PERSPECTIVES

Buying Time with Runnels: a Climate Adaptation Tool for Salt Marshes

Alice F. Besterman[1,2] · Rachel W. Jakuba[1] · Wenley Ferguson[3] · Diana Brennan[4] · Joseph E. Costa[5] ·
Linda A. Deegan[2]

Abstract

A prominent form of salt marsh loss is interior conversion to open water, driven by sea level rise in interaction with human activity and other stressors. Persistent inundation drowns vegetation and contributes to open water conversion in salt marsh interiors. Runnels are shallow channels originally developed in Australia to control mosquitoes by draining standing water, but recently used to restore marsh vegetation in the USA. Documentation on runnel efficacy is not widely available; yet over the past 10 years dozens of coastal adaptation projects in the northeastern USA have incorporated runnels. To better understand the efficacy of runnels used for restoration, we organized a workshop of 70 experts and stakeholders in coastal resource management. Through the workshop we developed a collective understanding of how runnels might be used to slow or reverse open water conversion, and identified unresolved questions. In this paper we present a synthesis of workshop discussions and results from a promising case study in which vegetation was restored at a degraded marsh within a few years of runnel construction. Despite case study outcomes, key questions remain on long-term runnel efficacy in marshes differing in elevation, tidal range, and management history. Runnel construction is unlikely to improve long-term marsh resilience alone, as it cannot address underlying causes of open water conversion. As a part of holistic climate planning that includes other management interventions, runnels may "buy time" for salt marshes to respond to management action, or adapt to sea level rise.

Keywords Runnel · Salt marsh · Sea level rise · Shallow water · Climate adaptation · Coastal restoration

Introduction

While for centuries salt marsh loss was driven by direct human alterations (Gedan et al. 2009), sea level rise (SLR) now poses one of the chief threats to salt marshes globally (FitzGerald and Hughes 2019; Bindoff et al. in press). Direct alterations (e.g., draining, filling) led to prolific reductions in the global inventory of salt marshes (Gedan et al. 2009;

Mcowen et al. 2017). Regulations have mitigated direct loss of wetlands along many temperate coastlines (Gedan et al. 2009; Bindoff et al. in press); however, legacy impacts from agriculture (Adamowicz et al. 2020) and mosquito ditching (Vincent et al. 2014; Burdick et al. 2020), in interaction with SLR (Raposa et al. 2017; Watson et al. 2017), continue to alter hydrology and stress vegetation. A primary manifestation of these stressors in Northwest Atlantic and Mississippi Delta marshes has been the expansion of unvegetated, shallow water features in marsh interiors, i.e., open water conversion (Barras et al. 2003; La Peyre et al. 2009; Vincent et al. 2014; Kearney and Turner 2016; Watson et al. 2017; Adamowicz et al. 2020) (Fig. 1c–e). Globally, salt marsh coverage is most extensive in low-lying temperate zones of the North Atlantic (Mcowen et al. 2017). Thus, marsh losses to open water conversion in this region are globally significant.

A surge in restoration of Northwest Atlantic salt marshes (specifically, northeastern USA) occurred over the past decade, funded by the US government in the


Communicated by Kenneth L. Heck

✉ Alice F. Besterman
abesterman@woodwellclimate.org

[1] Buzzards Bay Coalition, New Bedford, MA, USA

[2] Woodwell Climate Research Center, Falmouth, MA, USA

[3] Save the Bay, Providence, RI, USA

[4] Bristol Country Mosquito Control Project, Attleboro, MA, USA

[5] Buzzards Bay National Estuary Program, Mattapoisett, MA, USA

Fig. 1 a Salt marsh ecosystems support wildlife, e.g., great egrets (*Casmerodius albus*) and **b** saltmarsh sparrows (*Ammospiza caudacuta*). **c** Salt marsh interiors increasingly impacted by sea level rise, in interaction with legacy effects from human activity, are experiencing plant death and drowning. **d** Plant death is recent; with short, stressed vegetation and intact peat present with impounded water and **e** and algal mats. **f** Dead killifish (*Fundulus heteroclitus*) stranded when a shallow water area is drained
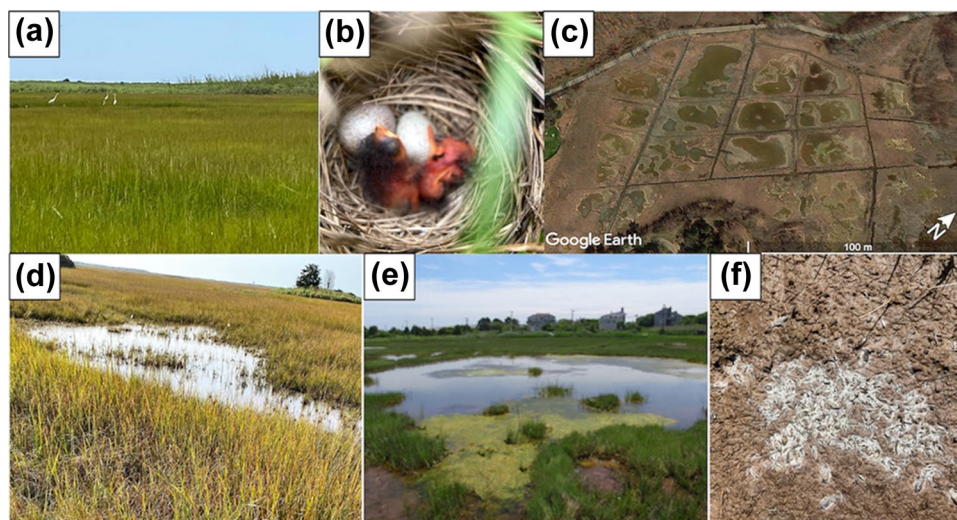
aftermath of Superstorm Sandy in 2012 (Babson et al. 2020). One technique involved digging "runnels"—small channels meant to drain standing water and promote revegetation (Wigand et al. 2017; Raposa et al. 2019; Babson et al. 2020; Perry et al. 2021; Wolfe et al. 2021). As of 2019, knowledge on runnel efficacy as a tool to build marsh resilience to SLR (defined here as the ability of a marsh to resist a state-change to open water) had not been widely shared. Recognizing the need to critically evaluate runnels as a climate adaptation tool, we organized a workshop on the runnel technique in early 2020. Workshop goals were to solicit expert opinion on the practice of runnelling and to build consensus around when and how to use runnels. This information was used to design our own experimental test of runnels, which is now underway. Seventy people including scientists, regulators, landowners, and resource managers from government, academia, and non-profits participated. During the workshop, scientists and managers presented outcomes from recent runnel-projects, followed by panel and small group discussions.

This paper is a product of the workshop, subsequent engagement among participants, and literature review. Three focal questions will be discussed here: (Q1) What problem do managers use runnels to address? (Q2) What is a runnel and how does it work? and (Q3) How effective are runnels? We focus on runnel use to mitigate marsh loss caused by interior shallow water expansion, describe runnel mechanics and present a case study, and discuss lessons learned on efficacy and remaining knowledge gaps. To address remaining gaps, we present specific research topics needing attention. We synthesized information from workshop presentations and subsequent engagement, and literature review to address all three questions. We used the case study to supplement our discussion of Q2 and Q3.

The Problem: Changing Hydrologic Dynamics

Pre-colonial salt marsh hydrology in the Northwest Atlantic featured networks of channels, as well as isolated pannes and pools (Redfield 1972). "Single-channel" hydrology included a primary channel intersected by dendritic tributaries draining the platform (Redfield 1972). Pannes and pools dot marsh platforms and create habitat for unique plants, fish, and waterbirds (Fig. 1a). These features were classically considered in dynamic equilibrium with the vegetated platform (Ewanchuk and Bertness 2004; Adamowicz and Roman 2005; Wilson et al. 2014; Adamowicz et al. 2020). Pannes are shallow depressions with waterlogged soils covered with sparse forbs and bare sediment that form and revegetate within a few years (Ewanchuk and Bertness 2004). Pools are deeper depressions that remain flooded and enlarge, and may drain and revegetate slowly over decades to centuries (Adamowicz and Roman 2005; Wilson et al. 2014).

A natural cycle of "pool recovery," also applicable to shallow pannes, has been documented in some marshes (Wilson et al. 2014; Mariotti 2016). Pools form and expand until they connect to a headward eroding creek that drains the pool. After pool drainage, the remaining bare peat area revegetates. Theoretical models suggest that a large tidal range (> 1 m), large sediment supply (> 70 mg/L), and accretion rates equal to or greater than relative SLR (RSLR) favor pool recovery (Mariotti 2016). Empirical work has shown that pool recovery can occur without satisfying all criteria (Smith and Pellew 2021). The idealized conditions promote pool recovery by maintaining marsh elevation such that unvegetated basins of pools and pannes are suitable for vegetative growth relative to

local sea level. If these unvegetated pool and panne basins become too low in relative elevation to revegetate, then pool recovery cannot occur. Without dynamic recovery, pool and panne expansion may lead to the conversion of marsh interiors to open water (Kearney and Turner 2016; Mariotti 2016; Himmelstein et al. 2021).

Recent studies report the expansion of bare and shallow water areas on marsh surfaces across the Northwestern Atlantic (Table 1). In Mississippi Delta marshes, the formation and expansion of interior pools has been contributing to marsh loss for decades (DeLaune et al. 1994; Barras et al. 2003). While pannes and pools were not classically considered to contribute to net marsh loss (Ewanchuk and Bertness 2004; Adamowicz and Roman 2005), observed increases in the number and size of unvegetated features suggest that marsh hydrology is not in equilibrium—and a trend toward net conversion of vegetated marsh to bare and shallow water areas (Table 1, and references therein). In North America, pre-colonial salt marsh hydrology was altered by colonial farmers, and later for mosquito control (Vincent et al. 2014; Adamowicz et al. 2020). Historic modifications to marshes including ditches, side-cast ditch spoils that form artificial levees, ditch-plugs, and embankments altered topography and hydrology, lowering resilience to current stress from SLR. Ditching lowered marsh platform elevations (Burdick et al. 2020), while artificial levees, embankments, and ditch-plugs created microtopographic impoundments that block natural water flow pathways (Vincent et al. 2014; Adamowicz et al. 2020; Wolfe et al. 2021). Increased inundation from SLR, in interaction with lowered platform elevations and impaired drainage, have lengthened hydroperiods and likely contributed to the recently observed increases in bare and shallow water areas within marshes (Fig. 2a–c) (Adamowicz et al. 2020; Himmelstein et al. 2021; Wolfe et al. 2021). Ecosystem managers have observed that vegetation communities in recently developed bare and shallow water areas do not resemble the diversity of "forb pannes" expected for northern New England (Ewanchuk and Bertness 2004, personal communication). Water depths in these recently developed areas appear intermediate between pannes and pools, creating conditions inhospitable to vegetation (Fig. 1d, e), but too shallow or hypersaline to support fish (Fig. 1f). Understanding whether marsh ecosystem function and resilience are being permanently altered depends on the trajectory of these shallow water areas.

Mariotti (2016) proposed a framework of shallow water expansion that described three scenarios: "pool recovery" (described above), "drowning," and "pool collapse." Drowning occurs when water becomes impounded on the marsh platform and stresses vegetation (Mariotti 2016). Plant death and peat subsidence follow, initiating a positive feedback cycle of water expansion (DeLaune et al. 1994; Chambers et al. 2019). Marsh drowning is predicted for marshes where

Table 1 Selected studies quantifying interior shallow water expansion in tidal marshes Table summarizes the findings on shallow water expansion, and any information provided by the study on background environmental characteristics is summarized

Location and study	Study findings	Environmental background
Buzzards Bay, MA (Costa and Weiner 2017, Costa, unpub. data)	Historical aerials of marshes between 1938 and 2009 show pannes generally expand and deepen (occasional drainage and revegetation in high marsh), while deeper pools only appear to expand	Elevations of 6 unditched marsh islands ranged from − 0.15 to 0.37 m above local sea level. Buzzards Bay in general has low turbidity; tidal ranges ~ 1 m or less, and majority of mainland marshes have been ditched
Narragansett Bay, RI (Watson et al. 2017)	Between 1972 and 2011, 20 out of 36 marshes lost marsh to interior ponding	Marsh median elevations 0.34–0.59 m NAVD88; tidal ranges < 1 m; low turbidity < 5 mg/L; SLR > 3.5 mm/yr. Majority of marshes have been ditched
Blackwater River, MD (Schepers et al. 2017)	Up to 21% of marsh area converted to pools between 1981 and 2010	Microtidal, spring tidal ranges ~ 0.2–1.0 m across system. Local RSLR 3.72 mm/yr; historical marsh accretion 1.7–3.6 mm/yr
Fire Island National Seashore, NY (Campbell and Wang 2019)	Doubling of pannes and pools between 1994 and 2015	Microtidal, mean tidal range 0.455 m; surface elevation tables indicate marshes not accreting sufficiently to keep pace with SLR
Chesapeake Bay and Delaware Bay (Taylor et al. 2020)	Increase of 14,000 ha of interior surface water in marshes between 1984 and 2014	Regional study including sites with different conditions, ranges not reported

Fig. 2 (**a**) Marshes with environmental conditions that favor conversion to open water are experiencing (**b**) longer periods of inundation on marsh interiors and (**c**) standing shallow water on marsh platforms (Winnapaug marsh, RI). If no action is taken, expanding shallow water leads to (**c1**) open water or (**c2**) pool collapse. If a (**c3**) runnel is created, (**d**) surface water should drain allowing revegetation (Winnapaug marsh, RI). But without changing the conditions that led to shallow water formation, (**e**) what is the long-term trajectory of runnel-adapted marshes?
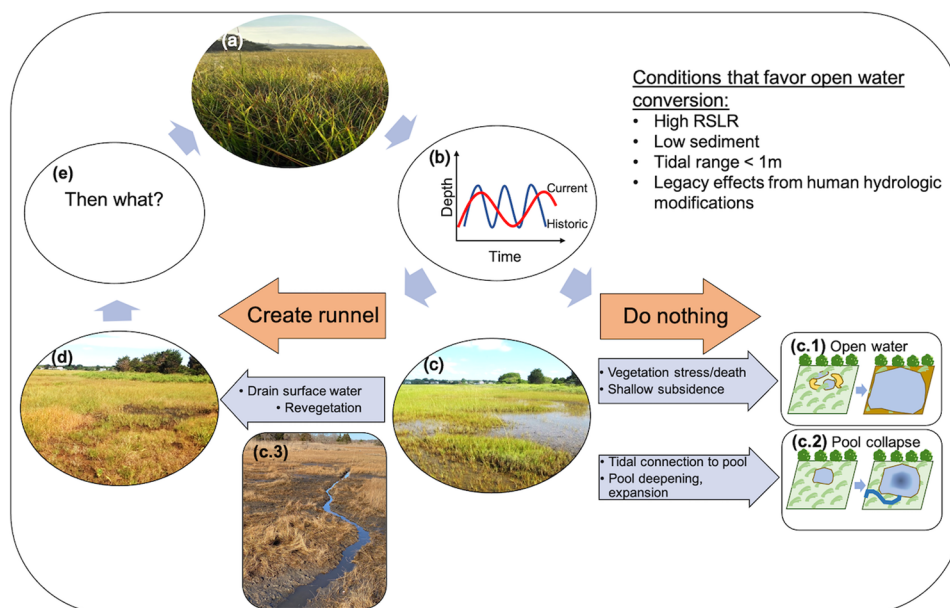
RSLR is high (> 10 mm yr^{-1}), and the entire platform is accreting at less than RSLR. Pool collapse involves a pool becoming tidally connected by a creek, similar to pool recovery, except the pool expands and deepens (Mariotti 2016; Schepers et al. 2020). This may occur when the pool basin elevation is too low relative to sea level for vegetation to recover. Instead of drainage, tidal water conveyed by the creek can erode unconsolidated material underlying pools (Schepers et al. 2020). Pool collapse happens when the marsh platform keeps pace with RSLR, but the basin of the pool does not, and can occur at lower rates of RSLR than drowning (5–8 mm yr^{-1}). Rates of RSLR capable of inducing pool collapse are higher than current SLR (3–4 mm yr^{-1}) experienced by the majority of marshes globally (Oppenheimer et al. in press). Other factors in addition to RSLR are important in determining whether a shallow water area proceeds along a collapse or recovery trajectory. The likelihood of pool collapse may increase with a smaller tide range, lower sediment supply, and when a larger volume of water is conveyed through the connecting creek (Mariotti 2016; Schepers et al. 2020). As a result, drowning and pool collapse are two mechanisms by which interior vegetated marsh areas convert to shallow water and contribute to marsh loss.

Researchers and resource managers have recently pointed to the widespread increase of interior shallow water as an indicator of marsh loss (Watson et al. 2017; Campbell and Wang 2019; Adamowicz et al. 2020; Schepers et al. 2020; Taylor et al. 2020; Duran Vinent et al. 2021; Himmelstein et al. 2021), and have responded with management actions to stop or slow open water conversion (Wigand et al. 2017; Raposa et al. 2019; Adamowicz et al. 2020; Babson et al. 2020; Perry et al. 2021; Wolfe et al. 2021). However, in some marshes an increase in standing surface water could represent a recovery of "natural" hydrology after marshes were historically over-drained by ditches, and some marshes show potential for pool recovery (Wilson et al. 2014; Smith and Pellew 2021). While these recovery scenarios may occur in some places, coasts with low sediment supply or small tidal ranges, and where legacy effects of agriculture and ditching have altered hydrology are unlikely to follow these trajectories (Mariotti 2016; Adamowicz et al. 2020; Wolfe et al. 2021). Rather, increasing periods of inundation will likely lead to drowning, pool collapse, or another mechanism of open water conversion (Fig. 2). In these marshes with low recovery potential, managers are using runnels as a low-cost tool to address increasing surface water and restore vegetation (Table 2 and Supplemental File 1).

Introducing Runnels: History and Progress

Runnels as a Climate Adaptation Tool

Runnels used for climate adaptation to SLR are a new application of an existing mosquito control technique used in the USA and Australia (Hulsman et al. 1989; Wolfe 1996). Historic mosquito ditches were excavated > 60 cm deep, causing peat oxidation and subsidence of the inter-ditch marsh platform (Burdick et al. 2020). More recently, mosquito control programs began constructing runnels that resembled natural channels to drain standing water (mosquito larvae habitat) and allow fish passage (mosquito larvae predators), with minimal impact to marshes (Hulsman et al. 1989).

A runnel is a small channel (generally ≤ 30 cm wide and deep) that drains standing water on the marsh surface. Runnels are constructed using hand-digging and low-ground

1495

Table 2 Selected runnel adaptation projects from the northeastern USA The table includes the year each project was initiated, project name and location, the closest NOAA tide station and tidal range and SLR reported for that station, primary motivations for the project, the number of runnels used in the project (if known), and general vegetation response to runnels

Year	Project name	NOAA station	Tidal range (m)	Sea level rise (mm/yr)	Motivations	No. of runnels	Vegetation response
2004	**Mile Creek Marsh A** 41.28, − 72.29	Bridgeport, CT	2.05	3.08	Revegetate dieback, mosquito control	2	After 2–5 years
2014	**Round Marsh** 41.51, − 71.37	Newport, RI	1.05	2.83	Phragmites control, mosquito control	8	After 1 year
2014	**Mile Creek Marsh B** 41.28, − 72.29	Bridgeport, CT	2.05	3.08	Revegetate dieback, wildlife habitat	4	After 1 year
2015	**Parker River NWR, Ditch Plug Removal** 42.78, − 70.81	Boston, MA	2.89	2.87	Revegetate dieback, low-marsh to high-marsh species	22	After 1 year
2017	**Reeds Beach at Cape May National Wildlife Refuge** 39.12, − 74.88	Cape May, NJ	1.48	4.88	Low-marsh to high-marsh species, wildlife habitat	40	After 1 year
2018	**Potters Pond** 41.38, − 71.53	Newport, RI	1.05	2.83	Revegetate dieback, mosquito control	7	After 2–5 years
2019	**Furbish Marsh Restoration** 43.28, − 70.58	Seavey Island, ME	2.47	2.05	Revegetate dieback, low-marsh to high-marsh species	2	No results yet
2019	**NFWF Gardiners** 40.69, − 73.27	Sandy Hook, NJ	1.43	4.15	Revegetate dieback, low-marsh to high-marsh species	40	After 1 year
2020	**Little Bay Conservation Area** 41.63, − 70.87	Newport, RI	1.05	2.83	Research and assessment, revegetate dieback	5	No results yet
2020	**Ocean View Farm** 41.52, − 71.00	Newport, RI	1.05	2.83	Research and assessment, revegetate dieback	7	No results yet
2020	**Smith Point** 40.74, − 72.88	Sandy Hook, NJ	1.43	4.15	Mosquito control, revegetate dieback	56	No results yet

pressure excavators or ditchers (Supplemental File 1) to follow topographical low areas, and only drain water within the rooting zone (Hulsman et al. 1989; Wigand et al. 2017). Runnels are similar in principle to tidal creek extension projects that connect an area of inundation to the tidal creek network, though tidal creek extensions are larger in scale than runnels (Raposa et al. 2019; Taylor et al. 2020; Wetland restoration at Farm Creek Marsh 2021). After observing rapid expansion of shallow water within northeastern US marshes, restoration ecologists began working with mosquito control agencies to use runnels for the dual purpose of mosquito abatement and marsh adaptation to SLR. Practitioners used the technique to target shallow water features that were expanding, had formed within the last few decades, and where an anthropogenic topographic feature was impairing water flow (ditch spoils, plugged ditches, embankments) (Wigand et al. 2017; Adamowicz et al. 2020; Perry et al. 2021; Wolfe et al. 2021). True ponds that remained flooded throughout the tide cycle, with unconsolidated sediments in the basin, and that appeared stable in dimension on decadal timescales, were not targeted with this technique (workshop communications). Save the Bay (STB), an environmental non-profit, launched a series of projects using runnels in Rhode Island (RI), USA beginning in 2010. In our 2020 workshop, resource managers reported projects on dozens of marshes across six northeastern US states, and another half dozen northeastern and mid-Atlantic marshes were under consideration for runnel-adaptation by land trusts, NGOs, and government agencies (Supplemental File 1). The majority of projects from the workshop, and one recently published study on runnels (Perry et al. 2021), have reported some vegetation recovery within 1–5 years (Table 2 and Supplemental File 1).

Runnel Case Study: Winnapaug Marsh, RI

An STB restoration project provides a case study on patterns of vegetation recovery. We selected this project because it has the longest monitoring record (8 years) of the STB

projects, including pre-treatment data. Habitat restoration using runnels can be summarized by three phases. Phase I: "Drainage" is characterized by a loss of standing surface water. Phase II: "Early colonizers" is characterized by bare sediment which is colonized by *Salicornia* spp. and *Spartina alterniflora*. Phase III: "High-marsh species" is characterized by *Distichlis spicata*, *Spartina patens*, and *Juncus gerardii* succeeding early colonizers.

Winnapaug back barrier salt marsh in RI (41.3306°N, −71.7684°W) is a grid-ditched marsh with significant surface water cover and platform degradation (Fig. 3). Tidal range at the nearest tide station in Newport, RI is 1.05 m; however, tidal amplitudes are restricted in back-barrier environments such as Winnapaug. Ditches were created during the 1930s, and peat spoils were placed along ditch edges, creating linear impoundments. Altered topography in combination with RSLR in RI (5.26 mm yr^{-1} between 1999 and 2015) (Raposa et al. 2017) led to the "waffle-maple-syrup" pattern (Adamowicz et al. 2020) seen in aerial imagery (Fig. 3). As of 2011, large mats of filamentous algae were growing in shallow water areas (Fig. 1e), and mosquito larvae were observed. Initial depths of shallow water areas ranged from a few centimeters up to about 25 cm and were generally less than 15 cm deep.

STB and Town of Westerly, RI, secured funds and permits to create runnels targeting shallow water areas. Environmental and vegetation monitoring was conducted prior to runnel creation in 2011, and post-implementation monitoring

was repeated in 2013–2015, 2017, and 2019. Initial hand excavation of a few small runnels began in summer 2012. In May 2013, STB and RI Department of Environmental Management's Mosquito Abatement Program used a low-ground pressure excavator to expand the runnel network, and volunteers hand dug smaller runnels. Clogged mosquito ditches were cleared, and the material was used to fill selected ditches and degraded areas. Hand digging continued in 2013–2014 to facilitate additional drainage. In total, around 33 runnels were created ranging from 2 to 8 m in length. Runnel widths ranged from 10 to 24 cm, and depths ranged from 10 to 18 cm.

Surveys of vegetation and surface water were conducted using quadrat sampling along transects (Roman et al. 2001). Vegetation and ground cover was estimated as percent cover of each transect (Fig. 3). In the text below, transect data was aggregated to present coverages by species or cover type for the entire marsh. Initially (2011), algal mats covered 44%, open water 14%, and bare peat 4.5% of the marsh platform. The marsh was dominated (57% cover) by *Spartina alterniflora*, a species which tolerates frequent inundation (Fig. 3). Less-flood tolerant, "high-marsh" species included *Distichlis spicata* (26%), *Spartina patens* (18%), and *Juncus gerardii* (2.7%).

Ecosystem responses to runnels proceeded as across the marsh (Fig. 4). During Phase I open water decreased to 5% by 2013, and was absent in 2014 across the entire marsh. Algal mats disappeared by 2013. During Phase II,

Fig. 3 Vegetation change shown as percent cover for Winnapaug transects (T1, T2, T3 bar plots) with runnelling (*excavation years) displayed over aerial imagery from 2011. Species in legend are ordered from most tolerant of inundation (purple — *Salicornia*) to least tolerant (yellow — *Iva*), and transects shown as black lines. White arrows indicate respective vegetation cover data. T2 and T3 had significantly greater surface water than T1 to begin with, and T1 had greater recolonization of high-marsh species than T2 or T3
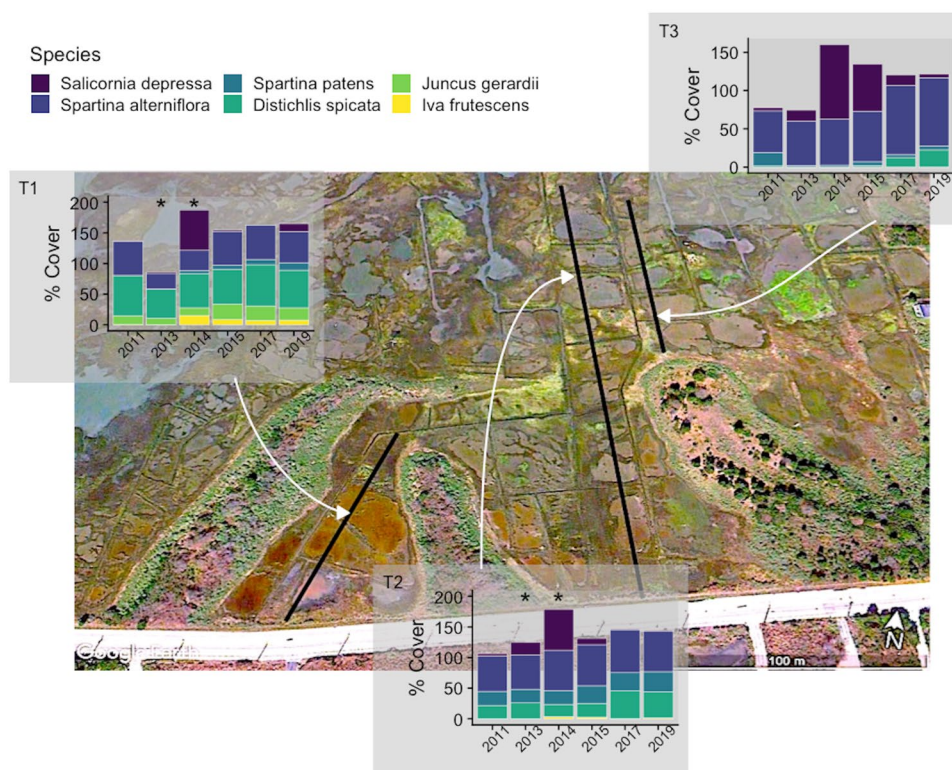
Fig. 4 Recovery of Winnapaug marsh after runnelling shown for **a** low elevation, T2 in Fig. 3, and **b** high elevation, T1 in Fig. 3, areas. Elevation was inferred from water table monitoring. Ground and species percent cover shown relative to runnel construction timeline on *x* axis. First dashed line (2012) indicates when digging began; excavation continued through 2014 ("Runnels complete"). Monitoring dates listed in text. Note that Phases I and II occurred simultaneously, and Phase III occurred sooner along the high-elevation transect
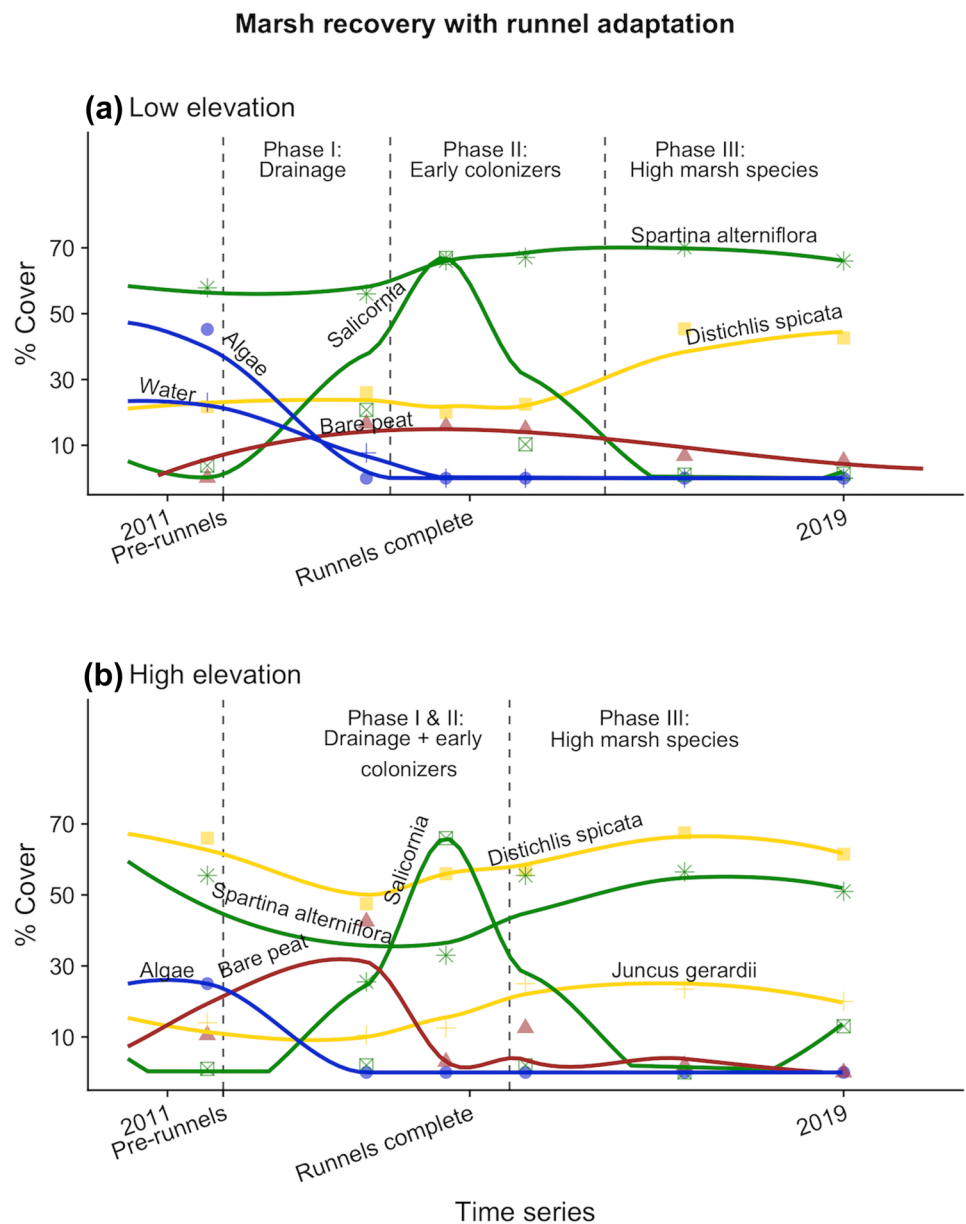
bare peat initially increased as water drained from the site (maximum of 26% by 2013), but then declined (3.2% by 2019) as areas were recolonized. *Salicornia depressa*, a flood-tolerant, early-colonizing species, increased rapidly from 3.3% prior to runnels to 73% in 2 years (2014). *Salicornia* then declined to 4.3% by 2019 as less flood-tolerant, high-marsh species increased. After 3 years (2015), Phase III-high marsh species began to increase. After 7 years (2019), *Distichlis* had increased to 42% cover, *Spartina patens* to 24% cover, and *Juncus* to 3.8%. *Spartina alterniflora* remained the dominant cover, increasing to 65% cover after 3 years, and 68% after 7 years (2019). The increase in vegetation, especially high-marsh species, suggests that

runnels have potential for short-term restoration of marsh plants.

While vegetation recovered across the marsh on the whole, responses differed across the marsh. Platform elevations along transect T1 were conducive to high-marsh species growth prior to runnel creation; as a result, draining the shallow water areas allowed bare peat to recolonize with high-marsh species quickly (Fig. 3). In contrast, the shallow water areas at the northern ends of transects T2 and T3 (Fig. 3) showed minimal response to runnels. Water levels decreased, but the features never fully drained and vegetation did not recover. As a result, T2 and T3 vegetation responses differed from T1 (Fig. 3). Based on water table monitoring

(Supplemental File 2), STB believes that basin elevations in some of the northern shallow water areas were too low in elevation for vegetation to recover. Long-term monitoring at this and other runnel project sites is important for assessing which marshes are good candidates for runnels, and how much time we can "buy" using this technique (a few years, decades, or more).

Efficacy of Runnels: Can We Buy Time for Salt Marshes?

Runnels have promise as a climate adaptation technique. Practitioner experiences shared at our 2020 workshop suggest that runnels will be most effective in higher elevation areas, and where peat is less degraded with root mats still intact and still firm within shallow water areas. Since only a few of the resource management projects that have used runnels included experimental designs or extensive data collection, work is still needed to statistically test runnel efficacy (Table 2 and Supplemental File 1). As a result, first-order questions remain on how responses in vegetation and the spatial footprint of a single runnel vary across a range of elevations, degradation levels (extent of elevation loss and peat decomposition within shallow water areas), tidal ranges, and suspended sediment concentrations. Key second-order questions remain as well, including when runnels risk triggering pool collapse, which ecosystem services are improved, if peat oxidation ever occurs, and, most critically, whether runnels can rebuild long-term resilience of salt marshes.

In 2020, our team launched an experiment to test runnels and address a set of these first- and second-order questions. We established 20 study sites (10 treatment, 10 control) at a range of elevations and degradation levels across two marsh complexes (Little Bay Conservation Area and Ocean View Farm; Table 2) using a replicated-BACI design. We are monitoring responses in vegetation, hydrology, soil properties, elevation, and decomposition, as well as a suite of baseline environmental characteristics. With this study, we will measure the rate of vegetation recovery and the spatial footprint of runnels and determine how hydrologic and vegetation responses differ along gradients of elevation and degradation. However, tidal range (~ 1 m) and suspended sediment concentrations (low) are similar at all our sites, so future studies will be needed to assess runnel efficacy along those gradients. Below, we discuss second-order questions on runnels, how our study is helping to answer them, and where future research is needed.

Avoiding Pool Collapse

Pool collapse could result from creating a runnel if the basin of a shallow water area is too low in elevation for

revegetation, if RSLR is faster than the accretion rate in the basin, or if too little suspended sediment is available for drained shallow water areas to accrete. While creating a runnel into a basin that is too low in elevation should be avoided, cases may exist where a resource manager does not have access to elevation data, or where the threshold elevation for revegetation is unknown. To avoid undesirable outcomes, practitioners manage water volumes flowing out of and into the shallow water area with sills, and by creating runnels in phases. Sills are shallow runnel segments that function as "speed bumps" to slow water velocity, and trap any unconsolidated material that could erode out of shallow water areas after runnel installation. Sills are created by shallower excavation or leaving unexcavated platform, while retaining positive drainage. Sills between runnel terminus and the connecting creek are important when larger tidal channels or high winds could expose bare soil to erosive energy. Runnel construction has been used as an adaptive management approach (Williams 2011), using phased construction to avoid erosion. After initial construction, managers evaluate drainage, and redistribute any sediment trapped in the runnel across the marsh platform. Removing unconsolidated sediment from the runnel avoids plugs forming, and conserves sediment which is a limited resource in many northeastern US marsh systems. Continued excavation is usually required to fully drain standing water. These approaches are encouraged by practitioners even when risks of pool collapse are not obvious, as they also help to avoid over-draining marshes with too much excavation at once. Long-term monitoring is still needed to determine whether pool collapse can be prevented in microtidal, low-sediment marshes with runnels.

Ecosystem Services

Observations suggest functional improvement of marshes after runnel construction. After 2016 excavated peat has been used to create small "islands" of elevated habitat for nesting saltmarsh sparrows (*Ammospiza caudacuta*; Fig. 1b), and other birds. Formerly these spoils were required under federal permits to be disposed in upland areas. Authors have observed sparrows nesting in areas drained by runnels shortly after creation. Small killifish (*Fundulus heteroclitus*) use runnels to access new foraging habitat, sometimes within hours of excavation. Invasive *Phragmites australis*, which outcompetes resident vegetation (but has a low salt tolerance), can be reduced in height and density by draining freshwater with runnels. Surveys of vegetation and wildlife before and after runnel creation are needed to understand the extent and longevity of habitat improvements. Research on other ecosystem services potentially affected by runnels

including nutrient cycling and organic matter storage is still needed.

Risk of Peat Oxidation

Over-draining marshes (e.g., with deep ditches) can lead to peat oxidation and platform subsidence (Burdick et al. 2020). Elevation losses worsen flooding stress from RSLR, and contribute to open-water conversion (Ganju et al. 2020). Runnels are designed to emulate naturally formed channels and avoid over-draining soils by only lowering water in the root zone. Water measurements from Winnapaug runnel sites show that the water table remained within a few centimeters of the soil surface (Supplemental File 2), suggesting minimal risk of over-draining from the shallow runnels used at this site. Further, one study found no increase in CO_2 emissions after installing runnels, suggesting that peat oxidation rates did not change (Perry et al. 2021). Over-draining soils could limit recovery of target vegetation species as well. A review of tidally restored marshes in Connecticut, USA found improved vegetation recovery when mean water tables were 24 cm below the marsh surface over cases with deeper water tables (29 cm below) (Warren et al. 2002). With a maximum runnel depth of 30 cm used in runnel projects (20 cm in some cases), we suggest that there is generally little risk of peat oxidation or creating inhospitable conditions for target vegetation species with runnels. Further research investigating how runnels impact soil saturation, decomposition, and resultant elevation is needed. In our current experimental work, we are measuring decomposition rates and monitoring hydroperiod, water table, elevation, and redox of marshes before and after runnel creation to address some of these questions.

Runnels and Long-Term Resilience of Marshes

Short-term responses to runnel construction are encouraging, but long-term ecosystem responses are uncertain. Runnels are proposed to imitate pool recovery in marshes where it may not otherwise occur by draining standing water and facilitating revegetation. They "buy time" for marshes to naturally adapt to SLR by vertical accretion or upland migration, or for additional intervention by managers to occur. However, empirical knowledge of critical thresholds past which marshes cannot recover from an open water conversion trajectory is sparse. As pool recovery occurs with large tidal ranges and high sediment loads, it remains unclear if runnels will be effective without these characteristics over the long term. The volume of tidal water flowing through a runnel and whether water drains fully from a shallow water area are critical variables that will define effectiveness — whether an area revegetates or deepens to a become a permanent pool

(Mariotti 2016; Schepers et al. 2020). Evident in this discussion is the need for a clear understanding of runnels from both theoretical and mechanistic perspectives.

Conditions that reduce marsh resilience to RSLR, specifically, low tidal range, low sediment supply, and low elevation capital, are not changed by runnel creation. Microtidal marshes are particularly vulnerable, where low tidal range corresponds with low sedimentation rates and reduced elevation capital as compared to mesotidal and macrotidal systems (Kearney and Turner 2016). These marshes are still vulnerable to drowning without an external sediment source (Kearney and Turner 2016; Ganju et al. 2020), or compensatory upland space to migrate (FitzGerald and Hughes 2019). Thus, runnels are unlikely to improve long-term marsh resilience to RSLR without additional adaptation strategies, e.g., marsh migration facilitation and sediment placement (La Peyre et al. 2009; FitzGerald and Hughes 2019). Facilitating marsh migration has been attempted by conserving marsh-adjacent habitats and removing human-made barriers. Other techniques include digging runnels into bordering freshwater wetlands, and "terracing" techniques have been proposed (Salt marsh bird conservation plan for the Atlantic Coast 2019). Sediment placement can compensate for low sediment supply by increasing elevation capital (La Peyre et al. 2009; Salt marsh bird conservation plan for the Atlantic Coast 2019). While additional adaptive action is likely required for marshes to persist, neither sediment placement nor marsh migration will be successful if marsh hydrology is severely compromised. Sediment additions into shallow water areas without facilitating drainage are unlikely to revegetate because waterlogged soil conditions will not support plant growth. Marsh migration into adjacent habitats occurs within a narrow range of elevation. If marsh just below this is stressed by flooding or has converted to open water, then migration is restricted. Runnels will not save salt marshes alone, but by helping salt marsh vegetation recover they complement other approaches.

Conclusions: Runnels and Resilience

The rate of marsh loss during the past few decades has raised alarm among managers, landowners, and communities who began urgently seeking solutions. In response to interior drowning of northeastern US marshes, ecosystem managers are using runnels to drain water and restore vegetation. Runnel projects proceeded without systematic examination of the conditions in which runnels are most effective or appropriate. Practitioners, regulators, and scientists alike have called for an evaluation of runnel projects to better inform management and funding decisions. Our workshop connected stakeholders and scientists to stimulate knowledge sharing on the runnel technique. In this paper,

we synthesized those discussions to document current consensus on runnel practice and known efficacy. While initial projects have shown runnels can facilitate revegetation of degraded marshes under some conditions, interactions with elevation and tide range are just a few of the factors needing assessment to include runnels in holistic adaptive planning to restore marsh habitat, and improve salt marsh resilience.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s12237-021-01028-8.

References

Adamowicz, S.C., and C.T. Roman. 2005. New England salt marsh pools: A quantitative analysis of geomorphic and geographic features. *Wetlands* 25: 279-288.

Adamowicz, S.C., G. Wilson, D.M. Burdick, W. Ferguson, and R. Hopping. 2020. Farmers in the marsh: Lessons from history and case studies for the future. *Wetland Science & Practice* 183–195.

Babson, A.L., R.O. Bennett, S. Adamowicz, and S. Stevens. 2020. Coastal impacts, recovery, and resilience post-Hurricane Sandy in the northeastern US. *Estuaries and Coasts* 43: 1603–1609. https://doi.org/10.1007/s12237-020-00809-x.

Barras, J. A., S. Vebille, D. Britsch, S. Hartley, S. Hawes, J. Johnston, P. Kemp, et al. 2003. Historical and projected coastal Louisiana land changes: 1978–2050: USGS Open File Report 03–334.

Bindoff, N.L., W. W. L. Cheung, J. G. Kairo, J. Arístegui, V.A. Guinder, R. Hallberg, N. Hilmi, et al. In press. 2019: Changing ocean, marine ecosystems, and dependent communities. In *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*, ed. H.-O. Pörtner, D. C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, et al.

Burdick, D.M., G.E. Moore, S.C. Adamowicz, G.M. Wilson, and C.R. Peter. 2020. Mitigating the legacy effects of ditching in a New England salt marsh. *Estuaries and Coasts* 43: 1672–1679. https://doi.org/10.1007/s12237-019-00656-5.

Campbell, A., and Y. Wang. 2019. High spatial resolution remote sensing for salt marsh mapping and change analysis at Fire Island National Seashore. *Remote Sensing* 11: 1107. https://doi.org/10.3390/rs11091107.

Chambers, L.G., H.E. Steinmuller, and J.L. Breithaupt. 2019. Toward a mechanistic understanding of "peat collapse" and its potential contribution to coastal wetland loss. *Ecology* 100: e02720. https://doi.org/10.1002/ecy.2720.

Costa, J.E., and M. Weiner. 2017. *Atlas of changes in salt marsh boundaries at selected islands in the West Branch of the Westport River, 1934–2016.* Buzzards Bay National Estuary Program Technical Report. Wareham, MA.

DeLaune, R.D., J.A. Nyman, and W.H. Patrick. 1994. Peat collapse, ponding and wetland loss in a rapidly submerging coastal marsh. *Journal of Coastal Research* 10: 1021–1030.

Duran Vinent, O., E.R. Herbert, D.J. Coleman, J.D. Himmelstein, and M.L. Kirwan. 2021. Onset of runaway fragmentation of salt marshes. *One Earth* 4: 506–516. https://doi.org/10.1016/j.oneear.2021.02.013.

Ewanchuk, P.J., and M.D. Bertness. 2004. Structure and organization of a northern New England salt marsh plant community. *Journal of Ecology* 92: 72–85. https://doi.org/10.1111/j.1365-2745.2004.00838.x.

FitzGerald, D.M., and Z. Hughes. 2019. Marsh processes and their response to climate change and sea-level rise. *Annual Review of Earth and Planetary Sciences* 47: 481–517. https://doi.org/10.1146/annurev-earth-082517-010255.

Ganju, N.K., Z. Defne, and S. Fagherazzi. 2020. Are elevation and open-water conversion of salt marshes connected? *Geophysical Research Letters* 47. https://doi.org/10.1029/2019GL086703

Gedan, K.B., B.R. Silliman, and M.D. Bertness. 2009. Centuries of human-driven change in salt marsh ecosystems. *Annual Review of Marine Science* 1: 117–141. https://doi.org/10.1146/annurev.marine.010908.163930.

Himmelstein, J., O. Duran Vinent, S. Temmerman, and M.L. Kirwan. 2021. Mechanisms of pond expansion in a rapidly submerging marsh. *Frontiers in Marine Science* 8: 704768. https://doi.org/10.3389/fmars.2021.704768.

Hulsman, K., P.E. Dale, and B.H. Kay. 1989. The runnelling method of habitat modification: An environment-focused tool for salt marsh mosquito management. *Journal of the American Mosquito Control Association* 5: 226–234.

Kearney, M.S., and R.E. Turner. 2016. Microtidal marshes: Can these widespread and fragile marshes survive increasing climate–sea level variability and human action? *Journal of Coastal Research* 32: 686. https://doi.org/10.2112/JCOASTRES-D-15-00069.1.

La Peyre, M.K., B. Gossman, and B.P. Piazza. 2009. Short- and long-term response of deteriorating brackish marshes and open-water ponds to sediment enhancement by thin-layer dredge disposal.

Estuaries and Coasts 32: 390–402. https://doi.org/10.1007/s12237-008-9126-8.

Mariotti, G. 2016. Revisiting salt marsh resilience to sea level rise: Are ponds responsible for permanent land loss? *Journal of Geophysical Research: Earth Surface* 121: 1391–1407. https://doi.org/10.1002/2016JF003900.

Mcowen, C., L. Weatherdon, J.-W. Bochove, E. Sullivan, S. Blyth, C. Zockler, D. Stanwell-Smith, et al. 2017. A global map of saltmarshes. *Biodiversity Data Journal* 5: e11764. https://doi.org/10.3897/BDJ.5.e11764.

Oppenheimer, M., B.C. Glavovic, J. Hinkel, R. van de Wal, A.K. Magnan, A. Abd-Elgawad, R. Cai, et al. In press. 2019: Sea level rise and implications for low-lying islands, coasts and communities. In *IPCC Special Report on the Ocean and Cryosphere in a Changing Climate*, ed. H.-O. Pörtner, D. C. Roberts, V. Masson-Delmotte, P. Zhai, M. Tignor, E. Poloczanska, K. Mintenbeck, et al.

Perry, D.C., W. Ferguson, and C.S. Thornber. 2021. Salt marsh climate adaptation: Using runnels to adapt to accelerating sea level rise within a drowning New England salt marsh. *Restoration Ecology*. https://doi.org/10.1111/rec.13466.

Raposa, K.B., M.L. Cole Ekberg, D.M. Burdick, N.T. Ernst, and S.C. Adamowicz. 2017. Elevation change and the vulnerability of Rhode Island (USA) salt marshes to sea-level rise. *Regional Environmental Change* 17: 389–397. https://doi.org/10.1007/s10113-016-1020-5.

Raposa, K.B., R.L. Weber, W. Ferguson, J. Hollister, R. Rozsa, N. Maher, and A. Gettman. 2019. Drainage enhancement effects on a waterlogged Rhode Island (USA) salt marsh. *Estuarine, Coastal and Shelf Science* 231: 106435. https://doi.org/10.1016/j.ecss.2019.106435.

Redfield, A.C. 1972. Development of a New England salt marsh. *Ecological Monographs* 42: 201–237. https://doi.org/10.2307/1942263.

Roman, C.T., M.J. James-Pirri, and J.F. Heltshe. 2001. Monitoring salt marsh vegetation: A protocol for the long-term coastal ecosystem monitoring program at Cape Cod National Seashore. Coordinated by the USGS Patuxent Wildlife Research Center, Coastal Research Field Station at the University of Rhode Island, Narragansett, RI 02882.

Salt marsh bird conservation plan for the Atlantic Coast. 2019. Atlantic Coast Joint Venture.

Schepers, L., P. Brennand, M.L. Kirwan, G.R. Guntenspergen, and S. Temmerman. 2020. Coastal marsh degradation into ponds induces irreversible elevation loss relative to sea level in a microtidal system. *Geophysical Research Letters* 47. https://doi.org/10.1029/2020GL089121.

Schepers, L., M. Kirwan, G. Guntenspergen, and S. Temmerman. 2017. Spatio-temporal development of vegetation die-off in a submerging coastal marsh. *Limnology and Oceanography* 62: 137–150. https://doi.org/10.1002/lno.10381.

Smith, J.A.M., and M. Pellew. 2021. Pond dynamics yield minimal net loss of vegetation cover across an unditched salt marsh landscape. *Estuaries and Coasts* 44: 1534–1546. https://doi.org/10.1007/s12237-020-00882-2.

Taylor, L., D. Curson, G.M. Verutes, and C. Wilsey. 2020. Mapping sea level rise impacts to identify climate change adaptation opportunities in the Chesapeake and Delaware Bays, USA. *Wetlands Ecology and Management* 28: 527–541. https://doi.org/10.1007/s11273-020-09729-w.

Vincent, R.E., D.M. Burdick, and M. Dionne. 2014. Ditching and ditch-plugging in New England salt marshes: Effects on plant communities and self-maintenance. *Estuaries and Coasts* 37: 354–368. https://doi.org/10.1007/s12237-013-9671-7.

Warren, R.S., P.E. Fell, R. Rozsa, A.H. Brawley, A.C. Orsted, E.T. Olson, V. Swamy, and W.A. Niering. 2002. Salt marsh restoration in Connecticut: 20 years of science and management. *Restoration Ecology* 10: 497–513. https://doi.org/10.1046/j.1526-100X.2002.01031.x.

Watson, E.B., C. Wigand, E.W. Davey, H.M. Andrews, J. Bishop, and K.B. Raposa. 2017. Wetland loss patterns and inundation-productivity relationships prognosticate widespread salt marsh loss for southern New England. *Estuaries and Coasts* 40: 662–681. https://doi.org/10.1007/s12237-016-0069-1.

Wetland restoration at Farm Creek Marsh. 2021. A Blackwater 2100 Project. Audubon Society. https://md.audubon.org/sites/default/files/farm_creek_marsh_083018-web.pdf. Accessed September 23.

Wigand, C., T. Ardito, C. Chaffee, W. Ferguson, S. Paton, K. Raposa, C. Vandemoer, and E. Watson. 2017. A climate change adaptation strategy for management of coastal marsh systems. *Estuaries and Coasts* 40: 682–693. https://doi.org/10.1007/s12237-015-0003-y.

Williams, B.K. 2011. Adaptive management of natural resources—framework and issues. *Journal of Environmental Management* 92: 1346–1353. https://doi.org/10.1016/j.jenvman.2010.10.041.

Wilson, C.A., Z.J. Hughes, D.M. FitzGerald, C.S. Hopkinson, V. Valentine, and A.S. Kolker. 2014. Saltmarsh pool and tidal creek morphodynamics: Dynamic equilibrium of northern latitude saltmarshes? *Geomorphology* 213: 99–115. https://doi.org/10.1016/j.geomorph.2014.01.002.

Wolfe, R. 1996. Effects of open marsh water management on selected tidal marsh resources: A review. *Journal of the American Mosquito Control Association* 12: 701–712.

Wolfe, R., P. Zarebicki, and W. Meredith. 2021. The evolution of salt-marsh mosquito control water management practices relative to coastal resiliency in the Mid-Atlantic and northeastern United States. *Wetlands Ecology and Management*. https://doi.org/10.1007/s11273-021-09817-5.