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SPECIAL ISSUE: CONCEPTS AND CONTROVERSIES IN TIDAL MARSH ECOLOGY REVISITED

Protecting People and Property While Restoring Coastal Wetland Habitats

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Abstract

Flood mitigation and protection of coastal infrastructure are key elements of coastal management decisions. Similarly, regulating and provisioning roles of coastal habitats have increasingly prompted policy makers to consider the value of ecosystem goods and services in these same decisions, broadly defined as "the benefits people obtain from ecosystems." We applied these principles to a study at three earthen levees used for flood protection. By restricting tidal flows, the levees degraded upstream wetlands, either by reducing salinity, creating standing water, and/or by supporting monocultures of invasive variety *Phragmites australis*. The wetlands, located at Greenwich, NJ, on Delaware Bay, were evaluated for restoration in this study. If unrestricted tidal flow were reestablished with mobile gates or similar devices, up to 226 ha of tidal salt marsh would be potentially restored to *Spartina* spp. dominance. Using existing literature and a value transfer approach, the estimated total economic value (TEV) of goods and services provided annually by these 226 ha of restored wetlands ranged from \$2,058,182 to \$2,390,854 y^{-1}. The associated annual engineering cost for including a mobile gate system to fully restore tidal flows to the upstream degraded wetlands was about \$1,925,614 y^{-1} resulting in a benefit-cost ratio range of 0.98–1.14 over 50 years (assuming no wetland benefits realized during the first 4 years). Thus, inclusion of a cost-effective mobile gate system in any engineering design to improve long-term flood resilience in the region would produce dual benefits of protecting people and property from major storms, while preserving and enhancing ecosystem values.

Keywords Tidal wetlands · Values · Restoration · Flood protection

Introduction

Natural and anthropogenic threats to coastal ecosystems are leading to seascape change at both local and continental scales (Gilby et al. 2021). To date, these threats have resulted in broad-scale habitat losses in these systems (Dobson et al. 2006; Coverdale et al. 2013), and ongoing climate change makes the future trajectory of the changes uncertain (Colombano et al. 2021). Coastal habitats support a diverse range of ecosystem services, for example through the direct and indirect support of biodiversity and secondary production. Recognition of the importance of wetlands to estuarine and coastal ecosystems has fueled increased interest in habitat repair and restoration (Weinstein et al. 2014; Waltham et al. 2021).

Flood mitigation and protection of coastal infrastructure are a key component of coastal management (Weinstein et al. 2007; Barbier et al. 2008; Hallegate et al. 2011; Jeroen et al. 2014; Sutton-Grier et al. 2015; Van Coppenolle et al. 2018). Policymakers have also been prompted to consider the regulating and provisioning role of tidal wetlands and related ecosystem services in their environmental management decisions, broadly defined as "the benefits people obtain from ecosystems" (MEA 2003; Turner et al. 2003; Schaefer et al. 2015). Included are the protection of wetlands that contribute to the success of regional fisheries and coastal ecosystem processes and functions such as flood mitigation, carbon sequestration, and groundwater recharge that warrant future protection

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(ICCATF 2011; zu Ermgassen, et al. 2021). Striking a balance between protection of people and property, and protection of the structural and functional characteristics of ecosystems, is essential to ensuring the persistence of ecosystem services that are derived from tidal wetlands into the future (ICCATF 2011).

Protection of people and property does not necessarily need to be exclusive of protection of tidal wetland functions. Small and simple enhancements to coastal protection infrastructure can provide ecological benefits (Sutton-Grier et al. 2015). While these enhancements come at some additional economic cost, it may be feasible to incorporate them when coastal protection structures are being maintained or upgraded. The decision to incorporate such enhancements often relies on assessment of the additional installation costs (and ongoing maintanence costs) against the potential benefits. Consequently, valuation of the additional ecosystem services that may be derived from these enhancements is essential to defining potential economic benefits, and building a business case for such improvements (zu Ermgassen, et al. 2021).

The northern Delaware Bay shoreline in New Jersey (NJ) is substantially modified with earthen dikes and levees many of which are in disrepair. In addition, these stuctures are aged, provide minimal flood protection at most locations, and because the restrict tidal flows upstream, have degraded extant wetlands. At the village of Greenwich, NJ, the subject of this paper, existing levees only provide reliable flood protection for a minimal 10-y storm (Guo et al. 2014; FEMA 2016). Enhancement of flood abatement structures to protect Greenwich and its surrounding farmlands against a 100-y storm is desired, but installation of the necessary infrastructure is expensive.

Here, we present a case study that explores the relative costs and benefits of "enhancing" flood mitigation at Greenwich while simultaneously achieving other positive environmental outcomes. Specifically, we address a *single element* of any proposed "blue print" for flood abatement at Greenwich or elsewhwere; whatever the cost of providing protection against the 100-y storm and including the effects of sea level rise, what would be the element cost of including *a mobile flood gate* in the designs that will allow unimpeded tidal flow upstream of the levees, and consequently allow restoration of these degraded wetlands? We assess the annual cost of adding and maintaining a mobile gate system at the three levees outlined above, compare this directly to the expected annual ecosystem services value provided by the restored wetlands, and evaluate benefit/cost ratios.

Study Location and Existing Hydraulic Conditions

Greenwich New Jersey, population 804, is located in a rural setting with a small village center surrounded by extensive farmland (Fig. 1). Levees surrounding Greenwich are located approximately 0.3 to 1.3 mi (0.5 to 2.1 km) from the village center and have various degrees of wetland disturbance upstream of their footprints. The longest, levee 51, extends ~ 1.6 km across the original tidal wetland (Fig. 1). Two, levees 48 and 50, have undersized and/or partially functional tidegates resulting in restricted tidal flow, and in the case of levee 48, extensive ponding at low tide upstream with loss of most emergent vegetation (Fig. 2a). The failure of a downstream dike (Pine Mount) at the confluence of the Cohansey River (Fig. 1), and consequent "piling up" of rerstricted tidal flows at levee 48 has also caused substantial erosion and loss of wetlands immediately below the levee (Fig. 2b). Impedance of tidal flows now also results in roadway flooding during spring tides at Levee 48 (Fig. 2c). Similarly, restricted tidal flow upstream of the tide gate at levee 50 has resulted in lower average salinities, allowing much of the interior marsh to become dominated by invasive *Phragmites australis* (Fig. 2d).

The absence of tidal flows above levee 51 (Fig. 1) has virtually eliminated upstream tidal marsh vegetation resulting in standing fresh water, extensive mud flats, and salt pannes (during drought conditions) (Fig. 2e). With the exception of the eroded area below levee 48, the tidal marshes downstream of the three levees are relatively undisturbed and are dominated by extensive coverage by *Spartina* spp.

Ecosystem Valuation Estimation

We applied a generic formula to estimate the monetary value (both use and non-use values where the latter are estimated; Clifton et al. 2014) for the Greenwich wetland system. Total ecosystem value (TEV) across all wetlands (n) in the Greenwich area was estimated using the formula:

$$\mathrm{TEV} = \sum_{k=1}^{n} \mathrm{A}(\mathrm{W}i) * \mathrm{V}(\mathrm{TEV}ki)$$

where $\mathrm{A}(\mathrm{W}_i)$ is the area of wetland i and $\mathrm{V}(\mathrm{TEV}k_i)$ is the estimated annual TEV (in 2018 USD) provided by wetland i.

Data for valuing coastal wetlands worldwide were developed by de Groot et al. (2012) and codified in the Ecosystem Services Value Database (ESVD; http://www.fsd.nl/esp/80763/5/0/50), a site than contains relevant data for 300 case studies. Additionally, Costanza et al. (2014) updated and summarized the results of de Groot et al. (2012) for 139 studies listed therein. Among the categories of wetland goods and services addressed (but not fully valued in the ESVD) are provisioning and regulating services, habitat services and cultural services.

The median value estimated from these studies, 12,163 international \$ ha^{-1} y^{-1} (equivalent to \$9974 USD ha^{-1} y^{-1}), or approximately \$10,579 ha^{-1} y^{-1} (2018 USD) is a useful first-order approximation for ecosystem

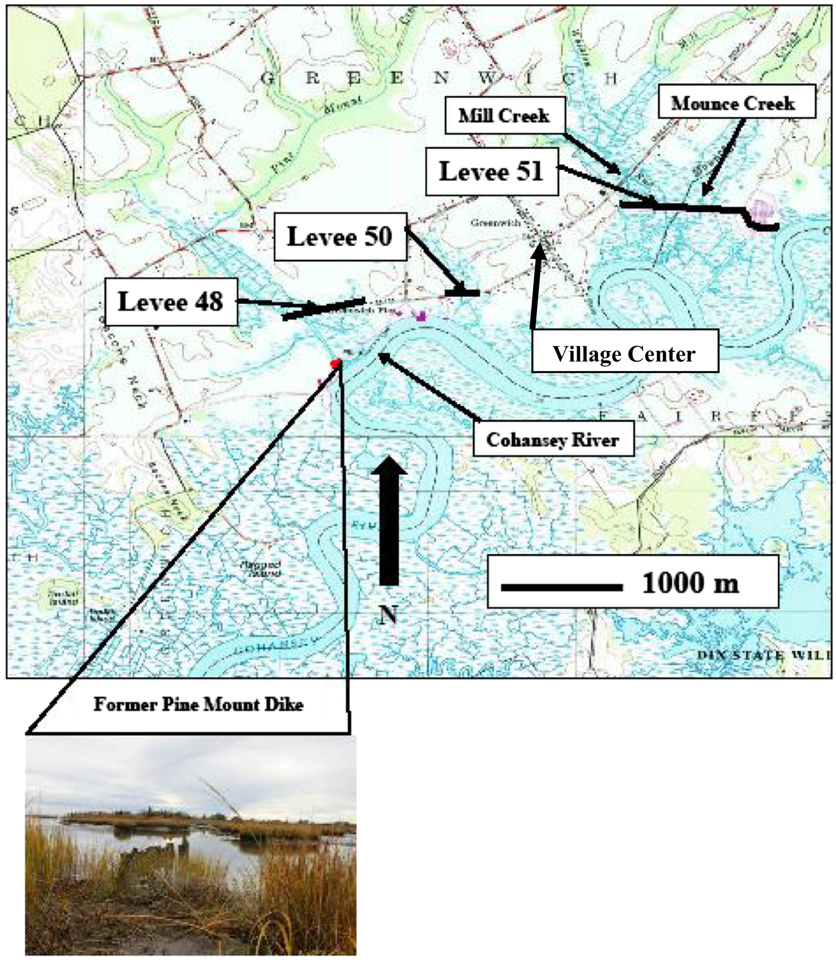

Fig. 1 Topographic map showing location of the three levees and associated tidal creeks at Greenwich, New Jersey 48, 50 and 51

valuation applied in this study. We supplemented this datum with an estimate for New Jersey wetlands based on Costanza and Mates (2007) who used 100 studies to calculate a median TEV of \$6847 ha^{-1} y^{-1} (2004 USD), or \$9107 ha^{-1} y^{-1}, in 2018 USD. These two values \$9107 and \$10,579, respectively, reflect the best available literature, and were used as estimates of TEV in the valuation analysis at Greenwich, considering applications of scale and local conditions. This led to an underlying assumption that a reasonable estimate for economic value of ecosystem goods or services at Greenwich could be inferred from the analysis of existing valuation studies at other sites.

The area of wetland to be restored was assumed to be equivalent to the tidal inundation areas upstream of the three levees. These areas were estimated from available tidal elevations at the creek mouths adjacent to the Cohansey River (NOAA 2018) (Fig. 1). Mean high water (MHW) and mean higher high water (MHHW) were calculated from predicted tidal heights on March 20 to April 19, 2018, as 1.70 m and 1.77 m above mean low low water (MLLW), respectively. Mean tide level (MTL) was also estimated as 0.88 m, but it was unknown how much the MTL at the levee was higher than the water level vertical datum (NAVD 88). Because the MTL observed at the Cohansey River was only slightly higher than the NAVD 88 datum (0.045 m), the calculated MHW and MHHW were thus approximately 0.84 m and 0.91 m above the latter. The inundation areas were subsequently mapped based on the MHHW and MHW and available LiDAR topographic elevation data for the sites (Fig. 3 a–f). These flooding limits likely reflect the area to be restored to *Spartina* spp. dominance, because the area flooded at this tidal height will include the presence of *Spartina patens* and *Distichlis spicata* in the high marsh zone. If unimpeded upstream tidal flow were reestablished through each levee, the potential wetland area restored would be about 181 ha. In addition, the 45 ha of mudflats below Levee 48 will likely revegetate once the hydroperiod and tidal sources of sediment were restored at the site (Teal and Weinstein 2002; see below). This combined level of restoration would increase the total wetland area dominated by *Spartina* spp. to about 226 ha, or about a 109% increase from estimated wetland areas between the dikes and the Cohansey River (Table 2).

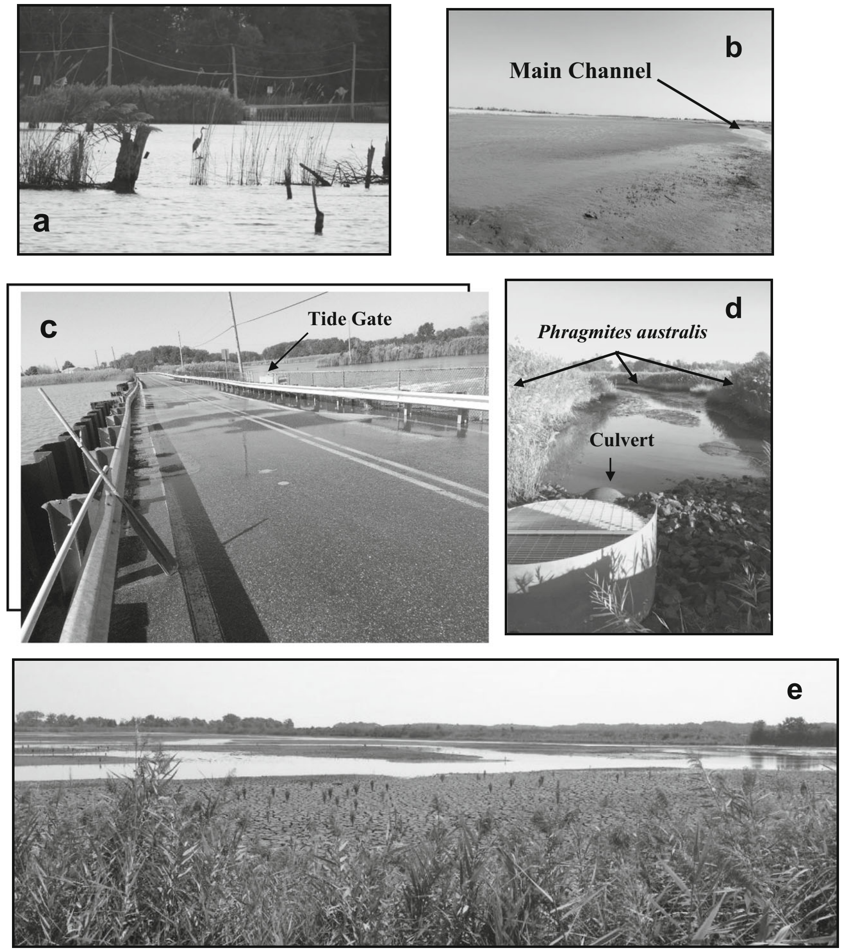

Fig. 2 **a** Open water and marsh fringe (**a**) upstream at Levee 48 at low tide created by retention of upland drainage and restriction of tidal flow; (**b**) mudflat area below levee 48 created by a downstream dike failure; (**c**) flooded roadway at high tide, Levee 48; (**d**) *Phragmites* dominated marsh upstream of levee 50; and (**d**) mudflats, pannes, and standing water upstream of levee 51

Using Eq. (1), and the parameters described above, the estimated range of annual TEV in 2018 dollars for the 226 ha of wetlands available for restoration at Greenwich was $2,058,182 and $2,390,854 (given the two estimates of TEV employed, Table 1). These are likely to be underestimates because not all goods and services for tidal wetlands at Greenwich can be valued.

Proposed Engineering Design for Mobile Tide Gates

Several mobile surge barrier designs were available to guide construction at the existing levees. The vertical lift gate was selected (Fig. 4), as it is a simple and straightforward design (Mooyaart et al. 2014). The gate is raised vertically from the sill at the channel bottom to open, and lowered to the sill to close. A tower with overhead cables, sheaves, and bull wheels are used to support the gate during operation.

Design parameters for the gates were obtained from existing data: mean high water (MHW), mean higher high water (MHHW), and mean tide level (MTL). For restoration of 100% of tidal flow, the width of the opening required would be the same as the width of the tidal creek. The latter were estimated from satellite imagery and historical photographs taken at the sites at low tide. Heights for the proposed mobile gates for the existing flood protection level were measured in the field as the distance between the creek bottom to the top of the road, levee, sheet piling, or berm, whichever was highest (Table 2).

Tidal creek widths required for restoration of full tidal flow at Levees 48 and 50 were estimated as 152.4 m and 9.14 m, respectively. The two historical tidal creeks at Levee 51, Mill and Mounce Creeks (Fig. 1), would require constructed widths of 30.5 m and 15.2 m respectively. The levee top surface elevation, tidal creek bottom elevation, mean tide level, and metal sheet piling or berm elevation within the required cross-sectional areas are shown in Fig. 5, a–d for the three levees across four tidal creeks. The water head was determined as the difference between water surface elevations at upstream and downstream sides of the gates. The downstream water surface elevation was assumed to be at the top elevation of the gate itself, and the upstream water surface elevation was assumed to be at the mean tide level (MTL). The estimated heights required

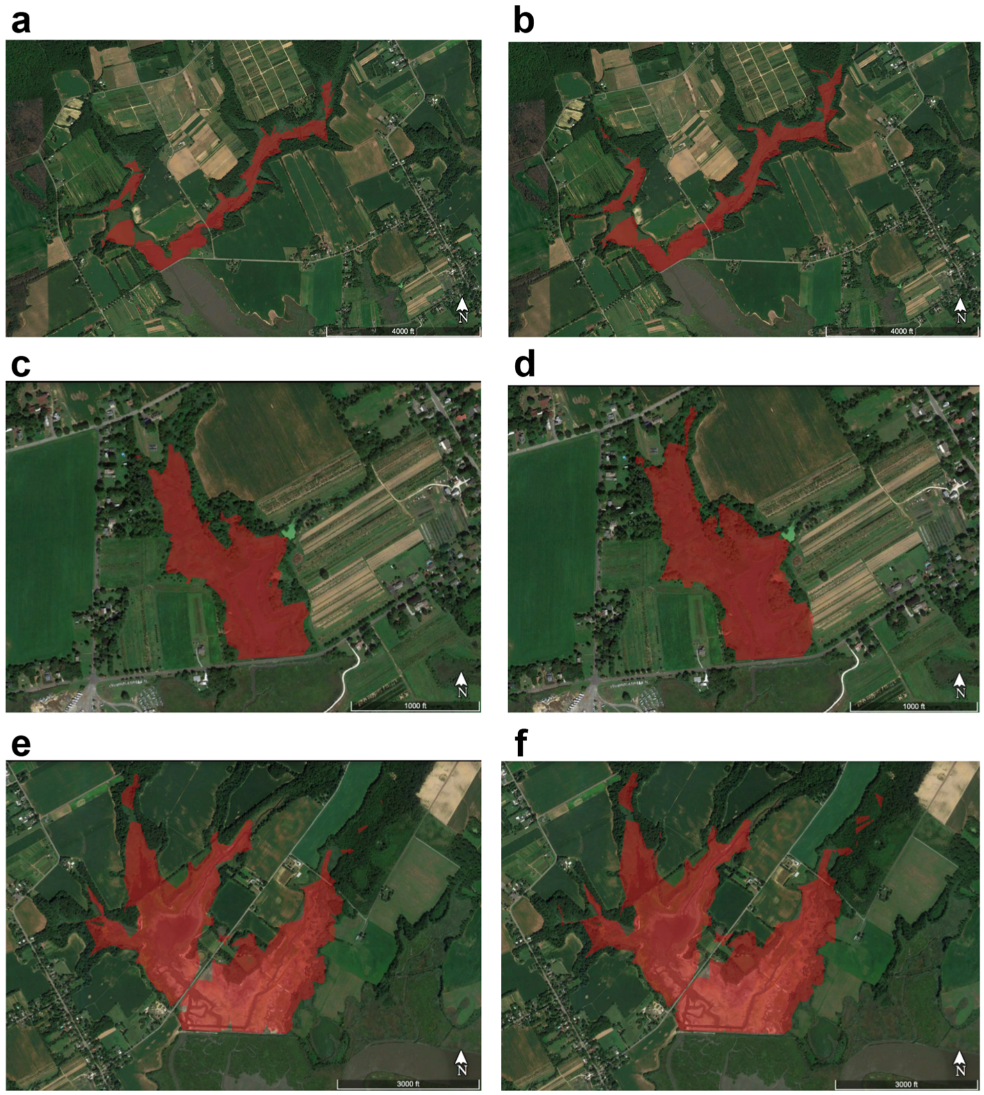

Fig. 3 Estimated tidal inundation (area shown in red) upstream of levees 48, 50 and 51 at mean high water (**a**, **c**, and **e**) and mean higher high water (**b**, **d**, and **f**)

for the mobile gates at the four levees were thus estimated to be 3.8 m (Levee 48), 5.4 m (Levee 50); 5.7 m (Levee 51—Mill Creek), and 5.7 m (Levee 51—Mounce Creek).

Table 1 Total economic value (TEV) of tidal wetlands that would be restored (1) by full flow of tides upstream of the levees 48, 50, and 51 located at Greenwich, NJ. An additional 45 ha of eroded wetlands downstream (d) of levee 48 has the potential to “self-engineer” to a restored state

Levee no.	Restoration hectares	Estimated TEV ($ $ha^{-1}y^{-1}$)	
		Low	High
48u	42	382,494	444,318
48d	45	409,815	476,055
50	11	100,177	116,369
51	128	1,165,696	1,354,112
Totals	226	2,058,182	2,390,854

Proposed Installation and Maintence Costs for Mobile Tide Gates, and Benefit/Cost

Estimating construction costs based on the actual design of a storm surge barrier for each levee location would be difficult and would require detailed knowledge of the general characteristics and dimensions of each component, including levees, closure structures, gates, and gate monoliths. Instead, we applied an approach proposed by the US Army Corps of Engineers (USACOE 2015), that considers average construction costs for storm surge barriers in various parts of the world. The method has also been used in other conceptual level studies (van Ledden et al. 2012). At Greenwich, costs were estimated from the barrier width, barrier height, and head (water level differential) acting on the barrier, as described above. This resulted in a cost of ~$31,000 m^{-3} of the volume calculated from the product of the three length

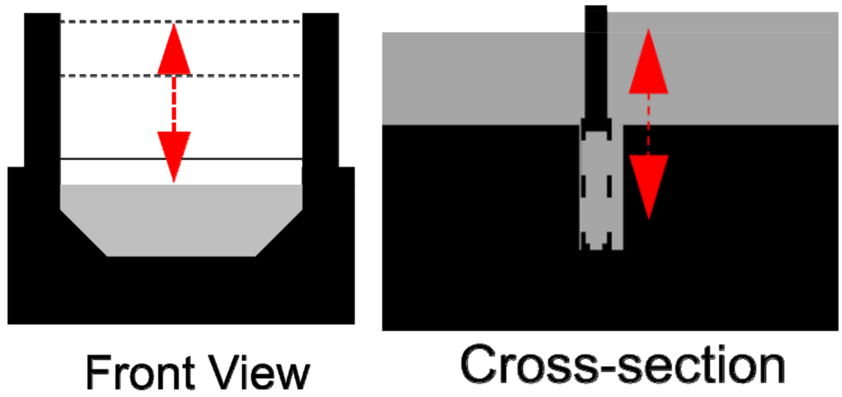

Fig. 4 Vertical lift gate design to allow unrestricted tidal flow through each levee

dimensions used in this study (van Ledden et al. 2012). After adjustment for inflation based on the Consumer Price Index (CPI-U) by the US Bureau of Labor Statistics, the construction cost was estimated to be \$33,975 m^{-3} in 2018 USD, and the costs for mobile gates at the three levees are shown in Table 3. In addition to the construction costs, parametric cost of 12% for engineering and design (E&D) and 10% for construction supervision and administration (S&A) were estimated. A contingency of 25% was also applied to the cost estimates.

Generally, any storm surge barrier will have substantial operation and maintenance costs. From maintenance numbers of three large barriers in the world (Thames Barrier, Maeslant Barrier, Eastern Scheldt Barrier), it has been estimated that the annual maintenance costs are approximately 0.5% of the first construction costs (van Ledden et al. 2012). A summary of the total first construction costs and the total annual costs are in Table 3. Total annual costs are estimated using a 50-year project life and annual interest rate of 3.5%. The 50-year life is commonly assumed for economic analysis of project benefits (USACOE 2015). They ranged from a minimum annual cost of \$80,370 per year (Levee 50) to a maximum of \$1,195,749 per year (Levee 48), at a total cost of \$1,925,614 (Table 3).

From the estimates outlined above, benefit (TEV)/cost (engineering) ratios ranged from 0.98 to 1.14. This suggested that use of mobile gates would be a viable option for restoring 226 ha of wetland at Greenwich.

Table 2 Tidal inundation areas (ha) upstream of Levees 48, 50, and 51 at mean high water (MHW) and mean higher high water (MHHW)

	Levee number			
	48	50	51	Total
MHW	39	11	123	173
MHHW	42	11	128	181

Discussion

Restoration Efforts on Delaware Bay

The senior author has been directly involved in tidal wetland restoration efforts on Delaware Bay since 1996. The 4050 ha (10,125 acres) of tidal salt marshes being restored are degraded wetlands that take two forms, (a) diked salt hay farms, some in continuous agriculture since the late 1800s (Weinstein et al. 2000) and (b) sites covered by near monocultures of the invasive variety of *Phragmites australis* (Saltonstall 2002; Weinstein et al. 2003). The restoration designs optimized use of natural site factors including channel size and configuration, drainage patterns, and ratio of marsh plain to open water to encourage natural restoration trajectories (Teal and Weinstein 2002). Suitable tidal exchange was achieved at formerly diked sites by excavating the mouth of the largest order tidal creeks (4th or 5th order) through the existing dikes. These were determined from creek cross sections of the same order at nearby reference marshes. At the *Phragmites* degraded sites, herbicide applications followed by prescribed burns removed virtually all standing plants and reopened the marsh plain to recolonization by vegetation, primarily *Spartina* spp. The restoration designs are summarized in Teal and Weinstein (2002) but were basically a “hands off” approach letting nature establish the restoration trajectories with minimal ecological or engineering intrusion.

The central focus of these restoration efforts addressed the linkage between the restored tidal marshes (adding about 3% of marsh area to the entire Bay) and the secondary production of finfish, shellfish, and their prey base. In a summary paper, Weinstein et al. (2012) described an ECOPath/EcoSim time series model (1996–2003) for the 11 sites that comprised the restoration effort (Weinstein et al. 2001). The model output indicated that restoration resulted in a net gain of 47.7 tons km^{-2} year^{-1} in system biomass production. The timeline for these restoration trajectories at the two sites located closest to Greenwich was met within 4 years (Weinstein

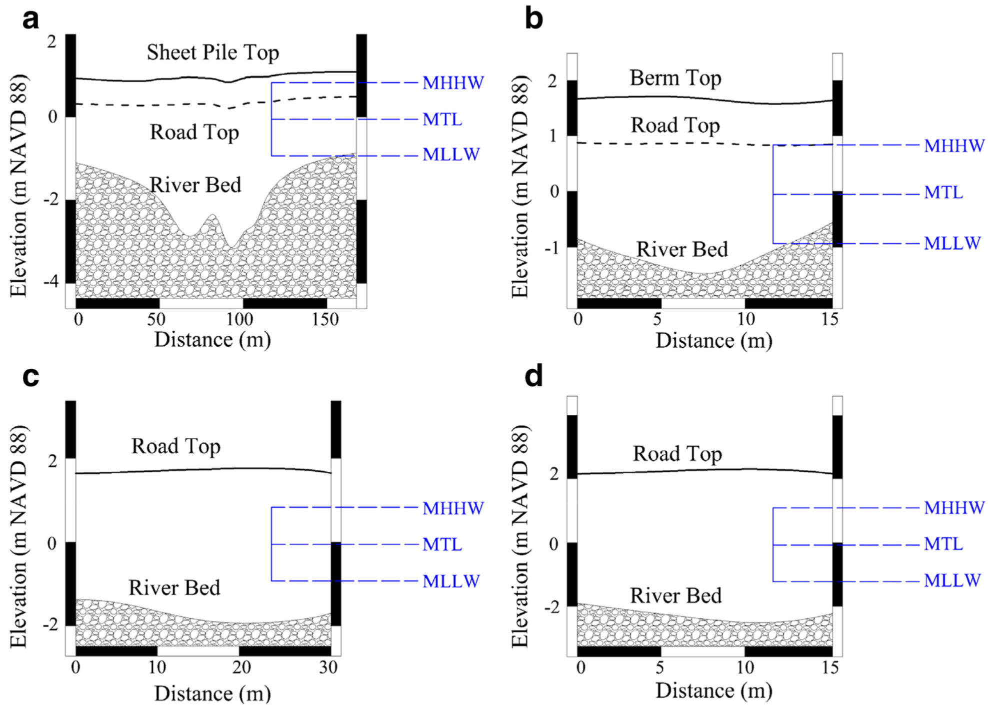

Fig. 5 Tidal creek bottom and levee (road) top elevation at Levees 48 (**a**), 50 (**b**), and 51 (**c**, Mill Creek, and **d**, Mounce Creek)

et al. 2019) when vegetation coverage by *Spartina* spp., drainage density and related tidal creek morphometrics converged with that of the reference marshes. The two sites, Brown's Run (196 ha) and Dennis Township Salt Hay Farm (227 ha), approximate the proposed area to be restored in this study (226 ha), and are located within 5 km and 30 km, respectively from the Greenwich sites.

At Greenwich, we anticipated that the restoration trajectories would become asymptotic in year four, and to be conservative, we applied the TEV values to a 46-year interval, for comparison to the engineering costs for installing the mobile gates with an initial 50 year "life expectancy."

Wetland Hydrology and the Presence of Invasive *Phragmites australis*

Wetland hydrology has been altered worldwide by the introduction of levees, floodgates, and culverts that restrict tidal amplitude and fragment habitats (Daiber 1986; Streever 1997). Such practices have degraded highly productive wetland ecosystems (Roman et al. 1984; Montague et al. 1987; Chambers et al. 1999), reduced fish and crustacean passage (Eberhardt et al. 2011), changed faunal assemblages (Raposa and Roman 2003; Kroon and Ansell 2006; altered food webs (Dick and Osunkoya 2000), mobilized acid sulfate soils, and

Table 3 First construction and annual costs (USD 2018) for mobile barrier with current coastal flood protection level at current levee heights. *E&D*, engineering and design; *S&A*, construction supervision and administration; *O&M*, operation and maintenance

Item	Unit cost	Levee 48	Levee 50	Levee 51 (Mill Creek)	Levee 51 (Mounce Creek)
	33,975 m^{-3}	18,300,000	1,230,000	6,600,000	3,340,000
Contingency	25%	4,575,000	307,000	1,650,000	835,000
Total construction		22,875,000	1,537,000	8,250,000	4,175,000
E&D	12%	2,745,000	184,500	990,000	501,000
S&A	10%	2,287,500	153,750	825,000	417,500
Total estimate first construction Cost		27,907,500	1,875,750	10,065,000	5,093,500
Annualized first costs		1,189,800	79,970	429,108	217,155
O&M	0.5%	5949	400	2146	1085
Total estimated annual average cost		1,195,749	80,370	431,254	218,241

reduced fisheries production and catches (Sultana and Thompson 1998).

Such marsh alterations are recognized as an "acute problem" (Chambers et al. 1999), and as the authors have noted, are often associated with an introduced variety of *Phragmites australis*, the m-haplotype (Saltonstall 2002). The latter has become a "signature [species] of tidal wetland alteration" (Chambers et al. 1999) especially in the middle and northern Atlantic coastal regions of North America, and has resulted in the conversion of *Spartina*-dominated systems to nearly monotypic stands of *P. australis* (Weinstein et al. 2003). *P. australis* is considered a weed species that alters both the structure and function of typical *Spartina* marshes by reducing plant biodiversity with concomitant reductions in habitat value for wildlife, waterfowl and other avian species, e.g., decreases in species richness, density, and abundance (Roman et al. 1984; Sinicrope et al. 1990; Weinstein and Balletto 1999; Raposa 2008; Dibble and Meyerson 2012, 2013).

A question remains, however, regarding the present ecological value of the *P. australis* dominated marsh surface upstream of levee 50 (~ 11 ha), and, what, if any, prior value can be assigned to this marsh? A review of the literature suggests that *P. australis* elevates the marsh (Weinstein and Balletto 1999; Rooth et al. 2003; Able and Hagan 2003) and dramatically reduces surface microtopography creating a virtually smooth marsh plain (Able et al. 2003). Larval fish traps placed flush with the marsh surface at *P. australis*-dominated sites, in Delaware Bay, captured significantly fewer larval and early juvenile common mummichogs (*Fundulus heteroclitus*) than traps placed at recovering sites dominated by *Spartina alterniflora* (Able et al. 2003).

P. australis' dominance at Delaware Bayshore and other brackish area of the upper Delaware Bay also appears to result in relatively poor growth in young-of-year nekton. By elevating the marsh surface, *P. australis* also negatively influences the marsh hydroperiod, reducing the flood stage and hindering the access to the marsh surface by nekton, and the exchange of materials carried by the tide. We previously conducted extensive studies of the flow of nutrients from dominant marsh vegetation including not only *P. australis* and *S. alterniflora*, but also microphytobenthos and phytoplankton, which attests to the relative value of tidal salt marshes in supporting the estuarine food web and secondary production. We note that nutrients from the C_3 plant, *P. australis* were indeed a source of C, N and S in the trophic spectrum of the upper Bay (Wainright et al. 2000; Currin et al. 2003; Litvin and Weinstein 2003, 2004; Litvin et al. 2014; Weinstein and Litvin 2016; Weinstein et al. 2000, 2005, 2019). But our previous work on *condition* of dominant finfish in *Phragmites*-dominated marshes on the Hudson River suggested that while somatic criteria, length and weight at age, were virtually identical in the two populations, common mummichogs (*Fundulus heteroclitus*) and white perch (*Morone americana*) captured at the *Phragmites* dominated site, these species were unable to lay down sufficient energy reserves in the form of triacyl glycerides (TAG) and free fatty acids for overwintering/migration at the end of the growing season (Weinstein et al. 2009, 2010). These differences were significant.

The latter effort was subsequently verified by Dibble and Meyerson (2012, 2013, 2016) who examined population dynamics of *Fundulus heteroclitus* captured in *Phragmites* dominated, tidally restricted marshes, and at undisturbed reference marshes, at several locations in New England. *F. heteroclitus* captured at the former sites also exhibited significantly reduced lipid reserves and increased lean dry (structural) biomass relative to fish collected at the reference locations, while fish in tidally restored marshes were equivalent across all metrics relative to those in reference marshes indicating that habitat quality was restored via increased tidal flushing. Moreover, reference marshes adjacent to tidally restored sites contained the highest abundance of young fish (ages 0–1) while tidally restricted marshes contained the lowest. These results indicated that *F. heteroclitus* residing in physically and hydrologically altered marshes are at a disadvantage relative to fish in reference marshes, but the effects can be reversed through ecological restoration. The authors also "detected a significant decrease in the proportion of actively spawning fish in restricted relative to paired [reference] marshes, but no difference between restored and paired unrestricted [reference] marsh fish."

The reduction in marsh coverage by *P. australis* and/or other freshwater wetland species with return of unimpeded tidal salt water flow has been well-documented (Sinicrope et al. 1990; Karberg et al. 2018). When tidal flows were reintroduced to a *Phragmites/Typha* degraded marsh in Connecticut (with undersized culverts), *Spartina alterniflora* dramatically reduced coverage of the former taxa in much of the open water area. For these reasons, and others cited above, we believe that the wetland functions and values of the restored *Spartina*-dominated marsh upstream of Levee 50 (< 5% of the total area to be restored) will substantially exceed the near zero values of the extant *Phragmites*-dominated marsh currently at the site, and the former values have consequently been used in our TEV estimates.

While tidal flows have been entirely eliminated at Levee 51, undersized culverts or floodgates like those located at Levees 48 and 50 (Fig. 2) reduced the original cross section of the creeks and have been implicated in degradation of ecological functions, including the now *P. australis* dominated area upstream of Levee 50.

Our previous studies on Delaware Bay also suggest that the eroded area immediately below Levee 48 would "self-engineer" to a *Spartina* spp. dominated tidal marsh with minimal site engineering (breaching dikes, initiating channel formation), and without vegetation plantings, once a natural hydroperiod and tidal flow induced sediment sources were reestablished (Teal and Weinstein 2002; Weinstein et al. 2000, 2014, 2019). As noted above, this was the case at the two sites closest to Greenwich, Brown's Run and Dennis Creek, both fully restored to *Spartina* spp. dominance within 4 years (see Fig. 6).

Other Valuation Challenges

Because ecosystems are complex, and not easily partitioned into functional units, estimating non-market values within such systems can be challenging (Costanza and Mates 2007). Additionally, the marginal cost per hectare of wetlands was generally expected to increase as wetland size decreased, and as a result, a single average or median value might "not be the same thing as a range of marginal values" (Costanza and Mates 2007). In this same context, the estimates used herein assumed spatial homogeneity of services within the ecosystem, and the general absence of dynamic interactions that might not reflect "real world conditions" (Costanza and Mates 2007).

Fig. 6 Wetland restoration at a 227 ha site located on the Delaware Bay Shore region. The site was fully revegetated from mudflats over a four-year period (courtesy PSEG)

Despite these challenges, Heal et al. (2005) concluded that the "general concepts seem to offer sufficient guidance for valuation to proceed with careful attention to the limitations of any ecosystem assessment." Although, local and short-term goods and services may be most easily observed and documented, any direct approach to the problem requires recognition of the potential for "double counting," aggregation errors, spatial and temporal dynamicism, and the need for careful framing of the assessment questions. Yet, applied objectively, valuation studies can provide useful information on the role of ecosystems in supporting human welfare (Howarth and Farber 2002). With the dramatic expansion of literature on the subject, value transfer is fast becoming a credible and practical way to inform decision making (Heal et al. 2005; Costanza and Mates 2007; Dodds et al. 2008).

The application of a literature-based approximation to "total economic value (TEV)" for tidal salt marsh goods and services at the Greenwich study sites should be considered with some caution. However, application of these values did support the estimation of benefits that may be derived from allowing unrestricted tidal exchange with the upstream wetlands, alongside works to improve flood relief to the community. By doing so, we have demonstrated that while people and property can be protected, the addition of mobile tidal gates to these protection structures will nearly double the wetlands goods and services values provided to Greenwich and the region.

Benefits/Costs' Ratio and Discounting

The range of annual TEV in dollars ($2,058,182 to $2,390,854) provided bounds on the valuation analysis used at Greenwich. As noted earlier, these are likely to be underestimates because not all goods and services for tidal wetlands can be valued. Annualized engineering costs for the mobile gates totaled $1,925,614 (Table 3). To be conservative, we used a 46-year period to estimate the TEV values at Greenwich (assuming that restoration asymptotes would be reached in year four), to compare against the 50-year period for engineering costs, and this resulted in the calculated B/C ratios of 0.98 to 1.14 over the 50 years. These ratios would *not* change if the extrapolated 50-y TEV/engineering values and costs were discounted at the same rate and applied herein. In essence, the TEV values for the wetlands at Greenwich can be considered to be stable in perpetuity and likely to increase in future generations (TEEB 2008). Whatever the maintenance or replacement costs for the mobile gates beyond the first 50 years, the US National goal of "no net loss" of wetlands (US Presidential Executive Order 11990) will ensure that the restored wetlands at Greenwich will stay intact.

Arrow (1966) noted that technological progress was likely to make future generations "richer" than those of the present generation, and consequently there was an argument for discounting the future at a rate "somewhere in the magnitude of 3%." There have been many counter arguments against discounting, which we chose not to apply here; e.g., Azar and Sterner (1996) argued that at the societal level, there is no ethical basis for using a time preference greater than zero. Heal et al. (2005) suggested further that "if one is convinced that future generations should be valued less than present generations, then a positive utility discount rate should be chosen; otherwise this rate should be zero."

From an ecological perspective, discounting may allow decision makers to give too little weight to the future, and we might continue the present course of habitat loss and degradation that will deny a sustainable future for humankind (Costanza et al. 1989). Central to these tenets is the notion of *scarcity* and the perception that marginal values of ecosystem goods and services will increase over time if these resources become scarcer, as may likely be the case for wetland goods and services in the future. Similarly, Goulder and Stavins (2002) adopted the view that the value of ecosystem goods and services be given equal weight when applied to present and future societies.

In any instance, discounting ecosystem services losses and the loss of vital ecosystem services needs to be critically reassessed, especially if irreversible changes are likely to occur. In this case, as Ring et al. (2010) noted, "we face an ethical decision rather than a purely economic one" Comments from TEEB (2008) further highlight the point:

> Applying a 3% discount rate over 50 years implies that we value a future ecosystem benefit to our grandchildren at only one-seventh of the current value that we derive from it. If our ethical approach sees our grandchildren valuing nature similarly to our generation, and deserving as much as we do, the discount rate for valuing such benefits over such a time period should be zero.

Greenwich's current levee system, at best, may obviate damage from a 10-y event. As do most coastal communities, the citizens of the township desire protection against a 100-y storm, but to do so, will likely be costly. We have addressed but a single component of the "blue print" for flood abatement at Greenwich; i.e., whatever the cost of providing protection against the 100-y storm, what would be the incremental cost of adding a mobile gate system, or similar, within the levees that will allow unimpeded tidal flow upstream? It is this framework, the cost-benefits of the TEV of 226 ha of restored wetlands at Greenwich can be compared against the engineering and maintenance costs of including a mobile gate system in any 100-year design at Greenwich. In addition, any secondary effects of the newly installed gate system such as potential changes to hydrodynamics that result in increased scour or erosion patterns would be evaluated and would employ distance-decay relationships (higher potential of impact closer to design structure and vice versa).

Conclusions

We provide a concise case study which synthesizes ecosystem service valuation with engineering considerations to estimate the value-added benefits of including mobile tide gates that restore unimpeded tidal flows upstream in flood mitigation enhancement works, against their costs. In the past four decades, ecosystem goods and services' valuation has been one of fastest-growing areas of research in environmental and ecological economics (Schaefer et al. 2015). The overarching goal has been to express the effect of a marginal change in ecosystem services provision in terms of trade off against other community values (consumer and/or citizen preferences like flood protection) (Turner et al. 2003). By doing so, policy formulation and the decision-making process can be improved especially for allocation of increasingly scarce resources among competing demands.

Increasingly, it is being shown that sustainable, multi-functional use of an ecosystem is usually not only ecologically sound, but also economically beneficial, both to local communities and to society as a whole (Balmford et al. 2002). However, monetary valuation should always be seen as a supplement to, not as a replacement for, ecological, social, and cultural values under consideration in the decision-making process. As coastal wetlands play a crucial role in sustaining local livelihoods, contributing to the local and regional economy (de Groot et al. 2006), the valuation process is increasingly critical for addressing trade-offs in the environmental review process. Using this approach, we have demonstrated that the potential economic return of restoring 226 ha of degraded wetlands at Greenwich is substantial and supports the inclusion of mobile gates in any planning effort to provide protection against major storms, e.g., those with a 100-y return period.

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