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Soil carbon consequences of historic hydrologic impairment and recent restoration in coastal wetlands

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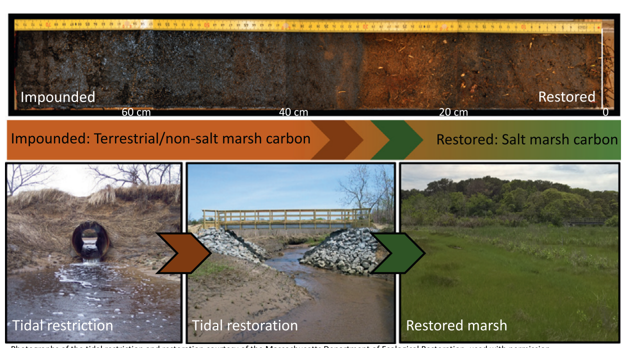
HIGHLIGHTS

- Hydrologic impairment is a common feature in coastal wetlands.
- Soil carbon was assessed at impounded, restored, and natural hydrology wetlands.
- Hydrologic impairment reduced carbon and vertical accretion by 30 to 70 %.
- Restored tidal exchange increased carbon and vertical accretion.
- Organic matter was the main contributor to new sediment at the restored sites.

GRAPHICAL ABSTRACT



Photographs of the tidal restriction and restoration courtesy of the Massachusetts Department of Ecological Restoration, used with permission.

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ABSTRACT

Coastal wetlands provide key ecosystem services, including substantial long-term storage of atmospheric CO_2 in soil organic carbon pools. This accumulation of soil organic matter is a vital component of elevation gain in coastal wetlands responding to sea-level rise. Anthropogenic activities that alter coastal wetland function through disruption of tidal exchange and wetland water levels are ubiquitous. This study assesses soil vertical accretion and organic carbon accretion across five coastal wetlands that experienced over a century of impounded hydrology, followed by restoration of tidal exchange 5 to 14 years prior to sampling. Nearby marshes that never experienced tidal impoundment served as controls with natural hydrology to assess the impact of impoundment and restoration. Dated soil cores indicate that elevation gain and carbon storage were suppressed 30–70 % during impoundment, accounting for the majority of elevation deficit between impacted and natural sites. Only one site had substantial subsidence, likely due to oxidation of soil organic matter. Vertical and carbon accretion gains were achieved at all restored sites, with carbon burial increasing from 96 ± 33 to 197 ± 64 g C m^{-2} y^{-1}. The site with subsidence was able to accrete at double the rate (13 ± 5.6 mm y^{-1}) of the natural complement, due predominantly to organic matter accumulation rather than mineral deposition, indicating these ecosystems are capable of large dynamic responses to restoration when conditions are optimized for vegetation growth. Hydrologic restoration enhanced elevation resilience and climate benefits of these coastal wetlands.

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1. Introduction

Tidal wetlands support the delivery of a wealth of ecosystem services, including habitat for birds, fish and other species, storm surge protection, coastal resilience to sea-level rise and recreational opportunities (Barbier et al., 2011; Narayan et al., 2017; Shepard et al., 2011; Xu et al., 2020). The marsh structure is the foundation of these ecosystem services and is composed of organic material, typically grown in situ, and mineral matter retained on the platform during periods of flooding. The organic marsh platform performs a key additional ecosystem service, namely long-term storage of atmospheric CO_2 taken up by marsh vegetation during photosynthesis and subsequently placed into a flooded, anoxic subsurface environment that is conducive to preservation (Neubauer and Megonigal, 2021; Redfield, 1972). This involvement in the global climate system via active cycling of CO_2 and other greenhouse gases (GHG) is referred to as blue carbon, with long-term carbon storage rates by coastal wetlands of 210 to 245 g C m^{-2} y^{-1}, resulting in a total global uptake of 10.2 Tg C y^{-1} (Chmura et al., 2003; Ouyang and Lee, 2014). The Intergovernmental Panel on Climate Change (IPCC) 2013 Wetlands Supplement provided guidance for inclusion of wetlands within national GHG inventories (IPCC, 2014). The U.S. first included wetlands in the 2017 inventory, noting that the GHG mitigation potential associated with wetland restoration was greatly underrealized due to low rates of coastal wetland restoration (Crooks et al., 2018; Gittman et al., 2019). In response to sea-level rise, wetland managers are determining whether and how to sustain coastal wetlands in the future (Kassakian et al., 2017), leading to an urgent need for wetland carbon cycle science to inform policies impacting these ecosystems.

Large areal conversions of tidal wetlands have historically occurred due to changes in land use (Gedan et al., 2009), resulting in an estimated ~0.48 million hectares of restricted and impounded wetlands in the continental U.S. where tidal exchange between the marsh and adjacent bay or estuary is restricted or blocked inadvertently as a consequence of road or other infrastructure construction, or intentionally as a result of diking and drainage (Fargione et al., 2018; Kroeger et al., 2017; Roman et al., 1984). Under these conditions, plant communities and soil biogeochemical processes are altered due to changes in water levels, flooding frequency, and salinity, impairing the ecological functions of salt marshes (Portnoy, 1999; Warren et al., 2002). For example, in New England substantial terrestrial run-off causes freshening behind impoundments and conversion of salt tolerant *Spartina*-dominated systems to monotypic stands of *Phragmites australis*, an invasive species without the same habitat provision as native salt marsh (Roman et al., 1984). Along the southeastern U.S. coast, impoundments for waterfowl also are fresher than their natural counterparts without managed hydrology (Montague et al., 1987). Conversely, areas without terrestrial freshwater input to coastal wetlands can have hypersaline ponding in impounded areas (Huff and Feagin, 2017). When tidal exchange is blocked, water table variability may occur due to changes in precipitation and evapotranspiration and may result in organic matter oxidation and subsidence during periods of lower water table as well as peat deposition during higher water tables. As a result, elevation loss may occur in impounded wetlands even if the water table is not lowered consistently throughout the impounded history (Bryant and Chabreck, 1998). Impounded conditions can follow historic drainage when the platform loses sufficient elevation that the water table rises relative to the land surface, as observed in modern times in the Herring River, MA, USA (Portnoy and Giblin, 1997). While impounded marshes may not lose elevation at the same rate as drained marshes, since the surface remains vegetated and the soil column flooded, they are, however, disconnected from tidal exchange and do not experience the press of sea-level rise or mineral sediment inputs from tidal exchange, effectively dampening the vertical accretion response (Anisfeld et al., 1999; Bryant and Chabreck, 1998; Kirwan and Temmerman, 2009; Morris et al., 2002). As a result carbon burial is suppressed compared to systems with natural hydrology (Suir et al., 2019). Currently, impounded coastal wetlands as described here are not covered in IPCC guidance on CO_2 emissions factors, so wetlands that fall within this managed category may not be fully accounted for in GHG inventories (IPCC, 2014, 2019).

While habitat impairment has driven much of the salt marsh restoration that has occurred to date, reduction in carbon uptake and enhanced methane emissions are increasingly considered as important co-benefits of ecosystem restoration (Crooks et al., 2018; Needelman et al., 2018). Indeed, coastal wetland restoration is seen as an important opportunity to enhance carbon capture through natural climate solutions (Fargione et al., 2018). The goals of this study are two-part. First, we assess the historical consequences of hydrologic manipulation on subsidence and associated soil carbon loss as well as suppression of carbon burial in impounded wetlands. These carbon emissions and reduced storage capacity represent the anthropogenic soil organic carbon emissions associated with impoundment. Second, we determine the impact of renewed tidal exchange on soil and carbon accretion following hydrologic restoration to assess the GHG mitigation potential related to soil carbon storage in restored wetlands. To do this, we compared paired natural salt marshes, with no known history of hydrologic manipulation, and restored marshes, where tidal exchange was reintroduced due to removal or enlargement of culverts, at five marshes on Cape Cod, MA, USA, that were restored between 5 and 14 years prior to sample collection. Marsh surface elevation, porewater levels and salinity, and creek water levels and salinity were compared between sites to assess if tidal restoration was successful. We further evaluate vertical and carbon accretion rates, as well as assess the source of soil organic matter. At each location, sediment age models were constructed to assess change over the past 100 years, covering the time each marsh was impounded and restored.

2. Methods

2.1. Experimental design

Salt marshes were identified where hydraulic connectivity was restored, typically by replacing an undersized culvert or other structure to enlarge the opening, allowing tidal exchange to occur with minimal impediment (Cape Cod Commission, 2001). Five marshes on the north shore of Cape Cod had paired restored and natural sites, with restored sites upstream from the former restriction, and natural sites in locations with no known history of altered hydrology (Fig. 1). The exception to this is Scusset Creek, where the closest unimpounded wetland was located across the Cape Cod Canal. Sites were selected within a small geographic range to mediate potential complicating factors such as connection to different water bodies and differences in geomorphology. At the time of sampling in 2015, these marshes had varying time since restoration that ranged from 5 to 14 years (Table 1). In this experiment, there are several factors that likely influence the impacts of impoundment and restoration, or our assessment of the impacts, including the following: 1) Each restored marsh site is upstream of the former restriction and the paired natural marsh sites. Thus, prior to impoundment, the restored marsh sites might have occupied a region of the marsh with greater freshwater influence, higher elevation, and potentially a greater percentage of high marsh species (Bertness, 1991; Redfield, 1972). This upstream-downstream relationship between restored and natural sites is a direct result of infrastructure that inhibits tidal exchange by bisecting marshes and characterizes many of the impounded marshes in New England (Karberg et al., 2018; Roman et al., 1984). 2) These sites have experienced different time scales of impoundment (from unknown to 71 to 137 years, Table 1). 3) While all restored sites displayed ecological characteristics of freshened marshes when impounded, especially the occurrence of *Phragmites australis*, the degree of tidal restriction, or how much of the water exchanged was impeded and the resultant water level behind the restriction, varied as a function of opening size and position, freshwater runoff volume, impounded marsh area, and tidal range. As a result, each marsh has a unique history of impoundment. A previous study reports plant biomass and carbon dioxide fluxes across these sites (Wang et al., 2020).

2.1.1. Study sites

Stony Brook consists of a freshwater stream with headwaters in a kettle pond that flows through a marsh to the ocean. The marsh was bisected by a

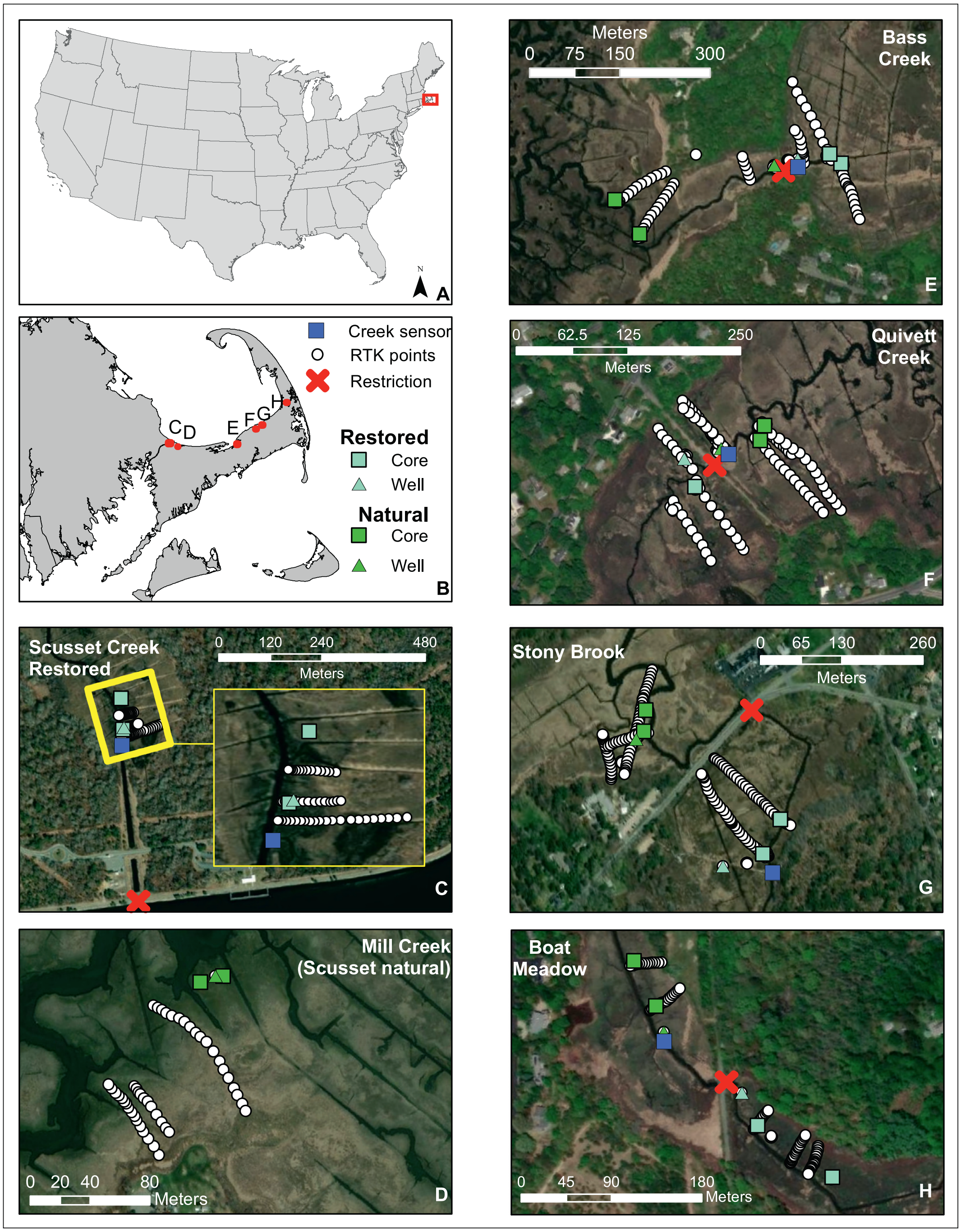

Fig. 1. Location map of the study site showing A) United States with study site indicated by red box, B) field sites on Cape Cod as numbered through the figure panels, C) Scusset Creek (restored) marsh, D) Mill Creek (Scusset Creek natural), E) Bass Creek, F) Quivett Creek, G) Stony Brook and H) Boat Meadow. Restored sites are shown in teal and natural sites in green. Cores are indicated by squares and wells by triangles. The red x marks the former restriction; the blue square is the location of the creek conductivity, temperature, and depth (CTD) sensor; and a white circle indicates the location of individual real-time kinematic geographic positioning system (RTK GPS) measurements. Maps and map images are the intellectual property of Esri and are used herein under license. Copyright © 2014 Esri and its licensors. All rights reserved.

Table 1
Restored marsh site characteristics, platform, and water level elevation differences between impounded and natural sites. Negative elevation difference values indicate the restored marsh value is lower than the natural marsh.

Marsh	Area restored (hectares)	Year impounded	Year restored	Years since restoration (as of 2015 sampling)	Mean difference in marsh platform elevation between restored and natural sites (cm)	Mean difference in water level elevation between restored and natural sites (cm)
Quivett Creek	4.5	Unknown	2005	10	10.4 ± 29.1	−2.6 ± 14.0
Bass Creek	15	Unknown	2008	7	3.2 ± 30.7	0.3 ± 14.3
Stony Brook	13	1895	2010	5	−18.5 ± 34.8	−9.6 ± 14.6
Boat Meadow	2.4	1865	2002	13	−14.1 ± 24.2	−0.8 ± 15.3
Scusset	20	1883	2001	14	−70.6 ± 20.5	−20.7 ± 27.8

bermed roadbed in 1895. Prior to restoration in 2010, two culverts were in place to allow tidal exchange, and one was replaced with a 5.5 m wide box culvert. Vegetation surveys prior to hydrologic restoration indicated significant *Phragmites* occurrence due to fresher conditions at the upstream portion of Stony Brook (Muramoto, 2011).

Bass Creek was a former salt marsh that was converted at an unknown time in the past into a cranberry bog by constructing a berm to which a culvert was added at an unknown date (Cape Cod Commission, 2001). An additional earthen berm that has been breached is in place approximately 100 m downstream of the restoration, with the date of placement and breaching unknown for that structure. In 2008 the culvert and berm were removed, and a bridge was built for the walking path, restoring the full tidal channel.

Quivett Creek was bisected during the construction of a road at an unknown time in the past (Cape Cod Commission, 2001). The road was subsequently abandoned due to flooding and became a walking path. In 2005 two culverts were removed and replaced with a bridge with an approximate 3 m wide opening. Prior to restoration, there was approximately 25 % cover by *Phragmites*.

Boat Meadow was restricted in 1865 with the construction of a railroad (Cape Cod Commission, 2001). Historically, this wetland was connected to an upstream kettle pond, however significant transportation infrastructure has altered the connectivity of the system. An additional earthen berm that appears to be a restriction occurs around 100 m downstream for which the date of construction and breaching are unknown. In 2002 a collapsed culvert was replaced, resulting in tidal flow to the region of the wetland that had been severely impounded. *Phragmites* are present upstream and downstream of the restriction, especially along the wetland edges.

The Scusset site includes Scusset Creek and the natural analogue, Mill Creek, across the Cape Cod Canal. Scusset Creek hydrology was altered beginning in 1883 with early development and later during the construction of the Cape Cod Canal, which was completed in 1940 (U.S. Geological Survey, 1996). The marsh was originally connected to Cape Cod Bay to the north, and when that area was developed, water flow was diverted to the south into the area that would later become the Cape Cod Canal. During construction of the Canal, beginning in 1930, the southern channel was blocked to a large extent by two small culverts. In 2001, the culverts were replaced with two 2 m box culverts outfitted with sluice gates to control water level and flow under the control of the U.S. Army Corps (US Army Corps of Engineers, 1996). In addition, at this restoration site the tidal creek was deepened and widened by dredging during hydrologic restoration. Restoration resulted in 20 ha of salt marsh vegetation compared to 5 ha prior to restoration, with invasive *Phragmites* remaining the dominant vegetation over the rest of the 120-hectare former salt marsh (Cape Cod Commission, 2001). Sampling for this study was done in the *Spartina* dominated area.

2.2. Sediment cores

Duplicate soil cores were collected from natural and restored marsh sites at five marshes in 2015 (Fig. 1, Table S1). At the time of collection, vegetation at the core sites consisted of *Spartina* species and other wetland species, including *Distichlis spicata* and *Juncus gerardi*. Cores were collected with a modified piston-coring system, whereby a core liner (11 cm diameter) was fitted with a gasketed piston that was placed on the sediment surface (Gonneea et al., 2019). The clear, sharpened core liner was pushed down into the marsh subsurface, while the piston was maintained at the marsh surface via tension on the piston. We visually observed the sediment surface to ensure that the soil column did not compact during collection. Peat recovery ranged from 56 to 168 cm and did not reach the basement glacial deposits. Cores were removed with a pulley system, capped, and returned to the Woods Hole U.S. Geological Survey laboratory the same day, and stored at 5 °C until processed, typically within 48 h. Cores were split vertically and sampled at 1 cm intervals to 30 cm and 2 cm intervals thereafter, except cores from Scusset Creek Marsh (SMB) and Mill Creek Marsh (MCA and MCB) which were sampled at 1 cm intervals to 45 cm and 2 cm thereafter. Sediment was weighed, frozen, and then freeze-dried, with dry bulk density (DBD) determined from the weight of a known volume of sediment after freeze-drying.

Sediment samples were prepared for radioisotope analysis by sealing approximately 5 g of dried, homogenized peat for three weeks and counted on a planar-type gamma counter for 24 to 48 h to measure ^{7}Be, ^{137}Cs, ^{210}Pb, and ^{226}Ra at 477, 661.6, 46.5 and 352 KeV energies respectively (Canberra Inc., USA). The top several sections from each core were counted within three weeks to capture ^{7}Be (53.28 day half-life), and then re-counted after three weeks. Detector efficiency was determined from EPA standard pitchblende ore in the same geometry as the samples. Activities of ^{7}Be, ^{137}Cs, and ^{210}Pb were decay corrected to time of collection. Suppression of low energy peaks by self-adsorption was corrected according to Cutshall et al. (1983). Excess ^{210}Pb was calculated by subtracting supported ^{226}Ra from total ^{210}Pb with a detection limit of 0.1 dpm g^{-1}.

Sediment ages were calculated using the continuous rate of supply ^{210}Pb age model, a variant on the advection-decay equation (Appleby and Oldfield, 1992; Goldberg, 1963). This age model assumes that ^{210}Pb down-core activity is a function of decay and variable sediment accretion. However, the full ^{210}Pb profile must be captured to prevent bias toward ages that are too old and accretion rates that are too low at depth (Binford, 1990). Vertical accretion rates were calculated as the age difference between each interval, with temporal resolution typically 2 to 10 years depending on marsh, treatment, and sediment age.

A 0.5-g aliquot of sediment from every-other sample depth was further ground in a ball mill and then analyzed for carbon (C) concentration and δ^{13}C at the U.C. Davis Stable Isotope Facility with an Elementar Vario EL Cube or Micro Cube elemental analyzer (Elementar Analysensysteme, Germany) interfaced to a PDZ Europa 20–20 isotope ratio mass spectrometer (Sercon Ltd., UK). Carbon density was calculated as the weight %C times the DBD; carbon accretion rates were calculated by multiplying the carbon density times the vertical accretion rate. All sediment core data are published in an accompanying data release (O'Keefe Suttles et al., 2021). Throughout the paper mean and standard deviation are reported. Difference between restored and natural sites and time periods were done in R (version 4.0.2) using the “t.test” function (R Core Team, 2020).

2.3. Water level and salinity

Wells were installed concurrently at restored and natural marsh sites to evaluate marsh porewater level and salinity in 2016 or 2019 (Fig. 1). Wells were screened (slot size ~3 mm) over 30 cm and installed to a depth of 35 cm, with a height of ~1 m off the marsh surface. InSitu AquaTroll 200 conductivity, temperature, and depth loggers were deployed to measure water level and salinity every 15 min. Conductivity was calibrated in the laboratory prior to deployment (50.0 mS/cm conductivity standard) and as needed in the field. Data were downloaded periodically (approximately monthly), sensors checked for fouling and calibration drift, and cleaned and calibrated as necessary prior to re-deployment. Water depths were corrected for barometric pressure using the WinSitu BaroMerge program (WinSitu Baromerge, 2019). Timing and duration of deployments varied throughout the study (Bass Creek: 4/21/16 to 12/21/16, Boat Meadow: 5/14–19 to 6/27/19, Quivett Creek: 3/18/16 to 12/21/16, Scusset and Mill Creeks: 7/2/19 to 10/24/19, and Stony Brook: 5/16/16 to 12/20/16), but a full spring-neap tidal cycle was captured at each marsh with further details available in accompanying data release (O'Keefe Suttles et al., 2022). In addition, a water level and conductivity logger (either an InSitu AquaTroll 200 or HOBO water level logger) was concurrently placed within the tidal creek during the study. Water levels were transformed to water elevations relative to the North American Vertical Datum of 1988 (NAVD88) based on elevation measurements of the sensors.

2.4. Elevation control, tidal datums and flooding duration

A Trimble RTK GPS was used to survey all core collection, well, and creek sensor sites. In addition, several transects across all marshes and treatments were conducted to evaluate marsh elevation ranges. All data were projected to the North American Datum of 1983 Massachusetts State Plane FIPS 2001 and elevations are given relative to NAVD88 with an elevation accuracy of 2–3 cm.

Flooding across the natural and restored marsh site was evaluated using the elevation surveys and the water level elevations. We assumed that the surface elevations measured at the transects across the marsh were representative of the larger marsh area. The flooding duration was calculated for each surveyed point as the time when the porewater level elevation was greater than the surface elevation. Note that this simple approach may both omit periods when the water level had drained at the well site, but standing surface water could still remain at certain locations on the marsh surface due to microtopography, as well as include periods categorized as flooded when water hadn't yet reached the entire marsh surface. We consider these omission and commission uncertainties to be small since the tidal range at all sites was sufficient to flood the marsh surface by 0.25 to 0.5 m and well data indicate porewater drainage occurred at all sites (O'Keefe Suttles et al., 2022).

3. Calculations

3.1. Accretion and soil organic carbon stock assessment

To assess the impact of impoundment and restoration on marsh elevation resiliency and soil organic carbon stocks, we outline the set of critical processes and how we evaluate them using dated soil cores. Carbon accretion occurs due to deposition and preservation of soil organic matter, with minor contribution from imported mineral sediment at these sites. Lower carbon accretion rates at impounded sites compared to analogous natural coastal wetlands results in avoided carbon sequestration. Subsidence is associated with organic matter oxidation upon lowering of the water table and has been reported across impounded and drained wetlands with a concurrent reduction in the relic soil organic carbon stock (Turner, 2004). Sediment records can integrate both subsidence and accretion if conditions promoting preservation versus decomposition vary through time, as might occur with water table fluctuations. In this study, a comparison of recent carbon accretion rates after restoration at previously impaired sites to rates at natural sites is used to assess the carbon burial capacity of restored wetlands. Though soil cores can provide imperfect elevation reconstructions due to decomposition or accretion that may be induced as environmental conditions change, such as salt to fresh and fresh to salt water conversion, they still provide a good integration of the impacts of environmental changes through time, and they are used in the analysis outlined below.

We first assess avoided carbon stock gains ($C_{avoided}$) due to suppression of carbon accretion rates during impoundment as follows:

$$C_{avoided} = (CAR_N - CAR_I)\Delta t \tag{1}$$

where CAR_N and CAR_I are the carbon accumulation rates (i.e. burial, g C m^{-2} y^{-1}) at the natural (N) and impounded (I) marshes, respectively, during the time the marsh was impounded and Δt is the time (years) of impoundment. Similarly, suppressed vertical accretion leads to avoided elevation gain ($E_{avoided}$) as follows:

$$E_{avoided} = (VAR_N - VAR_I)\Delta t \tag{2}$$

where VAR_N and VAR_I are the vertical accretion rates (mm y^{-1}) at the natural and impounded marshes, respectively, during the time the marsh was impounded. These values represent the unrealized carbon storage and elevation gain that are a consequence of impoundment.

We next assess soil carbon stock loss that occurs through subsidence if conditions during impoundment lead to decomposition of existing peat organic matter, soil compaction, and loss of elevation. Elevation differences between the natural and impounded/restored marshes reflect the accretion and elevation loss processes that have occurred in both the natural and restored marshes during impoundment and after restoration. The elevation difference between the restored and natural marshes at the time of impoundment (ΔE_i), restoration (ΔE_r), and at present (ΔE_p) is evaluated as follows:

$$\Delta E_t = E_{N,t} - E_{R,t} \tag{3}$$

where t is one of i (impoundment), r (restoration), or present (p) and $E_{N,t}$ and $E_{R,t}$ are the elevations of the natural (N) and restored marshes (R) at a given time, respectively. The present elevation of the marsh is determined by RTK GPS surveys across the marsh that integrate processes across space and provide a comprehensive evaluation of present elevations. Elevation differences between impounded/restored and natural marshes at the time of impoundment and restoration are derived from the dated soil cores. For this analysis, we assume that if at the time of restoration, the restored marsh has the same or higher elevation than the natural marsh ($\Delta E_r \leq 0$), no subsidence has occurred. However, if the restored marsh has an elevation deficit compared to the natural marsh ($\Delta E_r > 0$), it is then possible that subsidence has occurred at some point during impoundment. Subsidence is assessed by comparing the measured difference in elevation with that predicted by sediment vertical accretion rates as follows:

$$E_{subsidence} = \Delta E_r - E_{avoided} \tag{4}$$

If the measured elevation difference between natural and impounded marshes (ΔE_r) is the same or less than that predicted by accretion rate differences ($E_{avoided}$), we assume no subsidence and associated relic carbon stock loss has occurred. However, if the change in surface elevation is greater than that predicted by suppression of vertical accretion rates, we predict that subsidence has occurred and has been accompanied by a loss of soil carbon stock (i.e., g C m^{-2}) ($C_{subsidence}$) according to:

$$C_{subsidence} = C_d E_{subsidence} \tag{5}$$

where C_d is the soil carbon density. Note that comparison of historic carbon burial rates in restored and natural wetlands does not represent the full soil carbon benefits of restoration, which can only be assessed by comparing wetland future conditions under a no-restoration scenario. We do not attempt to model future conditions here; we compare restored and natural conditions to assess realized carbon burial benefits.

Finally, we assess loss of elevation capital due to impoundment. Elevation capital describes the elevation gains within the tidal frame that marshes maintain for optimal vegetative growth (Langston et al., 2021; Reed, 2002). If elevation was constant across the marsh platform, loss of elevation capital due to impoundment would simply be the elevation deficit between restored and natural marshes. However, restored marshes, particularly those that are closer to the marsh-upland border than their natural counterparts, may have had a higher elevation prior to impoundment than the natural counterpart. At such sites, while impoundment may not have resulted in lower elevations relative to the natural marsh, impoundment may still have resulted in lost potential elevation capital relative to elevation gains realized at the natural marsh, and ultimately relative to sea level. Loss of elevation capital was assessed by comparing the surface elevation differences between the natural and restored marshes at present to the surface elevation differences at the time of impoundment, plus any subsidence:

$$E_{capital} = \Delta E_p - \Delta E_i + E_{subsidence} \tag{6}$$

4. Results

4.1. Marsh elevation

At Quivett Creek and Bass Creek, the restored marsh site had a slightly higher mean elevation than the natural site (10.4 to 3.2 cm difference in mean elevation respectively, although only Quivett Creek had a statistically significant difference, $p < 0.05$) (Fig. S1, Table 1 and Table S2). At Boat

Meadow, Scusset Creek and Stony Brook, the restored marsh site had a significantly lower elevation ($p < 0.05$) and an average elevation difference across the marsh platform between the natural and restored sites of −14.1, −70.6 and − 18.5 cm respectively.

4.2. Marsh pore water and creek levels and salinity

Porewater salinity was lower at the restored marsh site compared to the natural site in Boat Meadow, Quivett Creek, and Stony Brook, while at Scusset Creek salinity was slightly higher and at Bass Creek salinity was markedly higher in the restored site (Fig. S2, Table S2). Salinity within the tidal creek was lower at every site and exhibited greater variability than porewater, with low salinity coincident with low water levels, indicative of upland inputs of terrestrial groundwater discharging through tidal creeks at low tide (Masterson and Walter, n.d.).

Mean water level elevation at the restored and natural well sites were within 1 cm at Bass Creek and Boat Meadow, while Quivett Creek displayed a slightly lower water elevation (2.6 cm) on the restored side (Table 1). Tidal restriction, where the high-water elevations are reduced, was more apparent at Stony Brook and Scusset Creek, where mean water elevations were 9.6 and 20.7 cm lower respectively (7 and 27 % of full tidal height), indicating incomplete hydrologic restoration. When the marsh was not flooded, the mean porewater level relative to the marsh surface, or how much of the marsh soil column is potentially exposed to oxic conditions, was nearly the same at the natural and restored sites at Bass Creek and Quivett Creek, while at Boat Meadow the natural site had a higher porewater level (Fig. S3, Table S2). Scusset Creek and Stony Brook porewater levels were higher at the restored site, resulting in a porewater level increase of 11.4 and 4.7 cm at the restored sites relative to natural sites, respectively, indicating less drainage of the soil column when the marsh is not flooded.

Flooding duration, or the percent of time that the surveyed points on the marsh platform were flooded, was determined for all sites ($n = 53$ to 214 individual elevation measurements, mean = 93). Flooding was similar at both sites in Bass Creek (restored = 60 ± 21 %, natural = 56 ± 16 %) (Fig. S4). At Quivett Creek, the natural site experienced greater flooding than the restored site, likely a result of reduced tidal range even after restoration since marsh elevation was similar (restored = 25 ± 24 %, natural = 53 ± 30 %). At all other sites the restored marsh experienced greater flooding, driven by lower surface elevation at the restored compared to natural sites (Boat Meadow restored = 82 ± 21 %, natural = 15 ± 16 %; Scusset Creek restored = 53 ± 33 %, natural = 10 ± 6 %; Stony Brook restored = 38 ± 35 %, natural = 30 ± 29 %).

4.3. Soil vertical and carbon accretion and carbon source

Age models extended on average 106 years (Fig. 2). Notably, age models for cores from two sites (restored Scusset Creek marsh and restored Quivett Creek marsh) were truncated at 54 and 78 years due to anomalies in the ^{210}Pb profiles. Vertical accretion and carbon burial were determined from the age models and assessed over recent (the year 2000 to the time of core collection) and historic (before the year 2000) time periods. Based on the temporal resolution of the age models and likely impact of the marsh grass root zone, we picked the oldest restoration date among the sites to represent historic and recent accretion. Vertical and carbon accretion varied considerably between sites and across time scales with mean and standard deviation reported (Table S2). Recent (since the year 2000) vertical accretion was greatest at the restored Scusset marsh (13.4 ± 5.6 mm y^{-1}) and lowest at the restored Stony Brook marsh (3.4 ± 0.9 mm y^{-1}) (Fig. 3). Historic vertical accretion rates (prior to 2000) were greatest at Boat Meadow natural marsh (5.2 ± 1.7 mm y^{-1}) and lowest at Bass Creek restored site, which would have been restricted during that time period (1.3 ± 0.4 mm y^{-1}). At every marsh, the natural site had a significantly ($p < 0.05$) higher historic vertical accretion rate than the then impounded site, while recent vertical accretion rates were not significantly different between natural and restored sites at Boat Meadow, Bass Creek, or Stony Brook, but were significantly higher at the natural Quivett Creek site and the restored Scusset Creek site compared to their counterparts (Table S2). Recent vertical accretion rates were significantly higher ($p < 0.05$) than historic rates at Bass Creek restored site, both Boat Meadow sites, the natural site at Quivett Creek, and the restored sites at Scusset Creek and Stony Brook (Table S2).

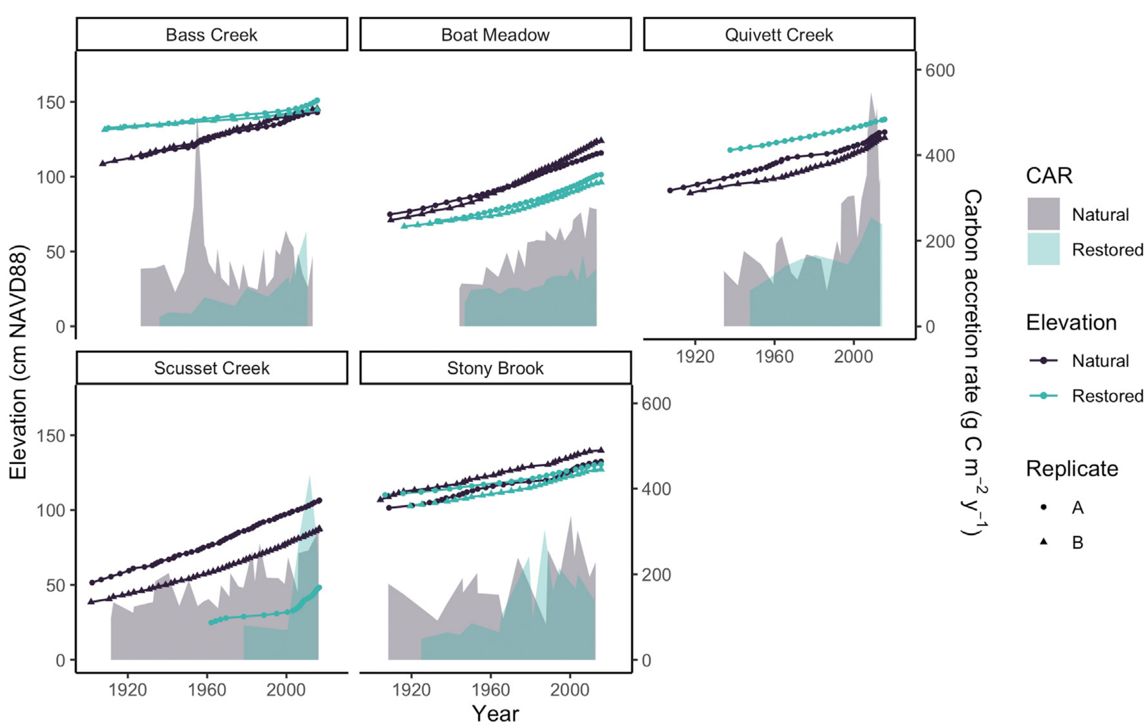

Fig. 2. Elevation trajectories for individual cores (lines) and concurrent carbon accretion rates (CAR, shaded regions) at restored and natural sites over the past 100 years. Restored data is shown in teal and natural marsh in dark grey, with carbon accretion rates in shaded teal and grey. Replicate indicates core A or B.

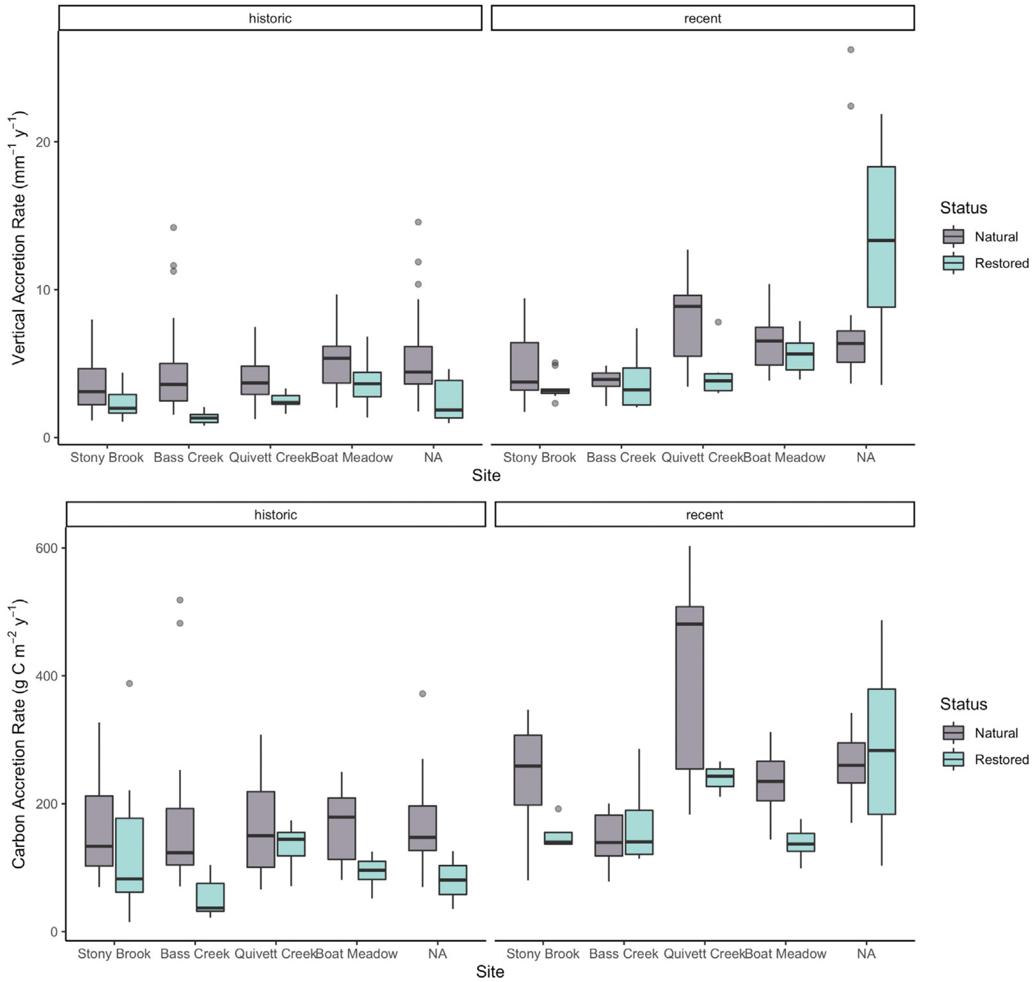

Fig. 3. Box and whisker plots of vertical (top) and carbon (bottom) accretion rates at restored and natural sites during historic (prior to the year 2000, left panel) and recent (since 2000, right panel) periods. These time periods were chosen to represent impounded and restored conditions. Rates at natural sites exceed restored sites except during the recent period at Scusset Creek.

Recent carbon accretion peaked at Quivett Creek natural marsh (391 ± 157 g C m^{-2} y^{-1}) and was lowest at the restored site at Boat Meadow 138 ± 27 g C m^{-2} y^{-1}) (Fig. 3). The lowest carbon accretion rate in the historic time period was at the then-impounded Bass Creek site (52 ± 29 g C m^{-2} y^{-1}), while historic rates at all five natural marsh sites were very similar around 165 ± 100 g C m^{-2} y^{-1}. Historic carbon accretion rates were significantly higher at natural compared to impounded sites at Bass Creek and Boat Meadow, while recent carbon accretion rates were higher at natural compared to restored sites at Boat Meadow and Quivett Creek ($p < 0.05$, Table S2). Recent carbon accretion rates were significantly higher than historic rates at both sites at Bass Creek, Quivett Creek and Scussett Creek ($p < 0.05$, Table S2).

Down-core profiles of carbon isotopes indicate that all restored sites did experience some organic matter inputs from non-salt marsh sources (i.e., terrestrial plants which photosynthesize via the C3 pathway with δ^{13}C between −25 and − 28 ‰, Fig. 4). At the Bass Creek restored site, inputs of salt marsh organic matter are evident at 30 cm, approximately the rooting depth of salt marsh grass, and gradually increase to nearly 100 % salt marsh organic matter in surface sediments. At the restored site in Boat Meadow, organic matter deeper than 50 cm has a salt marsh origin, with inputs from C3 sources evident up to 25 cm, when salt marsh carbon again dominates. Organic matter in one of the natural cores at Boat Meadow C isotopes reached −23 ‰ at 67.5 cm, indicating some input of terrestrial organic matter at this site. At both natural and restored sites in Quivett Creek, C isotopes range from −15 to −25 ‰ below the rooting zone, where salt marsh organic matter dominates, indicating terrestrial inputs in the past. Scusset Creek had terrestrial inputs from 50 to 25 cm, with salt marsh carbon dominating both above and below those depths. The restored site at Stony Brook, the most recently restored site, is mostly dominated by terrestrial carbon, although one core had salt marsh organic matter at depths >70 cm.

4.4. Elevation and soil organic carbon responses to impoundment and restoration

Avoided elevation gain during impoundment, which lasted 115 to 137 years, was between 15 and 37.5 cm (Table 2). Subsidence was evident at only one site, Scusset Creek, where 28.2 cm of the 70.6 cm elevation deficit between the restored and natural marshes could be attributed to subsidence. At the two additional sites with elevation deficits (Stony Brook and Boat Meadow), elevation deficits could be accounted for by reduced accretion at the restored site. However, comparison of the elevation difference between restored and natural marshes at the time of Impoundment and at present indicates that 12.6 to 67.7 cm of elevation capital was lost at each site (Table 2). Therefore, sites that started relatively high in the tidal frame, and remain higher today than the natural sites, showed evidence of lost elevation capital due to impoundment.

Suppressed soil carbon burial, as determined from the difference in carbon burial rates between the impounded and natural marshes integrated over the time of impoundment, ranged from 4125 to 14,125 g C m^{-2} (Table 2). At the one site where subsidence occurred, Scusset Creek, an additional 7615 g C m^{-2} of relict soil organic carbon was lost from the system.

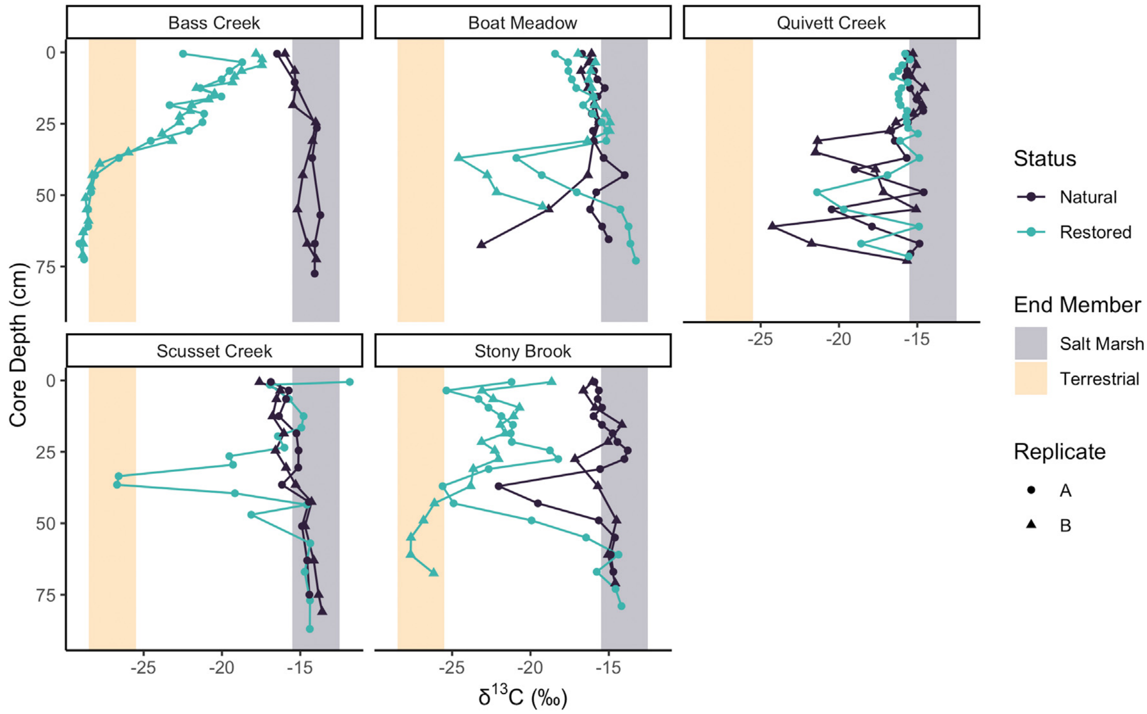

Fig. 4. Depth profiles of $\delta^{13}C$ of organic matter for restored (teal) and natural (dark grey) cores. Salt marsh (C4, grey) and terrestrial (C3, yellow) vegetation end members are indicated by shaded regions. Replicate indicates core A or B.

5. Discussion

5.1. Effectiveness of hydrologic restoration

A multitude of reasons exist for hydrologic restoration of wetlands, including habitat benefits, control of invasive species (particularly *Phragmites* at these sites), remediation of elevation loss (Karberg et al., 2018; Raposa, 2008; Warren et al., 2002) and, more recently, recognized climate benefits (Emery and Fulweiler, 2017; Kroeger et al., 2017; Surgeon Rogers et al., 2018). To evaluate restoration success, common metrics are assessed after remediation of tidal restriction and impoundment, including vegetation shifts, tidal range, and salinity upstream of the former restriction (Cadier et al., 2020). Additional metrics, including GHG cycling and carbon sequestration, are needed to assess the impact of restoration on climate mitigation benefits. After hydrologic restoration, at every site but Scusset Creek, salt marsh vegetation became the dominant species, as evidenced in both the current surface vegetation (Wang et al., 2020) and the isotopic composition of soil organic carbon (Fig. 4). This shift was likely due to renewed tidal exchange and increased seawater input to the marsh. Three restored sites in this study displayed full tidal restoration based on equivalent water elevations in restored and natural sites, while one (Stony Brook) displayed minimally reduced water elevation and one (Scusset Creek) still experienced reduced water elevation compared to the natural analogue. Notably at Scusset Creek, which has the largest area upstream of a tidal restriction (120 ha) in this study, renewed tidal flow resulted in 20 ha of *Spartina* vegetation (including where the samples for this study were collected), while the remainder of the wetland remains dominated by *Phragmites*. While water level elevation at Scusset Creek was not fully restored, tidal range has increased and mean water elevation decreased since impoundment (Fig. S5, U.S. Geological Survey (1996)).

Porewater salinity was lower at three of the restored sites compared to the natural analogues, although average salinity was 17 or greater at all sites (Table S2). Groundwater inputs at the wetland boundary are common and likely are the source of the lower salinity in upstream restored marshes (Masterson and Walter, n.d.). Two sites (Bass and Scusset Creeks) had higher salinity than the other sites, and at these locations the restored marsh was saltier than the natural marsh, potentially due to evapotranspiration and a longer water residence time higher in the estuary. At every site the creek salinity was lower and more variable than porewater salinity, with average salinity <10 at two tidal streams, again likely driven by groundwater becoming the dominant water source at low tide (Table S2). Tidal creek salinity was not a good indicator of porewater salinity, and caution should be used when monitoring wetland salinity in surface waters to infer biogeochemical processes, such as methane production and organic matter decomposition, within the marsh soil platform.

Table 2
Soil organic carbon consequences of impoundment.

Marsh	Avoided carbon burial during impoundment (g C m^{-2}) Eq. (1)	Avoided elevation gain (cm) Eq. (2)	Subsidence (cm) Eq. (4)	Carbon lost via subsidence (g C m^{-2}) Eq. (5)	Elevation capital lost (cm) Eq. (6)
Quivett Creek[a]	4125	17.5	na	na	16.9
Bass Creek[a]	14,125	37.5	na	na	24.5
Stony Brook	4945	15.0	na	na	20.8
Boat Meadow	9727	21.9	na	na	12.6
Scusset Creek	9912	29.5	28.2	7615	67.7

na means subsidence not found at these sites.
[a] Year of impoundment is unknown for Quivett and Bass Creeks. Since the majority of sites were impounded in the late 1800's on Cape Cod, here we use 125 years as the length of impoundment at these sites.

5.2. Resiliency consequences of marsh impoundment

Historically, vertical accretion was suppressed at all impounded sites, with natural marshes accreting 1.3 to 3.0 mm y^{-1} faster than their impounded counterparts, resulting in impounded accretion rates only 30 to 70 % of accretion rates at the natural analogues. This suppression of vertical accretion in the impounded marshes resulted in 17.5 to 37.5 cm of avoided elevation gain (Table 2). Despite accretion rates being lower during impoundment than corresponding natural rates, two marsh sites had higher modern elevations at the restored marsh site (Quivett Creek: 10.4 cm and Bass Creek: 3.2 cm higher than natural analogue). By comparing the natural and impounded marsh elevations at the time of impoundment with their elevations at the time of restoration, we can assess how much the impounded marshes lowered relative to the natural side, i.e. what the loss in elevation capital was (Eq. (6)). While Bass and Quivett Creek restored marshes are still higher than the natural marsh, they lost 16.9 to 24.5 cm of relative elevation capital during impoundment (Table 2). Since these sites sat higher in the tidal frame prior to impoundment than the other sites in this study, it is possible they were at the marsh-terrestrial border prior to tidal restriction. Supporting this are much lighter organic matter $\delta^{13}C$ at depth at Bass Creek, indicative of terrestrial carbon inputs rather than salt marsh inputs prior to impoundment, as shown at the other four sites (Fig. 4). Indeed, these restored marsh sites are likely an example of marsh restoration enabling coastal wetland migration into areas with sufficient elevation capital to successfully promote marsh habitat into the future.

At the three sites with a modern-day elevation deficit in the restored marsh (Boat Meadow: 14.1 cm, Stony Brook: 18.5 cm, and Scusset Creek: 70.6 cm, Table 1), it would take 88, 142 and 282 years respectively for the difference in accretion rates alone to yield the observed elevation difference between the natural and restored marsh surfaces. Both Stony Brook and Boat Meadow have been impounded long enough for suppressed accretion to result in the observed elevation deficit. This suggests that at Scusset Creek additional processes that lead to marsh subsidence contributed to elevation loss that resulted in the observed modern elevation deficit between natural and restored marshes. The 28.2 cm of subsidence accounts for ~50 % of the total elevation capital lost at this site.

Similar suppression of accretion rates and resulting elevation deficits in impounded wetlands have been observed at sites within National Wildlife Refuges managed for bird habitat in Louisiana (Bryant and Chabreck, 1998). There, accretion rates in natural sites were twice that of impounded sites, and elevation deficits were 10–50 cm. At several marshes along Long Island Sound, accretion was suppressed 22 to 50 % at wetland sites behind tide gates, where water drainage was allowed seasonally but incoming tides were kept out year round, compared to sites with natural hydrology (Anisfeld et al., 1999). Overall, impoundment resulted in loss of elevation capital, both from subsidence as well as reduced elevation building potential.

This has important implications for impounded marsh sites which could continue to drop in elevation both relative to natural marshes as well as to mean sea level. In this study, the largest accretion shortfall for impounded marshes would result in 30 cm of lost elevation capital relative to natural marshes over a hundred years. Elevation loss in impounded wetlands may have impacts beyond its current footprint for wetland resilience to sea-level rise since these areas may serve as critical migration corridors for coastal wetlands seaward of such structures. However, this migration capacity may have a limited time frame over which the ecosystem can utilize an area for wetland migration considering the acceleration of sea-level rise rates (Sweet et al., 2022). Finally, lower accretion and subsidence in wetlands behind structures that continue to control hydrology and decouple wetland biogeochemical feedbacks from sea-level rise may result in a transition from vegetated to open water environments, with resultant substantial loss of ecosystem services.

5.3. Resiliency consequences of marsh restoration

After restoration vertical accretion accelerated at all sites, however the magnitude of increase and relative accretion compared to the natural marsh counterpart varied across sites. At three sites, Boat Meadow, Stony Brook and Quivett Creek, vertical accretion rates in the natural marsh continued to outpace the restored marsh by 0.9 to 3.7 mm y^{-1}, indicating these marshes had not regained the full vertical accretion capacity of natural marshes. Platform elevation was a strong control on accretion rates across natural and restored sites (Fig. S6, $r^2 = 0.8$, $p < 0.05$). Accretion rates were nearly identical across restored and natural sites at Bass Creek. At one site, Scusset Creek, vertical accretion rates at the restored marsh were nearly double that of the natural marsh (13.4 ± 5.6 compared to 7.9 ± 6.0 mm y^{-1}). This site is an important case study in hydrologic restoration since the platform was quite low within the tidal frame due to a 70 cm elevation deficit compared to the natural counterpart (Fig. S5). Indeed, prior to restoration, water levels were so high that the platform would have been flooded a large percentage of the time. Restoration at this site was accomplished with a sluice gate system which is larger than the previous culvert and can control water levels as opposed to open culverts at the other sites where water level is a function of culvert size and elevation. After restoration, salinity was higher and water levels were lower, but only a partial restoration of tidal range was achieved in the restored marsh compared to the adjacent marsh (Fig. S5). The rapid accretion rates post restoration at Scusset Creek marsh indicate it is possible to elicit an acceleration in vertical accretion when conditions are optimal for salt marsh vegetation. Importantly, the accretion response was primarily attributed to organic matter deposition, rather than minerals as indicated by a low bulk density of recently accreted material (0.14 g cm^{-3}). The organic and mineral matter mixing model of Morris et al. (2016) predicts that 97 % of the volume of the newly accreted material at this site is from organic matter, with the remaining 3 % attributed to accretion from mineral matter.

A recent meta-analysis of sediment accretion following wetland restoration using the entire suite of Natural Climate Solutions, including vegetation planting, hydrologic restoration, managed retreat, and thin layer deposition, has posited that the success of coastal wetland restoration hinges on sediment availability (Liu et al., 2021). While Liu et al. (2021) did not differentiate the variety of restoration techniques within their analysis, they report that elevations high in the tidal frame had lower sediment accretion, as observed here. Liu et al. (2021) posited that sediment availability was the driver of accretion in restored wetlands, however, the wetland accretion reported here is dominated by organic matter, particularly the largest accretion gains at Scusset Creek, where tidal flooding is through a 2 m sluice gate, potentially limiting sediment transport. Importantly, organic matter contributions to accretion may not be fully captured in current models (e.g. Morris et al., 2002) under conditions that occur during wetland restoration, where rates of peat accumulation are much greater than predicted.

Similar drastic increases in vertical accretion have been reported for other sites after restoration, such as a two to three times increase in accretion at a Long Island Sound, CT, wetland (Anisfeld et al., 1999) and up to four times greater accretion in a restored San Francisco Bay, CA, wetland (Miller et al., 2008). At both of these restoration sites, organic matter dominated vertical accretion in the newly deposited peat. In Tillamook Bay, OR, low elevation restored wetlands had accretion rates double that of nearby low marsh and six times that of high marsh, while restored wetlands at higher elevation had lower accretion rates (Janousek et al., 2021), as observed here, although the role of organic matter was not reported. Restoration of salt ponds in Australia to salt marsh vegetation yielded enhanced carbon storage with time since vegetation establishment (9 years compared to 65) impacting rates more than elevation within the tidal frame or vegetation species (Gulliver et al., 2020). In the restored marshes in CA, CT, and OR, accretion rates were considerably greater than local sea-level rise rates. Drastic elevation gain is possible in restored marshes where platform elevation, water levels, and flooding regime induce a strong accretion response, likely by optimizing elevation for tidal flooding that promotes vegetation growth and organic matter preservation, akin to the vegetation and flooding dynamics observed in wetlands with natural hydrology (Cahoon et al., 2021; Kirwan and Guntenspergen, 2012; Morris et al., 2002). Not all restored marsh sites will meet this criterion, as evidenced by the sharp elevation increase only observed at one of the restored wetlands in this study, and the elevation dependence of post restoration vegetation productivity and vertical accretion shown here and elsewhere (Janousek et al., 2021; van Belzen et al., 2017). In addition, if an impounded marsh has experienced severe elevation loss, hydrologic restoration without additional elevation management actions could result in water levels that exceed vegetation tolerance and do not promote vegetated platform development. Thus, the elevation capital benefits of restoration may vary from minimal to substantial across coastal wetland tidal restorations with marsh elevation, as determined by subsidence and accretion, as well as the restored tidal range and water level, influencing potential benefits. Prior to coastal wetland restoration, consideration of the flooding frequency, which includes water and marsh surface elevation and tidal characteristics, and water elevations, may help identify whether the optimal environmental conditions will occur after hydrology is restored. Given the close relationship between vertical accretion and carbon storage, anticipated climate benefits related to peat formation also need to be evaluated within the context of accretion potential.

5.4. Climate consequences of marsh impoundment and restoration

The historical climate consequences of marsh impoundment reflect the environmental conditions experienced in each wetland after a tidal restriction was put in place and can include sequestered carbon that has been subsequently lost due to organic matter oxidation following drainage or other management action, as well as carbon burial that was avoided due to suppressed carbon accretion rates in impounded wetlands. Impounded wetlands are sites of decompositional disequilibrium, where active management of hydrology results in biogeochemical processes that are not at steady state, i.e. are rapidly changing or not in equilibrium. The resulting subsidence and accretion are likely largely driven by the water table position, which may fluctuate on both short (seasonal) and long (annual to decadal) time scales. Therefore, subsidence associated with loss of long-buried organic matter can co-occur with inputs of new soil organic matter, as seen in the soil profiles at these wetlands, where organic matter sourced from non-wetland species was deposited during impoundment. It is likely that the site that experienced net soil carbon release, Scusset Creek, was drained during a portion of the managed hydrology history, but we cannot assess the relative period of drained versus flooded conditions with the current record.

In the current study, we assess the net impact of impoundment on soil carbon including both suppressed carbon accretion and carbon loss associated with soil subsidence. Here only one wetland experienced soil organic carbon losses via subsidence of 7615 g C m^{-2}, equivalent to ~94 years of historic carbon accumulation. During impoundment, on average carbon burial was suppressed 42 % in impounded wetlands compared to natural wetlands (impounded: 96 ± 33 g C/m^2/y, natural: 165 ± 100 g C m^{-2} y^{-1}, $p \leq 0.001$), resulting in 8567 ± 4090 g m^{-2} of avoided carbon storage, equivalent to 52 years of carbon burial in natural marshes. Combining lost soil organic carbon and avoided carbon burial, these wetlands lost 25 to 106 years of comparable natural wetland carbon storage while they were impounded 115 to 137 years.

Since restoration, carbon storage rates have increased in the restored marsh areas (197 ± 64 g C m^{-2} y^{-1}) but are still 22 % lower on average than natural sites (254 ± 88 g C m^{-2} y^{-1}), although this difference is not significant (p = 0.28). Elevation within the tidal frame, rather than time since restoration, appears to be the most important driver of carbon burial rates in these marshes that were 5 to 14 years post restoration. Additional recovery of carbon burial rates is anticipated for restored wetlands that remain vegetated as they respond to increasing rates of sea-level rise, which results in enhanced carbon burial in coastal wetlands (Gonneea et al., 2019; Rogers et al., 2019).

6. Conclusions

Hydrologic manipulation is a ubiquitous feature of coastal wetlands. Impoundment resulted in suppressed elevation gain and carbon burial, with rates in impounded sites 40 % lower than their natural counterparts, resulting in a loss of elevation capital relative to both natural marsh systems, as well as sea level. During impoundment, subsidence, likely associated with wetland drainage at some point after tidal exchange was restricted, was observed at only one of five sites on Cape Cod, suggesting that while subsidence isn't universal, it can be significant when present. Restoration of hydrology was achieved either through enlargement of culverts or emplacement of sluice gates to control water exchange. Once hydrology was restored, coastal wetlands greatly increased vertical accretion, with elevation within the tidal frame being a strong control. One restored site was able to accrete at twice the rate of natural wetlands, with organic matter the dominant source of elevation gain. Carbon burial rates also increased after restoration, although they did not reach that of the natural wetlands within the 14 years post restoration studied here. Impounded marshes are frequently located landward of natural marshes and, as a result, these sites represent potential migration space of wetlands responding to sea-level rise. Restoration of tidal exchange in these wetlands successfully induced elevation gains. In addition, carbon burial capacity increased after restoration, enhancing the global warming mitigation potential of coastal wetlands.

CRediT authorship contribution statement

Meagan J. Eagle: Conceptualization, Methodology, Formal analysis, Investigation, Writing – original draft, Visualization, Project administration, Funding acquisition. **Kevin D. Kroeger:** Conceptualization, Project administration, Supervision, Writing – review & editing, Funding acquisition. **Amanda C. Spivak:** Conceptualization, Project administration, Methodology, Writing – review & editing, Funding acquisition. **Faming Wang:** Conceptualization, Writing – review & editing. **Jianwu Tang:** Conceptualization, Writing – review & editing, Funding acquisition. **Omar I. Abdul-Aziz:** Conceptualization, Writing – review & editing, Funding acquisition. **Khandker S. Ishtiaq:** Conceptualization, Writing – review & editing. **Jennifer O'Keefe Suttles:** Methodology, Formal analysis, Investigation, Data curation, Writing – review & editing. **Adrian G. Mann:** Formal analysis, Investigation, Data curation, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.157682.

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