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Ecological functions of an impounded marsh and three natural estuarine marshes along Woodbridge River, NY/NJ Harbor

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Abstract. An impounded estuarine marsh scheduled for restoration in 2003 and three nearby unimpounded marshes (*Spartina alterniflora*, *S. patens/Iva frutescens*, and *Phragmites australis*) in highly urbanized NY/NJ Harbor were studied to assess the impact of impoundment on marsh structure and function and to identify trajectories of ecosystem change following removal of the levee. Aboveground biomass was greater in the *Phragmites* and *S. alterniflora* marshes (706–809 g/m^2) as compared to the *S. patens/I. frutescens* and impounded marshes (378–588 g/m^2). Macro-organic matter (0–30 cm) was similar across all marshes (7452–9212 g/m^2). The *S. patens/Iva frutescens* marsh had the lowest aboveground biomass, but contained more plant species (2.8 species/0.25 m^2) and greater species diversity ($H = 1.33$) than the other marshes (1.3–2.0 species/0.25 m^2, $H = 0.16$–0.78). Rates of sediment and nutrient accumulation were lower in the impounded marsh (335 g sediment/m^2/yr, 97 g C/m^2/yr, 6.5 g N/m^2/yr, and 0.9 g P/m^2/yr) than in the reference marshes (422–1515 g sediment/m^2/yr; 111–160 g C/m^2/yr; 7–10 g N/m^2/yr; 1.6–2.8 g P/m^2/yr). Our results indicate that the impounded marsh does not contain the high species diversity of the high marsh, nor does it provide the same level of functions as naturally inundated marshes. Reintroduction of tidal inundation to the impounded marsh will enhance water quality benefits and favor development of *S. alterniflora* salt marsh community structure.

Keywords: functional assessment, tidal restriction, restoration, *Phragmites*, *Spartina alterniflora*

Introduction

The extent of salt marshes in North America has greatly declined since colonial times, due primarily to impacts from development (Mitsch and Gosselink, 2000). While direct impacts such as filling have been reduced since passage of the Clean Water Act in 1972, indirect impacts continue to have long-term effects in remaining salt marshes. The restriction of normal tidal flow related to the construction of roads, causeways, bridges, and impoundments results in loss of tidal flooding, lowering of the water table, drop in surface elevation, and decreased soil salinity (Roman *et al.*, 1984; Sinicrope *et al.*, 1990; Burdick *et al.*, 1997). This

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in turn leads to changes in the vegetative community, from a typical salt marsh characterized by *Spartina* spp. to a marsh dominated by *Phragmites australis* (common reed), a large, perennial, rhizomatous grass that forms nearly monotypic stands (Roman *et al*., 1984). *Phragmites* has invaded brackish and freshwater tidal wetlands throughout the coastal regions of North America in the last 200 years, outcompeting native vegetation and reducing species diversity (Galatowitsch *et al*., 1999; Meyerson *et al*., 2000; Rice *et al*., 2000). The spread of *Phragmites* has been linked to tidal wetland alteration, including disturbance of hydrologic cycles and nutrient regimes (Chambers *et al*., 1999; Bart and Hartman, 2000). Tidal restriction also reduces the water quality-related functions of salt marshes by reducing sedimentation and accretion rates (Anisfeld *et al*., 1999).

The densely populated area around the New York/New Jersey harbor is home to 17 million people (US Census, 2000). Once covered by vast estuarine marshes such as the Hackensack Meadows, population growth has reduced these marshes to isolated pockets surrounded by development. This has resulted in loss of wetland function in areas that are most in need of the water quality improvement and water storage that wetlands provide. Restoring degraded, impounded salt marshes in this highly urbanized area provides an opportunity to enhance local biodiversity as well as improve water quality functions. However, restoration of wetlands in urban landscapes is difficult because land clearing and conversion to impervious surfaces within the watershed leads to alteration of natural flow regimes and increased sediment, nutrients and invasive species (Ehrenfeld, 2000).

Ecological characteristics of vegetation and soils were measured in an impounded estuarine marsh and three unrestricted estuarine brackish-water marshes along the Woodbridge River, a tributary of the Arthur Kill, in Woodbridge, New Jersey (figure 1). The impounded marsh was dominated by *Phragmites*, and the three natural marshes were dominated by *S. patens/I. frutescens*, *S. alterniflora*, and *Phragmites*, respectively. The objective of the comparison was to evaluate the effects of impoundment on marsh structure and function and to identify possible trajectories of ecosystem change following removal of the levee and re-introduction of tidal inundation. Measured parameters included porewater salinity and sulfide, macrophyte production (aboveground biomass, macro-organic matter) and structure (species richness and diversity, stem density, number of flowering stems, stem height), and soil organic carbon, nitrogen, and phosphorus pools and accumulation.

Methods

Site history

There is a long history of human activity on the Woodbridge River. Native Americans frequented the area, following the river down to the Arthur Kill. The first gristmill was built on the river in 1671, followed by sawmills and firebrick factories in later years (Ernie Oros, Woodbridge River Watch, personal communication). Today the Woodbridge River corridor is almost entirely developed for commercial, industrial, and residential uses.

The impounded marsh was altered by ditching for mosquito control in the 1930s and the construction of a levee along the perimeter of the marsh in the 1960s (E. Oros, pers. comm.). Based on review of historical aerial photography (from 1932 to present), much

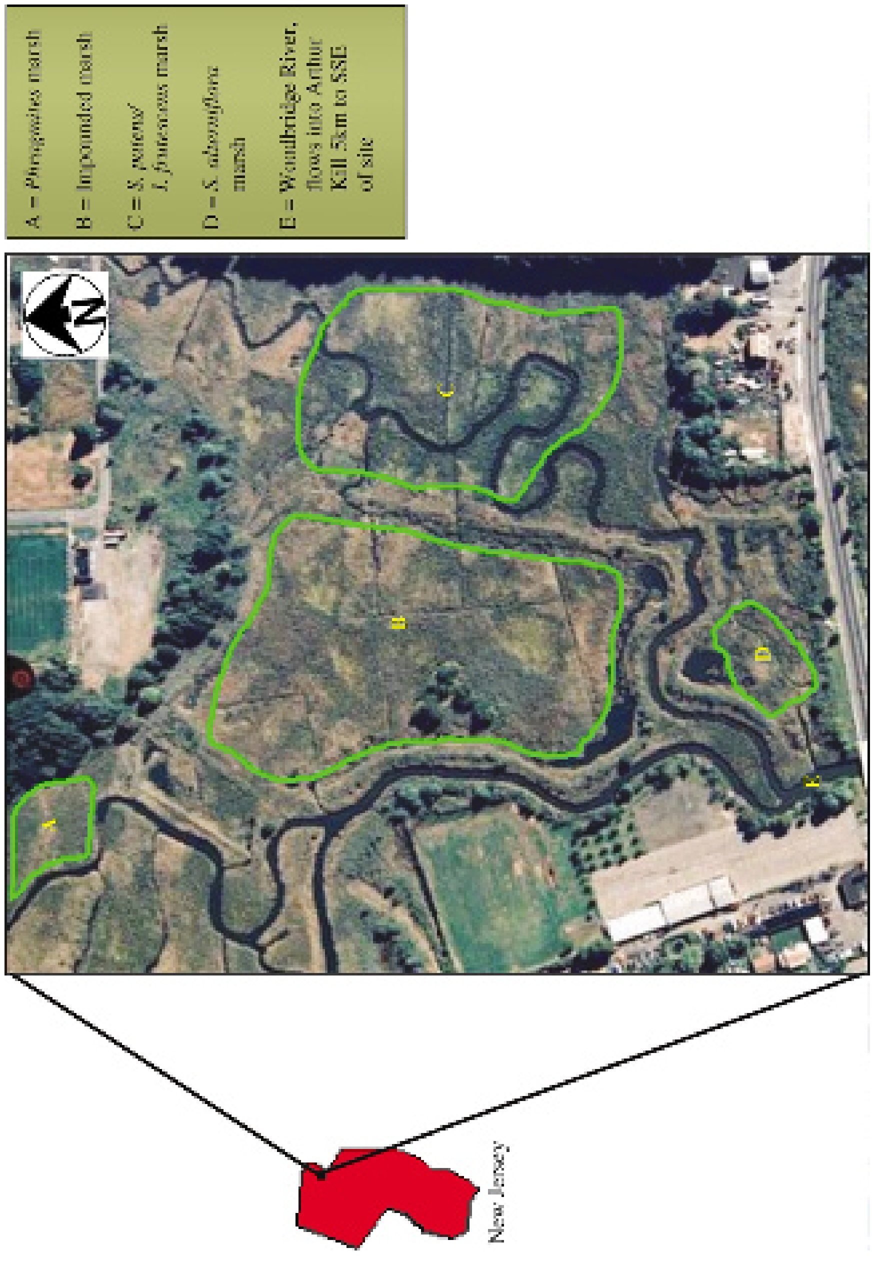

Figure 1. Aerial photograph of Woodbridge River, NJ, identifying impounded marsh and reference marshes.

of the marsh was historically dominated by high (*S. patens/Distichlis spicata/Juncus* sp.) marsh vegetation with some areas of *Iva frutescens* shrubs. It was most likely inundated only by spring and storm tidal flooding, as is the case for typical high marshes along the northeastern coast of the United States (Bertness, 1991). From the mid-1940s to the 1960s, the marsh was mowed to harvest salt hay (E. Oros, pers. comm.). Beginning in the 1970s, the native high marsh vegetation was slowly replaced by *Phragmites*, which began spreading on the levee and then along berms adjacent to the ditches.

Site description

The 25-acre impounded marsh is higher in elevation, at 3.3–4.3 feet above mean sea level (MSL), than the three reference marshes and supports dense stands of *Phragmites*, with vestigial high marsh meadows dominated by *D. spicata* and *S. patens*. Planned restoration of the marsh will involve removal of the levee, grading the marsh to lower the elevation to 1.5 ± 0.1 feet above MSL, reintroducing tidal inundation, and establishing cordgrass (*S. alterniflora*).

The reference marshes, dominated by *S. alterniflora*, *S. patens/I. frutescens*, and *Phragmites*, were chosen to represent alternative restoration endpoints. They exist along a gradient of decreasing tidal inundation, moving upstream along the tidal creek (figure 1). The *S. alterniflora* marsh covers approximately 6 acres and is located at the lowest elevation (0.7–2.3 feet above MSL), adjacent to the tidal creek, where it is inundated twice daily to a depth of 20 cm. This marsh represents the target plant community for the planned restoration. The *S. patens/I. frutescens* marsh covers approximately 10 acres and is located further upstream (3.0–3.3 feet above MSL). It is inundated by tidal water much less frequently and is dominated by *S. patens*, *I. frutescens*, and *D. spicata*. This marsh is representative of "high" marshes of the northeastern coast of the United States that are inundated only during spring or storm tides (Bertness, 1991). The *Phragmites* marsh, at 2.7–3.0 feet above MSL, is a 6-acre brackish marsh located still further upstream and dominated by dense stands of *Phragmites*.

Vegetation and soils were sampled near the end of the growing season, in September 2000, in the impounded marsh and in each of the three reference marshes. Vegetation was sampled again in September 2001. Surface and porewater salinity and sulfide were measured in September 2001 in all four marshes. All four marshes will be monitored to assess ecosystem development for at least three years following restoration of the impounded marsh.

Field sampling

Porewater

Porewater was collected from wells installed at low (streamside), medium (marsh plain), and high (backmarsh) elevations along three transects in each reference marsh in September 2001. At the impounded marsh, porewater was sampled at five points along each transect. The wells were made from PVC pipe and sampled water from a depth of 20 cm below the soil surface. Wells were capped to prevent inflow of rainwater and soil was tamped down around each well to prevent tidal surface water from running down the outside of the well and into the screening. Mean porewater sulfide of the *S. alterniflora* marsh was 15.8 ppm as compared to 0.02–0.05 ppm in the tidal creek, suggesting that contamination of wells with surface water was minimal. Water was pumped out of each well and fresh porewater was collected several hours later and analyzed for salinity and sulfide. Porewater samples were collected at high tide one day and the process was repeated at low tide the next day. Salinity was measured in the field using a refractometer. Sulfide in the porewater sample was preserved in the field with NaOH antioxidant buffer and analyzed for sulfide using a combination silver/sulfide electrode in the lab (APHA, 1998).

Vegetation

Vegetation was sampled using transects established at low, medium, and high elevations in each marsh. Five 0.25 m^2 plots were randomly clipped along each transect in September 2000 and 2001. The clipped material was separated by species, dead material was discarded, and the number of stems and flowering stems were counted. It was difficult to distinguish between *S. patens* and *Juncus gerardii* at the end of the growing season, so these two species were combined for aboveground biomass. Stem height was determined by measuring the length of the five longest stems of each species. Aboveground biomass was measured by drying the clipped material at 70°C and weighing to the nearest 0.1 g. Species diversity was calculated using the Shannon-Wiener Diversity Index (Begon *et al*., 1990).

In 2001, macro-organic matter was sampled by collecting a 30 cm deep soil core (8.5 cm diameter) from each clipped plot and subdividing the core into 2 depths (0–10 cm and 10–30 cm). The root material was separated from the soil by washing on a 2 mm diameter mesh screen. The roots and rhizomes remaining on the screen were dried at 70°C and weighed.

Soil organic carbon, nitrogen, and phosphorus

Soils were sampled in 2000 by taking a core (8.5 cm diameter by 30 cm deep) from each clipped plot. Cores were sectioned into 0–10 cm and 10–30 cm depths in the field. Soil material was air-dried, weighed (to determine bulk density), ground and sieved through a 2 mm-mesh diameter screen and was analyzed for organic carbon, total nitrogen and total phosphorus. Organic C and total N were determined using a Perkin-Elmer 2400 CHN analyzer. Analysis of bituminous coal (NIST standard no. 1632b) yielded values that were 98.1% of the certified value for carbon (76.86%) and 97.4% for nitrogen (1.56%). Total P was measured in nitric-perchloric digests (Sommers and Nelson, 1972) using the method of Murphy and Riley (1962). Acid digestion of an estuarine sediment standard (NIST standard no. 1646a) recovered 88.7% of the certified phosphorus content (270 μg/g). Bulk density was calculated from the air-dried weights by applying a moisture correction factor determined by drying a subsample at 70°C overnight. Soil organic C, N and P pools (0–10 cm and 10–30 cm depths) were calculated using depth-weighted bulk density and nutrient concentration measurements.

Soil nutrient and mineral accumulation

Rates of soil accretion were measured by collecting two cores (8.5 cm diameter by 30 cm deep) from each marsh, one at the streamside and one in the backmarsh. Each core was sectioned into 2 cm increments and each increment was air-dried, ground, and sieved through a 2 mm-mesh diameter screen. Samples were oven-dried at 70°C for 24 hours and 5–10 g subsamples were used for radiometric dating. Accretion rates were determined by gamma analysis of the 661 keV photopeak for ^{137}Cs using a low energy germanium detector (Canberra Industries, Meriden, CT). Cesium-137, produced from aboveground thermonuclear weapons testing, can be used as a marker horizon in soils. The ^{137}Cs maximum in the soil profile corresponds to the soil surface in approximately 1964, when atmospheric deposition was at its peak (Ritchie and McHenry, 1990). Only those profiles that contained well-defined, interpretable ^{137}Cs peaks (see, for example, figure 5) were used to calculate accretion rates.

Recent (post-1964) nutrient accumulation rates were calculated using soil accretion rates, bulk density, and carbon, nitrogen and phosphorus concentrations from the 0–10 cm depth. Mineral accumulation rates were calculated using soil accretion rates, bulk density, and the proportion of mineral matter (0–10 cm). Percent mineral content was calculated based on the assumption that organic matter is composed of 50% carbon.

Statistical analysis

Analysis of Variance (ANOVA) was used to test the null hypothesis that the measured parameters (plant community attributes, porewater salinity and sulfide, and soil characteristics) did not differ between the reference and impounded marshes (SAS, 1990; SPSS, 2000). Multiple mean comparisons were performed using Tukey's honestly significant difference (HSD) test (SPSS, 2000). All tests of significance were made at an alpha level of 0.05.

Results and discussion

Porewater

Porewater salinity was consistently greater in the *Phragmites*-dominated marsh as compared to the other marshes and significant differences in salinity between marshes were observed at both low and high tide. At low tide, mean salinity was significantly greater in the *Phragmites* marsh (24 ± 0.4 ppt) than in the other three marshes (16.1 ± 0.7 to 19.5 ± 1.2 ppt). At high tide, salinity was greatest in the *Phragmites* marsh (26.5 ± 0.4 ppt) and lowest in the *S. alterniflora* marsh (17.2 ± 0.9 ppt). The salinity of the river was 6 ppt at low tide and 17 ppt at high tide. Salinity was likewise consistently greater in all marshes at high tide than at low tide (figure 2(a)).

Porewater sulfide was consistently higher in the *S. alterniflora* marsh as compared to the other marshes. At low tide, the sulfide concentration was highest in the *S. alterniflora* marsh (15.8 ± 6 ppm), compared to 2.2 ± 0.8 ppm to 4.6 ± 1.3 ppm in the *S. patens/I. frutescens*, *Phragmites* and impounded marshes ($p < 0.05$). At high tide, sulfide concentration was again highest in the *S. alterniflora* marsh (9.7 ± 4 ppm) and was lowest in the *Phragmites* marsh (1.2 ± 0.2 ppm; $p < 0.05$). Like salinity, sulfide concentrations varied during the tidal cycle. Sulfide was higher in the three unimpounded reference marshes at low tide than at high tide, but no difference between tides was observed at the impounded marsh (figure 2(b)).

Our porewater data suggest that sulfide concentration, an indication of anoxic soil conditions, is more important than salinity in limiting the distribution of *Phragmites* since, of the four marshes, the *Phragmites* marsh contained the highest porewater salinity and the

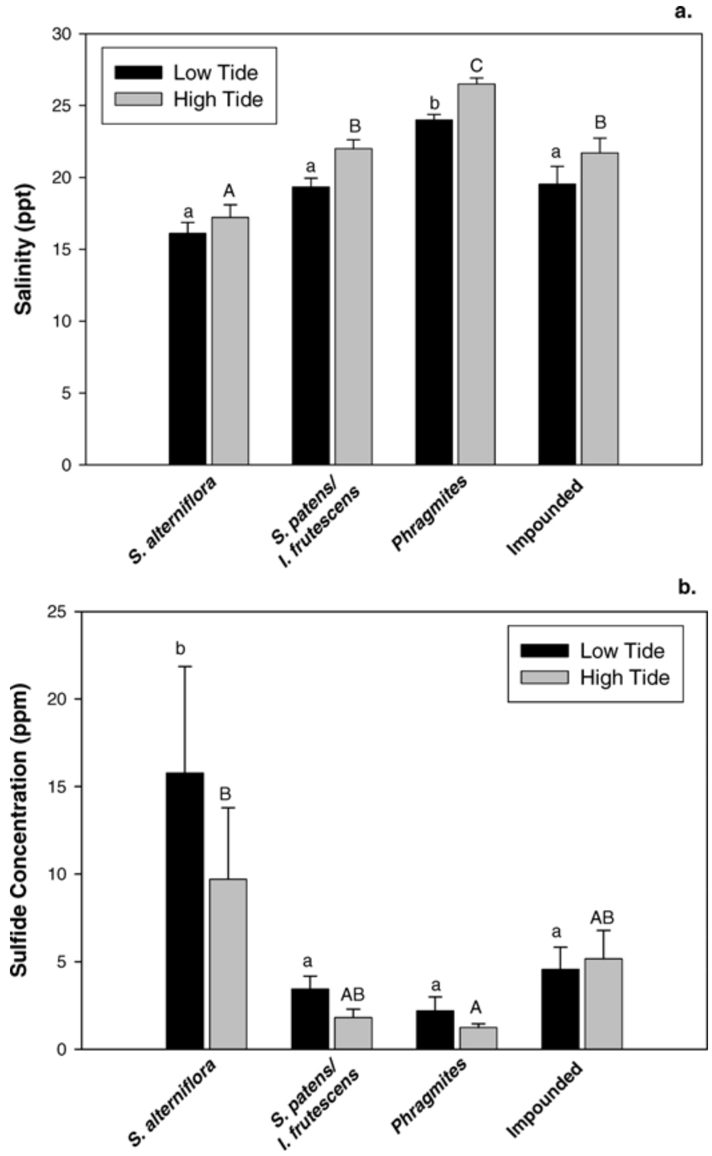

Figure 2. Porewater salinity (a) and sulfide (b) concentrations of impounded and reference marshes, September 2001. Means for each tide separated by different letters are significantly different ($p < 0.05$, according to Tukey's HSD test) (lower case letters are for low tide, capital letters are for high tide).

lowest sulfide concentrations. This is consistent with other studies. In a study conducted on the mouth of the Housatonic River in Connecticut, Chambers (1997) found that while *Phragmites* and *S. alterniflora* were growing throughout a porewater salinity range from 12 to 24 ppt, sulfide concentrations were always higher than 600 μM (19.2 ppm S^{2-}) in

S. alterniflora stands while *Phragmites* productivity was reduced where sulfide concentrations were higher than 400 μM (12.8 ppm S^{2-}). This was attributed to sulfide inhibition of the uptake of nitrogen by *Phragmites* relative to *S. alterniflora*, which is better adapted to sulfidic soil conditions (Chambers *et al.*, 1998).

This may explain the influence of elevation and tidal inundation on the distribution of *Phragmites*. *Phragmites* is typically found growing at higher elevations than *S. alterniflora* (Roman *et al.*, 1984; Chambers *et al.*, 1999). In our study, the *Phragmites* site was approximately 2 feet above MSL higher than the *S. alterniflora* site, and was inundated much less frequently. As our porewater data indicate, the absence of *Phragmites* in the *S. alterniflora* site may be linked more to the high sulfide concentrations associated with tidal inundation than with salinity. Additionally, several experimental studies have found that *Phragmites* germination and regrowth after disturbance is negatively affected by flooding and the associated lack of oxygen (Hellings and Gallagher, 1992; Wijte and Gallagher, 1996).

Vegetation

End of season standing crop biomass in 2000 ranged from 564 to 809 g/m^2, with no difference between the impounded and reference marshes (figure 3(a)). Aboveground biomass was somewhat higher in the *Phragmites* (809$\pm$150 g/m^2) and *S. alterniflora* (750 $\pm$ 50 g/m^2) marshes as compared to the high marsh (*S. patens/I. frutescens*, 564 $\pm$ 63 g/m^2) and impounded marsh (588 $\pm$ 57 g/m^2). In 2001, the *S. alterniflora* (706 $\pm$ 72 g/m^2) and *Phragmites* (709 $\pm$ 90 g/m^2) marshes had significantly more standing crop biomass than the *S. patens/ I. frutescens* (378$\pm$54 g/m^2) and impounded marshes (431$\pm$44 g/m^2) (figure 3(a)). *Spartina alterniflora* biomass was within the range reported for natural *S. alterniflora* marshes along the Northeast (NJ, CT, MA) coast (592–1592 g/m^2; Craft *et al.*, 1999). Aboveground biomass in the *Phragmites* and the high marsh were also within the ranges reported for Northeastern marshes dominated by *Phragmites* (1300–2400 g/m^2) and *S. patens*, *D. spicata*, and *J. gerardii* (199–694 g/m^2), respectively (Windham and Lathrop, 1999; Warren *et al.*, 2001; Windham, 2001).

The *S. patens/I. frutescens* marsh had significantly more stems (1281 $\pm$ 290 stems/m^2) than the other three marshes (98 $\pm$ 21 to 391 $\pm$ 151 stems/m^2; Table 1) in 2000. Higher stem density in the *S. patens/I. frutescens* marsh was attributed to the abundance of species with small diameter grass-like stems, such as *D. spicata*, *S. patens*, and *J. gerardii*. The dominant species in the other marshes (*S. alterniflora*, *P. australis*) possess stems of larger diameter, with correspondingly lower stem density.

There was a weak but significant inverse relationship between stem height and density of marsh vegetation ($R = -0.28$, $p < 0.01$). Mean stem heights were significantly lower in the *S. patens/I. frutescens* marsh (87$\pm$5 cm) as compared to the other marshes. The *Phragmites* marsh had the tallest stems (194 $\pm$ 13 cm) and the lowest stem density (98 $\pm$ 21 stems/m^2, Table 1). The preponderance of short stems in the *S. patens/I. frutescens* marsh was attributed to the dominance of *D. spicata* (74 $\pm$ 5 cm) and *S. patens/J. gerardii* (71 $\pm$ 5 cm), which have short growth forms. *Iva frutescens* (100–133 cm) and *S. alterniflora* (132 cm) were intermediate in height whereas *P. australis* exhibited the greatest mean stem heights (185–206 cm; Table 1).

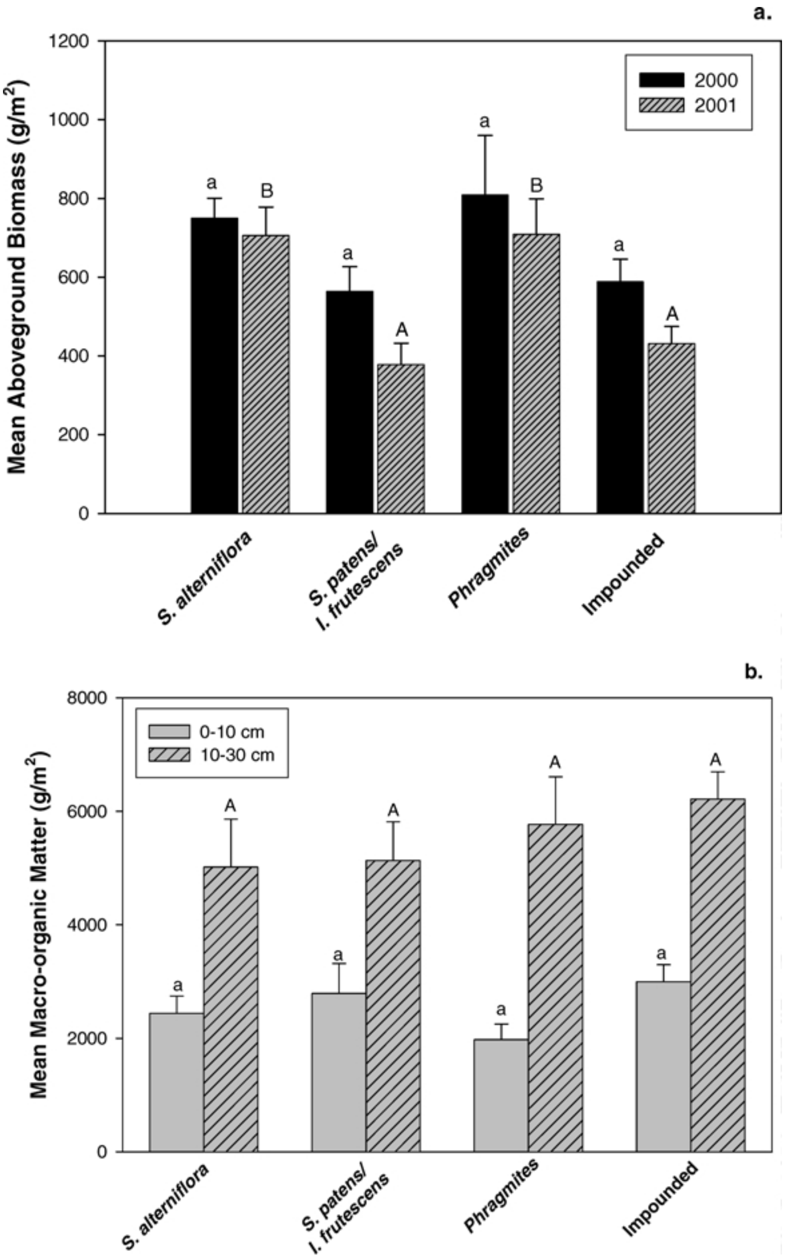

Figure 3. Mean aboveground biomass (g/m^2) (a) and mean macro-organic matter (g/m^2) (b) in impounded and reference marshes. Means separated by different letters are significantly different ($p \leq 0.05$, according to Tukey's HSD test). For aboveground biomass, lower case letters indicate differences between marshes in 2000 and capital letters indicate differences in 2001. For macro-organic matter, lower case letters indicate differences between marshes at the 0–10 cm depth and capital letters indicate differences between marshes at the 10–30 cm depth.

Table 1. Stem density, stem height, and number of flowering stems for impounded and reference marshes, September 2000. Different letters indicate a significant difference between marsh means ($p \leq 0.05$, according to Tukey's HSD test).

		By dominant species (with >15 stems/m²)				
	Mean	*D. spicata*	*I. frutescens*	*Phragmites*	*S. alterniflora*	*S. patens/ J. gerardii*
		Stem density (per m²) (±SE)				
S. alterniflora	295 ± 55a	73	–	–	221	–
S. patens/I. frutescens	1281 ± 290b	486	17	–	–	768
Phragmites	98 ± 22a	18	–	71	–	
Impounded	391 ± 151a	142	–	74	–	158

		By dominant species (>12% cover)				
	Mean	*D. spicata*	*I. frutescens*	*Phragmites*	*S. alterniflora*	*S. patens/ J. gerardii*
		Stem height (cm) (±SE)				
S. alterniflora	128 ± 4b	92	–	–	132	–
S. patens/I. frutescens	87 ± 5a	74	100	–	131	71
Phragmites	194 ± 13c		134	206	–	
Impounded	145 ± 15b	70	109	185	–	69

		By dominant species (>1 flowering stem/m²)					
	Mean	*Aster sp.*	*D. spicata*	*I. frutescens*	*Phragmites*	*S. patens*	*Scirpus robustus*
		Number of flowering stems (per m²) (±SE)					
S. alterniflora	1.1 ± 0.6a	–	–	–	–	–	–
S. patens/I. frutescens	109.6 ± 44.6b	–	34.4	14.1	–	58.4	1.3
Phragmites	21.1 ± 4.0a	–	–	1.9	19.2	–	–
Impounded	33.6 ± 9.3a	6.4	5.3	2.4	16.3	2.4	–

In addition to greater stem density, the *S. patens/I. frutescens* marsh had significantly more flowering stems (110 ± 45 per m^2; Table 1), as *S. patens*, *D. spicata*, and *I. frutescens* were in flower at the time of sampling. Flowering *S. alterniflora* stems were not observed at the *S. alterniflora* marsh in September 2000 although a few *S. patens* stems were flowering. A few lingering *P. australis* blooms were observed in the *Phragmites* and impounded marshes.

The quantity of macro-organic matter, the living and dead root and rhizome mat, did not differ between marshes (figure 3(b)). Macro-organic matter (0–30 cm) ranged from 7452 ± 768 g/m^2 in the *S. alterniflora* marsh to 9212 ± 510 g/m^2 in the impounded marsh. Across all marshes, macro-organic matter was greater in the subsurface layer (10–30 cm)

Table 2. Species richness, diversity, dominant species, and species list for impounded and reference marshes, September 2000. Shannon-Wiener species diversity was determined using the proportion of total biomass (by species) in clipped plots.

Marsh	Species richness (species/0.25 m^2)	Diversity	Dominant species	Species list (total)
S. alterniflora	1.27	0.16	*S. alterniflora*	*D. spicata, S. patens* (3)
S. patens/I. frutescens	2.80	1.33	*I. frutescens*	*S. patens, D. spicata, Scirpus robustus, S. alterniflora, S. cynosuroides, Aster* sp., *J. gerardii* (8)
Phragmites	1.47	0.40	*P. australis*	*I. frutescens, D. spicata, S. patens, J. gerardii* (5)
Impounded	2.00	0.78	*P. australis*	*I. frutescens, D. spicata, S. patens, Aster* sp., *Solidago* sp., *J. gerardii* (7)

than in the surface layer (0–10 cm). Part of this difference may be accounted for by the fact that the volume of the subsurface depth increment was twice that of the surface layer. When normalized to the same volume, the mean macro-organic matter of the subsurface layer was greater than the surface layer in the *S. alterniflora*, *Phragmites* and impounded marshes. Macro-organic matter in the *S. alterniflora* marsh was within the range reported for natural *S. alterniflora* marshes along the Northeast coast (2520–11,400 g/m^2; Craft *et al*., 1999).

Species diversity, calculated from the proportion of aboveground biomass in each quadrat in September 2000, was greatest in the high (*S. patens/I. frutescens*) marsh ($H = 1.33$) and lowest in the *S. alterniflora* marsh ($H = 0.16$; Table 2). The *S. patens/I. frutescens* marsh contained a total of eight species, with a mean richness of 2.8 species/0.25 m^2. The most abundant species were *I. frutescens*, *D. spicata*, *S. patens*, and *J. gerardii*, accounting for 89% of the total aboveground biomass. The *S. alterniflora* marsh only had a total of three species, with a mean richness of 1.27 species/0.25 m^2. *Spartina alterniflora* accounted for 96% of the aboveground biomass in this marsh. In the impounded marsh *P. australis* was the dominant species (77%), while *I. frutescens* was the next most abundant (14%). Species diversity was almost twice as high in the impounded marsh as in the *Phragmites*-dominated reference marsh, primarily due to the presence of high marsh meadows dominated by *D. spicata*, *S. patens*, *J. gerardii*, *Aster* sp. and *Solidago* sp. (Table 2, figure 4).

Soils

Bulk density was significantly higher in the *S. alterniflora* (0.36 ± 0.01 g/cm^3) and *P. australis* (0.38 ± 0.03 g/cm^3) marshes as compared to the *S. patens/I. frutescens* (0.21 ± 0.01 g/cm^3) and impounded marshes (0.21 ± 0.01 g/cm^3; Table 3). Higher bulk density reflects greater mineral sediment (e.g. sand, silt, clay) of higher particle density relative to soil organic matter (Craft *et al*., 1993). Bulk density also varied between depths in three of the four marshes. Bulk density typically increases with depth in mineral soils, as was observed in the *Phragmites* and impounded marshes. However, bulk density was higher in

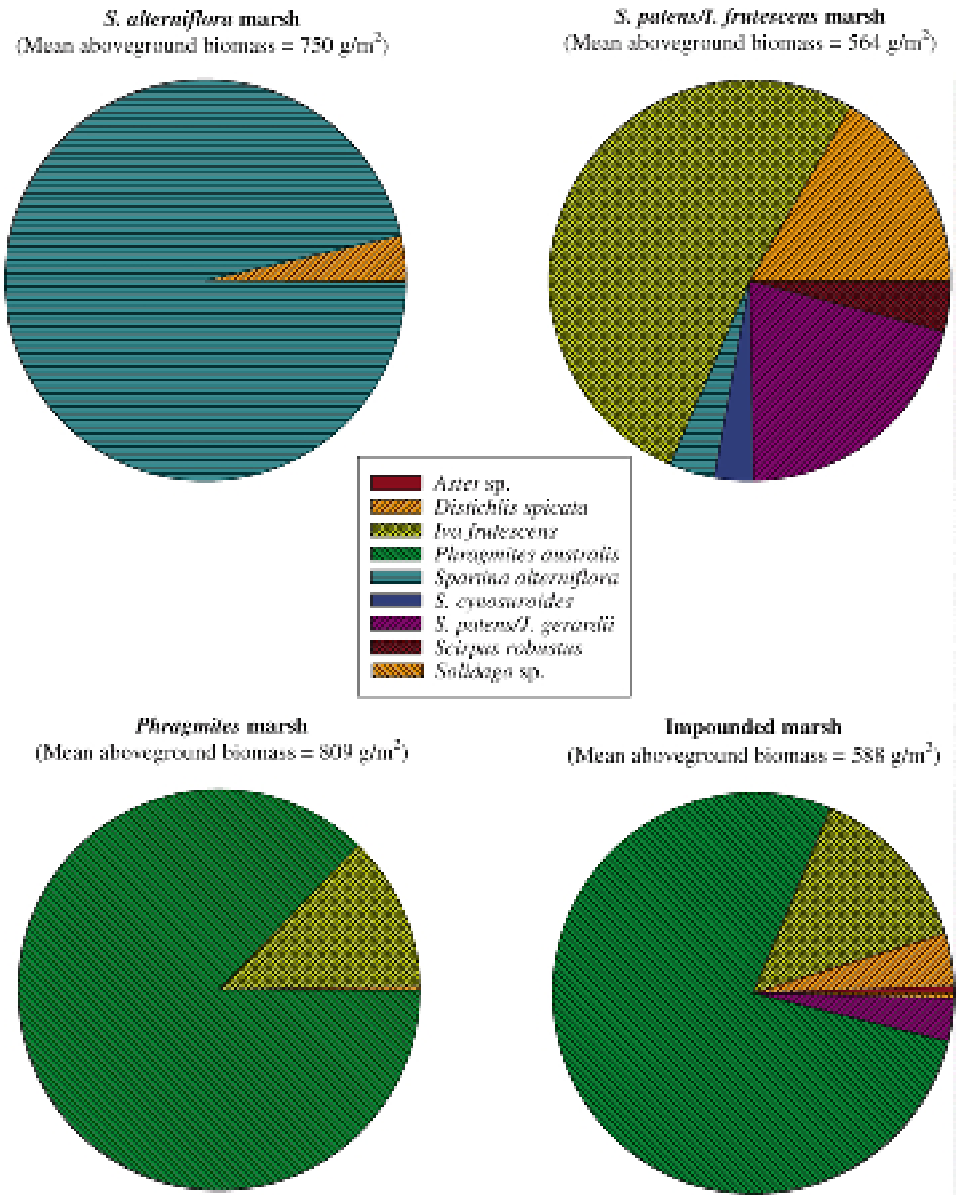

Figure 4. Vegetation composition, based on proportion of total biomass in clipped plots, in impounded and reference marshes, September 2000.

Table 3. Soil characteristics (bulk density, organic carbon, nitrogen, phosphorus; mean ± SE) of impounded and reference marshes, September 2000. Different letters indicate a significant difference between marsh means ($p \leq 0.05$, according to Tukey's HSD test).

	S. alterniflora	*S. patens/I. frutescens*	*Phragmites*	Impounded
0–30 cm				
Bulk Density (g/cm^3)	0.36 ± 0.01b	0.21 ± 0.01a	0.38 ± 0.03b	0.21 ± 0.01a
Organic Carbon (%)	7.50 ± 0.2a	16.89 ± 1.0c	11.63 ± 0.9b	16.99 ± 1.0c
Total Nitrogen (%)	0.47 ± 0.02a	1.03 ± 0.06c	0.71 ± 0.06b	1.04 ± 0.07c
Phosphorus (μg/g)	1288 ± 107a	1930 ± 205b	1364 ± 100a	1247 ± 125a
0–10 cm				
Bulk Density (g/cm^3)	0.38 ± 0.02c	0.21 ± 0.02ab	0.31 ± 0.04bc	0.19 ± 0.01a
Organic Carbon (%)	7.90 ± 0.3a	17.25 ± 1.4b	14.38 ± 1.1b	18.40 ± 1.1b
Total Nitrogen (%)	0.51 ± 0.02a	1.15 ± 0.08bc	0.94 ± 0.08b	1.23 ± 0.07c
Phosphorus (μg/g)	1535 ± 161a	2414 ± 330b	1633 ± 105a	1726 ± 164ab
C:N Ratio	17.9 ± 0.3a	17.4 ± 0.3a	18.4 ± 0.7a	17.4 ± 0.4a
N:P Ratio	9.2 ± 1.4a	12.4 ± 1.5a	13.0 ± 1.0ab	17.6 ± 1.5b
10–30 cm				
Bulk Density (g/cm^3)	0.35 ± 0.02b	0.21 ± 0.02a	0.45 ± 0.04b	0.24 ± 0.02a
Organic Carbon (%)	7.09 ± 0.3a	16.54 ± 1.5b	8.88 ± 1.0a	15.59 ± 1.7b
Total Nitrogen (%)	0.42 ± 0.02a	0.91 ± 0.08b	0.49 ± 0.06a	0.84 ± 0.08b
Phosphorus (μg/g)	1041 ± 114ab	1445 ± 178b	1095 ± 141ab	769 ± 71a
C:N Ratio	20.0 ± 0.4a	21.0 ± 0.3ab	21.7 ± 0.6b	21.7 ± 0.3b
N:P Ratio	10.2 ± 1.1a	17.5 ± 2.7ab	10.9 ± 1.3a	24.6 ± 2.6b

the 0–10 cm depth than in the 10–30 cm depth in the *S. alterniflora* marsh. This may be due to greater recent sedimentation in this marsh as compared to the other marshes (see "Sediment and Nutrient Accumulation"). Bulk density did not vary with depth in the *S. patens/I. frutescens* marsh.

Organic carbon was lowest in the *S. alterniflora* marsh (7.50 ± 0.2%), intermediate in the *Phragmites* marsh (11.63 ± 0.9%), and highest in the *S. patens/I. frutescens* (16.89 ± 1.0%) and impounded marshes (16.99 ± 1.0%; Table 3). Except for the *Phragmites* marsh, which had higher surface (14.4%) than subsurface (8.9%) organic C, there was no significant difference in soil organic C between depths. Soils of the *S. patens/I. frutescens* and impounded marshes are classified as "true" organic soils (Histosols) based on organic carbon content greater than 12% in the top 30 cm. Neither the *S. alterniflora* nor *Phragmites* marshes met the criteria for an organic soil (organic C = 12–18%, to a depth >30–40 cm). Soils of the *Phragmites* marsh are classified as Inceptisols (histic humaquepts) based on the presence of a surface (0–10 cm) layer rich in organic matter. The *S. alterniflora* soil was classified as an Entisol (hydraquent). Low soil organic C in the *S. alterniflora* marsh relative to the other marshes likely reflects greater tidal flushing that leads to enhanced decomposition, export of organic matter, and increased sediment deposition that "dilutes" soil organic matter.

Total nitrogen exhibited the same pattern as organic C, which is expected since C and N are closely bound in organic matter (Craft *et al*., 1991). The *S. alterniflora* marsh had the lowest total nitrogen ($0.47 \pm 0.02\%$), the *Phragmites* marsh had intermediate N ($0.71 \pm 0.06\%$), and the *S. patens/I. frutescens* and impounded marshes had the highest N content ($1.03 \pm 0.06\%$ and $1.04 \pm 0.07\%$, respectively; Table 3). Soil phosphorus was significantly higher in the *S. patens/I. frutescens* marsh (1930 ± 205 μg/g) than in the other marshes (Table 3). In all marshes, soil total N was greater in surface than subsurface soils. Many estuarine marshes are limited by N (Broome *et al*., 1975; Sullivan and Daiber, 1974; Valiela and Teal, 1974), so that higher surface soil N probably reflects efficient N retention by wetland biota (plants, microbes) in response to N limitation. Soil P also was greater in surface than subsurface soils at all marshes (Table 3).

Carbon to nitrogen ratio did not vary between marshes at the 0–10 cm depth (17.4 ± 0.3 to 18.4 ± 0.7). At the 10–30 cm depth, the C:N ratio was lower in the *S. alterniflora* marsh (20 ± 0.4) as compared to the *Phragmites* (21.7 ± 0.3) and impounded marshes (21.7 ± 0.3; Table 3). Soil N:P was significantly lower in the *S. alterniflora* marsh (9–10) as compared to the impounded marsh (18–25; Table 3). N:P was intermediate in the *S. patens/I. frutescens* (12–18) and *Phragmites* (11–13) marshes. There was no consistent difference in N:P between 0–10 cm and 10–30 cm depths. In studies on freshwater wetland vegetation, Koerselman and Mueleman (1996) found that N:P less than 15 indicates that N is the primary limiting nutrient, while a ratio higher than 15 indicates that P is limiting. This threshold has been applied to both freshwater and estuarine vegetation and soils (Bedford *et al*., 1999). Low N:P in the *S. alterniflora* marsh suggests that nitrogen limits productivity of this marsh. Higher N:P in the impounded marsh indicates a shift more towards phosphorus limitation.

Soil organic carbon pools (0–30 cm depth) ranged from 7,800 g/m^2 to 12,000 g/m^2, with significantly less carbon in the *S. alterniflora* marsh ($7{,}839 \pm 341$ g/m^2) as compared to the other marshes (9,398–11,929 g/m^2). Organic carbon pools were significantly greater in the *Phragmites* marsh ($11{,}929 \pm 584$ g/m^2) as compared to the other marshes (7,839–10,062 g/m^2). Nitrogen pools followed the same pattern, ranging from 480 g/m^2 to 710 g/m^2. Phosphorus pools ranged from a low of 70 ± 6 g/m^2 in the impounded marsh to a high of 154 ± 19 g/m^2 in the *Phragmites* marsh. Nutrient pools in the four marshes were higher than the ranges recorded for natural saltwater cordgrass marshes along the Atlantic coast of the United States (620-4,160 g C/m^2; 38–270 g N/m^2; 12–88 g P/m^2; Craft, 2001) and may reflect greater inputs of sediments and nutrients in stormwater from the highly urbanized watershed.

Sediment and nutrient accumulation

Cesium-137 exhibited interpretable profiles (a pronounced peak at depth, corresponding to 1964) in all four marshes (figure 5). Recent accretion rates based on ^{137}Cs were lowest in the impounded marsh (2.8 ± 0.28 mm/yr) and highest in the *S. alterniflora* marsh (4.7 mm/yr; Table 4, figure 5). The *Phragmites* (3.6 ± 0 mm/yr) and *S. patens/I. frutescens* (3.6 ± 0.55 mm/yr) marshes had intermediate accretion rates. These accretion rates were within the range reported for natural cordgrass marshes (3.7 ± 0.03 mm/yr) and restricted marshes (2.9 ± 0.03 mm/yr) in Connecticut (Anisfeld *et al*., 1999). Generally, streamside elevations

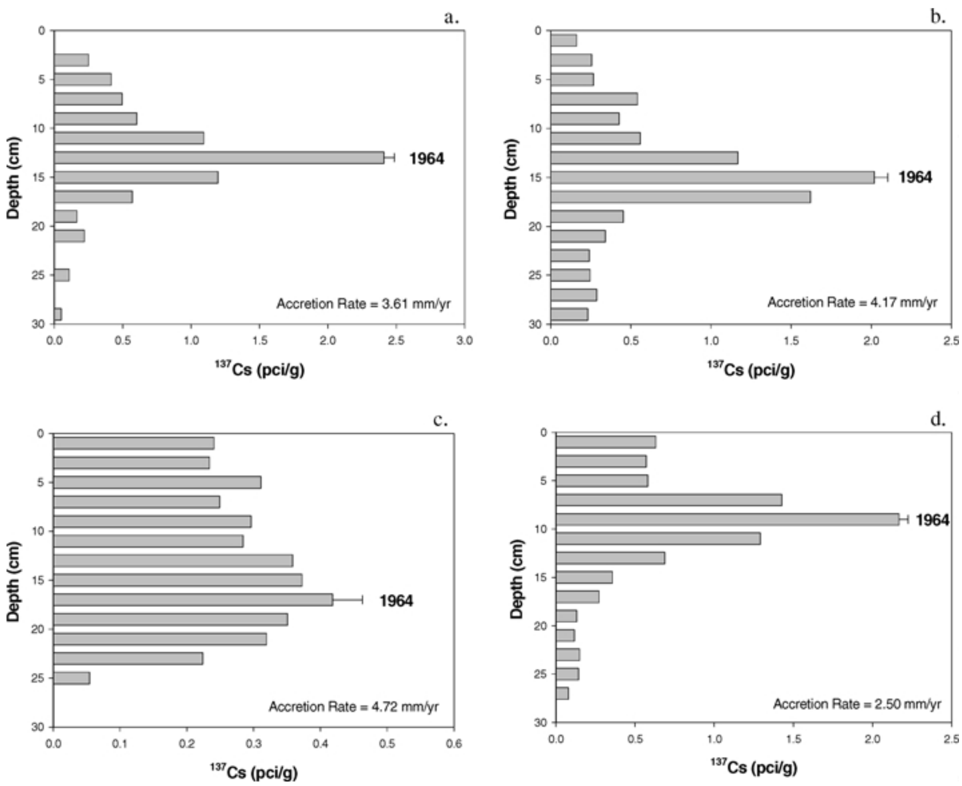

Figure 5. Representative ^{137}Cs profiles, from (a) *Phragmites* streamside, (b) *S. patens/I. frutescens* streamside, (c) *S. alterniflora* backmarsh, and (d) impounded backmarsh sites, September 2000.

had higher accretion rates than backmarsh elevations (Table 4), most likely due to greater tidal inundation and sediment deposition along the tidal creek (Craft *et al*., 1993).

Rates of mineral sediment accumulation based on ^{137}Cs were 2–5 times higher in the *S. alterniflora* marsh (1515 g/m^2/yr) as compared to the *Phragmites* (789 $\pm$ 0 g/m^2/yr), *S. patens/I. frutescens* (499 $\pm$ 77 g/m^2/yr) and impounded (335 $\pm$ 33 g/m^2/yr) marshes (Table 4). Our values are within the range (160–2670 g/m^2/yr) reported for natural Atlantic coast saltwater cordgrass marshes (Craft, 2001). High mineral accumulation in the *S. alterniflora* marsh relative to the other marshes reflects the high bulk density and low carbon content of the soil that results from greater sediment inputs to the marsh.

Recent rates of carbon accumulation ranged from 88 g/m^2/yr in the impounded marsh to 160 g/m^2/yr in the *Phragmites* marsh (Table 4). Nitrogen accumulation exhibited a similar pattern, ranging from 5.9 g/m^2/yr in the impounded marsh to 10.4 g/m^2/yr in the *Phragmites* marsh. Carbon and nitrogen accumulation rates are consistent with rates reported for natural Atlantic coast saltwater cordgrass marshes (21–215 g C/m^2/yr; 1.3–11 g N/m^2/yr; Craft, 2001). Phosphorus accumulation rates were highest in the *S. alterniflora* marsh (2.76 g/m^2/yr) and lowest in the impounded marsh (0.91 $\pm$ 0.09 g/m^2/yr). Phosphorus

Table 4. Sediment accretion and mineral and nutrient accumulation rates, based on ^{137}Cs, of the 0–10 cm depth of impounded and reference marshes, September 2000. We were unable to determine an accretion rate for the *S. alterniflora* streamside site.

	Accretion (mm/yr)	Mineral matter (g/m^2/yr)	Carbon (g/m^2/yr)	Nitrogen (g/m^2/yr)	Phosphorus (g/m^2/yr)
Phragmites (streamside)	3.61	789	160	10.4	1.81
(backmarsh)	3.61	789	160	10.4	1.81
S. alterniflora (streamside)	–	–	–	–	–
(backmarsh)	4.72	1515	142	9.3	2.76
S. patens/I. frutescens (streamside)	4.17	576	152	10.1	2.12
(backmarsh)	3.06	422	111	7.4	1.56
Impounded (streamside)	3.06	369	107	7.2	1.01
(backmarsh)	2.50	302	88	5.9	0.82

accumulation in the four marshes was higher than the range reported for natural Atlantic coast saltwater cordgrass marshes (0–0.4 g/m^2/yr; Craft, 2001), possibly due to higher erosion rates in this urbanized watershed, resulting in inputs of P adsorbed to clay particles.

The impounded marsh had the lowest rates of sediment, C, N, and P accumulation, reflecting the greater degree of isolation from tidal inundation as compared to the other marshes. In contrast, sediment and phosphorus accumulation were highest in the *S. alterniflora* marsh where low intertidal elevation and proximity to the tidal creek favor sediment and phosphorus deposition.

A review of other studies that compared the functioning of impounded/restricted versus natural marshes indicates that impoundment and tidal restriction generally results in lower sedimentation rates (Table 5). Impounded or restricted marshes consistently had lower accretion rates and lower organic matter accumulation as well. Impoundment thus

Table 5. Comparison of mean accretion, sedimentation, and organic matter accumulation rates for impounded/restricted and unimpounded estuarine marshes in New Jersey, Connecticut, and Louisiana.

		Accretion rate (mm/yr)	Sediment (g/m^2/yr)	Organic matter* (g/m^2/yr)
NJ (this study)	Impounded	2.8 ± 0.3	340 ± 35	200 ± 10
	Unimpounded	3.8 ± 0.3	820 ± 190	290 ± 18
CT (Anisfeld *et al.*, 1999)	Restricted	2.9 ± 0.03	1000 ± 200	230 ± 40
	Unrestricted	3.7 ± 0.03	680 ± 120	340 ± 30
LA–Fina LaTerre (Cahoon, 1994)	Impounded	0.7 ± 0.1	8 ± 2	20 ± 6
	Unimpounded	3.0 ± 0.9	337 ± 178	150 ± 40
LA–Rockefeller Refuge (Cahoon, 1994)	Impounded	1.2 ± 0.4	90 ± 40	20 ± 6
	Unimpounded	9.8 ± 1.1	2020 ± 270	670 ± 100

*Calculated assuming that organic matter is 50% organic C.

diminishes the water quality benefits of marshes by eliminating sediment (and P) deposition and reducing storage of N and P in soil organic matter. Impounded marshes are also more susceptible to subsidence, due to both decreased sediment and organic matter accumulation and increased decomposition of exposed sediments.

Conclusions

Our findings indicate that the impounded marsh in the urbanized Woodbridge River watershed does not provide the same level of water quality functions such as sediment and nutrient retention as adjacent natural marshes that experience tidal inundation, nor does it contain the high plant species diversity found in the high marsh. The *Phragmites* and *S. alterniflora* marshes had greater mean aboveground biomass than the *S. patens/I. frutescens* and impounded marshes, yet species diversity was greater in the *S. patens/I. frutescens* marsh because of the presence of species-rich high marsh meadows. The *S. alterniflora* marsh, which is flooded twice daily by the tides, had the highest rates of sediment and P accumulation of all four marshes, while the impounded marsh had the lowest accumulation rates. Planned reintroduction of tidal inundation by removal of the levee surrounding the impounded marsh will result in improved hydrologic connectivity and enhance water quality functions of the marsh by trapping sediment and sequestering N and P in the soil. Planting and natural revegetation of *S. alterniflora* will further favor the development of salt marsh community structure and food webs. Ongoing monitoring of the impounded marsh and reference marshes following restoration of tidal inundation will allow an assessment of salt marsh ecosystem development in a highly urbanized watershed.

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