Ocean & Coastal Management 149 (2017) 33–41

Contents lists available at ScienceDirect

Ocean & Coastal Management

journal homepage: www.elsevier.com/locate/ocecoaman

The impact of past management practices on tidal marsh resilience to sea level rise in the Delaware Estuary

Joseph A.M. Smith [a,*], Steven F. Hafner [b], Lawrence J. Niles [a]

[a] Niles & Associates, 109 Market Lane, Greenwich, NJ 08323, USA
[b] Stockton University Coastal Research Center, 30 Wilson Ave, Port Republic, NJ 08241, USA

ARTICLE INFO

Article history:
Received 12 August 2016
Received in revised form
28 August 2017
Accepted 13 September 2017

Keywords:
Tidal marsh
Environmental management
Ecological resilience
Tidal marsh reclamation

ABSTRACT

Defining appropriate management and conservation strategies to maximize tidal marsh resilience to sea level rise requires a clear understanding of the causes of marsh degradation. While sea level rise is a well-known threat to tidal marshes, current and past management practices on marshes can also greatly influence present-day marsh condition, resilience and future persistence. Using point-intercept analysis of maps and imagery, we assessed the past and current landcover and elevation of Delaware Estuary tidal marshes in New Jersey, USA. We estimated the historic extent of tidal marsh impoundment for agriculture and determined current marsh vegetation composition and elevation in areas that were and were not historically impounded. We estimate that more than half of all tidal marsh in the 36,539 ha study area had been historically impounded. A small fraction of this area remains impounded at present (7.6%). While tidal flow has since returned to formerly diked areas, marsh recovery has been incomplete. Overall 21.6% (4048.8 ha) of formerly impounded marsh has not revegetated, becoming open water after impoundment breaches. Marsh loss as a result of impoundment is also responsible for the loss of 2.3 km of adjacent shoreline beaches. Conversely, only 0.5% of marsh that was never impounded has converted to open water since 1931. This difference is likely due to dramatic elevation deficits caused by impoundment. Marsh elevation of current and formerly impounded areas (derived from LiDAR and validated with RTK GPS) is significantly lower than the elevation of marsh areas that were never impounded. Supporting this finding, the frequency of high marsh vegetation (an indicator of higher elevation)in vegetated formerly impounded areas is half that of areas that were never impounded. Marsh edge erosion and creek expansion have added an additional estimated 3836 ha to the amount of tidal marsh loss since 1931. Marsh transgression inland into forest and agricultural areas has resulted in estimated gains in marsh area of 2815 ha, offsetting a considerable proportion of losses. Given our results, we recommend the following management actions to maximize tidal marsh persistence in the Delaware Estuary: (1) Beneficial use of sediment to offset marsh elevation deficits resulting from historic impoundment, (2) Strategic land protection to maximize the potential for inland marsh migration, (3) Tidal flow restoration to remaining impounded areas in combination with the beneficial use of sediment to address elevation deficits. Determining the impacts to tidal marshes from past management practices makes it possible parse the relative contribution of relative sea level rise and site-level management, resulting in more targeted conservation strategies.

1. Introduction

As impacts from climate change-accelerated sea level rise threaten the continued existence of tidal marshes, many of these marshes have also been negatively impacted by current and former management and land use practices (Kennish, 2001; Silliman et al., 2009). Actions such as impoundment and tidal restriction (Roman et al., 1984), ditching (Bourn, 1950), open water marsh management (Wolfe, 1996), sediment removal (Hackney and Cleary, 1987) and hydrological alterations and diversions (Allison and Meselhe, 2010; Elias et al., 2003) can play an important role in determining present marsh condition and in turn, its resilience to sea level rise. Understanding the relative contributions of sea level rise and

* Corresponding author.
E-mail addresses: smithjam@gmail.com (J.A.M. Smith), Steven.Hafner@stockton.edu (S.F. Hafner), larry.niles@gmail.com (L.J. Niles).

https://doi.org/10.1016/j.ocecoaman.2017.09.010
0964-5691/

effects of management to tidal marsh losses, condition and resilience can allow for conservation and management strategies that are more targeted and effective.

In particular, identifying current and past management actions that negatively impact tidal marshes and threaten their future persistence (Gedan et al., 2009) can guide management at the site level to reverse these impacts through restoration (Weinstein et al., 2001). Conversely, if the cause of problems in marshes is assigned to sea level rise alone (Kirwan et al., 2010), practitioners may conclude that restoration actions will not correct the root problem (i.e. global carbon emissions) and therefore only represent a stop-gap on the way toward the inevitable outcome of marsh loss.

It is becoming clear that sea level rise alone may not necessarily spell the extinction of tidal marshes in many settings (Kirwan and Megonigal, 2013). Tidal marshes with higher marsh accretion rates and capacity for inland migration offer the potential for long-term persistence, particularly in estuaries with moderate to high tidal ranges (Kirwan et al., 2016). The Delaware Estuary, in concept, should represent a sea level rise-resilient tidal marsh system, with moderate (1.6–1.8 m) tidal range (Galperin and Mellor, 1990), high suspended sediment loads (Cook et al., 2007) and large frontage of undeveloped tidal marsh/upland ecotone (Smith, 2013) to allow for inland transgression (Kirwan et al., 2016). Despite these attributes, large acreages of these marshes have converted to open water and a significant proportion of the remaining marsh is in degraded condition (Kearney et al., 2002).

This paradox is not readily explained, but a consideration of past management of these marshes may offer insight. One important consideration is that large regions of Delaware Estuary tidal marsh were once impounded and drained for agricultural use (Philipp, 2005; Weinstein et al., 2000). Although the majority of these marshes are now not actively managed and are under tidal influence, the legacy of past impoundment may impact present and future resilience to sea level rise.

The practice of impounding and farming tidal marshes achieved large scale application in only a few places in North America (Nesbit, 1885), although large acreages of impounded tidal marsh are still extant northern Europe (Allen, 2000). In eastern North America, these places were the Delaware Bay (Nesbit, 1885), the Carolinas (Tompkins, 1987) and Canada Maritimes (Butzer, 2002; van Proosdij et al., 2013). The geographic, cultural and tidal context of these regions made landscape-scale impoundment of marshes for agricultural production both feasible and cost effective.

Impounded Delaware Estuary tidal marshes were used to grow field crops and "salt hay", which included a mix of high marsh plant species that were used for fodder and other purposes (Sebold, 1992). At least 6000 ha of marshes were impounded Salem County, New Jersey alone by the mid-1800s (Cook, 1870). This is in addition to comparable areas in the two other New Jersey counties bordering the bay (Cumberland, and Cape May) as well as along Delaware's coastline (Phillip, 1995; Sebold, 1992). To date, no comprehensive mapping of historically impounded marshes has been made.

While much of Canada Maritimes' "dykelands" are still intact, with 32,350 ha of impounded tidal marsh managed by provincial governments (van Proosdij et al., 2013), impoundment management was always a private enterprise in the Delaware Bay (New Jersey Legislature, 1911). As a result, under-engineered dikes, ditches and sluices needed constant maintenance and required close cooperation among adjacent landowners (Sebold, 1992; Vorst, 1977). When economic conditions changed beginning with the Great Depression and continuing through World War II, resources were too scarce to invest in impoundment maintenance and dikes began to lapse (Sebold, 1992; Stutz, 1992). Beyond the Depression, the practice became less economically viable over time. This coincided with the era of wetland conservation that saw state and federal governments, along with non-profits take over ownership of tidal marshes. With the exception of one large multi-site project that incorporated explicit restoration goals and actions (Teal and Weishar, 2005), these new conservation lands reverted to tidal flow in ad-hoc coastal realignment (Esteves, 2013) as dikes were allowed to lapse.

When tidal marshes are impounded, tidal flow is prevented from entering the marsh and drainage systems within the impoundment further dry the marsh to allow for farming activity. This change in hydrology exposes the marsh soils to air and the underlying peat soils begin to break down resulting in a loss of surface elevation (Portnoy, 1999; Roman et al., 1984; Warren, 1911; Weinstein et al., 2000; Weinstein and Weishar, 2002). Compaction from equipment, vegetation removal and tidal and sediment deprivation all contribute to further decrease elevation and work against the process of marsh vertical accretion in response to sea level rise (Bryant and Chabreck, 1998; Warren and Niering, 1993).

Impounded marshes behind dikes frequently fall to surface elevations that are too low to support natural marsh vegetation once dikes are removed (Weinstein and Weishar, 2002). Some marshes of the Delaware Bay have lost from 60 cm to 1.2 m of elevation after impoundment (Warren, 1911; Weinstein et al., 2000). Elevation loss likely varies as a result of landscape context and period during which the areas was impounded. The dramatic effects on elevation resulting from impoundment coupled with the large acreage of the Bay's marshes that were under this form of management suggests that the majority of the Delaware Bay's marshes may be, in many cases, significantly lower elevations than marshes that were never impounded. In some cases dike breaches have led to considerable subsequent losses in marsh area as a result of these elevation deficits (Weinstein et al., 2000).

To improve upon the management and restoration of Delaware estuary tidal marshes, we test the hypothesis that past impoundment explains present-day variation in tidal marsh elevation and vegetation composition. We do this by first determining a baseline historical extent of tidal marsh impoundment and second, examining whether these past practices have impacted present day marsh vegetation composition and elevation. Furthermore we estimate net change in tidal marsh area since 1931 by quantifying marsh conversion to open water and increases in area via marsh migration into upland. Our goal is to develop a revised tidal marsh conservation management model that incorporates both a perspective on climate change-induced causes of tidal marsh degradation and loss as well as those caused by past management practices (Almeida et al., 2014) which manifest themselves in present day marsh degradation and vulnerability to sea level rise.

2. Methods

2.1. Study area

We defined our study area as the New Jersey Delaware Bay tidal marsh extent in 1931 (Fig. 2). To delineate this area, we used National Wetlands Inventory (NWI) map areas (Wilen and Bates, 1995) classified as "Estuarine and Marine Wetland" as a starting point. We then digitized and added areas omitted by NWI that had been lost between 1931 and the creation/updating of NWI maps in the 1990s and 2000s. These area were either (1) interior marsh that had converted to mud/open water or (2) bay-fringing marsh lost to edge erosion (Phillips, 1986). To arrive at a total estimate of marsh edge erosion, we also mapped areas that had eroded since NWI mapping and present using 2015 imagery (NJ Office of Information Technology, Office of Geographic Information Systems Orthoimagery 2015).

We further mapped NWI areas of commission that classified areas of current open water and mudflat as estuarine and marine wetlands to compute a total estimate of marsh lost to this cover type since 1931.

To understand changes in marsh area at the upland and tidal marsh interface, we mapped areas of upland conversion to tidal marsh. We did this by first estimating conversion of forest to tidal marsh between 1931 and the creation of NWI maps (see below methods *historic and present marsh cover*). To estimate change since the creation of NWI maps, using 2015 imagery we digitized polygons representing the conversion of agricultural areas and NWI-coded "freshwater forested/shrub wetland" to tidal marsh.

2.2. Historic and present marsh cover

We generated 1000 random points across the study area to estimate both historic and present marsh cover (Nowak and Greenfield, 2010). At each point we classified land cover of the marsh surface for historic (1888–1931) and present (2006–2015) time frames. For the historic period, we primarily classified marsh as impounded (i.e. diked "reclaimed" marsh used for farming) or unimpounded. We made historic landcover decisions based on 1888 topographic maps (Vermeule, 1888) which mapped impounded marsh in certain areas (particularly in Salem county and along tidal rivers) and 1931 aerial imagery (NJ Office of Information Technology (NJOIT), Office of Geographic Information Systems (OGIS)) where salt hay and upland crops grown on reclaimed impounded tidal marshes look distinct from surrounding undiked marshes (Fig. 1). The 1931 imagery has a pixel resolution of 2 m and a root mean squared error (RMSE) of 11 m (Smith, 2013). We examined hydrology to inform classification as well because linear ditches (Fig. 1, distinct from mosquito grid ditches) in marshes during the 1880–1931 are also indicative of impoundment. We used LiDAR (2008 USGS South New Jersey County Project) and the South Jersey Levee Inventory (USDA Natural Resources Conservation Service, 2010) to look for evidence of existing and remnant dikes around impoundments.

For the present time frame, we assigned a broader range of classifications. We used 2006 true-color growing season imagery (USDA National Agriculture Imagery Program, 0.3 m pixel resolution aerial photography) to assign vegetation classifications and more recent 2015 imagery (NJOIT OGIS orthoimagery, 0.3 m pixel resolution aerial photography) to assign open water classifications (creek expansion, bay-edge erosion, and other open water/mud flats).

Vegetation categories had unique spectral signatures that are easily distinguishable in true color aerial imagery during growing season. These categories are low marsh (*Spartina alterniflora*), high marsh (comprised primarily of *Spartina patens*, *Distichilis spicata* and *Juncus gerardii)*, *Phragmites australis* marsh and a freshwater tidal marsh category comprised of a diverse array of species with wild rice (*Zizania aquatica*) dominant or co-dominant (Westervelt et al., 2006). This plant community occurs in the upper reaches of tidal creeks and in the upper reaches of the estuary. Additionally we noted where points fell into remnant or more recently created impounded marsh.

We used the past and present landcover classification at each random point to estimate percent cover of cover types across the study area. To estimate error around landcover percentages, we bootstrapped these frequency data with 1000 sampling iterations (Scheiner, 1998).

For categorical analyses of past vs. present landcover patterns based on percent cover estimates we used fisher's exact chi square test across all points with Bonferroni correction for multiple comparisons (Rice, 1989).

For the two categories for which it was possible (edge erosion and marsh loss to open water), we created an independent estimate of area change by hand-digitizing polygons over imagery for validation purposes.

2.3. Elevation patterns

We examined elevation patterns of tidal marsh across the study area using a LiDAR-derived digital elevation model for southern New Jersey (NOAA, 2008). There are known issues with LiDAR measurement error in tidal marsh, specifically it overestimates

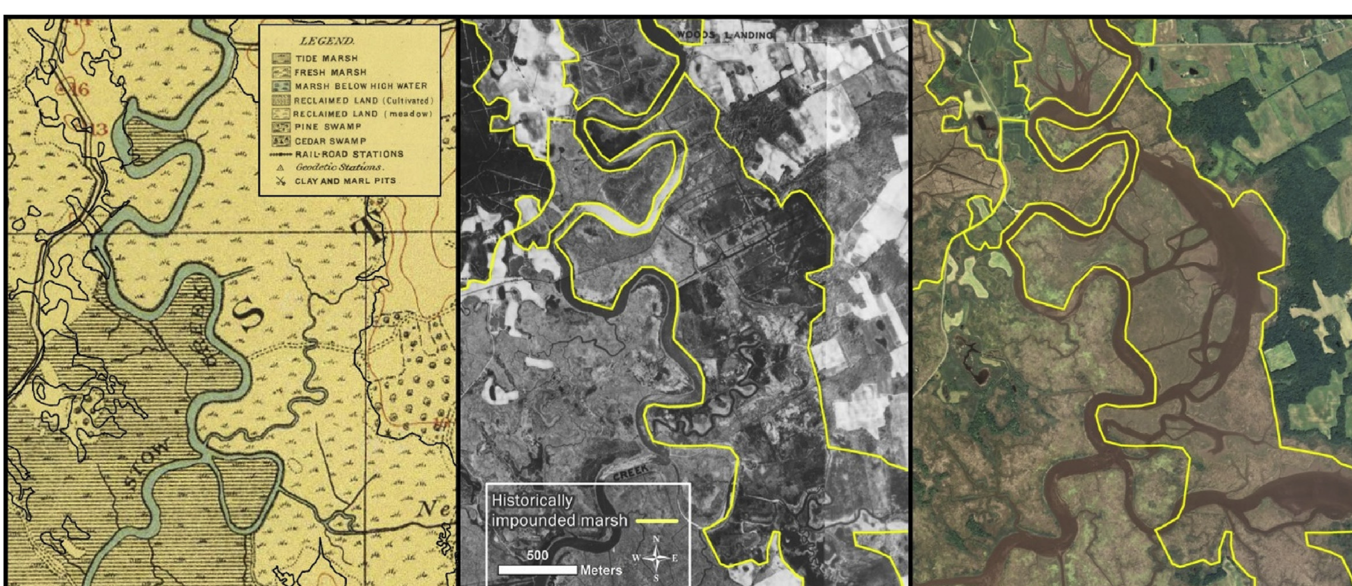

Fig. 1. Illustration of tidal marsh impoundment in (A) 1888 Vermeule maps where impounded marsh was mapped as "reclaimed land", (B) in 1931 aerial imagery when the impounded marsh was still under management, showing darker-colored managed vegetation and linear drainages and (C) 2006 NAIP imagery showing partial vegetation recovery, channel expansion and the loss of tidal marsh area to mudflat after farming ceased and impoundments breached. Location of this site within the study area is illustrated in Fig. 2.

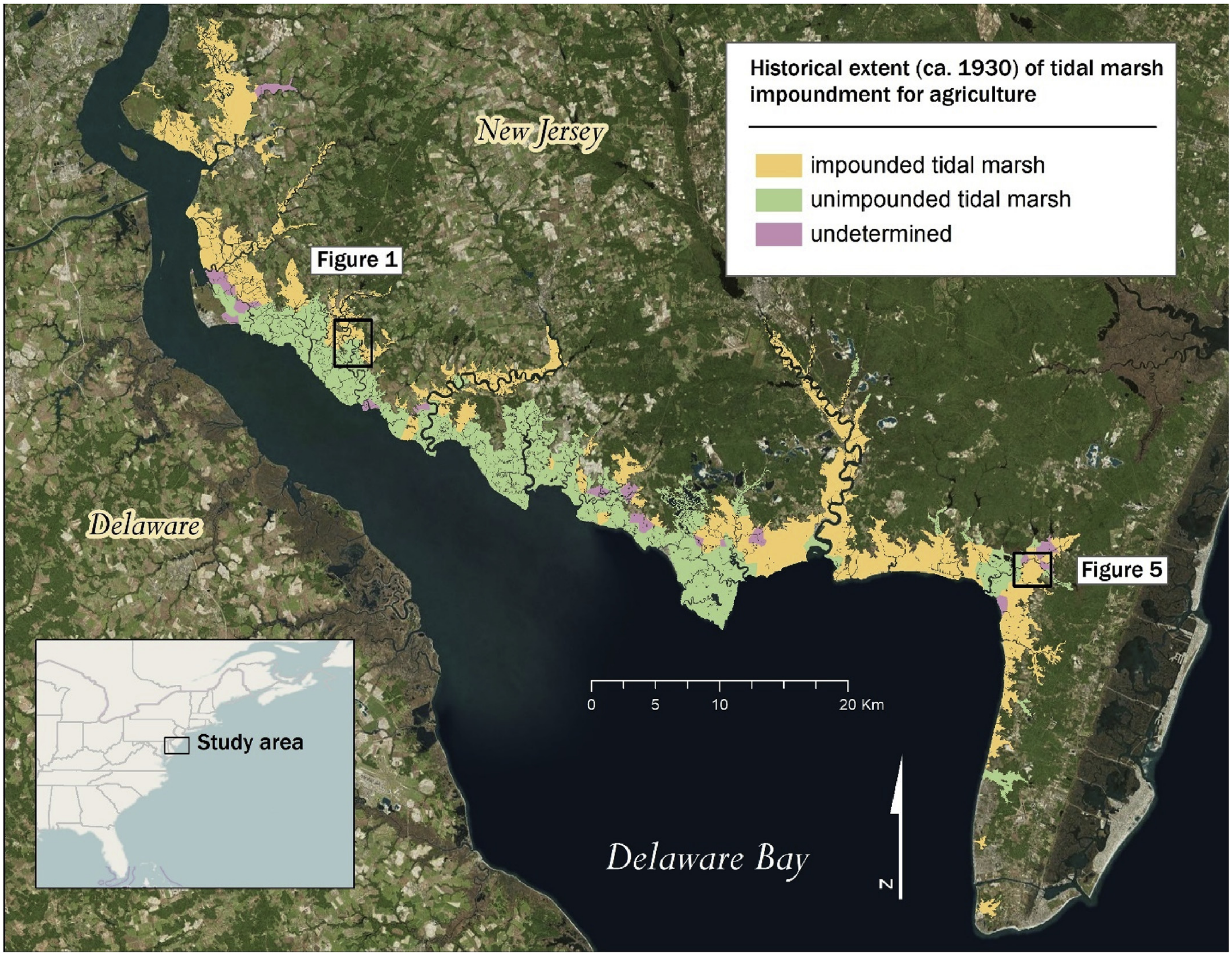

Fig. 2. Map of estimated historical extent of impounded tidal wetlands in the New Jersey portion of the Delaware Bay. Map is based on the point intercept classification of 1000 random points across the study area. For display purposes, these points are represented as Voronoi polygons. Undetermined areas are points where the marsh could not be confidently classified as impounded or unimpounded in the historical period between 1888 and 1931.

elevation in areas of taller, denser vegetation because point returns from above-ground vegetation are more likely to be misclassified as ground points (Chassereau et al., 2011; Millard et al., 2013; Schmid et al., 2011). To understand the magnitude and patterns of LiDAR elevation measurement error with this particular LiDAR dataset and study region, we collected a set of RTK GPS elevation measurements and characterized vegetation at each point ($n = 329$) in April and July 2015 at a 55 ha Delaware Bay tidal marsh study site (centroid at 39.19704, −74.9958). Points were arranged along 16 transects ranging from 200 m to 800 m long, spaced approximately 30 m apart. Transects spanned from lower elevations of open water and mudflat to the upper elevation range of salt marsh vegetation. At the point of each elevation measurement we recorded maximum vegetation height and the dominant plant species. Cover categories included mud/open water, low marsh (*Spartina alterniflora*) and high marsh (comprised of *Spartina patens*, *Distichlis spicata* and *Juncus gerardii*).

We used Classification and Regression Trees (CART, De'ath and Fabricius, 2000) to classify elevation thresholds that correspond to unvegetated intertidal areas, low marsh and high marsh. We conducted separate analyses with elevation data from RTK and LiDAR data sets in order to understand variation in classification thresholds and accuracy between the two data sources. We grew classification trees to the point where they described upper and lower thresholds for the three vegetation categories and we assessed predictive accuracy with ROC AUC. Using the elevation-vegetation thresholds we defined using CART, we then used chi-square analysis to examine the distribution of observations across these three elevation categories (unvegetated, low marsh, high marsh) between historically impounded and unimpounded marsh.

To examine patterns of elevation with respect to historic marsh management, we extracted LiDAR DEM data at each random point used for cover analysis. We divided these data into four categories: (1) marsh that was formerly impounded, (2) the proportion of this marsh that is currently vegetated and (3) marsh that was never historically impounded. We defined a fourth category by identifying several active impounded salt hay farms that still existed in 2008 when LiDAR data were collected (4 discrete sites comprising approximately 350 ha). We generated 200 additional random points in these areas in order to estimate elevation of actively managed impounded marsh to gain insights into how elevation varies between active vs historically farmed tidal marsh.

3. Results

3.1. Study area

The area we defined as our historical basis for the extent of tidal marsh on the New Jersey portion of the Delaware Estuary in 1931 encompasses 36,539 ha (Fig. 2). This includes 32,243 ha defined as estuarine and marine wetland in the National Wetlands Inventory. Added to this is 3118 ha of interior tidal marsh that converted to open water prior to NWI mapping and 1177 ha of tidal marsh at the bay's edge that was lost to erosion between the 1931 and the creation of NWI maps.

3.2. Historic and present marsh cover

For the historic period, of the 1000 points, we classified 513 as impounded farm, 381 as unmanaged tidal marsh, 51 as tidal marsh with undetermined management, 43 as forest, 12 in miscellaneous other categories. Overall, based on point intercept analysis, a minimum of 50.8% (47.8–53.9 95% CI) of the tidal marsh in our study area was historically impounded for farming (Fig. 2). This translates an area of 18,562 ha.

A comparison between historic impounded and unimpounded marshes across the range of present day categories (Table 1) indicates that historically impounded marsh in present day has significantly lower coverage of both low and high marsh and significantly higher coverage of freshwater tidal marsh, *Phragmites australis*, and interior marsh mudflat/open water when compared to marsh that was never historically impounded.

Conversion of interior marsh to open water was the single greatest source of marsh loss from 1931 to present. This form of loss was almost exclusively confined to areas that were formerly impounded, representing approximately 11% of historic marsh area (Fig. 3., 4055.8 ± 1113.7 SD ha). An independent estimate of the area based on polygon mapping of open water loss produced an estimate of 3634 ha. In cases where impoundment boundaries coincided with the bay shoreline, these losses also include the sandy beaches that rest upon the bay's fringing saltmarshes. Of these beaches, 2.3 km have been lost to open water where low elevations behind the beach berm caused the beach to disintegrate, dispersing sand across the intertidal area after dike breaches.

Added to this is marsh lost across all historic categories due to bay-edge erosion (4.7%, 1717 ± 255.7 SE ha). An independent estimate of edge erosion based on polygon mapping totaled 1624 ha. Creek and ditch lateral expansion represented and additional 5.8% decrease in tidal marsh area (2119 ± 255.7 SE ha). Tidal marsh area increased via inland migration into uplands, totaling approximately 2816 ha. This includes 1425 ± 219.2 SE ha gained between 1931 and the creation of NWI maps and an additional 1391 ha since the NWI maps were initially produced (Fig. 3).

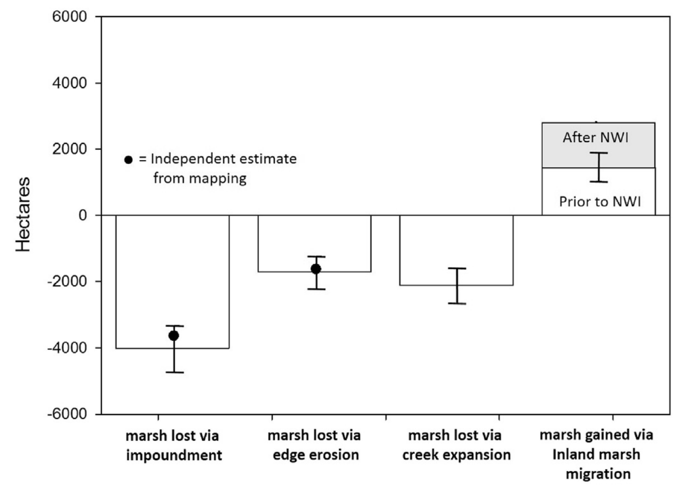

Fig. 3. Estimates of marsh loss and gains between 1931 and 2015. Estimates are based on the present-day percentage of random point intercepts in each category across the historical baseline tidal marsh study area. 95% CI are calculated from bootstrapped frequency data. Independent point estimates from mapping are based on mapped polygons.

3.3. Vegetation-elevation patterns – correspondence between LiDAR and rtk data

LiDAR and RTK GPS elevations were highly correlated ($R^2 = 0.93$, DF = 1, $p < 0.0001$). On average LiDAR overestimated true elevation by 11.4 cm (±0.65 SE). Positive measurement bias increased as vegetation height increased ($R^2 = 0.44$, DF = 1, $p < 0.0001$). Average vegetation height at RTK GPS elevation points for *Spartina alterniflora* was 95 cm (±2.3 SE), while average high marsh vegetation height was 32.7 cm (±5.6 SE). Vegetation height and elevation (RTK GPS) are negatively correlated ($R^2 = 0.66$, DF = 1, $p < 0.0001$, $n = 100$), so that elevation estimates of tidal marsh in the lower range of elevations will have the tallest vegetation and the greatest LiDAR-derived positive-biased elevation error.

Vegetation-elevation thresholds via CART were very similar for both LiDAR and RTK GPS elevation data. For LiDAR, mud and open water predominated at elevations below 0.43 m (AUC 0.93), vegetated low marsh occurred between 0.43 and 0.92 m (AUC 0.86) and high marsh vegetation began to occur at elevations above 0.92 m (AUC 0.84). Thresholds for RTK GPS data were nearly identical, with mud and open water at elevations below 0.43 m (AUC 0.99), vegetated low marsh between 0.43 and 0.94, (AUC 0.95) and high marsh occurring at elevations above 0.94 (AUC 0.86).

Including mudflat and open water, formerly impounded tidal marsh areas (mean 0.43 m ± 0.51 SD) are more than 0.4 m below ($Z = -11.03$, $p < 0.0001$) the elevation of marshes that were never impounded (0.84 ± 0.22 SD). Excluding mudflat and examining the subset of formerly impounded marsh that has revegetated, the

Table 1
A comparison of present-day tidal marsh composition for marshes that were either historically impounded or unimpounded. Percentages in columns represent the present-day vegetation cover within each historic landcover category. Headings with an asterisk significant differences in present-day coverage of that category between historically impounded vs unimpounded marsh ($\alpha = 0.00625$, Fishers exact test 2 × 2 comparisons).

Historic Landcover	Current Landcover (percent cover of historic area)							
	low marsh*	high marsh*	freshwater tidal marsh*	phragmites australis*	mud flat/ open water*	coastline edge erosion*	tidal channel lateral expansion	impounded
impounded and farmed tidal marsh	36.3%	1.0%	4.5%	19.5%	21.6%	3.1%	6.4%	4.9%
unimpounded tidal marsh	62.5%	17.1%	0%	5.0%	0.5%	7.9%	5.2%	1.6%
p-value	<0.0001	<0.0001	<0.0001	<0.0001	<0.0001	0.0021	0.48	0.0087

elevation of the formerly impounded marsh is still significantly lower (by 10 cm, $Z = -4.76$, $p < 0.0001$) than never-impounded marsh (Fig. 4). Given our results of increasing positive bias at lower elevations with taller vegetation, these differences represent a minimum estimate of elevation deficits in formerly impounded areas.

Areas that were still impounded and farmed when 2008 LiDAR data were collected had elevations which were well below (mean 0.18 m ± 0.22 SD) those necessary to support vegetated marsh in the absence of dikes (Fig. 4). This was illustrated by dike breaches at one site subsequent to 2008 when farmed high marsh became open water and mudflat (Fig. 5).

Using the elevation-vegetation thresholds we defined above with CART analysis, we found significant differences between historically impounded and unimpounded marsh across three elevation-vegetation categories (high marsh, low marsh and unvegetated). Based on the random points used for landcover estimation, formerly impounded revegetated marsh (Fig. 4) had significantly fewer observations in high marsh elevation class (20.1 vs 43.1%, $X^2 = 24.1$, $p < 0.0001$) and significantly more observations in low marsh (69.3 vs 52.3%, $X^2 = 14.2$, $p = 0.0002$) and unvegetated elevation classes (10.5 vs 4.6%, $X^2 = 6.3$, $p = 0.01$) than marsh that was not historically impounded.

4. Discussion

We estimate that more than half Delaware Estuary tidal marshes were subject to impoundment for farming from early colonial period through the early 20th century. By comparing formerly farmed marshes with marshes that were never impounded, we demonstrate that impoundment has left lasting impacts, with important implications for tidal marsh resiliency to sea level rise. The primary impact is lower marsh surface elevation, which in turn has resulted in a reduction of high marsh vegetation and the loss of approximately 4000 ha of marsh to open water, comprising the largest single contributor to marsh loss in the study area. The inconsistent recovery of these marshes is related not only to elevation, but also potentially to a site's wave exposure, tidal flow velocities and the arrangement and density of remnant creek channels and ditches which may limit the accumulation of sediment to elevations that can support vegetation (Weinstein et al., 2000).

We confirm with elevation data from both former and contemporary impounded marsh that significant elevation deficits result from the practice of impoundment and farming. Given the bias of LiDAR-derived elevation data in taller vegetation which occurs at lower elevations, the true elevation deficit of formerly impounded, currently vegetated marsh is likely greater than reported here. Vegetated formerly impounded marsh is typically covered with tall *Spartina alterniflora* ranging between 50 and 150 cm in height (Kreeger et al., 2015). Therefore, the deficit may be 30 cm or more, based on the error evident in our field validation, rather than the 10 cm deficit apparent with LiDAR data.

The elevation deficits we documented using LiDAR for actively impounded areas (70 cm, having shorter vegetation with less positive bias) fall into the range found in other studies. In a diked salt marsh in Cape Cod, Massachusetts, Portnoy and Giblin (1997) found a 90 cm deficit compared with surrounding unaltered marsh while Roman et al. (1984) found a 20–40 cm deficit in impounded salt marshes in Long Island Sound, Connecticut.

Other sources of marsh loss include shoreline edge erosion and creek expansion, totaling approximately 3800 ha. At least some of this creek expansion may be the result of channel adjustment to increases in water volume and velocity as tides returned to previously impounded marshes (Williams et al., 2002). Much of the losses from erosion and channel widening are offset by a gain in marsh area via inland marsh migration, an estimated 2815 ha. This suggests that in the absence historic impoundment practices, Delaware Bay marshes would have experienced considerably less net loss.

Indeed, tidal marshes that were never impounded have experienced less change over the 1931–2015 time frame of our analysis. Although marsh area was lost to edge erosion and creek expansion, in contrast with formerly impounded marsh, relatively little area has been lost via conversion of marsh surface to open water. These natural marshes also hold the majority of the estuaries' high marsh.

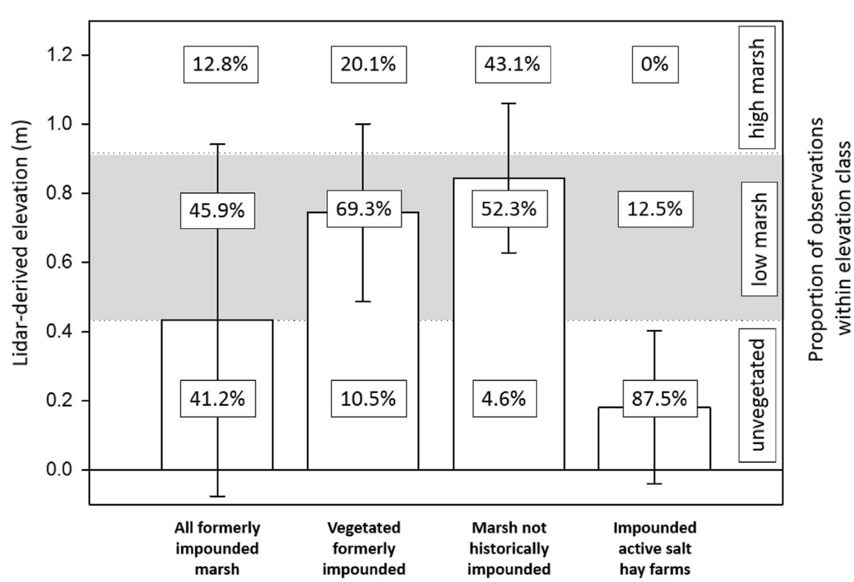

Fig. 4. LiDAR-derived elevation for formerly and actively impounded marsh compared with marsh regions that were never impounded. Percentages indicate the percent of observations the fell into three elevation categories derived from CART analyses of elevation-vegetation relationships, indicating significantly lower frequency of elevations that can support high marsh in formerly impounded areas. For areas that were still actively impounded in 2008, the majority of observations fell into the elevation category that is too low to support any tidal marsh vegetation.

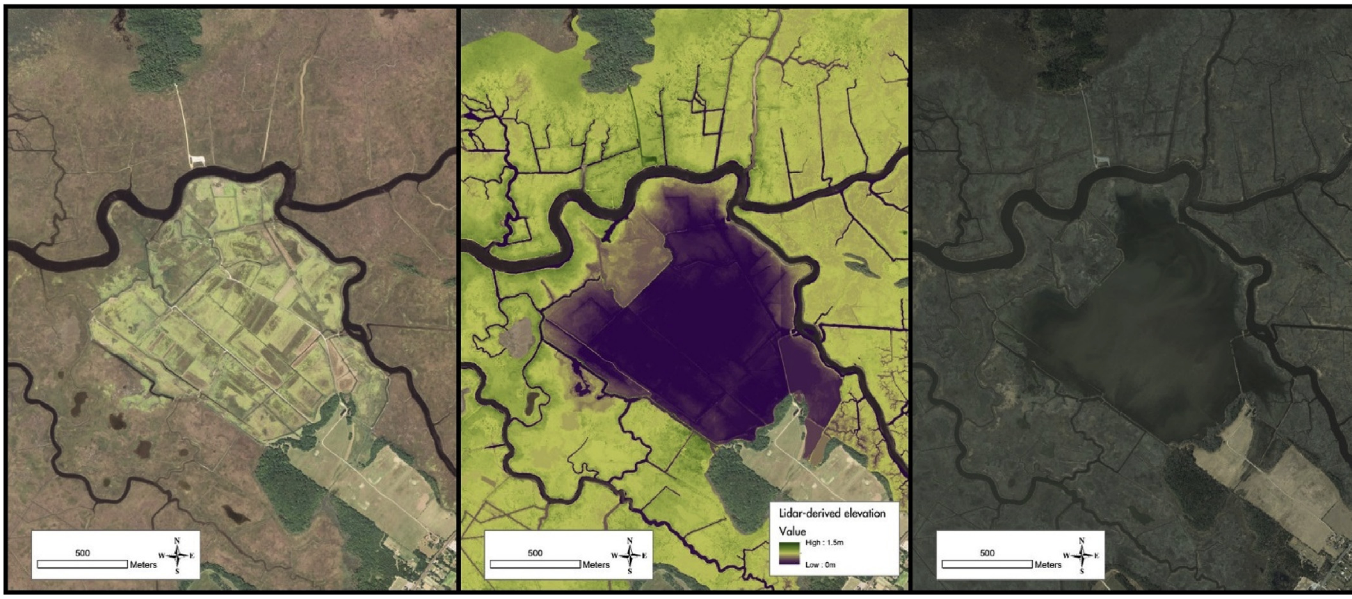

Fig. 5. Image sequence depicting an impounded salt hay farm that was (A) active in 2008, showing light green high marsh vegetation (dominated by *Spartina patens* and *Distichlis spicata*) that contrasts with darker green/brown surrounding *Spartina alterniflora* marsh (B) A LiDAR digital elevation model from the same year shows significantly lower than surrounding unimpounded marshes. After Hurricane Sandy, dikes suffered catastrophic breaches and reintroduction of tidal flow that resulted in (C) conversion of the area to open water and mud flat. Location of this site within the study area is illustrated in Fig. 2. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Being at the upper extreme of the elevational range where marshes exist, high marsh is an indicator of marsh resilience to sea level rise. Increased variation in topography (Temmerman et al., 2004) and vegetation types increases overall species diversity and high marsh is the primary breeding habitat for several tidal marsh endemic birds and invertebrates (Greenberg et al., 2006; Tiner, 2013).

Our results indicate that high marsh habitat is greatly reduced in the Delaware estuary as a result of past impoundment practices. Based on our observations in areas that were never impounded, we estimate that the minimum reference ratio of high marsh to low marsh in restoration design be 1:5. It should be noted that a significant portion of existing unimpounded high marsh was and is mowed for salt hay (Smith unpublished data). We are unable to quantify the impacts to marsh condition of this practice, but assuming some degree of degradation resulting from decades and perhaps centuries of mowing, 1:5 must be considered a minimum ratio of the former extent of high marsh habitat.

Formerly impounded areas have significantly higher coverage of freshwater tidal marsh and *Phragmites australis* when compared with marsh that was never impounded. This difference is not due to the past impoundment of marshes *per se*, rather it reflects the geographic pattern of siting impoundments in lower-salinity settings along tidal river courses and in the upper bay where these vegetation communities form. It is notable that almost all of the present-day freshwater tidal marsh in our study area was formerly impounded. This suggests that present-day freshwater tidal marshes may differ significantly in character from such marshes that were never impounded.

Sea level rise in the mid-Atlantic is among the highest rates on the east coast (Engelhart et al., 2009). While this translates to significant wetland losses as coastlines maintain equilibrium with sea level (Kirwan and Murray, 2008; Wilson and Allison, 2008), it also can translate to significant inland gains of marshes at tidal wetlands encroach on upland habitats (Smith, 2013). Conversion of upland to tidal marsh between 1990 and present has occurred at more than double the rate (23 ha/yr) compared to the period between 1931 and 1990 (9.3 ha/yr). The accelerated pace of sea level rise may in part be responsible for the more rapid conversion of upland to tidal marsh in recent years. A complementary explanation is that many impounded marsh areas also artificially protected uplands from tides and sea level rise. Subsequent dike breaches then caused the sudden transition of uplands to tidal wetlands as sea level reached its proper level along the upland/marsh ecotone. The large connected interface between uplands and wetlands across the largely forested and agricultural landscape has allowed for considerable offsetting of wetland losses in the Delaware Estuary. Nonetheless impediments to inland tidal marsh transgression do exist in many places along the interface zone that prevent these offsets from reaching their maximum potential (USDA Natural Resources Conservation Service, 2010).

While inland marsh transgression gains are critical for offsetting marsh loss, it is important to note that the character of these inland marshes that form beneath dying forest and inundated agricultural land are different than the outer fringe marshes that they replace — with, for example, a higher cover of *Phragmites australis* (Anisfeld et al., 2016; Smith, 2013) and potentially poorer habitat quality for tidal marsh dependent vertebrates. As part of an overall tidal marsh conservation strategy, complementary management actions that focus on retaining and restoring existing tidal marsh are necessary to maintain overall habitat diversity and function.

Our results indicate that the primary source of marsh degradation in the Delaware Estuary is not related to climate change and sea level rise. It is instead the result of management decisions. As these impounded marshes came into the hands of state, federal and non-profit ownership, the dominant management strategy was to allow dikes to breach and permit the passive recovery (Elliott et al., 2007) of tidal flow to marshes (Almeida et al., 2014; Slavin and Shisler, 1983). While these sweeping changes exposed a much larger marsh area to tidal inundation and has increased estuarine productivity and function, the results of this study confirm that the recovery of marsh area has been erratic (Phillip, 1995; Weinstein et al., 2000). Furthermore the long term persistence of these recovering marshes are in question because, although vertical accretion of Delaware Estuary marshes can likely keep pace with the current rate of relative sea level rise (Boyd et al., 2017; McDowell, 2017), they likely cannot also recover the large elevation deficits that our results show are the legacy of impoundment.

One notable example of active restoration was the PSEG power company-funded Estuary Enhancement Project (Teal and Peterson, 2005) which restored tidal flow to three formerly impounded marsh sites as part of special conditions for a power-generating permit. The project strategically breached dikes and dredged new tidal channels in impounded marshes designed to promote accelerated accretion (Teal and Weishar, 2005). The marshes largely revegetated but still remain today at elevations considerably below those of marshes that were never impounded. The architects of this restoration project recognized these elevation deficits and proposed as a solution sediment addition, but this tactic was not part of the restoration (Weinstein and Weishar, 2002).

Overall Delaware estuary marshes are shown to be relatively resilient. With the exception of impounded areas, little marsh has been lost to vertical drowning, considerable gains are being made inland to offset horizontal loss, and the moderate tidal range and suspended sediment load (Cook et al., 2007; Sommerfield and Wong, 2011) support adequate accretion (Boyd et al., 2017; McDowell, 2017) necessary to match the current (and perhaps an accelerated) pace of sea level rise (Kirwan et al., 2016).

On the negative side, more than half of these marshes were impounded, which has reduced tidal marsh resilience to sea level rise. These effects are largely the result of elevation deficits. Restoration techniques of sediment addition has the potential to offset these deficits. More than three million cubic meters of dredged sediment are removed from the Delaware Estuary annually (Delaware Estuary Regional Sediment Management Workgroup, 2013) and the beneficial use of such sediment for marsh restoration is emerging as a mainstream practice (DeLaune et al., 1990; Mendelssohn and Kuhn, 2003; Yozzo et al., 2004). Such an approach to restoration is the only hope for recovering and ensuring the resilience of tidal marshes impacted by past impoundment. Although “thin-layer application” (6–8″) is a widely used term for this practice (Ford et al., 1999), for the Delaware Estuary marshes, in many cases the depth of sediment needed will be much greater from 1 to 2' or more. Understanding that current degradation is in large part related to the past management of these marshes empowers us to develop a new management paradigm of marsh restoration to ensure their resilience as sea level rise progresses and accelerates.

To minimize net loss of wetlands and to ensure the long-term persistence of tidal marshes in the region, we recommend that managers (1) institutionalize the use of dredged sediment for marsh restoration in order to recover elevation and increase habitat diversity, (2) maximize the potential for inland marsh migration through strategic land protection at the upland/tidal marsh interface and (3) restore tidal flow and elevation in remaining impounded or tidally restricted areas where feasible to maximize tidal marsh area and the potential for inland migration.

5. Conclusion

The results presented here illustrate that research and management in Delaware Estuary tidal marshes must explicitly account for the past history of impoundment in order to understand the broader impact of sea level rise apart from that of site-level management. This distinction yields the insight that conversion of marsh to open water is limited primarily to formerly impounded marshes. For those areas that were not historically impounded, current evidence suggests that Delaware Estuary tidal marshes are relatively resilient to sea level rise in their capacity to accrete vertically and expand horizontally inland. For formerly impounded marshes, these resilient traits also suggest that the investments in restoration will continue to pay dividends because natural feedbacks in the system will allow projects to maintain themselves once the dramatic elevation deficits that are the legacy of impoundment are corrected.

More broadly, we argue that tidal marsh conservation and management decision-making must explicitly consider the relative contributions of past and present management actions to marsh ecological integrity and resilience apart from the effects of climate change. Practices such as tidal range alteration (Swanson and Wilson, 2008), tidal restriction (Roman et al., 1984), impoundment (Bryant and Chabreck, 1998) and ditching (LeMay 2007) all can play an important role in influencing the integrity and long-term trajectory of tidal marshes. Disentangling the effects of site-level management from those of climate change on marsh condition can have an important influence on scientists' and managers perceptions of the resilience of tidal marshes to sea level rise. For example, if the impacts of site-level management actions are ignored or discounted, researchers and managers may overestimate the deleterious impacts of sea level rise. This interpretation may in turn alter the choice of actions taken (or not taken) to manage and restore tidal marshes. A refined understanding of tidal marshes that parses the effects of site-level management actions from the broader impacts of sea level rise and other factors will result in greater utility of research results along with more targeted and effective management strategies.

Acknowledgements

We thank Joseph Atzert, Mark Caltagirone and Nicholas Van Wattingen for assistance with fieldwork. We thank Kenneth Strait of PSEG for sharing his salt marsh restoration experiences and for providing access to field sites. This project was possible with grants from the National Fish and Wildlife Foundation (Project ID: 2300.14.043429) to American Littoral Society and the Dupont Clear into the Future program to Conserve Wildlife Foundation of New Jersey. We thank Tim Dillingham and Capt. Al Modjeski of American Littoral Society and David Wheeler and Stephanie Feigin of Conserve Wildlife Foundation of NJ for their support.

References

Allen, J.R.L., 2000. Morphodynamics of holocene salt marshes: a review sketch from the atlantic and southern North sea coasts of Europe. Quat. Sci. Rev. 19, 1155–1231.
Allison, M.A., Meselhe, E.A., 2010. The use of large water and sediment diversions in the lower Mississippi River (Louisiana) for coastal restoration. J. Hydrol. 387, 346–360.
Almeida, D., Neto, C., Esteves, L.S., Costa, J.C., 2014. The impacts of land-use changes on the recovery of saltmarshes in Portugal. Ocean Coast. Manag 92, 40–49.
Anisfeld, S.C., Cooper, K.R., Kemp, A.C., 2016. Upslope development of a tidal marsh as a function of upland land use. Glob. Change Biol. n/a-n/a).
Bourn, W.S., 1950. Some Biological Effects of Ditching Tidewater Marshes. U.S. Fish and Wildlife Service, Washington D.C.
Boyd, B.M., Sommerfield, C.K., Elsey-Quirk, T., 2017. Hydrogeomorphic influences on salt marsh sediment accumulation and accretion in two estuaries of the U.S. Mid-Atlantic coast. Mar. Geol. 383, 132–145.
Bryant, J.C., Chabreck, R.H., 1998. Effects of impoundment on vertical accretion of coastal marsh. Estuaries 21, 416–422.
Butzer, K.W., 2002. French wetland agriculture in Atlantic Canada and its European roots: different avenues to historical diffusion. Ann. Assoc. Am. Geogr. 92, 451–470.
Chassereau, J.E., Bell, J.M., Torres, R., 2011. A comparison of GPS and lidar salt marsh DEMs. Earth Surf. Process. Landf. 36, 1770–1775.
Cook, G.H., 1870. Annual Report of the State Geologist of New Jersey, for 1869. New Jersey Geological Survey, Trenton, NJ.
Cook, T.L., Sommerfield, C.K., Wong, K.-C., 2007. Observations of tidal and springtime sediment transport in the upper Delaware Estuary. Estuar. Coast. Shelf Sci. 72, 235–246.
De'ath, G., Fabricius, K.E., 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecology 81, 3178–3192.
DeLaune, R.D., Pezeshki, S.R., Pardue, J.H., Whitcomb Jr., J.H., W.H.P, 1990. Some

influences of sediment addition to a deteriorating salt marsh in the Mississippi river deltaic plain: a pilot study. J. Coast. Res. 6, 181–188.
Delaware Estuary Regional Sediment Management Workgroup, 2013. Dredging and Dredged Material Management, Delaware Estuary Regional Sediment Management Plan White Paper.
Elias, E., Stive, M., Bonekamp, H., Cleveringa, J., 2003. Tidal inlet dynamics in response to human intervention. Coast. Eng. J. 45, 629–658.
Elliott, M., Burdon, D., Hemingway, K.L., Apitz, S.E., 2007. Estuarine, coastal and marine ecosystem restoration: confusing management and science – a revision of concepts. Estuar. Coast. Shelf Sci. 74, 349–366. http://dx.doi.org/10.1016/j.ecss.2007.05.034.
Engelhart, S.E., Horton, B.P., Douglas, B.C., Peltier, W.R., Törnqvist, T.E., 2009. Spatial variability of late Holocene and 20th century sea-level rise along the Atlantic coast of the United States. Geology 37, 1115.
Esteves, L.S., 2013. Is managed realignment a sustainable long-term coastal management approach? J. Coast. Res. 933–938.
Ford, M.A., Cahoon, D.R., Lynch, J.C., 1999. Restoring marsh elevation in a rapidly subsiding salt marsh by thin-layer deposition of dredged material. Ecol. Eng. 12, 189–205.
Galperin, B., Mellor, G.L., 1990. A time-dependent, three-dimensional model of the Delaware Bay and River system. Part 1: description of the model and tidal analysis. Estuar. Coast. Shelf Sci. 31, 231–253.
Gedan, K.B., Silliman, B.R., Bertness, M.D., 2009. Centuries of human-driven change in salt marsh ecosystems. Annu. Rev. Mar. Sci. 1, 117–141.
Greenberg, R., Maldonado, J.E., Droege, S., McDonald, M.V., 2006. Tidal marshes: a global perspective on the evolution and conservation of their terrestrial vertebrates. BioScience 56, 675–685.
Hackney, C.T., Cleary, W.J., 1987. Saltmarsh loss in southeastern North Carolina lagoons: importance of sea level rise and inlet dredging. J. Coast. Res. 3, 93–97.
Kearney, M.S., Rogers, A.S., Townshend, J.R.G., Rizzo, E., Stutzer, D., Stevenson, J.C., Sundborg, K., 2002. Landsat imagery shows decline of coastal marshes in Chesapeake and Delaware Bays. Eos Trans. Am. Geophys. Union 83, 173–177.
Kennish, M.J., 2001. Coastal salt marsh systems in the U.S.: a review of anthropogenic impacts. J. Coast. Res. 17, 731–748.
Kirwan, M.L., Guntenspergen, G.R., D'Alpaos, A., Morris, J.T., Mudd, S.M., Temmerman, S., 2010. Limits on the adaptability of coastal marshes to rising sea level. Geophys. Res. Lett. 37, L23401.
Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature 504, 53–60.
Kirwan, M.L., Murray, A.B., 2008. Tidal marshes as disequilibrium landscapes? Lags between morphology and Holocene sea level change. Geophys. Res. Lett. 35, L24401.
Kirwan, M.L., Temmerman, S., Skeehan, E.E., Guntenspergen, G.R., Fagherazzi, S., 2016. Overestimation of marsh vulnerability to sea level rise. Nat. Clim. Change 6, 253–260.
Kreeger, D., Moody, J., Katkowski, M., Boatright, M., Rosencrance, D., 2015. Marsh Futures: Use of Scientific Survey Tools to Assess Local Salt Marsh Vulnerability and Chart Best Management Practices and Interventions (No. PDE Report No. 15–03). Partnership for the Delaware Estuary, Wilmington, DE.
LeMay, L.E., 2007. The Impact of Drainage Ditches on Salt Marsh Flow Patterns, Sedimentation and Morphology. College of William and Mary, Rowley River, Massachusetts (M.S. thesis).
McDowell, C., 2017. Marsh Sediment Accumulation and Accretion on a Rapidly Retreating Estuarine Coast. University of Delaware.
Mendelssohn, I.A., Kuhn, N.L., 2003. Sediment subsidy: effects on soil–plant responses in a rapidly submerging coastal salt marsh. Ecol. Eng. 21, 115–128.
Millard, K., Redden, A.M., Webster, T., Stewart, H., 2013. Use of GIS and high resolution LiDAR in salt marsh restoration site suitability assessments in the upper Bay of Fundy, Canada. Wetl. Ecol. Manag. 21, 243–262.
Nesbit, D., 1885. Tide Marshes of the United States. Govt. Print. Off., Washington.
New Jersey Legislature, 1911. Compiled Statutes of New Jersey: intoxication liquors - replevin. Soney & Sage.
NOAA, 2008. U.S. Coastal Lidar Elevation Data.
Nowak, D.J., Greenfield, E.J., 2010. Evaluating the national land cover database tree canopy and impervious cover estimates across the conterminous United States: a comparison with photo-interpreted estimates. Environ. Manage 46, 378–390.
Philipp, K.R., 2005. History of Delaware and New Jersey salt marsh restoration sites. Ecol. Eng. 25, 214–230.
Phillip, K., 1995. Tidal Wetlands Characterization - Then and Now (Report submitted to Delaware River Basin Commission).
Phillips, J.D., 1986. Spatial analysis of shoreline erosion, Delaware Bay, New Jersey. Ann. Assoc. Am. Geogr. 76, 50–62.
Portnoy, J.W., 1999. Salt marsh diking and restoration: biogeochemical implications of altered wetland hydrology. Environ. Manage 24, 111–120.
Portnoy, J.W., Giblin, A.E., 1997. Effects of historic tidal restrictions on salt marsh sediment chemistry. Biogeochemistry 36, 275–303.
Rice, W.R., 1989. Analyzing tables of statistical tests. Evolution 43, 223–225.
Roman, C.T., Niering, W.A., Warren, R.S., 1984. Salt marsh vegetation change in response to tidal restriction. Environ. Manage 8, 141–149.
Scheiner, S., 1998. Design and Analysis of Ecological Experiments. CRC Press.
Schmid, K.A., Hadley, B.C., Wijekoon, N., 2011. Vertical accuracy and use of topographic LIDAR data in coastal marshes. J. Coast. Res. 27, 116–132.
Sebold, K., 1992. From Marsh to Farm: the Landscape Transformation of Coastal New Jersey. US Department of the Interior, National Park Service, Washington D.C.
Silliman, B.R., Grosholz, E., Bertness, M.D., 2009. Human Impacts on Salt Marshes: a Global Perspective. Univ of California Press.
Slavin, P., Shisler, J.K., 1983. Avian utilisation of a tidally restored salt hay farm. Biol. Conserv. 26, 271–285.
Smith, J.A.M., 2013. The role of Phragmites australis in mediating inland salt marsh migration in a mid-atlantic estuary. PLoS ONE 8, e65091.
Sommerfield, C.K., Wong, K.-C., 2011. Mechanisms of sediment flux and turbidity maintenance in the Delaware Estuary. J. Geophys. Res. Oceans 116, C01005.
Stutz, B., 1992. Natural Lives, Modern Times: People and Places of the Delaware River, first ed. (Crown, New York).
Swanson, R.L., Wilson, R.E., 2008. Increased tidal ranges coinciding with Jamaica Bay development contribute to marsh flooding. J. Coast. Res. 1565–1569.
Teal, J., Weishar, L., 2005. Ecological engineering, adaptive management, and restoration management in Delaware Bay salt marsh restoration. Ecol. Eng. 25, 304–314.
Teal, J.M., Peterson, S.B., 2005. Introduction to the Delaware Bay salt marsh restoration. Ecol. Eng. 25, 199–203.
Temmerman, S., Govers, G., Meire, P., Wartel, S., 2004. Simulating the long-term development of levee–basin topography on tidal marshes. Geomorphology 63, 39–55.
Tiner, R.W., 2013. Tidal Wetlands Primer. University of Massachusetts Press.
Tompkins, M.E., 1987. South Carolina's diked tidal wetlands: the presisting dilemmas. Coast. Manag. 15, 135–155.
USDA Natural Resources Conservation Service, 2010. South Jersey Levee Inventory. USDA Natural Resources Conservation Service Assisting New Jersey Department of Environmental Protection Bureau of Dam Safety and Flood Control.
van Proosdij, D., Perrott, B., Carrol, K., 2013. Development and application of a geotemporal atlas for climate change adaptation in bay of fundy dykelands. J. Coast. Res. 1069–1074.
Vermeule, C.C., 1888. New Jersey State Map.
Vorst, J.V., 1977. Cedar Swamp Creek: Stories and Sketches of the Area self-published).
Warren, G., 1911. Tidal Marshes and Their Reclamation. U.S. Dept. of Agriculture, Washington D.C.
Warren, R.S., Niering, W.A., 1993. Vegetation change on a northeast tidal marsh: interaction of sea-level rise and marsh accretion. Ecology 74, 96–103.
Weinstein, M.P., Philipp, K.R., goodwin, P., 2000. Catastrophes, near-catastrophes and the bounds of expectation: success criteria for macroscale marsh restoration. In: Concepts and Controversies in Tidal Marsh Ecology. Kluwer Academic Publishers, New York, pp. 777–804.
Weinstein, M.P., Teal, J.M., Balletto, J.H., Strait, K.A., 2001. Restoration principles emerging from one of the world's largest tidal marsh restoration projects. Wetl. Ecol. Manag. 9, 387–407.
Weinstein, M.P., Weishar, L.L., 2002. Beneficial use of dredged material to enhance the restoration trajectories of formerly diked lands. Ecol. Eng. 19, 187–201.
Westervelt, K., Largay, E., Coxe, R., McAvoy, W., Perles, S., Podniesinski, G., Sneddon, L., Walz, K., 2006. A Guide to the Natural Communities of the Delaware Estuary: Version I.
Wilen, B.O., Bates, M.K., 1995. The US Fish and Wildlife Service's national wetlands inventory project. Vegetation 118, 153–169.
Williams, P.B., Orr, M.K., Garrity, N.J., 2002. Hydraulic geometry: a geomorphic design tool for tidal marsh channel evolution in wetland restoration projects. Restor. Ecol. 10, 577–590. http://dx.doi.org/10.1046/j.1526-100X.2002.t01-1-02035.x.
Wilson, C.A., Allison, M.A., 2008. An equilibrium profile model for retreating marsh shorelines in southeast Louisiana. Estuar. Coast. Shelf Sci. 80, 483–494.
Wolfe, R.J., 1996. Effects of open marsh water management on selected tidal marsh resources: a review. J. Am. Mosq. Control Assoc. 12, 701–712.
Yozzo, D.J., Wilber, P., Will, R.J., 2004. Beneficial use of dredged material for habitat creation, enhancement, and restoration in New York–New Jersey Harbor. J. Environ. Manage 73, 39–52.