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Coastal wetland response to sea level rise in Connecticut and New York

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A R T I C L E I N F O

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A B S T R A C T

The persistence of salt marshes in the landscape depends on their ability to accommodate rising sea level and minimize additional flooding stress. We use sediment cores and water level data from 14 marshes in Connecticut and New York to evaluate how marsh accretion, mineral and organic accumulation, carbon storage, and hydroperiod have changed from 1900 to 2012. We observe a regional acceleration in marsh accretion beginning around 1940, although marsh accretion did not reach parity with sea level rise for several additional decades. Despite a rise in marsh accretion from 1.0 mm yr^{-1} circa 1900 to 3.6 mm yr^{-1} at present, the marsh surface has lost elevation relative to tidal datums. Declining relative elevations have led to increased tidal flooding, particularly in high marsh settings. As flooding increased, organic matter accumulation accelerated at all marshes. Accelerating mineral deposition was only observed in areas of short-form *Spartina alterniflora*. Mineral and organic sediment accumulation co-limit accretion, but organic accumulation was the stronger limiting factor, suggesting that marsh response to sea level rise in the region is sensitive to processes affecting rates of belowground production and decomposition. Marsh carbon storage over the period of study averaged 84 g C m^{-2} yr^{-1}, increasing as accretion accelerated. If marshes remain spatially intact as sea levels rise, these results suggest that marshes have the capacity to become even greater C sinks.

1. Introduction

Coastal wetlands face an uncertain future as sea level rises. Low-end estimates of sea level rise (SLR) suggest that a global increase in mean sea level (MSL) of 19–83 cm is likely by 2100 (Church et al., 2013). In the New York City region, interactions between eustatic sea level rise, glacial isostatic adjustment, and ocean circulation contribute to rates of SLR that are higher than the global average (Horton et al., 2010; Boon, 2012). Given the primacy of tidal flooding in organizing coastal wetlands, the dramatic changes in sea level anticipated during the coming decades have the potential to alter wetland structure and function.

Salt marshes persist in the face of sea level rise by making horizontal and vertical adjustments. Marsh migration into upland zones represents the former category, as it is driven primarily by a horizontally-expanding intertidal zone rather than by vertical accretion or sediment accumulation. However, in many urbanized estuaries, opportunities for horizontal migration are severely limited by roads, seawalls, and other hard structures.

Vertical marsh accretion increases marsh surface elevation and buffers plant communities against flooding stress, a requirement if marshes are to maintain their current spatial extent and vegetation distributions. Tidal sediment deposition and belowground production of roots and rhizomes directly contribute volume to the peat and actively raise the marsh surface (Reed, 1995; Nyman et al., 2006).

Aboveground plant biomass also contributes to marsh accretion, albeit indirectly through litterfall or increased sediment trapping. If plants occupy elevations that are supraoptimal for productivity, additional flooding will increase shoot biomass, thereby reducing water velocities and causing tide-borne sediment to fall out of suspension and accumulate on the marsh surface (Morris et al., 2002). The magnitude of the marsh accretion response varies with suspended sediment concentration, tidal flooding regime, the rate of sea level rise, and marsh canopy response to changes in flooding (Kirwan et al., 2010; Morris et al., 2013).

There are limits to the processes that allow marshes to accommodate sea level rise. Both aboveground and belowground productivity can be inhibited and decline after reaching a flooding

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stress threshold (Morris et al., 2002; Kirwan and Megonigal, 2013). Provided sufficient sediment supply, mineral contributions to accretion theoretically increase steadily with inundation, even after organic matter (OM) contributions begin to decline (Kirwan and Megonigal, 2013).

These processes are also affected by regional-scale dynamics, particularly those that influence sediment supply to the coastline. During the late twentieth century, ambient sediment supply along the coastline of the northeastern US declined slightly as land use and land cover changed (Kirwan et al., 2011; Weston, 2014). Altered sediment regimes constrain marsh response to sea level rise and increase marsh vulnerability to submergence.

Salt marshes can sequester carbon ("blue carbon") at rapid rates (Chmura et al., 2003; McLeod et al., 2011). Because the magnitude and direction of the marsh C sink is directly attributable to vegetation (Macreadie et al., 2013), disparities between marsh accretion and sea level rise have the potential to reduce blue carbon storage by contributing to vegetation loss.

In this paper we combine sediment cores and water level data to measure how marsh accretion, organic and mineral sediment accumulation, carbon storage, and marsh hydroperiod have changed since 1900. We seek to understand how marshes in Connecticut and New York have responded to changes in sea level during the twentieth century, and to assess whether vulnerabilities related to sediment supply may be adversely affecting the resilience of coastal wetlands to sea level rise.

2. Methods

2.1. Research sites

Sediment cores and water level data were obtained for 14 marshes in New York Harbor and Long Island Sound (LIS; Fig. 1). The sites were diverse in terms of adjacent land cover, size, and tidal range, although all were located within close, if not immediate, proximity to human-modified coastlines.

Marsh vegetation cover in the region is not homogenous. Some marshes, particularly in western LIS and New York Harbor, have a high marsh platform dominated by *Spartina patens* and *Distichlis spicata*, with only a thin band of tall-form *Spartina alterniflora* along the creekbank, following the classical upland-to-bay sequence described by Miller and Egler (1950). Other wetlands in the region consist primarily of a platform vegetated by short-form

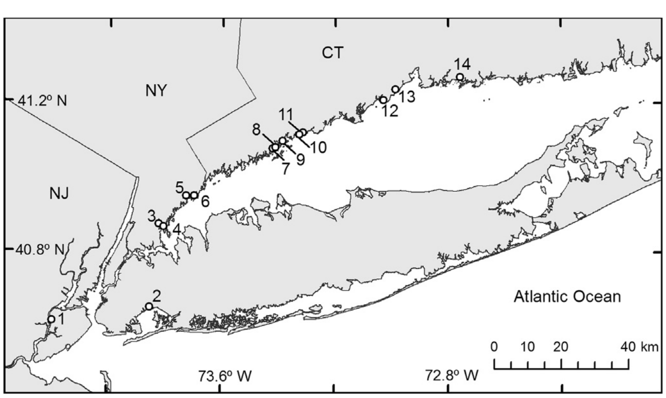

Fig. 1. Locations of 14 marshes studied. 1: Saw Mill Creek, Staten Island, NY; 2: Spring Creek, Queens, NY; 3: Hutchinson River, Bronx, NY; 4: Pelham Bay Park, Bronx, NY; 5: Otter Creek, Mamaroneck, NY; 6: Marshlands Conservancy, Rye, NY; 7: Village Creek, Norwalk, CT; 8: Harborview, Norwalk, CT; 9: Canfield Island Cove, Norwalk, CT; 10: Sherwood Island State Park, Westport, CT; 11: Greens Farms, CT; 12: Gulf Pond East, Milford, CT; 13: Oyster River, Milford, CT; 14: Banca marsh, Branford, CT.

S. alterniflora, with only small patches of true high marsh (Niering and Warren, 1980). Both marsh types were included in this study.

2.2. Sediment cores

At each site, exploratory sediment cores were used to characterize subsurface peat features and identify a coring location that was representative of the site and not subject to obvious disturbances. A single sediment core was then collected from each site and used for geochemical analysis, except at one site where three cores were collected in different marsh zones. Sediment cores were collected with a 15 cm diameter polyvinyl chloride (PVC) coring tool constructed of 1.5 or 2 cm PVC rings secured together with electrical tape, allowing for easy sectioning (Benoit and Rozan, 2001). This large diameter was intended to minimize compression and other sampling artifacts, and to provide abundant material for analysis.

Sediment compression was quantified by measuring vertical displacement of the marsh surface after the coring tool was inserted. Peat compression during coring averaged 5% of core depth, comparable to values reported elsewhere (e.g., Anisfeld et al., 1999; Callaway et al., 2012). We did not correct for compression, effectively assuming the compression occurs below the ^{210}Pb zone.

In total, sixteen sediment cores were collected from fourteen marshes. Two of the cores were collected in 2010, twelve in 2012, and two in 2013. All cores were collected on the marsh platform in areas representative of the dominant site vegetation. Because of differences in vegetation between sites, ten sediment cores were taken in areas of short-form *Spartina alterniflora*, while six cores were taken in areas occupied by *Spartina patens* or a mix of *S. patens* and *Distichlis spicata* (hereafter referred to as *S. patens*). Differences in results between vegetation types are noted, where they exist.

At each site, a local elevation benchmark was established. The surface elevation of each sediment core was recorded relative to the local benchmark using a Topcon GPT-3200 total station (vertical accuracy of 2 mm).

Cores were sectioned into 1.5–2 cm intervals for physical and geochemical analysis. Each core section was weighed before and after drying to 60 °C, and was then pulverized in a ball mill. Subsamples from each core section were re-dried and weighed at 60 °C, at 105 °C, and following combustion at 500 °C. Elemental and geochemical measurements were made on subsamples dried to 60 °C to minimize artifacts on analytes of interest (Hg, N; not presented in this study). We used the sample-specific 60°C–105 °C conversion factor to report masses and concentrations on a moisture-free (105 °C) basis.

Bulk density (g cm^{-3}) was measured as the moisture-free mass per unit volume. Organic content (%) was measured as the percent mass loss during combustion. Conversely, mineral content (%) was measured as the ashed residue of a sample relative to the moisture-free mass.

Organic carbon content was measured on a ThermoFinnigan carbon–nitrogen analyzer following treatment with 6 normal HCl to remove carbonates. Replicates, blanks, and standard reference material (SRM) were run with every 10 samples. Mean SRM recovery was 99 ± 1% (±1 SE) of expected C content. The mean coefficient of variation for replicate measurements was 1%.

Activities of ^{210}Pb, ^{214}Pb, and ^{137}Cs were measured by gamma ray spectrometry with a low-background Ge detector. Activities were measured at energies of 46.5, 352.7, and 661.7 keV, respectively, and were corrected for background, detector efficiency, and self-absorption (Cutshall et al., 1983). Dried, ground samples were sealed in 115 cm^3 cans, or for smaller samples, 10 mL scintillation vials. Sealed samples equilibrated for at least 21 days before analysis. This equilibration period allowed supported ^{210}Pb (^{210}Pb