concentrations are very low for all the marshes with concentrations varying in the sequence East Creek > Accabonac Harbor (control) > Hubbard Creek > Flax Pond > Accabonac Harbor (treated) > Frost Creek > West Pond. Total sulfur values are quite high for East Creek, Accabonac Harbor and Hubbard Creek (273 – 574 µmol/g), and are lower in Frost Creek, West Pond, and Flax Pond (187 – 471 µmol/g).

The degree of pyritization (DOP) is a useful parameter to monitor sulfur dynamics in the sediment. This parameter is usually expressed as a fraction. The higher the DOP, the higher the amount of reactive iron that is bound to sulfur in the sediment (mostly as FeS_2) and the lower the capability of the sediment for sequestering more sulfur. In turn, this can lead to higher concentrations of sulfide accumulating in the pore water when sulfate reduction is active. In other words, the DOP is a measure of how able the marsh is to remove sulfide from the pure water geochemically through iron sulfide precipitation. High values of DOP indicate that most of the reactive iron is tied up as pyrite and is unable to effectively "titrate" additional production of hydrogen sulfide.

The degree of pyritization (DOP) is expressed as follows:

$$DOP = \frac{Fe_{Pyrite}}{Fe_{Pyrite} + Fe_{Leach}} \qquad (3)$$

where

$$Fe_{pyrite} = \frac{(Total\ Sulfur) - AVS}{2} \qquad (4)$$

and Fe_{Leach} is the amount of Fe leached in 1 N HCl for 24 hrs.

In general, the marshes of the Peconic Bay system show higher degrees of pyritization than the marshes on the north shore of Long Island (Table 2). Of this latter group, West Pond shows a higher degree of pyritization followed by Frost Creek, East Creek, and Flax Pond. On the east end of Long Island, the treated area of Accabonac Harbor has a higher a degree of pyritization than the control area. Figure 18 shows the pore water sulfide concentration versus percent of degree of pyritization for some of these marshes. As expected, the higher the degree of pyritization, the higher the concentration of sulfide in the pore water. One notable exception is the control area of Accabonac Harbor (sampling point 2) which shows the opposite behavior. This may be due to the fact that the health of Accabonac Harbor is the poorest of all the marshes studied. The concentration of sulfide in the pore water may be influenced more strongly by factors other than the degree of pyritization.
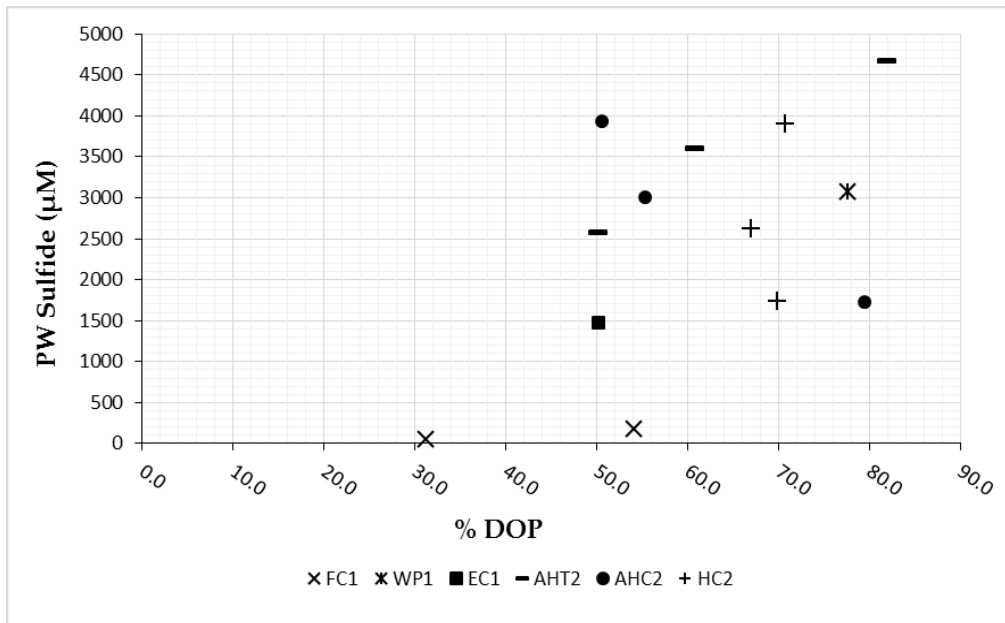
Figure 18. Pore water sulfide versus percent degree of pyritization

b) *Marsh accretion*

In order to be able to survive, marshes must accrete at a rate equal or higher than the rate of sea level rise. The global rate of sea level rise is increasing and marshes may have difficulty keeping pace with this increase. Marsh accretion depends on several factors such as the amount of inorganic sediment and organic debris available for accretion on the marsh surface, and the growth and accumulation of marsh roots on the subsurface and some other factors. All these factors are controlled by the hydrology of the marshes. In addition, marshes may experience episodes of subsidence (due to compaction for example). A marsh stressed by a phytotoxin like sulfide may experience increased difficulty keeping pace with sea level rise. Comparing the average accretion rates of the marshes studied with the average sea level rise at the Battery (0.28 cm/y, 1856 - 2014) and Montauk (0.32 cm/y, 1947 – 2014) provides a means of determining how

well the marshes are doing with respect to sea level rise. On this basis, Mashomack, East Creek, Hubbard Creek, and Frost Creek seem to be able to keep pace with sea level rise. West Pond, and Flax Pond are in a more uncertain situation and Accabonac Harbor is clearly well below rates of sea level increase. The ^{210}Pb-derived accretion chronologies permit more detailed examination of a marsh's ability to keep pace with sea level. For example, Table 3 lists average accretion rates for both 1973-2003 and 2003 – 2009 for the marshes studied. The former dates were chosen because they capture the period over which the marsh loss statistics in Table 1 were compiled. Although the post-2003 accretion rates are in many cases similar to those of the 1973-2003 period, in some cases, declines are evident (e.g. at sites in East Creek, Flax Pond and West Pond). Results from SET emplacements, which provide accretion rates over seasonal to interannual time scales, may help resolve these trends.

Some marshes, such as Accabonac Harbor, seem to have had low accretion rates in comparison with average sea level rise rates from dates as far back as 1900, leading us to question why they have not drowned before. There are different possibilities. On one hand, coastal development was not so aggressive in the first half of the 20th century in many of these areas, thus allowing salt marshes to migrate upland more easily. On the other hand, the cores taken for the measurement of the accretion rates were from mid- to low-marshh areas. Because sea level has increased since the last glaciation, areas that are today low marsh may have been high marsh a hundred years ago. Thus, a transition from high to low marsh may explain why these marshes continue to exist.

c) *Comparison with other East Coast marshes*

One of the common characteristics for the marshes of the east coast of the United States is that smooth cordgrass (*Spartina alterniflora*) is the dominant plant in the lower part of the marsh and the middle and high marsh areas are dominated by plants such as salt meadow hay (*Spartina patens*) and saltgrass or spikegrass (*Distichlis spicata*). It is important to note that, although the roots of the marsh plants live all year, the above-ground parts of the plants die back in the fall in the mid-Atlantic and New England marshes. In southern marshes, the plants are growing all year round (Butler and Weis, 2009). There are also some other factors to take into account moving from north to south along the east coast of the United States that explain some of the differences observed in the characteristic features of each marsh region. From north to south, thermal forcing (higher temperatures), high-marsh soil salinity (due to more evaporation), the role of salinity in marsh plant zonation, occurrence of salt pans, the role of positive feedbacks, and salt tolerance of plants all increase in the marshes (Bertness and Pennings, 2000).

Northeast salt marshes have been heavily altered by humans since colonial times. This alteration is directly linked to the fact that the northeast coast of the United States has been and remains a heavily populated area (Gosselink and Baumann, 1980). Indeed, it was not until the 1960s that legislation meant to protect coastal wetlands was passed.

On Long Island, salt marshes are found on both the north and south shores and in the Peconic estuary (eastern end). The north shore borders the Long Island Sound which offers protection and a more ideal environment for the development of salt marshes. The south shore is open to the Atlantic Ocean but its several barrier islands offer protection for the sand beaches on the oceanfront and for the development of salt marshes. Tidal ranges may be quite different depending on the area. For example, tidal range varies from 0.2 m in Bayshore in the south shore, to intermediate values of 0.7 m near Montauk to 1.3 m in western Long Island Sound. Areas with larger tidal range tend to be dominated by the tall form of smooth cordgrass (*Spartina alterniflora*) and areas with a smaller tidal range favor the short form of *Spartina alterniflora* and to have a more extensive high marsh area (data taken from the Department of Environmental Conservation).

In comparison with other marsh systems of the U.S. northeast coast such as the Great Sippewissett Marsh in Falmouth, Massachusetts and the Great Marsh of Delaware, the marshes of Long Island are present in multiple sites of different aerial extent, but are generally smaller than the Sippewissett Marsh and the Great Marsh of Delaware. For example, the Great Sippewissett Marsh covers an area of about 500 acres (Traver, 2006), it is located behind two barrier beaches and has not been extensively ditched. The Great Marsh of Delaware is even larger, ~2000 acres, but has been heavily ditched for mosquito control (Meredith et al. 1982).

Marshes of New England such as Sippewissett are often characterized by organic-rich peat substrates as a consequence of basins with a small drainage capacity and relatively low loads of suspended sediment (Chapman, 1960). In the southern U.S., drainage basins are often larger, which increases the sediment load and leads to more extensive marshes with higher inorganic sediment contents (Roman et al. 2000).

Summary and conclusions

In summary, aerial data confirms that many of Long Island marshes are experiencing substantial marsh loss (Table 1). Pore water hydrogen sulfide concentrations in Long Island marshes commonly exceed a threshold value of 4000 µM in the late summer/early fall periods and this may be a contributing factor to marsh loss. Some marshes studied (e.g. Accabonac Harbor, Hubbard Creek) show high degrees of pyritization suggesting long-term exposure to elevated sulfide and possibly limited ability to cope with seasonal high levels of sulfide. With the exception of the Accabonac Harbor marsh sites, the studied marshes generally have accreted to keep pace with sea level (1973-2003) but may be showing decreasing ability to do so since 2003.

An important rationale for this study was to determine whether geochemical or dynamical (accretion) characteristics of a given marsh could help explain the marsh loss statistics in Table 1. East Creek was chosen as a "control" site because, in the period 1974/5 to 2003, it showed no loss of marsh area. However, on three of the four sampling times, East Creek showed pore water hydrogen sulfide ~4000 µM or greater. However, its degree of pyritization was toward the low end of the range, suggesting that reactive iron was available to precipitate the dissolved sulfide, and its accretion chronologies show it to be keeping pace with sea level rise. In contrast, Accabonac Harbor has experienced considerable loss of intertidal marsh and showed relatively high hydrogen sulfide (>3000 µM) on both late spring and summer samplings. Moreover, the accretion chronologies show accretion rates that are clearly below the rate of sea level rise. Heavy ditching of the marsh for mosquito control may have played a role in depressing the rates of marsh accretion. Taken together, these data suggest that loss of the Accabonac Harbor marsh is likely to continue. Further monitoring of both pore water geochemistry and accretion rates (via SET) is recommended.

Final thoughts

North America has lost about 70 percent of salt marsh since the arrival of the European colonizers (Traver, 2006). Long Island also has experienced significant marsh loss and this loss has increased since the beginning of 20th century and especially in the last 50 years. Eutrophication of coastal waters may be a factor contributing to marsh loss. Stressed marshes may have increasing difficulty keeping pace with rates of sea level rise. The present study has focused on exposure of marsh grasses to phytotoxins such as sulfide and marsh accretion relative to sea level rise as two factors contributing to salt marsh loss in Long Island. Follow-up monitoring of these marshes, the addition of more salt marsh study areas, new aerial data, sampling of the low marsh areas and inclusion of Surface Elevation Tables (SETs) data are all necessary steps for a more complete evaluation of the health and trend of Long Island salt marshes.

In the 1920's, lead additives were introduced as octane-boosters for gasoline. Those who dared to raise their voices against the use of lead in gasoline found themselves heavily confronted by the petrochemical industry. Dr. Robert A. Kehoe was a spokesman for the lead industry. Regarding the petition from certain groups of stopping the use of lead in gasoline, he said:

"They (industry leaders) have expressed themselves repeatedly not so much as being interested in opinions as being interested in facts, and if it can be shown … than an actual danger to the public [occurs] as a result of the treatment of the gasoline with lead, the distribution of gasoline with lead in it will be discontinued from that moment."

He added:

"It is a thing which should be treated solely on the basis of facts." (U.S. Public Health Service, 1925, p. 70)

This was later called the Kehoe's *show-me rule*. Lead use in gasoline continued for 60 years more after these statements by Mr. Kehoe and we are still paying the consequences (Grandjean, 2013, p. 31). Ignorance has been used and is still used as a justification to do harmful things to the environment and to ourselves. Like Kehoe many today ask for proofs/facts that justify the cessation of certain actions that benefit them. We know that there are different stressors that contribute to salt marsh loss and, at different marshes and/or in different moments, each one of them has an impact that is variable depending of the multiple and complex biogeochemical conditions of a certain marsh at a fixed point in time (i.e. some marshes are able to tolerate better higher concentrations of sulfide than others). But, independently of this variable impact, at this point, we can apply the *show-me rule* of Kehoe. For this reason, we must protect and preserve these rich ecosystems.

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Appendices:

Appendix Table A1: Pore water data

Marsh	Collection Date	Coordinates Latitude Longitude	Depth (cm)	pH	Salinity	H_2S (µM)	NH_4^+ (µM)	PO_4^{3-} (µM)
East Creek	10/3/2008	40° 51.906′N 73° 42.661′W EC 1	2	-	-	43.5	13.8	6.48
			6	-	-	1472	67.7	41.3
			12	-	-	3083	137	50.0
			20	-	-	-		-
			30	-	-	3491	174	29.4
			40	-	-	1476	166	92.3
			50	-	-	3497	275	42.3
		40° 51.908′N 73° 42.675′W EC 2	2	-	-	1454	9.57	77.7
			7	-	-	2461	16.6	59.7
			12	-	-	3249	77.9	65.2
			25	-	-	4980	90.7	49.0
		40° 51.889′N 73° 42.697′W EC 3	12	-	-	2240	344	134
			20	-	-	3282	245	78.7
			30	-	-	1713	210	85.5
	5/18/2009	40° 51.911′N 73° 42.659′W EC 1	2	-	25.3	3.23	7.86	-
			7	-	-	13.5	10.71	2.90
			12	-	25.6	815	99.3	28.7
			20	-	-	1253	138	51.6
		40° 51.904′N 73° 42.678′W EC 2	2	-	27.8	6.30		
			7	-	-	9.38	7.14	6.77
			12	-	25.0	93.0	30.7	26.1
			20	-	-	443	161	42.9
		40° 51.890′N 73° 42.704′W EC 3	2	-	24.4	5.79	7.86	
			7	-	-	4.43	40.7	5.81
			12	-	24.7	773	218	94.8
			20	-	-	1375	295	88.1
			30	-	-	1744	400	79.4
	9/28/2009	40° 51.910′N 73° 42.658′W EC 1	7	6.45	-	5568	100	21.9
			12	6.46	24.3	6221	104	16.5
			20	6.4	25.1	5606	106	17.7
			30	6.43	-	4856	109	16.8
		40° 51.904′N 73° 42.681′W EC 2	2	6.83	25.5	2795	22.9	59.4
			7	6.55	-	4727	136	79.7
			12	6.62	28.0	6777	274	60.0
			20	6.83	-	6788	348	71.9
			30	6.57	-	6583	390	82.6
		40° 51.888′N 73° 42.703′W EC 3	2	-	24.8	1181	-	93.5
			12	6.53	27.8	5563	72.9	21.6
			20	6.46	-	5282	136	16.5
			30	6.51	-	3718	144	29.4
	4/22/2010	- - EC 1	2	-	24.9	142	75.9	38.6
			7	6.93	-	2880	323	52.5
			12	7.03	22.9	4434	368	46.5
			20	-	-	3200	168	53.7
			30	7.10	-	4867	374	50.6
		40° 51.906′N 73° 42.677′W EC 2	2	6.71	23.7	0	3.71	30.3
			7	-	-	143	45.8	85.4
			12	-	24.2	2475	121	123
			20	7.23	-	4970	382	71.2
			30	-	25.1	4764	279	100
		40° 51.891′N 73° 42.706′W EC 3	2	6.53	24.4	948	66.8	46.8
			7	-	-	497	94.4	60.2
			12	7.01	-	2031	172	81.7
			20	7.27	25.8	3492	245	53.5
			30	-	-	3698	266	59.9

Appendix Table A1: Pore water data (continuation)

Marsh	Collection Date	Coordinates Latitude Longitude	Depth (cm)	pH	Salinity	H$_2$S (µM)	NH$_4^+$ (µM)	PO$_4^{3-}$ (µM)
West Pond	20/05/2009	40° 53.307′N 73° 38.295′W	12	4.78	25.0	2541	264	24.5
		WP 1	20	-	-	3081	309	26.5
		40° 53.303′N 73° 38.310′W	12	-	24.5	682	53.6	16.8
		WP 2	2	-	21.5	1.69	143	20
		40° 53.301′N 73° 38.327′W	7	-	-	362	6.43	9.03
			12	-	22.7	667	7.86	9.35
		WP 3	20	-	-	589	7.14	13.2
	29/09/2009	40° 53.306′N 73° 38.293′W	2	6.54	26.3	890	4.29	4.84
			7	6.59	-	3469	68.6	10.6
		WP 1	12	-	25.1	4543	148	172
			20	6.66	-	5531	202	197
			30	-	-	3815	214	16.8
		40° 53.302′N 73° 38.309′W	2	6.39	24.8	2638	19.3	4.84
			7	6.45	-	3734	50.7	6.77
		WP 2	12	6.54	26.3	4651	110	6.45
			20	6.55	-	4354	123	14.5
			30	6.52	-	4662	159	20.6
		40° 53.298′N 73° 38.328′W	7	6.67	-	5935	346	17.4
		WP 3	12	-	25.3	3259	209	12.6
			30	6.53	-	4468	99.3	11.6
	4/20/2010	40° 53.305′N 73° 38.197′W	2	6.39	24.3	480	19.8	13.2
			7	6.69	-	996	91.6	53.7
		WP 1	12	-	25.0	2502	208	24.0
			20	6.72	-	4981	269	25.7
			30	6.67	-	4975	-	-
		40° 53.303′N 73° 38.314′W	2	6.14	23.0	834	-	2.16
			7	-	-	1479	4.21	12.8
		WP 2	12	6.49	-	1885	76.6	7.45
			20	6.82	-	1046	71.6	29.1
			30	-	-	917	55.1	11.2
		- -	2	-	25.3	105	17.4	10.3
			2K	6.65	-	182	0	
			7	-	-	1506	113	21.2
			7R	6.65	-	3579	389	6.71
			12	6.90	23.7	4153	410	15.3
		WP 3	12K	6.80	-	3649	0	-
			20	6.81	-	4196	392	13.7
			20R	-	-	4699	427	18.9
			20K	-	-	3784	-	-
			30	-	-	3129	-	-
Frost Creek	5/18/2009	40° 54.263′N 73° 35.597′W	2	4.91	21.1	59.6	-	-
			7	4.89	-	180	0	10.6
		FC 1	12	-	-	698	35.0	4.52
			50	-	-	1141	116	20.3
		40° 54.255′N 73° 35.630′W	2	-	23.9	37.1	14.3	-
		FC 2	20	4.95	23.6	144	6.43	3.00
		40° 54.250′N 73° 35.644′W	2	4.89	23.7	773	95.7	11.9
		FC 3	7	-	-	538	59.3	10.3
			12	-	25.3	787	104	15.2
	9/29/2009	40° 51.910′N 73° 42.658′W	2	6.36	20.0	933	0.714	0.323
			7	6.64	-	3507	5.00	1.61
		FC 1	12	6.51	26.1	3766	12.1	6.13
			20	6.5	-	3955	50.0	9.35
			30	6.45	-	3718	80.7	8.39
		40° 54.254′N 73° 35.630′W	2	6.5	21.4	987	4.29	6.77
			7	6.46	-	2066	11.4	4.19
		FC 2	12	6.58	23.0	3394	65.0	5.16
			20	6.49	-	3771	68.6	4.52
			30	6.50	-	3809	116	8.39
		40° 54.249′N 73° 35.644′W	2	6.49	24.0	2433	15.7	4.84
			7	6.48	-	3728	27.1	3.87
		FC 3	12	6.48	25.0	3755	50.0	3.55
			20	6.48	-	4333	115	4.52
			30	6.48	-	5946	157	5.16

Appendix Table A1: Pore water data (continuation)

Marsh	Collection Date	Coordinates Latitude Longitude	Depth (cm)	pH	Salinity	H$_2$S (µM)	NH$_4^+$ (µM)	PO$_4^{3-}$ (µM)
Flax Pond	10/6/2008	40° 57.862′N 73° 08.882′W FP 1	2	-	-	308	22.1	24.6
			6	-	-	2525	52.1	31.5
			12	-	-	3422	86.4	48.4
			30	-	-	2444	87.1	34.8
			40	-	-	1647	86.4	33.9
			50	-	-	1957	99.3	21.1
		40° 57.873′N 73° 08.847′W FP 2	7	-	-	3387	18.1	16.5
			25	-	-	3533	111	45.2
		40° 57.889′N 73° 08.843′W FP 3	4	-	-	1288	109	69.7
			12	-	-	3085	77.1	72.6
	5/19/2009	40° 57.859′N 73° 08.877′W FP 1	2	-	26.8	2.20	1.43	0.968
			7	-	-	248	33.6	37.7
			20	-	27.2	557	41.4	-
		40° 57.874′N 73° 08.849′W FP 2	2	-	27.4	405	38.6	11.0
			7	-	-	833	52.1	-
			12	-	27.1	633	47.1	22.6
			20	-	-	1735	149	30.6
		40° 57.887′N 73° 08.820′W FP 3	2	-	-	3.23	5.00	-
			7	-	-	7.33	43.6	13.5
			12	-	-	3.74	5.00	-
Hubbard Creek	5/27/2010	40° 54.520′N 72° 33.942′W HC 1	2	6.05	-	4.14	-	-
			7	6.41	-	889	2.43	-
			12	6.17	-	2084	18.6	-
			20	6.54	-	3566	164	-
			30	6.63	-	4855	139	-
		40° 54.493′N 72° 33.935′W HC 2	2	6.52	-	2634	17.2	-
			7	6.59	-	1737	26.2	-
			12	-	-	3904	120	-
			20	6.62	-	4995	258	-
			30	6.61	-	3792	184	-
		40° 54.469′N 72° 33.946′W HC 3	2	6.13	-	927	-	-
			7	6.30	-	1020	3.50	-
			12	6.17	-	2026	14.7	-
			20	6.30	-	3512	70.7	-
			30	6.64	-	4774	166	-
	9/29/2010	40° 54.522′N 72° 33.942′W HC 1	2	6.54	-	2552	28.9	-
			7	6.92	-	2375	19.2	-
			12	6.61	-	5570	176	-
			20	6.61	-	5827	196	-
			30	6.5	-	6098	205	-
		40° 54.491′N 72° 33.941′W HC 2	2	6.4	-	47.248	106	-
			7	6.84	-	511	99.3	-
			12	6.37	-	2646	37.3	-
			20	6.53	-	4811	0.429	-
			30	-	-	5476	0.286	-
		40° 54.469′N 72° 33.945′W HC 3	2	6.22	-	27.5	-	-
			7	6.4	-	314	9.29	-
			12	6.55	-	1522	76.4	-
			20	6.6	-	3928	262	-
			30	6.63	-	4737	-	-
Mashomack	8/8/2008	41° 2.598′N 72° 17.468′W MS 1	2	-	-	3378	33.9	23.1
			7	-	-	1479	34.1	28.5
			12	-	-	3572	33.1	7.16
			25	-	-	618	36.7	27.1
			35	-	-	38.8	1.93	-
			45	-	-	39.0	97.1	14.3
		41° 2.595′N 72° 17.482′W MS 2	8	-	-	1326	50.4	56.5
			10	-	-	1465	81.4	59.4
			15	-	-	1211	28.3	28.9
			18	-	-	943	83.6	37.4
			36	-	-	695	264	53.2
		41° 2.600′N 72° 17.502′W MS 3	6	-	-	3017	101	41.9
			14	-	-	4675	109	25.9
			24	-	-	142	115	90.6
			34	-	-	2382	307	33.2
			44	-	-	3287	273	22.5

Appendix Table A1: Pore water data (continuation)

Marsh	Collection Date	Coordinates Latitude Longitude	Depth (cm)	pH	Salinity	H_2S (µM)	NH_4^+ (µM)	PO_4^{3-} (µM)
Accabonac Harbor T	5/26/2010	41° 1.590′N 72° 8.931′W AH T1	2	6.2	-	443	-	-
			7	-	-	-	-	-
			12	6.81	-	2909	209	-
			20	6.72	-	2787	289	-
			30	3.67	-	3129	363	-
		41° 1.578′N 72° 8.934′W AH T2	2	-	-	2589	216	-
			7	6.36	-	3612	266	-
			12	6.43	-	4679	355	-
			20	6.41	-	4846	389	-
			30	-	-	4157	404	-
		41° 1.565′N 72° 8.929′W AH T3	2	6.38	-	2607	150	-
			7	-	-	3837	289	-
			12	6.69	-	4012	505	-
			20	-	-	2733	507	-
			30	6.6	-	5513	589	-
	10/19/2010	41° 1.590′N 72° 8.930′W AH T1	2	4.6	-	0	-	-
			7	5.66	-	0	6.43	-
			12		-	826	107	-
			20	6.9	-	767	311	-
			30	6.74	-	3829	401	-
		41° 1.576′N 72° 8.933′W AH T2	2	6.72	-	2572	143	-
			7	6.69	-	4712	314	-
			12	6.51	-	5560	361	-
			20	6.41	-	6073	426	-
			30	6.54	-	6384	424	-
		41° 1.566′N 72° 8.931′W AH T3	2	6.56	-	4855	206	-
			7	-	-	6561	449	-
			12	-	-	6828	471	-
			20	6.62	-	7035	454	-
			30	6.58	-	7612	499	-
Accabonac Harbor C	5/26/2010	41° 1.668′N 72° 8.850′W AH C1	2	-	-	63.1	15.1	-
			7	6.54	-	3175	98.6	-
			12	-	-	2454	91.4	-
			20	6.34	-	1467	90.7	-
			30	6.50	-	4495	191	-
		41° 1.665′N 72° 8.832′W AH C2	2	-	-	1733	62.0	-
			7	-	-	3936	274	-
			12	-	-	4211	438	-
			20	-	-	3003	-	-
			30	6.59	-	4413	467	-
		41° 1.661′N 73° 8.804′W AH C3	2	-	-	4391	48.4	-
			7	6.59	-	5022	106	-
			12	-	-	4242	186	-
			20	6.37	-	4923	259	-
			30	6.36	-	4486	256	-
	10/19/2010	41° 1.680′N 72° 8.841′W AH C1	2	-	-	4086	164	-
			7	7.02	-	5836	286	-
			12	-	-	5126	384	-
			20	6.90	-	6803	434	-
			30	6.50	-	5856	340	-
		41° 1.658′N 72° 8.833′W AH C2	2	-	-	1985	49.1	-
			7	-	-	5195	531	-
			12	6.89	-	5925	637	-
			20		-	6547	644	-
			30	6.85	-	6985	534	-
		41° 1.660′N 72° 8.806′W AH C3	2	6.05	-	1985	288	-
			7	6.02	-	5195	690	-
			12	-	-	5925	721	-
			20	-	-	6547	580	-
			30	-	-	6985	383	-

Appendix Table A2: Radiochemical data

Accabonac Harbor C2 (41° 1.665´N 72° 8.832´W // 5/26/2010)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	84.6	0.168	13.4 ± 0.5	0.48 ± 0.07	13.1 ± 0.5	0.00 ± 0.07
1-2	82.7	0.191	7.17 ± 0.38	0.36 ± 0.05	6.89 ± 0.39	0.00 ± 0.02
2-3	81.1	0.209	6.68 ± 0.32	0.14 ± 0.02	6.62 ± 0.32	0.00 ± 0.01
3-4	78.6	0.242	14.5 ± 0.33	0.28 ± 0.03	14.4 ± 0.33	0.09 ± 0.01
4-5	77.2	0.260	11.5 ± 0.36	0.22 ± 0.02	11.4 ± 0.36	0.10 ± 0.02
5-6	78.1	0.249	12.3 ± 0.40	0.33 ± 0.04	12.1 ± 0.41	0.03 ± 0.01
6-7	79.5	0.233	6.03 ± 0.30	0.32 ± 0.04	5.78 ± 0.31	0.64 ± 0.05
7-8	79.4	0.234	7.38 ± 0.33	0.60 ± 0.06	6.87 ± 0.34	2.68 ± 0.09
8-9	78.7	0.245	5.08 ± 0.24	0.61 ± 0.05	4.52 ± 0.25	1.73 ± 0.07
9-10	77.1	0.265	4.43 ± 0.29	0.59 ± 0.06	3.89 ± 0.30	0.35 ± 0.04
10-12	78.6	0.244	3.29 ± 0.20	0.31 ± 0.03	3.01 ± 0.21	0.05 ± 0.01
12-14	82.2	0.199	1.85 ± 0.17	0.62 ± 0.06	1.24 ± 0.18	0.01 ± 0.00
14-16	83.6	0.181	1.25 ± 0.16	0.20 ± 0.02	1.06 ± 0.17	0.04 ± 0.01
16-18	80.0	0.227	0.95 ± 0.17	0.65 ± 0.07	0.30 ± 0.19	0.10 ± 0.02
18-20	75.3	0.288	2.20 ± 0.15	1.14 ± 0.06	1.07 ± 0.16	0.11 ± 0.01
20-22	76.3	0.275	1.22 ± 0.14	0.69 ± 0.05	0.54 ± 0.15	0.60 ± 0.04

Accabonac Harbor T2 (41° 1.578´N 72° 8.934´W // 5/26/2010)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	80.0	0.227	27.1 ± 0.5	0.33 ± 0.04	27.1 ± 0.5	0.09 ± 0.01
1-2	79.9	0.228	20.0 ± 0.4	0.57 ± 0.04	19.7 ± 0.4	0.15 ± 0.02
2-3	80.1	0.225	15.6 ± 0.5	0.42 ± 0.04	15.4 ± 0.5	0.13 ± 0.02
3-4	79.6	0.232	11.6 ± 0.4	0.75 ± 0.06	11.0 ± 0.4	0.08 ± 0.01
4-5	82.4	0.195	7.67 ± 0.33	0.22 ± 0.02	7.55 ± 0.34	0.14 ± 0.02
5-6	83.5	0.181	6.45 ± 0.40	0.57 ± 0.06	5.96 ± 0.41	0.17 ± 0.03
6-7	83.3	0.184	3.83 ± 0.37	0.16 ± 0.02	3.72 ± 0.38	0.36 ± 0.04
7-8	82.5	0.194	4.32 ± 0.34	0.55 ± 0.06	3.83 ± 0.35	0.40 ± 0.04
8-9	83.0	0.189	5.47 ± 0.35	0.58 ± 0.07	4.96 ± 0.36	1.90 ± 0.09
9-10	82.6	0.194	5.27 ± 0.36	1.11 ± 0.10	4.22 ± 0.38	1.27 ± 0.08
10-12	81.6	0.206	3.72 ± 0.25	1.17 ± 0.08	2.58 ± 0.27	0.38 ± 0.04
12-14	79.0	0.239	3.51 ± 0.31	0.84 ± 0.09	2.71 ± 0.32	0.01 ± 0.00
14-16	80.3	0.223	2.08 ± 0.22	0.40 ± 0.06	1.70 ± 0.23	0.11 ± 0.02
16-18	83.3	0.185	0.86 ± 0.07	0.34 ± 0.03	0.52 ± 0.08	0.10 ± 0.01

Hubbard Creek 2 (40° 54.494´N 72° 33.939´W // 5/27/2010)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	84.6	0.168	19.1 ± 0.5	0.17 ± 0.02	19.1 ± 0.5	0.24 ± 0.03
1-2	85.4	0.159	17.7 ± 0.5	0.25 ± 0.03	17.7 ± 0.5	0.01 ± 0.00
2-3	85.6	0.155	14.9 ± 0.5	0.86 ± 0.11	14.2 ± 0.5	0.00 ± 0.01
3-4	84.3	0.171	15.5 ± 0.6	0.22 ± 0.04	15.4 ± 0.6	0.00 ± 0.01
4-5	85.0	0.164	13.4 ± 0.6	0.95 ± 0.11	12.6 ± 0.6	0.03 ± 0.00
5-6	85.2	0.161	14.2 ± 0.6	0.30 ± 0.05	14.1 ± 0.6	0.00 ± 0.00
6-7	85.6	0.157	12.3 ± 0.6	0.41 ± 0.10	12.0 ± 0.6	0.00 ± 0.02
7-8	86.0	0.151	9.73 ± 0.69	0.34 ± 0.06	9.49 ± 0.70	0.00 ± 0.03
8-9	86.8	0.143	8.16 ± 0.45	0.68 ± 0.11	7.57 ± 0.46	0.44 ± 0.05
9-10	86.7	0.144	7.98 ± 0.44	0.52 ± 0.05	7.54 ± 0.44	0.51 ± 0.05
10-12	85.9	0.153	6.43 ± 0.23	0.53 ± 0.05	5.96 ± 0.24	1.53 ± 0.05
12-14	86.0	0.152	5.85 ± 0.53	0.61 ± 0.09	5.30 ± 0.54	0.56 ± 0.08
14-16	84.7	0.167	2.73 ± 0.32	0.38 ± 0.08	2.38 ± 0.33	0.17 ± 0.03
16-18	83.5	0.183	3.15 ± 0.42	0.58 ± 0.09	2.60 ± 0.44	0.00 ± 0.03
18-20	82.5	0.195	3.45 ± 0.35	0.87 ± 0.09	2.61 ± 0.36	0.08 ± 0.02
20-22	86.8	0.143	2.32 ± 0.30	0.35 ± 0.06	1.99 ± 0.30	0.00 ± 0.03

East Creek 1 (40° 51.906´N 73° 42.661´W // 10/3/2008)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	81.8	0.203	10.2 ± 0.4	0.64 ± 0.09	10.1 ± 0.4	-0.11 ± 0.03
1-2	81.5	0.207	10.5 ± 0.5	0.59 ± 0.11	10.5 ± 0.5	-0.26 ± 0.12
2-3	82.4	0.193	10.9 ± 0.7	0.75 ± 0.09	10.7 ± 0.7	-0.29 ± 0.14
3-4	80.7	0.216	5.91 ± 0.36	0.43 ± 0.07	5.80 ± 0.39	0.18 ± 0.03
4-5	79.8	0.228	7.48 ± 0.37	0.85 ± 0.09	7.01 ± 0.40	-0.20 ± 0.06
5-6	80.4	0.221	6.48 ± 0.31	0.56 ± 0.05	6.26 ± 0.33	0.08 ± 0.01
6-7	80.5	0.218	4.58 ± 0.21	0.85 ± 0.08	3.94 ± 0.24	0.10 ± 0.01
7-8	78.9	0.240	6.31 ± 0.26	1.11 ± 0.07	5.50 ± 0.28	0.01 ± 0.00
8-9	77.8	0.257	3.57 ± 0.22	1.23 ± 0.06	2.48 ± 0.24	0.26 ± 0.03
9-10	79.1	0.239	1.94 ± 0.14	1.32 ± 0.07	0.65 ± 0.17	0.51 ± 0.03
10-12	80.6	0.219	3.66 ± 0.17	1.19 ± 0.06	2.61 ± 0.20	0.80 ± 0.04
12-14	77.3	0.263	3.65 ± 0.27	1.32 ± 0.09	2.47 ± 0.30	1.71 ± 0.08
14-16	76.0	0.281	2.79 ± 0.21	1.52 ± 0.09	1.35 ± 0.24	1.02 ± 0.06
16-18	73.5	0.316	2.94 ± 0.22	1.20 ± 0.07	1.84 ± 0.24	0.16 ± 0.02
18-20	72.8	0.325	3.19 ± 0.24	1.72 ± 0.08	1.55 ± 0.26	0.07 ± 0.01

East Creek 2 (40° 51.908´N 73° 42.675´W // 10/3/2008)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	78.7	0.233	9.39 ± 0.78	1.62 ± 0.19	8.25 ± 0.85	0.52 ± 0.56
1-2	79.2	0.239	10.3 ± 0.4	1.01 ± 0.10	9.90 ± 0.45	0.01 ± 0.00
2-3	78.0	0.252	10.4 ± 0.4	1.09 ± 0.13	9.84 ± 0.48	0.36 ± 0.31
3-4	78.5	0.248	9.33 ± 0.59	1.73 ± 0.19	8.06 ± 0.65	0.20 ± 0.03
4-5	79.8	0.232	8.14 ± 0.38	1.30 ± 0.12	7.27 ± 0.43	0.21 ± 0.11
5-6	79.6	0.232	8.01 ± 0.48	0.72 ± 0.08	7.74 ± 0.52	0.22 ± 0.12
6-7	79.5	0.234	5.43 ± 0.24	0.90 ± 0.07	4.81 ± 0.27	0.15 ± 0.02
7-8	75.0	0.296	6.33 ± 0.26	1.16 ± 0.08	5.49 ± 0.29	0.17 ± 0.05
8-9	71.6	0.344	6.89 ± 0.40	1.69 ± 0.12	5.52 ± 0.45	0.03 ± 0.01
9-10	74.8	0.296	5.64 ± 0.32	1.41 ± 0.10	4.49 ± 0.35	0.09 ± 0.01
10-12	76.2	0.277	5.59 ± 0.24	1.10 ± 0.07	4.77 ± 0.27	0.06 ± 0.01
12-14	74.1	0.310	3.88 ± 0.24	1.52 ± 0.09	2.51 ± 0.27	0.09 ± 0.02
14-16	75.7	0.284	3.07 ± 0.22	1.37 ± 0.08	1.80 ± 0.25	0.32 ± 0.03
16-18	75.0	0.297	3.30 ± 0.24	1.36 ± 0.08	2.06 ± 0.27	0.97 ± 0.05
18-20	73.7	0.314	3.28 ± 0.21	1.63 ± 0.08	1.75 ± 0.24	0.50 ± 0.04
20-22	72.1	0.338	1.65 ± 0.16	1.27 ± 0.07	0.40 ± 0.18	0.37 ± 0.03
22-24	69.9	0.370	1.85 ± 0.17	1.50 ± 0.07	0.37 ± 0.20	0.04 ± 0.01

East Creek 2B (40° 51.904´N 73° 42.678´W // 5/21/2009)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	75.4	0.289	11.3 ± 0.4	1.00 ± 0.09	10.8 ± 0.5	0.01 ± 0.00
1-2	72.7	0.327	10.3 ± 0.4	0.99 ± 0.09	9.80 ± 0.43	-0.03 ± 0.01
2-3	76.0	0.281	7.97 ± 0.39	1.19 ± 0.12	7.10 ± 0.42	-0.21 ± 0.06
3-4	78.6	0.245	6.03 ± 0.30	0.83 ± 0.08	5.45 ± 0.33	-0.10 ± 0.02
4-5	79.8	0.229	6.45 ± 0.30	1.28 ± 0.10	5.42 ± 0.33	-0.23 ± 0.11
5-6	78.3	0.250	5.49 ± 0.32	2.01 ± 0.14	3.65 ± 0.36	-0.08 ± 0.04
6-7	75.0	0.294	6.36 ± 0.35	1.73 ± 0.11	4.85 ± 0.38	0.23 ± 0.03
7-8	73.1	0.320	6.79 ± 0.31	1.58 ± 0.09	5.46 ± 0.34	0.02 ± 0.00
8-9	74.0	0.309	5.14 ± 0.24	1.69 ± 0.10	3.62 ± 0.27	-0.05 ± 0.02
9-10	74.3	0.302	5.27 ± 0.28	1.30 ± 0.09	4.16 ± 0.31	0.08 ± 0.01
10-12	72.1	0.334	3.46 ± 0.20	1.08 ± 0.07	2.50 ± 0.23	0.06 ± 0.01
12-14	72.9	0.323	3.30 ± 0.21	1.73 ± 0.10	1.65 ± 0.24	0.04 ± 0.01
14-16	75.5	0.289	4.04 ± 0.25	1.43 ± 0.08	2.74 ± 0.27	0.22 ± 0.02
16-18	73.8	0.312	2.97 ± 0.19	1.55 ± 0.07	1.48 ± 0.21	0.08 ± 0.01

Flax Pond 1 (40° 57.862´N 73° 8.882´W // 9/3/2008)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	73.5	0.316	10.2 ± 0.3	0.45 ± 0.04	10.5 ± 0.3	0.05 ± 0.01
1-2	72.7	0.327	8.17 ± 0.27	0.67 ± 0.04	8.03 ± 0.30	0.03 ± 0.00
2-3	74.1	0.307	3.98 ± 0.16	0.61 ± 0.04	3.62 ± 0.18	-0.03 ± 0.00
3-4	72.8	0.325	4.07 ± 0.24	0.92 ± 0.07	3.37 ± 0.27	0.13 ± 0.02
4-5	57.9	0.565	3.20 ± 0.16	0.77 ± 0.04	2.60 ± 0.18	0.07 ± 0.01
5-6	60.0	0.527	3.20 ± 0.22	1.52 ± 0.07	1.80 ± 0.25	0.05 ± 0.01
6-7	60.1	0.527	2.32 ± 0.20	0.71 ± 0.05	1.73 ± 0.22	0.07 ± 0.02
7-8	74.0	0.307	2.78 ± 0.24	1.07 ± 0.08	1.83 ± 0.27	0.03 ± 0.01
8-9	71.4	0.343	2.36 ± 0.16	1.18 ± 0.06	1.27 ± 0.18	0.12 ± 0.01
9-10	72.5	0.330	3.00 ± 0.23	1.13 ± 0.08	2.01 ± 0.26	0.11 ± 0.02
10-12	77.8	0.255	2.54 ± 0.22	0.43 ± 0.04	2.26 ± 0.24	0.25 ± 0.03
12-14	73.4	0.317	3.17 ± 0.19	1.17 ± 0.06	2.16 ± 0.22	0.56 ± 0.04
14-16	72.0	0.337	1.05 ± 0.07	0.67 ± 0.04	0.41 ± 0.09	0.39 ± 0.03
16-18	67.8	0.401	2.38 ± 0.16	1.15 ± 0.07	1.32 ± 0.18	0.13 ± 0.02
18-20	67.1	0.412	1.60 ± 0.14	1.37 ± 0.07	0.25 ± 0.17	0.03 ± 0.00
20-22	73.8	0.312	2.11 ± 0.19	1.63 ± 0.09	0.51 ± 0.23	-0.07 ± 0.03
22-24	79.0	0.241	1.31 ± 0.16	0.54 ± 0.05	0.83 ± 0.18	-0.13 ± 0.04
24-26	80.5	0.221	1.08 ± 0.11	1.32 ± 0.07	-0.24 ± 0.13	-0.26 ± 0.23

Flax Pond 2 (40° 57.873´N 73° 8.847´W // 9/3/2008)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	74.0	0.309	9.95 ± 0.42	1.04 ± 0.10	9.58 ± 0.46	0.04 ± 0.01
1-2	72.3	0.333	8.79 ± 0.43	0.97 ± 0.08	8.40 ± 0.47	0.08 ± 0.01
2-3	72.7	0.327	6.06 ± 0.23	0.61 ± 0.04	5.85 ± 0.25	-0.10 ± 0.04
3-4	76.7	0.271	9.03 ± 0.31	0.78 ± 0.06	8.87 ± 0.34	-0.01 ± 0.00
4-5	76.4	0.276	7.58 ± 0.24	0.58 ± 0.04	7.52 ± 0.27	0.27 ± 0.02
5-6	74.9	0.297	4.85 ± 0.20	1.04 ± 0.06	4.10 ± 0.22	-0.08 ± 0.01
6-7	71.1	0.352	5.87 ± 0.24	1.06 ± 0.06	5.17 ± 0.26	0.13 ± 0.02
7-8	61.2	0.508	4.40 ± 0.18	1.19 ± 0.05	3.45 ± 0.21	0.22 ± 0.02
8-9	65.9	0.429	3.87 ± 0.18	1.12 ± 0.05	2.96 ± 0.20	0.13 ± 0.01
9-10	75.3	0.288	3.64 ± 0.27	0.75 ± 0.07	3.11 ± 0.30	-0.03 ± 0.01
10-12	77.0	0.266	2.60 ± 0.22	0.55 ± 0.04	2.21 ± 0.25	0.53 ± 0.04
12-14	76.1	0.279	1.58 ± 0.13	0.75 ± 0.06	0.89 ± 0.16	0.64 ± 0.05
14-16	76.2	0.278	1.94 ± 0.18	0.84 ± 0.06	1.19 ± 0.21	0.49 ± 0.04
16-18	70.5	0.360	2.19 ± 0.16	0.99 ± 0.05	1.30 ± 0.19	0.08 ± 0.01

Flax Pond 3 (40° 57.889´N 73° 8.843´W // 9/3/2008)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	69.0	0.380	11.2 ± 0.5	0.82 ± 0.09	10.9 ± 0.5	-0.20 ± 0.18
1-2	74.0	0.309	8.77 ± 0.36	0.67 ± 0.06	8.56 ± 0.38	NA
2-3	75.2	0.289	3.12 ± 0.21	0.35 ± 0.04	2.93 ± 0.22	-0.07 ± 0.01
3-4	74.1	0.305	4.34 ± 0.38	0.60 ± 0.07	3.96 ± 0.41	0.04 ± 0.01
4-5	72.3	0.331	5.88 ± 0.32	0.62 ± 0.05	5.56 ± 0.34	0.14 ± 0.02
5-6	67.4	0.402	4.41 ± 0.22	0.38 ± 0.04	4.27 ± 0.24	0.09 ± 0.01
6-7	59.8	0.519	5.06 ± 0.30	0.75 ± 0.05	4.56 ± 0.32	0.18 ± 0.02
7-8	65.3	0.439	3.10 ± 0.18	0.64 ± 0.04	2.60 ± 0.19	0.27 ± 0.02
8-9	69.9	0.368	3.28 ± 0.17	0.53 ± 0.04	2.91 ± 0.18	0.10 ± 0.01
9-10	71.4	0.345	5.54 ± 0.27	0.72 ± 0.06	5.10 ± 0.29	0.12 ± 0.02
10-12	73.0	0.322	4.41 ± 0.29	0.47 ± 0.04	4.17 ± 0.31	0.23 ± 0.03
12-14	73.4	0.318	1.73 ± 0.16	0.48 ± 0.04	1.33 ± 0.17	0.33 ± 0.03
14-16	78.2	0.251	2.33 ± 0.19	0.51 ± 0.04	1.92 ± 0.20	0.32 ± 0.03
16-18	77.0	0.267	2.34 ± 0.16	0.62 ± 0.05	1.82 ± 0.18	0.26 ± 0.03
18-20	75.1	0.295	2.29 ± 0.19	0.61 ± 0.04	1.78 ± 0.20	0.67 ± 0.04
20-22	72.3	0.334	2.38 ± 0.18	0.75 ± 0.05	1.73 ± 0.20	0.87 ± 0.05
22-24	67.1	0.407	2.27 ± 0.18	1.11 ± 0.09	1.23 ± 0.21	0.13 ± 0.03
24-26	61.5	0.505	1.65 ± 0.15	0.95 ± 0.05	0.75 ± 0.16	0.08 ± 0.01
26-28	67.8	0.401	1.04 ± 0.09	0.77 ± 0.05	0.29 ± 0.11	0.27 ± 0.03

Frost Creek 1 (40° 54.263´N 73° 35.597´W // 5/18/2009)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	77.2	0.263	13.2 ± 0.5	0.51 ± 0.05	13.2 ± 0.5	0.22 ± 0.29
1-2	70.5	0.357	5.02 ± 0.29	0.69 ± 0.06	4.49 ± 0.31	0.01 ± 0.02
2-3	77.0	0.264	8.16 ± 0.33	0.88 ± 0.06	7.55 ± 0.35	0.04 ± 0.07
3-4	77.5	0.259	7.55 ± 0.31	0.50 ± 0.04	7.31 ± 0.32	0.07 ± 0.11
4-5	77.8	0.254	10.1 ± 0.47	0.38 ± 0.04	10.0 ± 0.49	0.03 ± 0.05
5-6	78.2	0.250	9.04 ± 0.40	0.71 ± 0.06	8.63 ± 0.42	0.08 ± 0.12
6-7	81.2	0.211	8.70 ± 0.45	0.06 ± 0.01	8.96 ± 0.47	0.28 ± 0.28
7-8	81.6	0.206	4.16 ± 0.28	0.07 ± 0.01	4.24 ± 0.29	0.07 ± 0.09
8-9	83.5	0.183	4.93 ± 0.34	0.48 ± 0.07	4.62 ± 0.36	0.20 ± 0.27
9-10	82.7	0.192	6.08 ± 0.42	1.04 ± 0.11	5.23 ± 0.45	0.57 ± 0.58
10-12	79.4	0.234	4.85 ± 0.32	0.80 ± 0.07	4.20 ± 0.34	1.78 ± 0.74
12-14	81.3	0.211	1.60 ± 0.16	0.88 ± 0.06	0.75 ± 0.18	0.92 ± 0.70
14-16	79.7	0.231	2.80 ± 0.19	1.01 ± 0.06	1.85 ± 0.21	0.09 ± 0.18
16-18	77.4	0.261	2.38 ± 0.18	1.02 ± 0.06	1.41 ± 0.19	-0.02 ± 0.04
18-20	81.2	0.212	1.03 ± 0.10	0.79 ± 0.05	0.25 ± 0.12	-0.18 ± 1.28
20-22	83.6	0.183	0.93 ± 0.13	0.59 ± 0.06	0.36 ± 0.15	-0.20 ± 0.00
22-24	84.3	NA	NA	NA	0.31 ± 0.14*	NA
24-26	84.2	0.175	0.79 ± 0.12	0.55 ± 0.04	0.25 ± 0.13	-0.04 ± 0.07
26-28	85.9	0.154	-0.12 ± 0.04	0.12 ± 0.02	0.00 ± 0.04	0.00 ± 0.00
28-30	82.7	0.194	1.39 ± 0.20	0.98 ± 0.07	0.43 ± 0.22	-0.10 ± 0.60

West Pond 1 (40° 53.307´N 73° 38.295´W // 5/20/2009)

Depth (cm)	% Water	Dry Bulk Density	^{210}Pb (dpm/g)	^{226}Ra (dpm/g)	^{210}Pb Excess (dpm/g)	^{137}Cs (dpm/g)
0-1	74.4	0.300	11.3 ± 0.5	0.71 ± 0.10	11.0 ± 0.55	0.00 ± 0.03
1-2	77.5	0.256	10.3 ± 0.5	0.67 ± 0.08	10.0 ± 0.56	0.00 ± 0.07
2-3	78.9	0.238	6.41 ± 0.39	0.70 ± 0.06	5.94 ± 0.41	0.00 ± 0.00
3-4	79.7	0.230	7.59 ± 0.36	0.74 ± 0.08	7.11 ± 0.38	0.05 ± 0.01
4-5	81.5	0.205	6.85 ± 0.36	0.38 ± 0.05	6.72 ± 0.38	0.00 ± 0.00
5-6	80.0	0.226	4.84 ± 0.32	0.53 ± 0.06	4.48 ± 0.34	0.01 ± 0.00
6-7	78.5	0.246	3.64 ± 0.34	0.78 ± 0.08	2.97 ± 0.36	0.09 ± 0.01
7-8	77.7	0.255	4.91 ± 0.33	0.72 ± 0.08	4.36 ± 0.36	0.00 ± 0.02
8-9	76.4	0.275	5.91 ± 0.37	0.85 ± 0.08	5.26 ± 0.39	0.13 ± 0.02
9-10	79.2	0.236	4.84 ± 0.31	0.81 ± 0.09	4.19 ± 0.34	0.07 ± 0.01
10-12	81.3	0.207	3.71 ± 0.25	0.42 ± 0.05	3.43 ± 0.27	0.15 ± 0.02
12-14	83.2	0.185	1.60 ± 0.23	0.71 ± 0.06	0.93 ± 0.25	0.33 ± 0.05
14-16	79.3	0.235	2.10 ± 0.17	0.73 ± 0.06	1.42 ± 0.19	0.71 ± 0.04
16-18	76.8	0.268	2.53 ± 0.20	0.98 ± 0.06	1.61 ± 0.21	0.08 ± 0.01
18-20	81.1	0.213	1.87 ± 0.25	0.76 ± 0.08	1.16 ± 0.27	0.15 ± 0.02
20-22	76.4	0.275	1.44 ± 0.18	0.94 ± 0.06	0.52 ± 0.20	0.10 ± 0.01
22-24	74.9	0.297	2.27 ± 0.23	0.86 ± 0.07	1.47 ± 0.25	0.00 ± 0.13
24-26	79.3	0.237	1.14 ± 0.21	0.84 ± 0.09	0.31 ± 0.24	0.00 ± 0.01
26-28	80.8	0.219	1.12 ± 0.21	0.50 ± 0.06	0.64 ± 0.22	0.00 ± 0.04