IS SEA LEVEL RISE ALTERING WETLAND HYDROLOGY IN HUDSON RIVER VALLEY TIDAL MARSHES?

A Final Report of the Tibor T. Polgar Fellowship Program

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ABSTRACT

Sea levels are rising globally, and the eastern U.S. is a hotspot for accelerated sea level rise. Across the mid-Atlantic region, coastal marshes are converting to open water as marshes lose elevation and tidal channels and ponds expand due to loss of marsh macrophyte habitat. Marsh groundwater hydrology is a major control on ecological zonation in coastal marshes and is being altered by rising sea levels; thus, it is important to consider both tidal and groundwater dynamics in order to understand marsh resilience in the context of climate change. This study examines changes in groundwater hydrology over 20 years in Piermont Marsh, a mesohaline tidal marsh in the Hudson River Estuary, New York by comparing water and marsh surface elevation data collected in 1999 and in 2019. This study found that the frequency of marsh flooding and tidal influence on the marsh water table has increased. Although marsh elevation has gained at a similar rate as sea level rise, the marsh is flooded more frequently because the high tides are increasing more rapidly than sea level, causing an increase in tidal range. This tidal range expansion is also likely deepening marsh creeks which increases tidal gradients to the marsh interior and the influence of tides on the water table. Although the marsh was flooded more frequently, groundwater levels were actually lower relative to the marsh surface in the marsh interior, which might be linked to the expansion of *Phragmites australis* due to increased evapotranspiration rates. Future research should focus on the impact of *P. australis* on marsh hydrology, especially in the context of accelerated sea level rise.

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INTRODUCTION

Sea levels are rising globally (Church and White 2006) and the eastern U.S. is a hotspot for accelerated sea level rise (SLR). Between 1950 and 2009, SLR rates between Boston and Cape Hatteras were three to four times higher than the global average, a pattern arising not only from geologic factors, but from ocean temperature, density, and circulation patterns (Sallenger et al. 2012). In fact, the rates of SLR acceleration in the eastern U.S. are among the greatest in the world, and in recent years, actual water levels have averaged more than 10 cm greater than predicted heights (Sweet and Zervas 2011).

Across the mid-Atlantic region, coastal marshes are in decline. Analyses conducted for a 'State of the Estuary' report indicates that Delaware Bay has lost an acre per day of tidal wetlands between 1996 and 2006 (PDE 2012), and a recent analysis of Long Island tidal wetlands shows a loss rate exceeding 10% between the early 1970s and early 2000s (Bowman 2015). Similar rates and patterns of losses have also been estimated for Chesapeake Bay and southern New England (Kearney et al. 2002; Smith 2009; Watson et al. 2017), suggesting that these patterns and trends are regionally widespread. These losses of wetland habitat are symptomatic of marsh drowning (e.g., Blum and Roberts 2009) and include marsh retreat, edge erosion, marsh island loss, and the development and enlargement of die-back areas found on the marsh platform (Figure 1). While some of these wetland losses are eventually expected to be offset by marsh migration into uplands, intensive development of adjacent areas in the high population density mid-Atlantic region acts as a barrier for upslope transgression of intertidal habitats, exacerbating threats of SLR to coastal marsh survival.

The widespread loss of marsh habitat loss across the East Coast is being driven by accelerated SLR. Marsh vegetation losses have been observed to be a function of elevation.

Marshes that sit far below the level of the high tide convert to tidal flats much faster than higher elevation coastal wetlands (Watson et al. 2014), suggesting increasing inundation as a significant driver of loss. In addition to outright loss (e.g., Hartig et al. 2002), shifts from high to low marsh vegetation have been found to occur where marsh accumulation falls below rates of SLR (Warren and Niering 1993), illustrating that current flooding levels are detrimental to the survival of marshes. New evidence from mesocosm experiments shows that increased inundation reduces marsh macrophyte productivity (Watson et al. 2017).

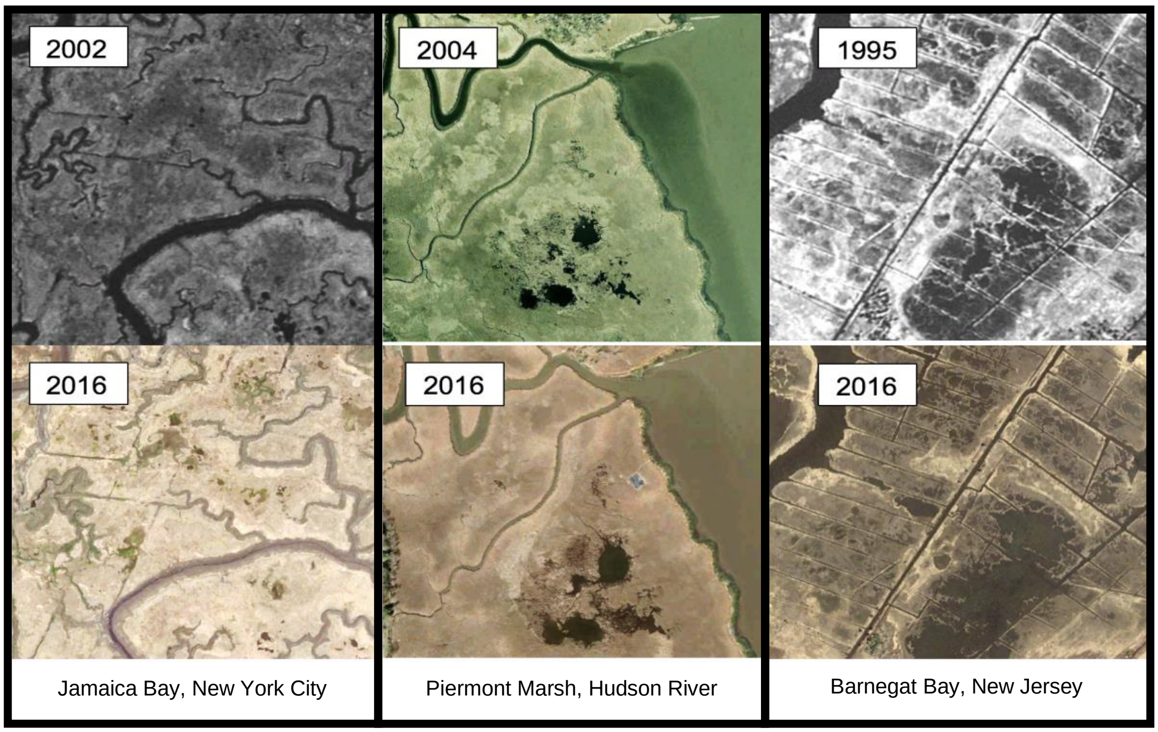

Figure 1. Expansion of open water in Mid-Atlantic tidal wetlands. Satellite images from the turn of the century and 2016 illustrate an increase in unvegetated extent over time, primarily through the expansion of interior ponds and the widening of tidal channels. Aerial images: Google Earth Pro.

In addition to increasing flooding in tidal marshes, SLR is changing shallow groundwater hydrology in coastal systems (Wang et al. 2017; Rotzell and Fletcher 2013). Tidal groundwater exchange is a vital driver of ecosystem function and ecological zonation in coastal marshes. In the marsh interior, freshwater shallow aquifers are perched on top of saltwater inputs from ocean systems and are influenced by forcing from tidal creeks that essentially push the freshwater up at high tide (Wilson et al. 2011). The amplitude of tidally influenced groundwater fluctuations is expressed as tidal efficiency, which is the fraction of tidal flux in the water table relative to the tidal range in the oceanic boundary. Due to the restricted length of a tidal period and the low permeability of marsh sediments tidal efficiency on the marsh platform decreases with distance from tidal creeks (Harvey and Odum 1990; Williams et al. 2002; Wilson et al. 2011). Rising sea levels can have a compounded influence on marsh groundwater elevations and fluctuations (Bjerklie et al. 2012). This hydrological regime is a major driver of marsh macrophyte distribution (Morris 1995; Moffett et al. 2012; Xin et al. 2013; Wilson et al. 2015).

As hydrological regime is a driver of marsh macrophyte zonation, changes in marsh hydrology due to SLR can result in vegetation die-off (Smith et al. 2012) which drives peat collapse and ponding (Delaune et al. 1994). The temporal expansion of marsh pools has also been correlated to hydrological conditions (Schepers et al. 2017; Sandi et al. 2018). The loss of marsh vegetation contributes to marsh instability by reducing sedimentation rates and causing erosion (Coleman and Kirwan 2019). Consequently, macrophyte die-back can result in the formation of increasing areas of open water pools in marsh habitats (Figure 1) which in the context of SLR are unlikely to convert back to vegetated land (Mariotti 2016). Highly flooded portions of salt marshes are poor habitats for *Phragmites australis* (Chambers et al. 2003) so when native species die back due to increased flooding ponded areas will likely stay bare (Mariotti 2016). While *P. australis* can be problematic for local plant diversity (Silliman and Bertness 2004) the species may be contributing to greater rates of marsh accretion than *Spartina alterniflora* and preventing erosion of marsh sediments (Rooth and Stevenson 2000).

Changes in marsh hydrology due to SLR is causing wide scale habitat loss of high priority conservation species such as birds and fish (Bilkovic et al. 2012; Thorne et al. 2015; Hunter et al. 2015). Marsh birds such as the globally threatened saltmarsh sparrow (*Ammospiza caudacuta*) depend on high-marsh vegetation for nesting sites. Rising water tables and increased flooding frequency will not only reduce the amount of habitat available to these birds (Hunter et al. 2015), but higher tides will likely result in increased frequency of nest flooding and egg loss (Bayard and Elphick 2011) and consequent species decline and extirpation or extinction (Rush et al. 2009; Correll et al. 2016; Field et al. 2016; Rosencranz et al. 2018). Increased open water area and channel connectivity in marsh habitats can increase predator access to resident marsh fish (Torio and Chmura 2015) and eventually as marshes convert to open water to the complete loss of salt marsh pond habitat (Thorne et al. 2015; Crotty, et al. 2017). Consequently, it is important to understand how groundwater levels in coastal wetlands are being altered by SLR across multiple scales and ecological groups.

While assessments of wetland vulnerability to SLR have traditionally focused on wetland elevation relative to the tides and wetland elevation change or predicted changes using spatial models (Raposa et al. 2016; Watson et al. 2017), this study examines the change in the water table and tidal influence over time by leveraging the data from previous research. In 1999, Montalto et al. (2006) mapped the hydrology of a brackish tidal marsh within the Hudson River estuary by assessing a variety of variables including topography, water table elevation, and hydroperiod and found low levels of tidal influence and an increase in water table elevation within the marsh interior. Twenty years later, this study revisited the site and methods used by Montalto et al. (2006) to evaluate the change in marsh hydrology over time due to SLR and its impact on marsh macrophyte distribution and soil stability. In addition to these goals, this study attempts to determine if there are water table characteristics associated with conversion to open water by comparing current areas of marsh die-back with surrounding areas of healthy marsh. The following hypotheses were tested:

1. Hydrology changed at Piermont Marsh from 1999 to 2019.
2. Due to changes in hydrology over time, such as SLR and the expansion of *P. australis* populations at Piermont marsh, tidal influence on the water table extends further into the marsh interior in 2019 than in 1999.
3. The expanding ponds in the center of the marsh have different hydrological regimes than vegetated areas of the marsh. Specifically, the ponded areas have a more static water table than vegetated areas, i.e. less tidal influence on the water table.

METHODS

Study Site

Piermont Marsh is a tidal marsh located within the Hudson River Estuary in New York, United States (Figure 2). It is comprised of 417 hectares on the western side of the Governor Mario M. Cuomo Bridge in Rockland County, about 40 km north of New York City. The site is comprised of brackish tidal marshland and shallows at the mouths of Sparkill and Crumkill Creeks and includes intertidal flats and uplands. The marsh experiences diurnal mesohaline tides is bisected by tidal creeks and channels. The marsh interior is dominated by salt marsh ponds in which the spotfin killifish (*Fundulus luciae*) was first reported within the Hudson River estuary (Yozzo and Ottman 2003). The marsh is located at the southernmost edge of the Hudson River National Estuarine Research Reserve (HRNERR) and is designated as a Significant Coastal Fish and Wildlife Habitat by the New York State Department of State (NYSDOS 2012) and a Critical

Environmental Area by Piermont Village (NYSDEC 1985). The site's vegetation is dominated by *Phragmites australis* but retains some small patches of native marsh species in the interior. Local land managers and residents anecdotally report that species diversity on the marsh has decreased over the last few decades (NYSDEC 2017).

Study Design

Hydrologic measurements were collected over the summer of 2019 on Piermont Marsh. These data were compared to hydrological measurements made in 1999, which focus on describing the marsh water table from a large tidal channel to the marsh interior (Montalto et al. 2006). The aim of this study was to compare current tidal flooding and groundwater table levels with measures made in 1999, in order to identify how SLR has altered marsh hydrology. In addition, ground water levels were monitored within and adjacent to the expanding open water area in the center of the marsh (Figure 2).

Water Table Measurements

Three tidal gauges were installed within the tidal creek (Figure 2; Table 1). Onset Hobo 20UL water level loggers were suspended inside perforated pipes and attached to cinderblocks which were placed in the center of the channel. Loggers were programmed to take a water elevation measurement every ten minutes. In order to determine ground water levels throughout the marsh, seven water level loggers were installed along a gradient from the tidal channel to the upland, replicating the measures conducted by Montalto et al. (2006). The location of Montalto's wells were established by georeferencing maps from Montalto et al. using Google Earth 7.3.2.5776 and were confirmed visually by Franco Montalto in the field.

Wells were constructed by suspending a pressure transducer within a 7.5 cm diameter perforated PVC pipe lined with screening to prevent sediment from entering the well. A hole was excavated with an augur and the well was placed inside it. The surface of the well was vented to the atmosphere. A concrete collar was installed at the marsh surface around the well in order to prevent the preferential flow of water down the side of the well. Seven wells were installed along the original transect, perpendicular to the creek, and two additional wells were installed along the same transect into the ponded area at the center of the marsh (Figure 2, Table 1).

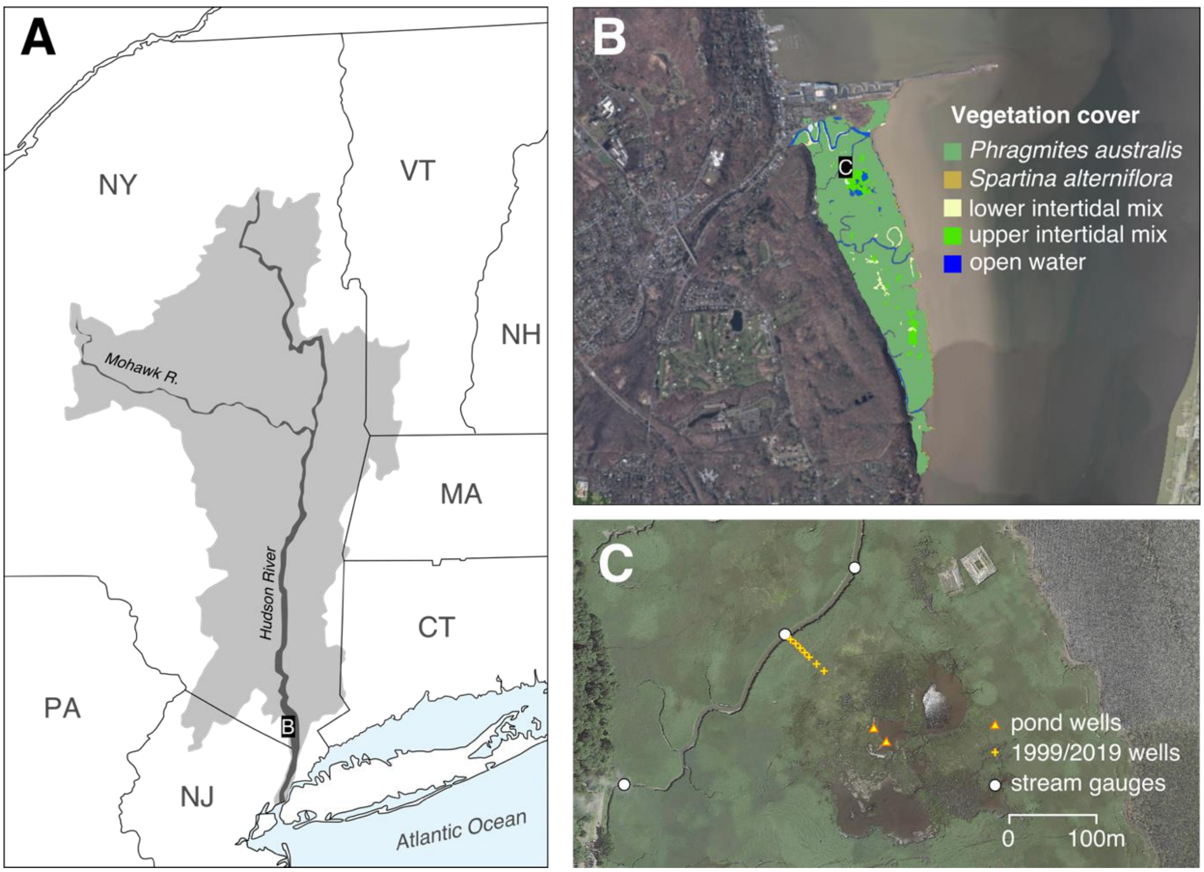

Figure 2. Map of study area showing (A) the Hudson River Valley with the location of Piermont Marsh; (B) Vegetation cover at Piermont Marsh which is dominated by the non-native *Phragmites australis;* (C) the location stream gauges (denoted as circles) as well as groundwater wells along the a transect perpendicular to a tidal channel and in two areas of expanding ponded water at Piermont Marsh. The location of map insets (marked 'B' on map A and 'C' on map B) is shown.

Wells were installed 5 May 2019 and water levels were downloaded three times between May 2019 and 12 August 2019 using a Hobo Optic USB Base Station. An atmospheric pressure logger was also deployed to correct pressure transducer data for variations in atmospheric pressure. The absolute elevation of the top of each well was measured using RTK-enabled static GPS measurements from Leica GNSS GS14 rover units and static measures using an AX1202 GG base station unit in order to reference water levels to the NAVD88 vertical datum. Reference water levels were measured each time data was collected. The distance from the top of the well to the water surface was measured by hand using a meter stick and the time in GMT-4 was recorded. To relate marsh elevation with water elevations, GPS surveys were conducted along the transect using a Leica GNSS GS14 rover unit.

Table 1. Location of stream gauges (SG) and groundwater (GW) wells.

	Type	Distance from creek (m)	Latitude	Longitude	2019 Marsh Elevation (m, NAVD88)	1999 Marsh elevation (m, NAVD88)
1	SG	0	41.036750°	-73.909667°	-	-
2	SG	0	41.036083°	-73.910611°	-	-
3	SG	0	41.034556°	-73.912833°	-	-
1	GW	1 (levee)	41.036061°	-73.910498°	0.72	0.61
2	GW	6	41.036023°	-73.910455°	0.72	0.63
3	GW	12	41.035980°	-73.910411°	0.73	0.64
4	GW	18	41.035941°	-73.910361°	0.75	0.65
5	GW	24	41.035899°	-73.910317°	0.77	0.66
6	GW	36	41.035812°	-73.910233°	0.72	0.68
7	GW	48	41.035717°	-73.910142°	0.80	0.69
8	GW	135	41.035155°	-73.909433°	-	-
9	GW	164	41.034957°	-73.909261°	-	-

Data Analysis

Although rates of SLR at the Battery, NY are reported at 2.87 mm yr^{-1}, (NOAA 2020), these long-term rates do not capture more recent sea level trends. For this reason, simple linear trends were calculated using monthly tide station data (from the Battery tide gauge, NY) to estimate changes that have occurred between 1999 and 2019 in terms of tidal flooding. Monthly mean sea level, monthly mean high water, and monthly mean low water were obtained, and trends estimated using linear regression (NOAA 2020). The two studies' position in the 19-year metonic cycle were also examined using monthly water elevation levels (1980-2020) from the NOAA Battery tidal gauge (NOAA 2020).

Pressure transducer data was post-processed using HOBOware Pro (Ver. 3.7.16 , Onset Computer Corporation, Bourne, MA) using reference water levels collected in the field and were corrected for atmospheric pressure using the HOBOware barometric compensation assistant. Barometric data was obtained from the Hudson River National Estuarine Research Reserve, due to inconsistences in data from the barologger (NERRS 2019). Raw water elevation data from 1999 was analyzed in concert with the 2019 data. Water level data from 1999 were converted from the NVGD29 to NAVD 88 datum using NOAA VDatum v4.0.1 (NOAA 2019) prior to analysis. The transducer in well seven experienced three brief malfunctions from 30 May to 3 June 2019, which resulted in inaccurate elevation measurements for a total of 19.5 hours. These data were excluded from the analysis. In 1999, Montalto also experienced malfunctions at well seven. These data were corrected by Montalto into smoothed six-hour increments using average water elevation measurements and calculated error and calibrated using regression (Montalto et al. 2006). No other well transducers appeared to have malfunctioned.

Changes in surface flooding were computed using data from channel gauges installed in 1999 and 2019 (5 May – 12 August). Marsh flooding was calculated as the percentage of time that water levels exceeded the average marsh elevation (0.75 m NAVD in 2019 and 0.65 m NAVD in 1999). The number of tides that flooded the marsh per month and the average maximum flooding depth were identified for 1999 and 2019. Sub-surface flooding (6 April 1999 – 26 May 1999; 5 May 2019 – 30 June 2019) was calculated as the percentage of time that the groundwater table exceeded thresholds of 5 and 10-cm below the marsh surface. Although groundwater data were compared over the same season (spring), which is important due to the strong seasonal variability in sea level in the Mid-Atlantic (Figure 3), dates could not be completely matched (6 April 1999 – 26 May 1999 vs. 5 May 2019 – 30 June 2019) due to limited data in 1999. Comparing mean sea level to April and May to mean sea level during May and June suggests a minimal difference in sea level over this time period (2 cm).

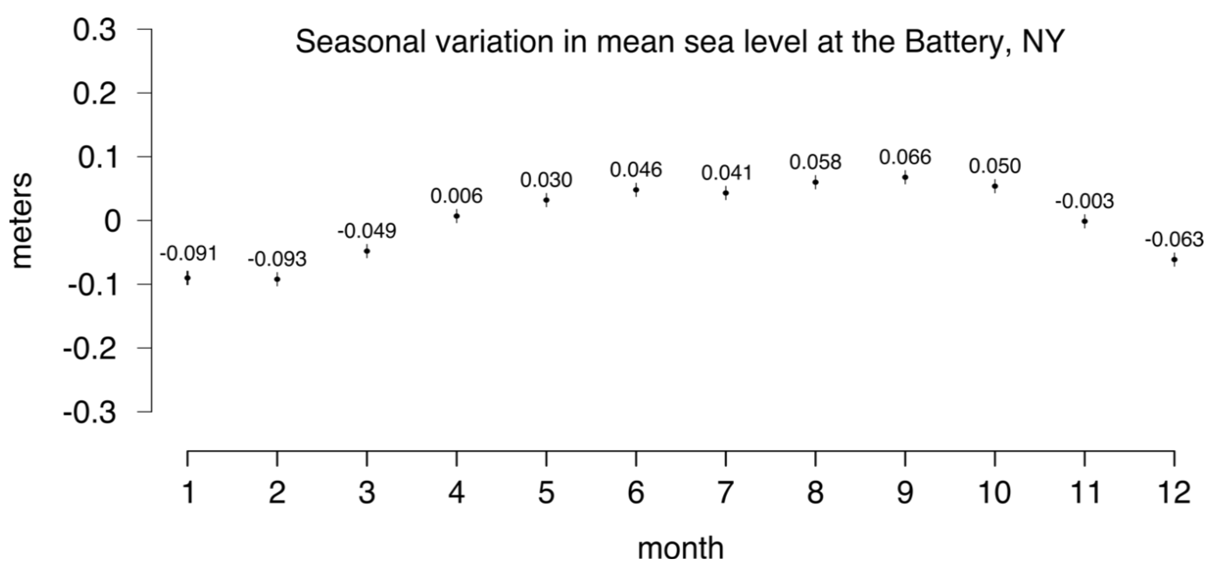

Figure 3. Average seasonal tidal elevation at the Battery, NY: 1920 to 2020. Monthly tidal variation from at the NOAA Battery tidal gauge (NOAA 2020). These data show the seasonal cycle in tides and illustrate that it is appropriate to directly compare the 1999 and 2019 data despite a one month offset. The seasonal cycle difference between April and May is 2.4 cm and the difference between May and June is 1.6 cm.

Tidal efficiency (TE) is defined as the ratio of the amplitude of groundwater fluctuations in a coastal aquifer to the amplitude of tidal fluctuations at the ocean boundary. Range of tide and tidal efficiency (*TE*) were calculated for 1999 and 2019 well data empirically as:

$$TE = \frac{s_w}{s_t}$$

where s_w is the daily range of water-level fluctuation in a well tapping the groundwater table and s_t refers to the daily range of tide (Ferris et al. 1962). Because the 1999 channel gauges were exposed at low tide (Montalto et al. 2006), *TE* was calculated based on range of tide reported from the NOAA tide gauge at the Battery, NY (NOAA 2020). Data analysis and calculations were performed in Excel (Microsoft, version 16.35) and in R 3.6.1 (R Core Team 2019) using package FSA (Ogle et al. 2020).

RESULTS

Tidal data

Examination of water levels at the Battery shows that measures (in spring 2019 and 1999) occurred during normal and not anomalous tidal periods (Figure 4). Because both studies occurred 20 years apart, and the chief metonic cycle is 18.6 years, both studies occurred at similar points in the cycle. From linear regression, it was estimated that from 1999 to 2019 mean high water at the Battery increased at an average rate of 7.5 mm yr^{-1}, mean sea level increased by 4.5 mm yr^{-1}, and mean low water increased by 2.0 mm yr^{-1} (Figure 5; Table 2).

Table 2. Tidal trends at the Battery, NY. Rates of mean water elevation change from April 1999 to April 2019, the equations used to calculate rates (*x* = year; *y* = water level in meters), and the overall mean change in tidal range.

	Rate of increase (mm yr^{-1})	Regression Equation	r^2	p	Total change
Mean monthly high water	7.6	$y = 7.56\text{x}10^{-3}x - 13.68$	0.09	$4.38\text{x}10^{-12}$	15.1 cm
Mean monthly sea level	4.5	$y = 4.52\text{x}10^{-3}x - 8.24$	0.08	$1.12\text{x}10^{-5}$	9.0 cm
Mean monthly low water	2.0	$y = 1.98\text{x}10^{-3}x - 3.86$	0.02	0.05	4.0 cm
Tidal range	5.6	Tidal range = MHW - MLW	-	-	11.1 cm

Comparing the channel gauge records with marsh elevations in 1999 and 2019 suggest that tidal flooding has increased. The average marsh elevation along the transect in which the wells were emplaced was flooded 3.2% of the time in 1999, and 9.9% of the time in 2019. In 1999, the marsh flooded on average 12 times per month between May and August. The average flooding duration was 2.1 ± 0.80 hours (mean ± standard deviation), and the marsh flooded on average to a depth of 8.7 ± 5.4 cm. In contrast in 2019, the marsh flooded on average 29 times per month between May and August. The average flooding duration in 2019 was 2.6 ± 1.1 hours, and the marsh flooded on average to a depth of 12 ± 8.8 cm. While marsh flooding was limited to spring tides during both time periods, high tides resulted in marsh flooding about 2.4 times more frequently in 2019 than 1999.

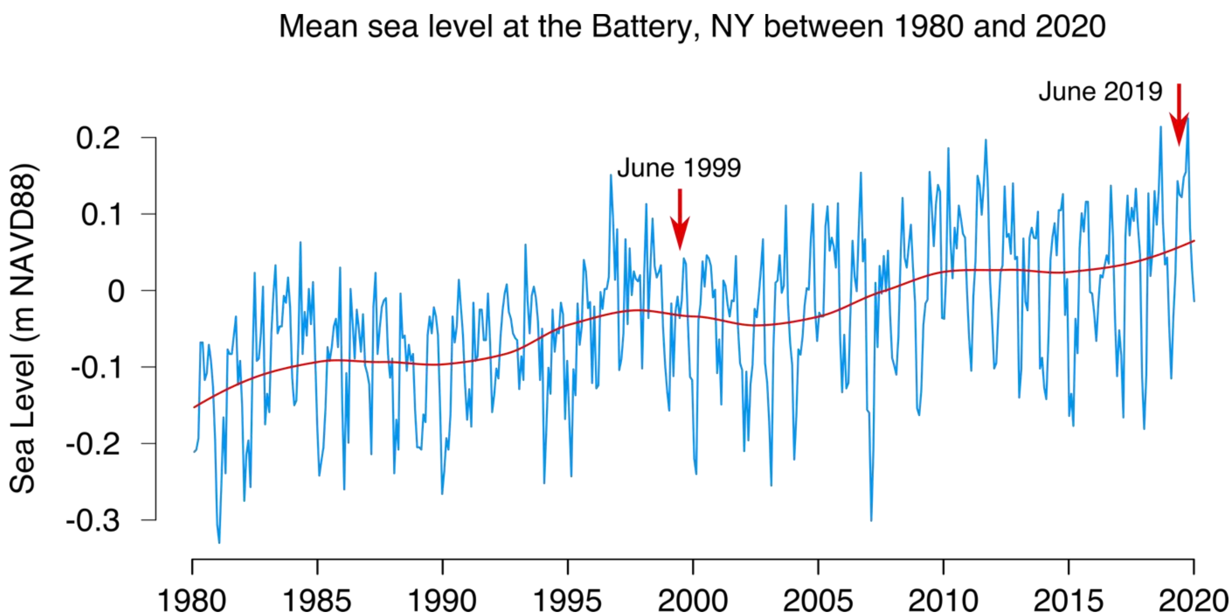

Figure 4. Mean sea level at the Battery: 1980 to 2020. This figure illustrates seasonal changes in water level, upward trends in sea level, and periodic changes in tides due to astronomic and oceanographic forcings (NOAA 2020). A loess smoothing curve (span=0.40) shows multi-decadal sea level fluctuations. Because sampling periods were 20 years apart, they are during approximately the same relative positions within the 18.6 year metonic cycle.

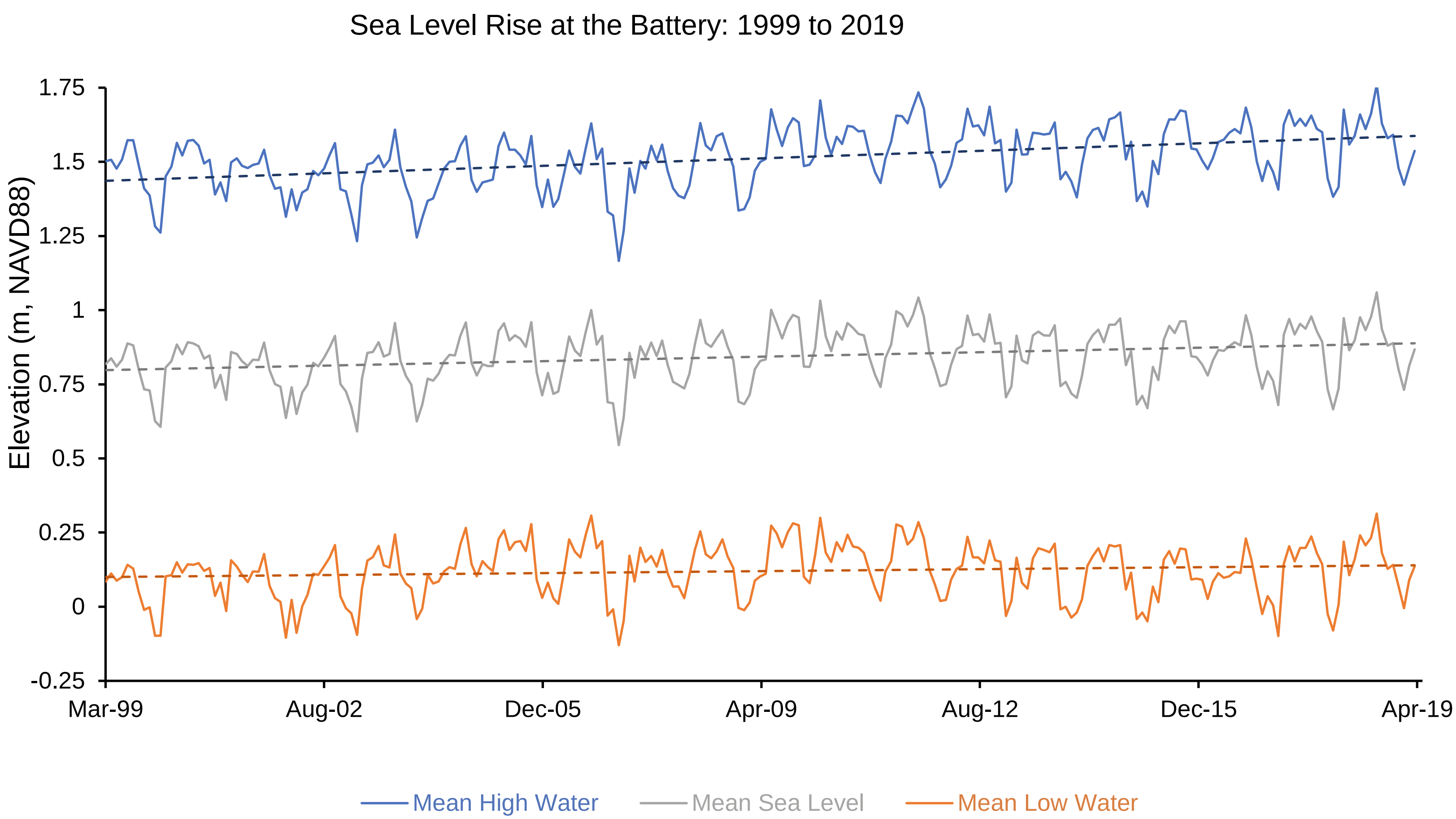

Figure 5. Linear trends in tidal elevations at the Battery: 1999 to 2019 (NOAA 2020). Mean monthly elevations at the Battery between 1999 and 2019 evaluated with linear regression. Rates of rise in mean high water was 7.6 mm yr^{-1}; the rate of sea level rise was 4.5 mm yr^{-1}, and the rate of mean low water rise was 2.0 mm yr^{-1} (Table 2).

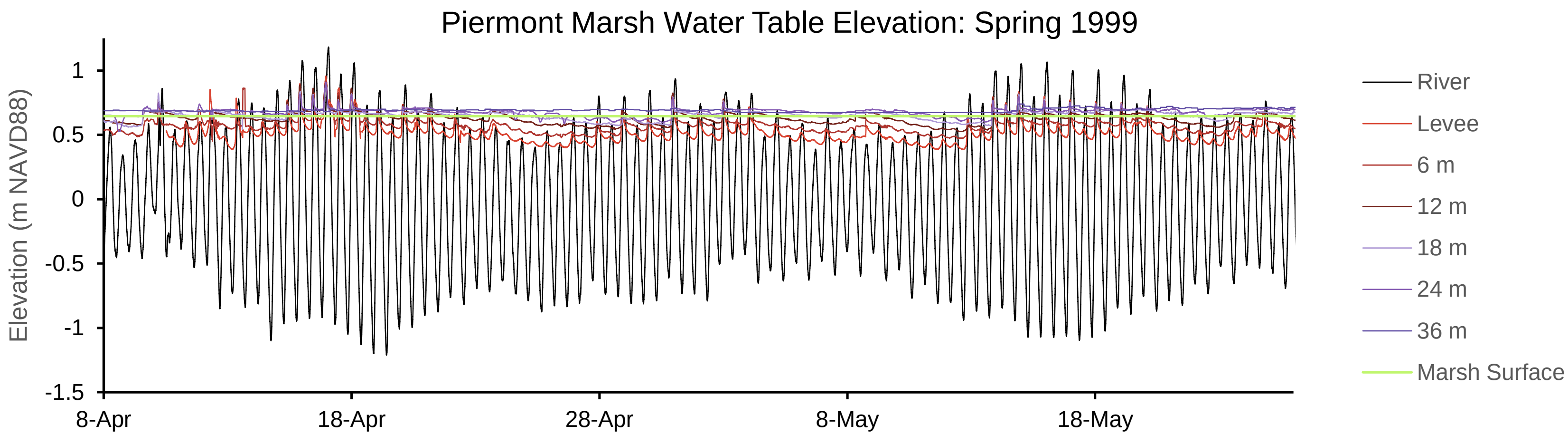

Figure 6. Water table and marsh surface elevations at Piermont Marsh: 1999. Water table and marsh surface elevations from April through May 1999. Tidal elevation reference from the NOAA Battery gauge is in black (NOAA 2020), and each Piermont Marsh wells are represented by a gradient from red to purple, reds are closer to the tidal creek and purples are farther from the creek. Marsh surface is shown in green.

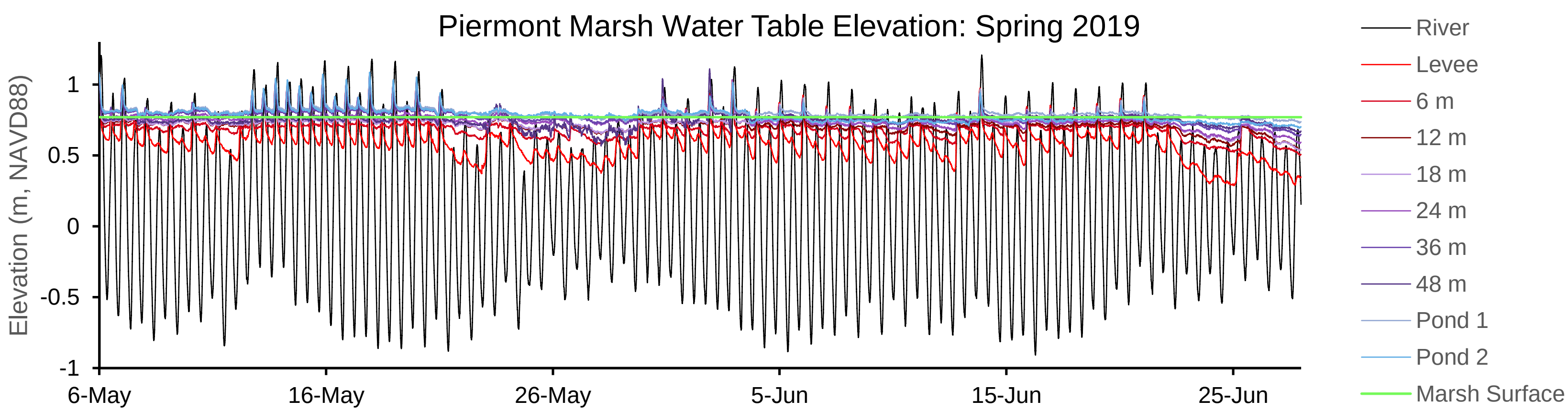

Figure 7. Water table and marsh surface elevations at Piermont Marsh: 2019. Water table and marsh surface elevations from May through June 2019. Tidal elevation reference from the NOAA Battery gauge is in black (NOAA 2020), and each Piermont Marsh wells are represented by a gradient from red to purple, reds are closer to the tidal creek and purples are farther from the creek. Marsh surface is shown in green.

Water table data

Water table elevation, tidal range, and tidal efficiency were all greater in magnitude in 2019 than in 1999 (Figures 8-10). The median maximum water in all wells were at least 10 centimeters higher in elevation in 2019 than in 1999, with the greatest difference at the levee, where water levels were 20 cm greater in 2019 than in 1999. While the median minimum water was higher in all wells in 2019 than in 1999, the difference was of a smaller magnitude than the mean maximum water values. Median minimum water elevation differences ranged from 0.4 cm to 1.3 cm, with the largest magnitudes at the levee and six meters in from the creek. In 2019, median minimum water showed a similar increase in elevation with distance from the creek as in 1999. In both maximum and minimum tides, the interquartile range of the data was generally greater in 2019 than in 1999 (Figure 8).

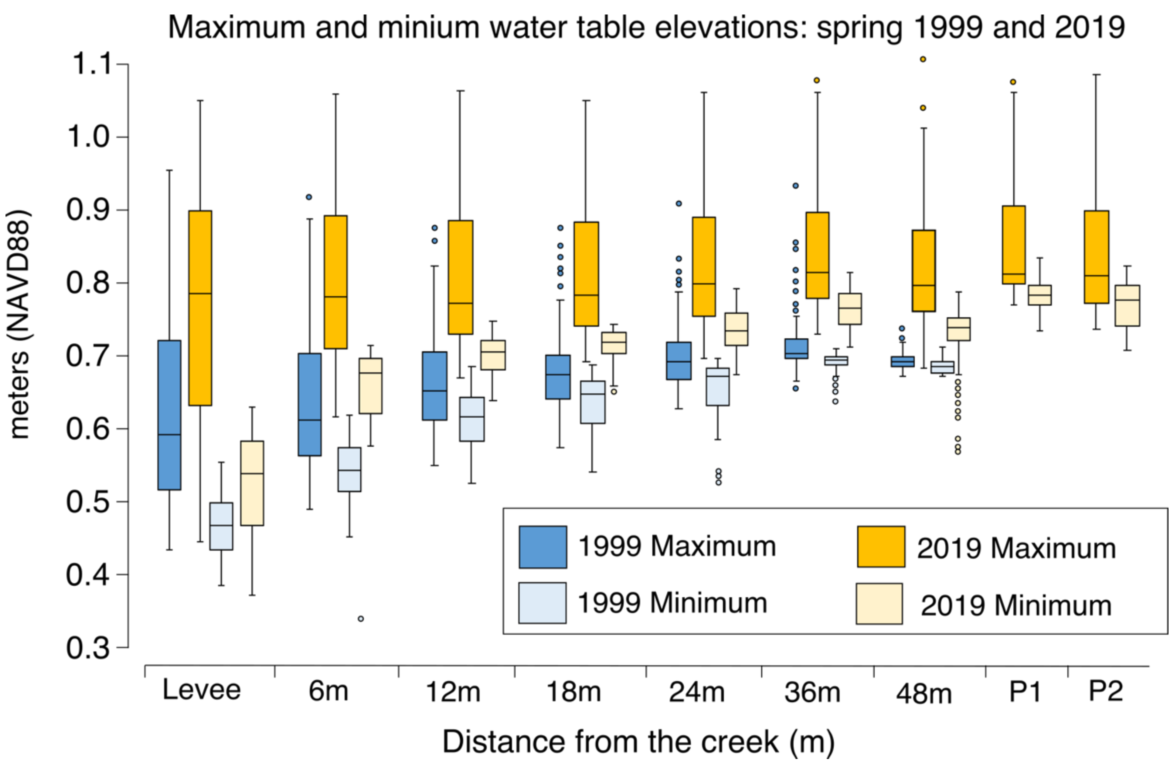

Figure 8. Maximum and minimum water table elevations at Piermont Marsh. Semidiurnal water table elevations across Piermont Marsh in the spring of 1999 and 2019. 1999 data are in blue and 2019 data are in orange. Dark colors are high water data and light colors are low water data.

While surface flooding was found to increase consistently over the last 20 years, the position of water table relative to the marsh varied with channel proximity. In 1999, the mean water table position was below the marsh surface (e.g. 10-20 cm of depth) adjacent to the tidal channel but increased to near the marsh surface (to e.g. 0-2 cm of depth) in the marsh interior. In 2019, the same general trend was found, with lower water tables at the channel edge and increasing water tables in the marsh interior; however, in comparison with 1999, the percentage of time that the water table was at the marsh surface in 2019 in the marsh interior decreased (Figure 9), resulting in decreasing prevalence of root zone flooding.

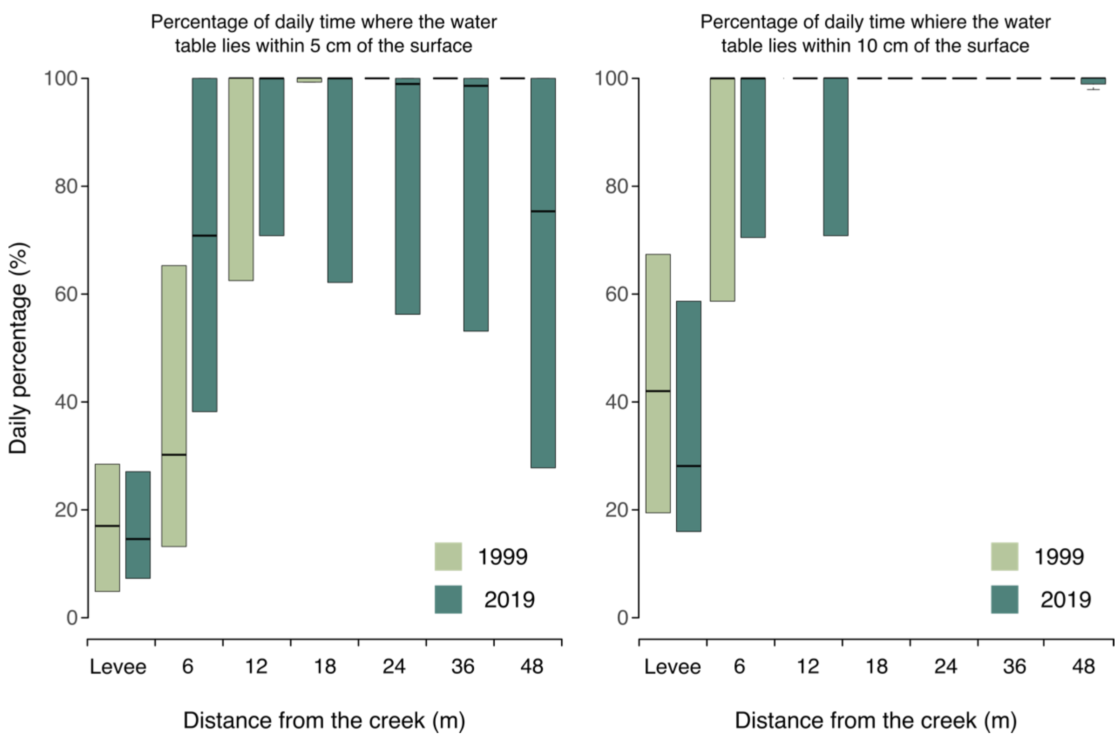

Figure 9. Water table elevation relative to marsh surface at Piermont Marsh. The percent of daily time in which the water table was within 5 and 10 cm of the marsh surface elevation along the gradient from the channel edge to marsh interior. The values indicate median and inter-quartile range.

The tidal influence on groundwater fluctuations showed a similar trend in both years, that is decreasing with distance from the tidal channel, but the magnitude of tidal influence is higher in all wells in 2019 than in 1999. The greatest difference is at the levee, with a difference of 0.68 and the smallest difference is 48 m from the creek, with a difference of 0.16 (Figure 10).

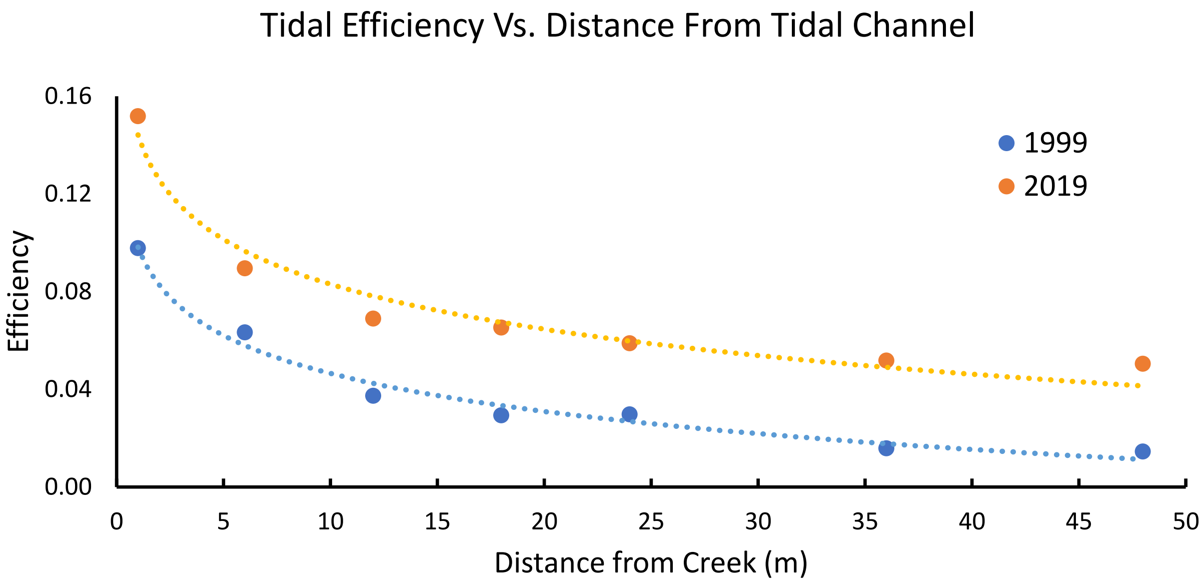

Figure 10. Tidal influence on water table fluctuations at Piermont Marsh. A comparison of tidal efficiency in the spring 1999 and 2019. Tidal efficiency is the amplitude of the groundwater fluctuation expressed as a fraction of the tidal range in the Hudson River. 1999 data is in blue and 2019 data is in orange.

The wells installed in ponds in the interior of the marsh are experiencing less tidal influence than those wells closer to the tidal creek. While there is a steady decrease of mean tidal efficiency up to 48 meters into the vegetated marsh, the curve seems to have bottomed out in the ponded marsh interior. Pond 1 and pond 2 have almost identical median efficiencies, at 0.03, and the spread of the data at both wells is similar, at 0.04 in the interquartile range. All wells in the vegetated areas had an interquartile range at or more than 0.05.

DISCUSSION

Hydrology has changed in Piermont Marsh over the last 20 years. Tidal range has expanded both in the marsh and at the Battery in the Hudson River, and tidal influence is extending farther into the marsh in 2019 than in 1999. Mean sea level and marsh elevation have risen at similar rates, but the frequency and magnitude of high water flooding the marsh has increased, which is likely driving changes in hydrology and thus the ecological zonation of marsh habitats. While marsh flooding has increased over time, the water table was lower relative to the marsh surface. In addition to SLR, a likely driver of the observed changes in the groundwater table is the expansion of *P. australis* into the marsh interior, which could be altering soil porosity, hydraulic conductivity, and evapotranspiration rates. Finally, the ponded areas in the center of Piermont Marsh are experiencing less tidal influence than the vegetated areas. These distinct hydrological regimes may be shifting ecological zonation and consequent peat collapse and expansion of open water.

Tidal range expansion has primarily been driven by increased elevations of high tides, rather than by changes low tide elevations. In Piermont Marsh, the magnitude of the difference of mean high tides in 1999 to 2019 is greater in all wells than the magnitude of the difference of mean low tides. For example, 2019 mean high tide at the levee is 19.4 cm higher than the mean high tide in 1999, but the mean low tide is only 7 cm higher, which is less than the 12 cm average increase in marsh surface elevation. This trend is apparent along all wells in the transect and can also be observed at the NOAA Battery gauge in the Hudson River. While sea level at the Battery rose at a rate of 4.6 mm yr^{-1} from 1999 to 2019, monthly mean low water has risen by 1.9 mm yr^{-1} and mean monthly high water has risen by 7.56 mm yr^{-1}, expanding the mean tidal range by more than 10 cm. Increasing tidal ranges introduce more hydraulic energy in tidal creeks, which can result in deeper channels with steeper banks (Allen 2000; Williams and Orr 2002; Williams et. al 2002). This creek geometry increases the tidal gradient to the marsh interior, resulting in greater tidal influence on the marsh water table (Allen 2000; Wilson and Gardner 2006; Wilson et al. 2011). The interactions of increased tidal range, marsh stability, and resilience to SLR are debated in the scientific community (Osgood 2000; Kirwan and Guntenspergen 2010; Pickering et al. 2012; Balke et al. 2016; Cahoon et al. 2019).

Increased tidal range driven by higher high tides are contributing to another major trend observed in this study, which is increased tidal influence on the marsh groundwater table, particularly in the marsh interior. In 1999, tidal influence did not propagate more than 20 meters into the marsh but in 2019 tidal influence continued far into the marsh interior. As can be observed in Figure 10, in 1999 mean tidal efficiencies at and past 18 meters are consistently around 0.015. In 2019 tidal efficiencies are higher at all wells than in 1999, and do not level out until far into the ponded marsh interior. In addition, the pond wells in 2019 experienced tidal efficiencies around 0.025, higher than the interior wells measured in 1999. As a consequence of the change in tidal range and increased tidal efficiency in the marsh interior, the increasing trend of mean high tide with distance that was observed in 1999 is less apparent in 2019. This is because in 1999, only the highest tides were propagating into the marsh interior, which caused the mean to increase due to lack of influence from low tides. In 2019, more tides of all magnitudes are propagating into the marsh interior, causing the mean maximum tide to be fairly consistent across the entire transect; however, the mean minimum tides increase in magnitude with distance from the creek both in 1999 and in 2019, resulting in a perched water table in the marsh interior in both years.

As the frequency and magnitude of water table fluctuations determine the eco-hydrological zonation of marsh macrophyte habitat (Moffett et al. 2012; Xin et al. 2013; Wilson et al. 2015), the observed changes in hydrology are altering plant communities across the marsh. Another likely factor is the expansion of *P. australis* into the marsh interior. *P. australis* could have a compounding impact on marsh macrophyte community distribution, because not only does it crowd out native competitors, but dense *P. australis* populations also alter groundwater hydrology (Windham and Lathrop 1999; Chambers et al. 2003). While there is not extensive research on the impact of *P. australis* on groundwater hydrology, there is evidence that the *P. australis* root mat generally increases hydraulic conductivity in marsh sediments (Baird et al. 2004; Saaltink et. al. 2019) and that the presence of extensive stands of *P. australis* can lower the water table due to increased evapotranspiration (Windham and Lathrop 1999; Windham et al. 2001). *Phragmites* has been expanding across Piermont Marsh since at least the 1960's (Winogrond and Kiviat 1997) and anecdotal evidence and observations of satellite images show that *P. australis* distribution has greatly expanded at Piermont Marsh over the last 20 years.

The ponded areas in the Piermont Marsh transect are experiencing less tidal influence than the vegetated area. While mean groundwater fluctuations continued to decline along an exponential curve from the levee to 48 meters into the marsh, they appear to have leveled out in the ponded marsh interior. Wilson et al. (2015) identified four primary eco-hydrological zones in Atlantic salt marshes, characterized by 1) short form *Spartina alterniflora* and 2) tall form *Spartina alterniflora* in the low marsh, 3) *Salicornia* zone in the high marsh, and 4) a *Juncus* zone adjacent to the uplands. The hydrological regime observed at Piermont marsh in 2019 resembles the *Spartina* zones in vegetated areas and the *Salicornia* zone in the ponds. As defined by Wilson et al. (2015) the *Salicornia* zone is primarily driven by upward flow of groundwater during neap tides without significant discharge in between high-water events, resulting in hypersaline zones in the marsh interior. This hydrological pattern was observed in the ponded areas at Piermont Marsh (Figure 11) and may be compounded by increased frequency and amplitude of high water and consequent increased tidal influence. Hydrological regime in the intact section of the marsh suggest that the water table is being forced up at high water, but is also draining at low water, allowing for greater flushing in vegetated areas than in the ponded areas (Figure 12).

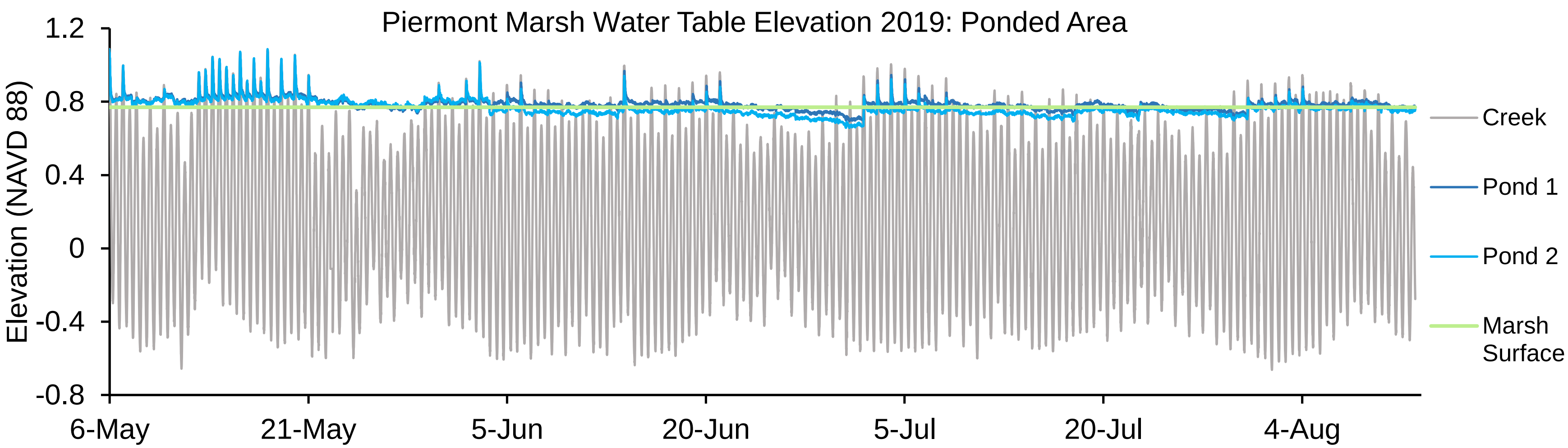

Figure 11. Water table fluctuations at Piermont Marsh: Ponds. Water table and marsh surface elevation in the ponded area from May to August 2019. Tidal channel water surface elevations are in grey and wells located inside ponds are in blue. Marsh surface is shown in green. The ponded areas are located far into the marsh interior. Water table fluctuations indicate upward forcing of ground water at high water, but little drainage at

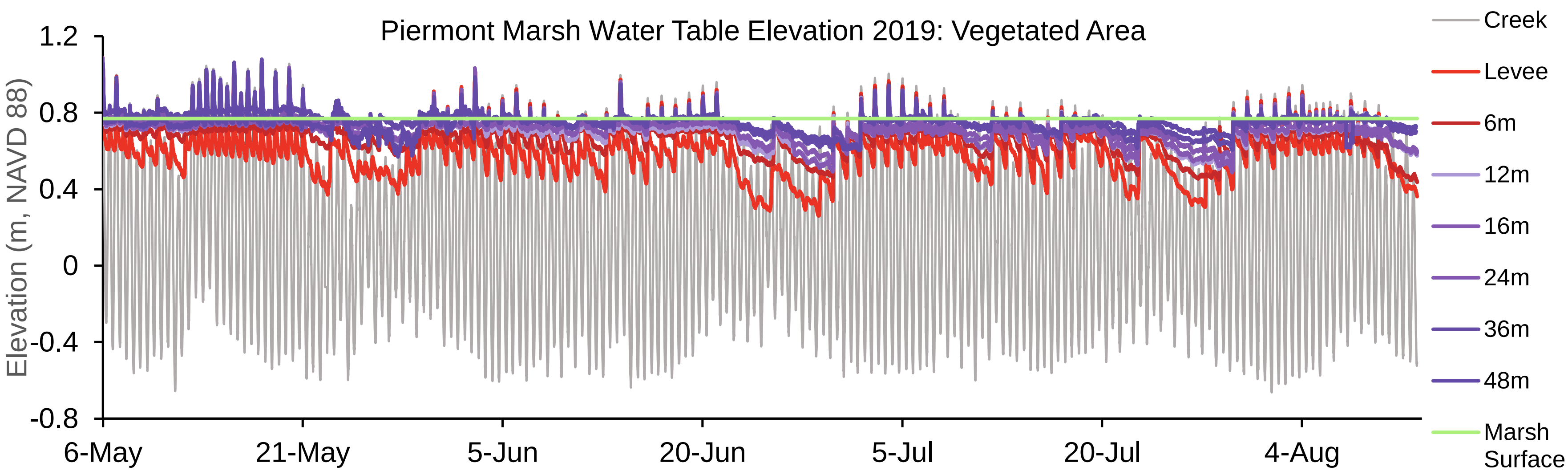

Figure 12. Water table fluctuations at Piermont Marsh: Vegetated. Water table and marsh surface elevation in the vegetated area from May to August 2019. Tidal channel water surface elevations are in grey, reds data points are closer to the tidal creek and purple are farther from the creek. Marsh surface is shown in green. Water table surface elevation in the vegetated areas indicates both upward forcing from high water and drainage during low water.

Future Research

Tidal range dynamics have changed dramatically from 1999 to 2019 at Piermont Marsh and at the Battery in the Hudson River. Other studies have found similar trends globally and locally (Pickering et al. 2012; Mawdsley et al. 2014; Balke et al. 2016; Pickering et al. 2017; Talke et al. 2018) and some tidal datums have been updated to reflect changing tidal dynamics due to accelerated SLR (Bamford 2013; Wang and Myers 2016); however, a comprehensive review of mean high water and mean low water trends in the United States has not been published since 2003 (Flick et al. 2003). A more comprehensive analysis of regional tidal range and SLR should be performed in order to more fully understand changing tidal dynamics in the Mid-Atlantic region and to inform marsh and coastal management into the future.

Further research into the extent and impact of *Phragmites australis* at Piermont Marsh is necessary. The authors of this report hope to perform a spatial analysis on *P. australis* distribution from the mid-1996 to 2015, and to perform porosity tests on sediments from Piermont Marsh in order to better understand how invasion by *Phragmites* my alter marsh groundwater levels. Generally, more research is necessary to understand how *P. australis* alters hydrology in coastal marshes, particularly in regard to hydraulic conductivity and porosity.

Finally, this study is only a comparison of one particular year to another, so while these data do show that hydrology in Piermont Marsh has changed over twenty years, these data do not reveal the rate of change. In addition, extensive statistical analysis of these data is not possible with only two time-steps to compare. Additional years of monitoring will clarify the findings of this study and make possible an evaluation of the statistical merit of these findings.

CONCLUSION

Hydrology changed dramatically at Piermont Marsh from 1999 to 2019. The tidal range in the Hudson River has expanded, over the last 20 years primarily due to rapid increase of mean high-water elevations. Larger tidal ranges in tidal creeks are increasing the tidal gradient into the marsh interior which results in greater amplitudes of water table fluctuations further from the creek. Tidal influence is now extending further into the marsh interior and while the marsh is flooding more frequently, the water table elevation is lower relative to the marsh surface. Expanding ponded areas in the marsh interior are experiencing tidal forcing during high water but are not draining at low water. Altered hydrology may be impacting the ecological zonation of marsh macrophytes causing vegetation die-back, peat collapse, and the increasing areas of open water across the marsh. SLR, the expansion of tidal range, and increased populations of *P. australis* are likely important drivers of the changes observed in Piermont Marsh. Further research is recommended on tidal range expansion in the Northeastern United States and on the impact of *P. australis* populations on marsh hydrology and ecological zonation.

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