Hydrological Restoration of Salt Marshes

A report prepared for the Coastal Resources Division, Georgia Department of Natural Resources

Margaret Myszewski and Merryl Alber

Georgia Coastal Resources Council

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Hydrological Restoration of Salt Marshes

Background

Salt marshes are intertidal habitats that are regularly flooded with salt water. They are characterized by salt-tolerant vegetation and are typically located in transitional zones between estuaries and upland areas. Salt marshes provide key ecosystem services including shoreline protection, enhanced biological productivity, increased biodiversity, improved water quality, and carbon sequestration (Craft 2022; EPA 2020; Hopkinson et al. 2019; Lang et al. 2024). These services, along with facilitating activities such as commercial and recreational fishing, tourism, and birdwatching, generate substantial economic benefits (Table 1). For example, saltwater recreational fishing, much of which depends on salt marshes as nursery grounds, contributed $155 million to Georgia's GDP in 2022, supported 3,200 local jobs, and added significant sales and labor income to coastal communities (Frimpong 2023).

Table 1. Values of ecosystem services of tidal marshes

Ecosystem service	Examples of human benefits	Average value (Adj. 2025 $ ha^{-1} yr^{-1})
Disturbance regulation	Storm protection and shoreline protection	$4,432
Waste treatment	Nutrient removal and transformation	$15,011
Habitat/refugia	Fish and shrimp nurseries	$440
Food production	Fishing, hunting, gathering, aquaculture	$660
Recreation	Hunting, fishing, birdwatching	$1,838
Total		$22,595

Dollar values were adjusted for inflation from original data, presented in 2007 dollars (Costanza et al. 1997). The adjustment was done with the U.S. Department of Labor Inflation Calculator, which uses the Consumer Price Index to correct values through time. (Adapted from Gedan, Silliman, and Bertness 2009)

Despite their importance, salt marshes and other coastal wetlands are threatened. Results from the most recent U.S. Fish and Wildlife Service wetlands survey (2009-2019) confirm the continuation of a nationwide, 70-year trend of conversion of vegetated saltwater wetlands (including salt marshes) to non-vegetated saltwater wetlands (i.e., mud flats, beaches, shorelines) (Lang et al. 2024). Although Georgia marshes are relatively high in the tidal frame (Langston et al. 2021) and have not experienced large losses over the past decades (Burns et al. 2020), a recent study identified areas of high vulnerability along the Georgia coast (Runion et al. 2025). It is therefore important to evaluate potential opportunities for salt marsh restoration.

Historically, many salt marshes were cut off from tidal flooding due to hydrologic restrictions. These were installed for flood protection, reclamation of land for agricultural and other uses, and/or for mosquito control. Although these interventions were implemented for practical purposes, they can significantly reshape salt marsh ecosystems, potentially leading to long-term environmental consequences, as described in Table 2.

Table 2: Potential effects of tidal restrictions on upstream wetlands and resources.

Resource/ Function Affected	Proximate Cause	Potential Upstream Effects
Vegetation	Reduced salinity	Invasion of invasive species alter vegetative community structure
Water Quality	Reduced tidal flushing	Decreased ability to remove pollutants; promote conditions that favor harmful bacteria.
Salt Marsh Specialist Bird Species	Vegetation change; loss of tidal wetlands	Loss of breeding habitat
Fish and Shellfish	Reduced tidal inundation; increased water velocity	Limited habitat availability and restriction of movements between upstream and estuarine habitats
Resiliency to Storm and Flood Events	Loss of tidal wetlands	Loss of wave attenuating and shoreline stabilizing effects of coastal wetlands.
Sedimentation and Subsidence	Reduced tidal inundation	Reduced vertical sediment accretion rate and marsh elevations

(Adapted from EPA 2020)

These barriers alter flooding patterns, which can shrink salt marsh areas, impede pollutant filtration, promote invasive species, and degrade habitat for salt marsh-dependent wildlife. Additionally, tidal restrictions may reduce salinity, and in some cases convert salt marshes into freshwater systems or open water, fundamentally changing habitat conditions (Purinton and Mountain 1996). The effects of tidal restrictions on salt marshes are discussed in further detail in Part 1 below.

Growing concern for salt marsh loss along with the accompanying ecological and economic services they provide has created more interest in wetland restoration. Habitat restoration can be defined as assisting the recovery of degraded ecosystems to restore biological structure and function (Gann et al. 2019). In salt marshes this can often be achieved by hydrologic restoration, which involves removing barriers that impede water flow or installing structures that facilitate hydrological connection (Gann et al. 2019).

Once tidal flooding is re-established by hydrological restoration activities, the salt marsh can often recover without further intervention (Craft 2022). However, measurements need to be taken to ensure the tidal regime and level of flooding (relative to the salt marsh surface) is appropriate to support native vegetation and access for macroinvertebrates, nekton, and other fauna (NASEM 2017). This involves consistent, long-term tracking of multiple indicators from the restored salt marsh and comparison with those from appropriate reference salt marshes to determine the success of the restoration project (Zhao et al. 2015). Although there is currently no uniform definition of successful salt marsh hydrological restoration, the Society of Ecological Restoration defines full hydrological recovery as “the state or condition whereby, following restoration, all key ecosystem attributes closely resemble those of the reference model” (Gann et al. 2019). These key attributes include appropriate species composition, intact community structure, optimal physical conditions, robust ecosystem functions, and proper tidal exchanges (Gann et al. 2019).

This report highlights the importance of hydrological connectivity in sustaining salt marsh ecosystems and identifies critical factors for successful hydrological restoration, focusing primarily on projects on the US east coast. The report is divided into three sections. Part One is an overview of tidal hydrology, the effects of tidal restrictions, and approaches for hydrological restoration. Part Two summarizes scientific studies that examined the effects of hydrological restoration on the indicators most frequently used in restoration projects to assess salt marsh functionality. Part Three provides information on data gaps and further research needs.

Part 1. Hydrological Restrictions in Salt Marshes

Inundation

Hydrology is a key factor shaping ecological processes in salt marshes. Inundation patterns define the vertical boundaries of the intertidal zone, which lies between subtidal (always inundated) and upland areas (only inundated by extreme events such as hurricanes), with different organisms dominating areas of low versus high elevation (Fig. 1). Tidal flooding also facilitates the exchange of sediment and other materials between intertidal and estuarine areas. The extent and frequency of tidal inundation is therefore a major structuring variable for wetland communities, affecting salinity, soil moisture, and redox potential (Crain, Gedan, and Dionne 2009). The degree of hydrological connectivity strongly influences species richness, vegetation patterns, and productivity by regulating microtopography and habitat conditions (Gelwick et al. 2001). Highly connected tidal zones support greater exchanges of water, nutrients, sediments, and organisms, enhancing biodiversity and ecological resilience, thus highlighting the critical role of connectivity in wetland function (Chen et al. 2023).

Fig. 1. Zonation of saline tidal marsh vegetation in relation to tide levels. (Craft 2022)

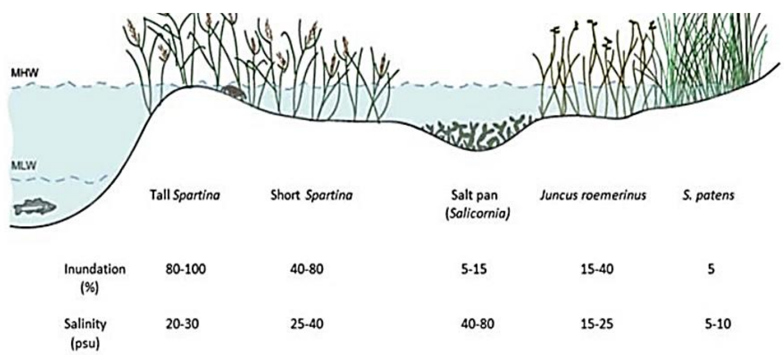

Fig 2. Culvert installed following restoration of a salt marsh at Jacob's Point, Narragansett Bay, Rhode Island. **(Photo: Save the Bay)**

Reintroducing hydrological connectivity typically involves removing or modifying tidal restrictions to replicate the natural tidal regimes found in undisturbed salt marshes (Twomey et al. 2024). Below are descriptions of the most common types of tidal restrictions, including culverts, dikes, and ditches, along with remediation approaches.

Culverts

A culvert is a structure that directs water under roads, railways, or other obstacles (Fig. 2). Originally built with large granite slabs, modern culverts are usually cylindrical and made of corrugated metal. In coastal regions, culverts can be used to allow tidal water to continue to flow into and out of salt marshes (EPA 2020).

Effects on salt marshes

Reduced tidal flooding in salt marshes due to poorly installed or undersized culverts causes multiple harmful impacts. If salt water flow is blocked then salt and flood-tolerant species can be displaced by plants typically found in fresher and drier conditions (Roman and Burdick 2012). Culverts can also result in lower sediment delivery, declining marsh surface elevation, loss of native salt marsh vegetation, and invasion by non-native or upland plant species (Eberhardt et al. 2011). Studies at a tidally restricted salt marsh creek in New England showed that an undersized culvert increased water velocity, which significantly reduced mummichog passage between upstream and downstream areas and resulted in separated (although similar) nekton populations (Eberhardt et al. 2011). Additionally, restricted tidal flow can disrupt or eliminate the export of organic matter from marshes (Purinton and Mountain 1996, Fig. 3). Correll et al. (2017) used a large dataset of bird surveys in coastal marshes from Maine to Virginia spanning 18 years and found that the best explanation for avian population declines was the presence of culverts. Birds classified as salt marsh specialists maintained their populations in marshes with no road crossings but declined in marshes restricted by downstream road crossings. The authors speculated that this outcome may have resulted from culverts limiting sediment accretion by reducing sediment supply to the marsh, leading to habitat loss for specialist species such as the Saltmarsh Sparrow.

Fig 3. Culvert sited too high in the bank to allow for full tidal range. (Photo: Cape Cod Commission, 2001)

Remediation

The negative impacts of culverts on salt marsh hydrology can be reduced or eliminated by replacing, removing, or repairing them, especially when older culverts have failed due to breakage or being undersized. However, culverts must be maintained or they can become clogged with debris, oysters, or other organisms, and if they are undersized they can continue to restrict tidal flow. Additionally, water flow velocities through culverts must be monitored to ensure they are suitable for fish passage (Craig et al. 2010).

Fig 4. Historic rice impoundment, Beaufort County, South Carolina (Photo: Henry De Saussure Copeland)

Dikes

Dikes are artificial barriers, usually made of dirt, that are built to limit or prevent the flow of water for land management. Dikes are often used for flood control purposes where they are built along rivers or coastal shorelines to prevent floodwaters from flowing onto land. Dikes can also be used to retain water through the creation of impoundments where water levels, water exchange, and salinity are artificially controlled using tide or flap gates

(Kennish 2001, Fig. 4). The degree of water exchange between the impounded marsh and a larger body of water depends on the purpose of the impoundment. Waterfowl impoundments, for example, generally limit water exchange to a greater degree than those managed for fisheries (Carswell et al. 2015; Robinson and Jennings 2014). The Duck Pond, which is located on the north end of Sapelo Island, Georgia, is separated by a dike into a northern and a southern pond. The northern pond is influenced by tidal waters resulting in a salt marsh plant community, while the southern pond, primarily filled by rainwater and freshwater creeks, was managed as a waterfowl impoundment where water exchange was controlled through a flash board riser (Woods and Gordy 2015).

Dikes can also be used to drain water from a salt marsh to create land for agricultural and residential purposes. Once the dike is constructed, water from the newly isolated marsh area is drained creating farmland that can be used for crop fields, pastureland, or development (Gedan et al. 2009, Fig. 5). In the northeastern states, early Dutch settlers installed systems of dikes in salt marshes that protected the land from saltwater inflow and allowed easy access to salt hay meadows (Butler and Weis 2009). Similarly, in 1948, Howard Coffin diked an area of salt marsh on Sapelo Island for use as a pasture for his dairy herd. However, the site converted to an unvegetated salt pan instead of the desired pastureland, and in 1956 the dike was breached by the State of Georgia's Marine Institute, allowing reintroduction of tidal flooding (Craft 2001; Craft 2023).

Fig 5. Example of dike built to drain a salt marsh for farmland **(Photo: Isen Sno)**

Effects on salt marshes

Dikes disrupt salt marsh systems by interfering with natural tidal exchange (Butler and Weis 2009). When used to form impoundments, dikes are also linked to lower dissolved oxygen, reduced turbidity, and blocked species movement (Carswell et al. 2015). For example, over 16,200 ha of salt marshes along the Indian River Lagoon in Florida, were restricted using dikes by the early 1970s as a form of mosquito control. Isolation from the Lagoon cut off aquatic access by transient estuarine species that used the salt marshes for nurseries and feeding. One study of these marshes found that the number of fish species present dropped from 16 to 5 after diking. Wetland vegetation within some diked marshes was eliminated while others developed into freshwater systems (Brockmeyer et al. 1997).

Dikes also reduce inorganic sediment input, lowering sediment accretion (Kennish 2001). Soils in diked and drained salt marshes can become aerated, increasing both organic matter decomposition and soil compaction, often lowering salt marsh elevations below mean sea level (Portnoy and Giblin 1999). Studies in the Chenier Plain of southwestern Louisiana in 1994, showed that diked salt marshes had surface elevations 20–30 cm lower than natural salt marshes, due to isolation from sediment delivery, storm effects, and oxidation (Bryant and Chabreck 1998). Fish community composition can also be affected by dikes. A study in the Nemours Plantation on the Combahee River, South Carolina, compared the diversity and abundance of fishes between diked and impounded salt marshes managed for waterfowl with those managed for recreational fishing. They found that, while both management types supported similar numbers of species, waterfowl impoundments supported greater proportional abundance of dominant species. The authors concluded that, by eliminating frequent tidal exchange, waterfowl impoundments may create an environment that ecologically mimics the marsh interior, thereby favoring marsh-interior species. In contrast, the daily tidal circulation in fish impoundments supports the habitat requirements of marsh-edge and marsh-subtidal specialists (Carswell et al. 2015).

Similarly, Robinson and Jennings (2014) compared resident fish communities in two waterfowl impoundments in Beaufort County, South Carolina to observations from fish assemblages sampled at 12 sites in North Carolina (Cape Fear estuary) and South Carolina (North Inlet estuary). Results showed that the fish assemblages in the impoundments were dissimilar to those in the tidal creeks and that tidal influence was the factor responsible for these differences. They speculated that the lack of tidal connectivity between the impoundments and the salt marsh for most of the year limited the number of species that could colonize the impoundments to those available during the fall flooding period.

Remediation

The hydrological reconnection of salt marsh habitat through breaching or removal of dikes can remedy the negative ecological impacts caused by resulting hydrological isolation. Although long-term maintenance requirements are much lower than with culvert replacement, the size of the dike breach must be adequate to prevent scour, and the design should accommodate for potential scour and sedimentation (Craig et al. 2010).

Dike breaches have also been shown to be successful when accompanied by channel restoration. For example, Able et al. (2000) compared fish response in a restored former salt hay farm to a natural marsh in Lower Delaware Bay. Restoration of the site consisted of six dike breaches and construction of 5,500 m of channels and creeks inside the diked salt marsh area that ranged from 700 to 1,300 m in length. Three years following restoration there were major improvements in species richness, composition and abundance of fishes (see Part 2).

Fig *6*. Example of Rotational Impoundment Management.
(https://www.irlspecies.org/misc/impoundments.php)

The negative effects of dikes can also be managed by the installation of water control structures such as tide gates or flap gates, which may be used where strict management of water levels is required (i.e., flood and mosquito control) (Craig et al. 2010). An example of the use of tide gates for mosquito management is Rotational Impoundment Management (RIM), a mosquito control strategy that seasonally manages water levels in diked salt marshes to prevent mosquito breeding (Fig. 6). During the summer breeding season, impounded areas are flooded to the minimum levels needed to prevent egg laying by salt marsh mosquitoes, while in the winter and spring, culverts allow water exchange with the surrounding estuary. In Florida's Indian River Lagoon, RIM was implemented using four culverts to reconnect diked salt marshes: two with flap gates to hold tidal water while still allowing organism movement, and two with flashboard risers to maintain adequate flooding without damaging vegetation (Brockmeyer et al. 1997).

Ditches

A ditch is a small to moderate trench created to channel water. A ditch can be used to drain water from low-lying areas, or to channel water from a more distant source for plant irrigation. In the southeast, the earliest ditches were constructed to drain freshwater wetlands to facilitate rice and cotton planting. Later ditches were dug to allow for road construction and maintenance (Heynen and Hardy, 2023). Historically, ditches have been used for agricultural and mosquito control purposes. In New England, agricultural ditching dates from the 17th century when colonists used the salt marsh as a source of food and bedding for livestock (Burdick et al. 2020). Rice farming in 18th and 19th century South Carolina and Georgia used a system of drainage ditches to divide inland fields along tidal freshwater rivers and tributaries into small plots and canals which provided irrigation (Fig. 7). Floodgates, located in the freshwater portion of the tidal streams, controlled the flow of water into the irrigation ditch. The gates were placed so that water flowed into the ditches at high tide and was shut off at ebb tide. The gates could be opened at low tide to drain the fields (Adair and Engler 1955). Beginning in 1904, intensive grid ditching of salt marshes was used for mosquito control on the eastern seaboard (Fig. 8). This method of ditching involves the digging of ditches at regular intervals on salt marsh surfaces to remove standing water and control mosquito populations by draining off pooled waters, allowing fish to feed on mosquito larvae during high tides (Kennish 2001; Gedan et al. 2009).

Fig 7. Ditching in the Mulberry rice plantation, Moncks Corner, S.C.
(Photo: Photo Library of Congress)

These ditch networks are still visible today, as is evident on Sapelo Island which has an extensive network of drainage ditches constructed over a period of more than 100 years, starting in the early 1800s and continuing into the 1940s and '50s (Heynen and Hardy 2023). Cotton (2004) found that drainage patterns in a former 18th-century rice plantation located along the Ogeechee River remained highly altered 50 and 100 years after restoration and did not exhibit substantial changes over time.

Fig 8. Historic mosquito ditches at Assateague Island National Seashore
(Photo: National Park Service)

Effects on salt marshes

Ditching disrupts salt marsh hydrology by removing soil and creating artificial levees that disrupt natural water flow (Besterman et al. 2022). Ditches also alter tidal inundation and drainage patterns, impacting critical processes like plant growth, decomposition, and sediment dynamics (Burdick et al. 2020). Spoil levees often block natural tidal flow, reducing sediment delivery and marsh accretion in addition to causing saline intrusion into freshwater areas (Kennish 2001). In regions where marsh peat accumulates (e.g., New England), mosquito ditches can drain peat several meters from the marsh edges, depending on depth and tidal elevation (Crain et al. 2009).

Historic ditch networks can have present day impacts. Little Saint Simons Island's salt marshes were historically modified in the 1930s by installing parallel grid ditches to enhance larvivorous fish habitat, but this alteration has since impeded tidal flushing, increased sediment deposition, and caused dieback of species such as *Spartina alterniflora* during droughts (Cohen-Kivett 2020). Likewise, the ditches on Sapelo Island are currently in a state of disrepair and are increasing flood risk for the Island's inhabitants by connecting otherwise disconnected low-lying areas, allowing high tide waters to migrate further inland than they would naturally (Heynen and Hardy 2023).

Remediation

Fig 9. Construction of ditch plug for wetland restoration. (Photo: USFWS)

A ditch plug is a structure, often made of earthen fill, placed within a ditch to block or redirect the flow of water (Kennish 2001). Ditch-plug creation involves excavating marsh soil from an upstream portion of the ditch and using it to plug the seaward end (Fig. 9). Ditch plugging is used to restore tidal hydrology in ditched and drained salt marshes for the purpose of slowing or blocking water flow in order to restore natural marsh function by increasing water retention, improving habitat quality, and enhancing sediment and nutrient retention. When used to remediate salt marshes ditched for mosquito control in Maine, plugging increased surface water habitat for larvivorous fish to control mosquitoes while also providing habitat for wading birds and waterfowl (Vincent et al. 2014).

Fig 10. A hand-dug runnel drains excess water to restore tidal hydrology. (Besterman et al. (2021)

Runnel construction is a ditch remediation method being used in New England to drain artificial water pools resulting from poorly maintained ditches that were originally dug for agricultural and mosquito control purposes (Delgado and Pau, 2024). Although such ponds are rare in Georgia's salt marshes, they are common in the northeast (Burns et al. 2020). Runnels are shallow (typically under 30 cm wide and deep) and are made by hand or with low-impact equipment to drain surface water while maintaining root zone moisture (Besterman et al. 2022, Fig. 10). Although runnels do not directly address the underlying causes of artificial pooling, and their long-term effectiveness is uncertain, they can play a useful role in re-establishing hydrologic connection to allow regular tidal flooding and support the revegetation of shallow waterlogged pools (McKown et al. 2024).

Part 2. Evaluation of Hydrological Restoration

Hydrological restoration

Salt marsh hydrological restoration commonly focuses on removing tidal restrictions to reestablish natural tidal exchange, with the goal of restoring their structure and function (Neckles et al. 2002). Key objectives of such projects include restoring native vegetation to the salt marsh, improving fish habitat connectivity, sustaining salt marsh platform elevation, and enhancing nutrient storage in the soil (NASEM 2017). Ecological responses to salt marsh restoration are highly site-specific and strongly influenced by pre-restoration or tide-restricted conditions (Raposa 2008). This underscores the importance of monitoring the biological and/or physical indicators that will most effectively measure the extent of salt marsh recovery over time.

Hydrology, soil and sediments, vegetation, nekton, and birds are often used as indicators of salt marsh tidal restoration success (Table 3). These indicators provide essential information on whether a restoration project is functioning as intended and whether its goals and objectives are being achieved. They also supply valuable data that can guide restoration management and improve the design of future restoration efforts (NASEM 2017). According to a review of 23 salt marsh restoration projects, the most frequently assessed indicators measured ecosystem function, including species growth and productivity, survivorship, habitat function, physical attributes, biological threats, and physico-chemical variables (Bayraktarov, et al. 2020). NOAA has developed standardized monitoring indicators that the agency requires recipients of NOAA Restoration Center Grants to use. Monitoring indicators for hydrological restoration projects (e.g., projects that restore water elevations and flows) include measurements of land elevation and water level, whereas monitoring indicators for salt marsh restoration include tidal inundation, percent native vegetation cover and survival, and presence of invasive species (NOAA 2024).

Table 3. Common indicators used to evaluate ecosystem development of restored salt marshes (Adapted from Craft 2022).

Vegetation	Plant cover
	Stem height & density
	Above/belowground biomass
	Number of species
	Species diversity
Fauna	Epifauna species density/diversity/number
	Trophic structure
	Nekton species density/diversity/number
	Avian species density/diversity/number
Soils	Bulk density
	Organic matter content
	Total nitrogen
	Accretion and sedimentation
Porewater	Salinity
	Conductivity
	Sulfide

The sections below describe the biological, chemical/ physical indicators commonly used to determine the success of hydrological restoration on salt marshes along with results from relevant studies.

Biological indicators

The presence of native salt marsh vegetation and wildlife is a significant indicator of a functional ecosystem, and as such, is frequently measured to determine restoration success.

Vegetation

Vegetation change is one of the most commonly used indicators in wetland restoration because the response time is often rapid, changes are easily tracked, and variations in vegetation are associated with everything from variations and migrations in microbial and invertebrate communities to soil conditions to sediment accumulation.

<u>Culvert removal/replacement</u>

- In Massachusetts, Buchsbaum et al. (2006) monitored the change in salt marsh vegetation after a tidal restoration project replaced an undersized culvert with a larger one. Four marsh grasses, *Phragmites*, *Typha*, *Spartina sempervirens*, and *Spartina alterniflora* accounted for 85% of the compositional differences between pre- and post-restoration conditions. The first three species, which all have low salt tolerance, declined after restoration (*Phragmites* by 18% in one year and *Typha* by 20% over three years), while *S. alterniflora* increased exponentially into the vacated areas. Although *Phragmites* and *Typha* experienced steep initial declines, their abundances stabilized after 2-3 years.

- Eagle, et al. (2022), studied five salt marshes in Cape Cod, Massachusetts where tidal exchange was restored by removal or enlargement of undersized culverts between 5 and 14 years prior to sample collection in 2015. At the time of collection, vegetation at the study sites consisted of *Spartina* and other wetland species, including *Distichlis spicata* and *Juncus gerardi*. Following restoration, *Spartina* became the dominant species at 4 of the 5 study sites, likely due to renewed tidal exchange and increased seawater input to the marsh. The fifth site, a salt marsh with the largest area of tidal restriction (120 ha) before culvert removal, supported 20 ha of *Spartina* vegetation, while the rest was dominated by *Phragmites*. Although the site was not fully restored 14 years following culvert removal, the tidal range had increased.

<u>Rotational Impoundment Management</u>

- Brockmeyer et al. (1997) conducted a review of research on the effectiveness of Rotational Impoundment Management in tidally restricted salt marshes along Florida’s Indian River Lagoon. They found that vegetation recovery can be rapid once tidal exchange is restored. For example, following the installation of 13 culverts to reconnect a 1,215-ha of diked salt marsh in the Merritt Island National Wildlife Refuge, salt-tolerant vegetation cover increased by over 1,000% and freshwater species decreased by 74% within three years. Vegetation recovery was not always consistent, however. At a salt marsh site that had been tidally restricted for three years, opening culverts to water exchange led to immediate recovery of some salt-tolerant plants, but after seven years, total plant cover remained far below the original 75%.

<u>Dike breaching by flap gate removal</u>

- From 1999 to 2001, Cotton (2004) monitored vegetation recovery at the Tucker mitigation site on the Ogeechee River, Georgia to assess the effects of flap gate removal. The Tucker site is located on a former 18th-century rice plantation and was historically modified with dikes and flapgates for rice farming and subsequently for waterfowl management and hunting. Pre-restoration in 1996, only two flap gates remained, between 1996 and 1999, two natural dike breaches occurred, and in 1999 the remaining flap gates were removed to restore hydrologic connectivity. A nearby marsh that was also part of the original plantation but had naturally reverted to a functional brackish marsh was used as a reference.

  Sampling occurred at two locations: Site B, near a natural dike breach, and Site R, close to the Ogeechee River. Prior to restoration, both sample sites had about 85% vegetation cover and supported typical brackish marsh species. Site B was dominated by *Juncus effusus* and *Typha* species, while Site R primarily contained *Spartina* species. Post-restoration, the sample sites had only 8.5% vegetation cover, consisting solely of *Zizaniopsis miliacea*. By October 2001, two years after flap gate removal, vegetation cover in the impoundments increased to approximately 20%. New species, including *Spartina alterniflora*, *Spartina cynosuroides*, and *Amaranthus cannabinus*, began to establish, with greater recovery observed at Site B, likely due to better tidal flushing. *S. alterniflora* also appeared in the reference marsh, possibly in response to increased salinity from regional drought conditions.

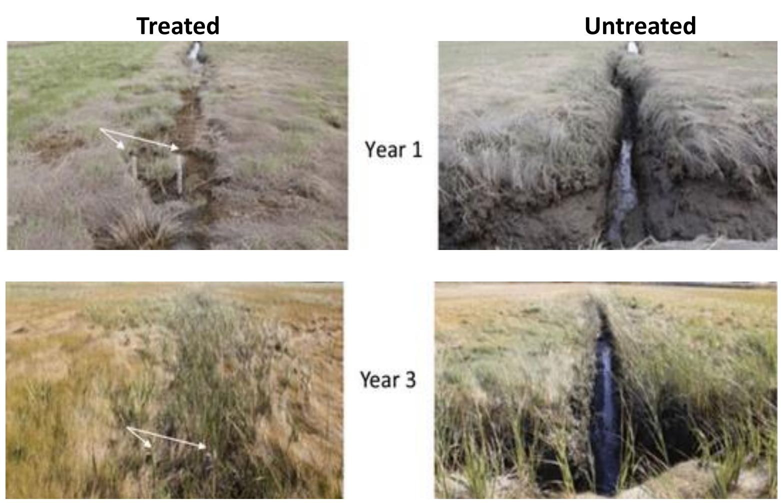

Fig 11. Comparison of treated (with braided hay placement) and untreated ditches. Upper row: immediately following hay treatment. Lower row: three years later, showing recolonization of marsh grasses. (Adapted from Burdick et al. 2020)

<u>Ditch remediation</u>

- In a study by Burdick et al. (2020), following placement of a 15–20 cm layer of braided mowed hay onto nine ditched New England salt marshes (2015-2017), only *Spartina alterniflora* colonized the ditch centers during the following three years. (Fig. 11) In treated plots, vegetative cover was approximately 8% greater than in untreated ditched reference plots in 2015 and increased to about 20% greater two years later. Average stem density in treated ditches increased from 20 shoots/m^2 in 2015 to over 60 shoots/m^2 by 2016, whereas no significant changes were observed in the control ditches. These results indicate that the salt hay treatment was able to stimulate recolonization of native vegetation.

Ditch plugging

- A study by Adamowicz and Roman (2002) assessed how plugging historic drainage ditches influenced three salt marshes at the Rachel Carson National Wildlife Refuge in Maine. At two marshes, plugs were installed in the spring of 2000 and monitored in 1999 (pre-plug) and 2000–2001 (post-plug); the third marsh was plugged in 1998 and monitored in 1999–2000 without before-plug data. In all plugged sites, water-table levels rose closer to the marsh surface and the extent of standing water upstream of plugs increased markedly. This hydrologic change drove a vegetation shift at two sites from high-marsh species such as *Spartina patens* to more flood-tolerant *Spartina alterniflora*, thus indicating that wetter substrates favor low-marsh communities. The authors concluded that ditch plugging effectively re-wet marsh platforms and altered vegetation composition, but they cautioned that wetter conditions may reduce marsh grass productivity and resilience to sea-level rise. Consequently, they recommended that ditch plugging be treated as an experimental restoration technique, with at least a decade of continued monitoring necessary to fully understand long-term marsh development and ecosystem function.
- Using three barrier beach salt marshes along the Gulf of Maine, Vincent et al. (2014) examined the effects of ditching and ditch-plugging on plant characteristics by comparing them to natural creek and pool habitats. Results indicated that plant characteristics were significantly different between ditch-plug and natural pools. Species richness, diversity, and biomass were significantly lower in ditch-plug habitat compared with all other habitats, and plant cover averaged only 30% in habitat adjacent to ditch-plugs, which was significantly lower than the natural reference habitat.

Runnel installation

- At Winnapaug, Rhode Island, a back-barrier salt marsh degraded by 1930s grid-ditching and peat spoil impoundments, Besterman et al. (2022) evaluated the effects of runnel installations that took place between 2012 and 2019 to address surface water cover and platform loss. Before the creation of runnels, the salt marsh was dominated by *Spartina alterniflora* (57%). Less salt-tolerant, high-marsh species also present included *Distichlis spicata* (26%), *Spartina patens* (18%), and *Juncus gerardii* (2.7%). Following runnel construction, the flood-tolerant pioneer *Salicornia depressa* rose to 73% cover by 2014 but then declined to 4.3% by 2019 as less flood-tolerant high-marsh species recovered. By 2019, *Distichlis* cover rose to 42%, *Spartina patens* to 24%, and *Juncus* to 3.8%, while *Spartina alterniflora* remained dominant, increasing to 68%. These results suggest that runnels support short-term recovery of both flood- and drought-tolerant marsh vegetation.

- McKown et al. (2024) studied runnels in 17 high salt marshes across Maine, Massachusetts, and Rhode Island, where abandoned agricultural ditches had led to pool formation and habitat loss. They found that runnels reversed the expansion of waterlogged areas, resulting in annual vegetated gains of 1.55% and a total net gain of 2.08 ha of salt marsh surface post-restoration, while nearby reference marshes continued to decrease.
- In 2022, Sullivan et al. (2025) examined 11 salt marsh sites in Massachusetts and Rhode Island to evaluate the effectiveness of installing runnels to drain artificial pools formed by historical ditching. Each salt marsh site included three treatment types: impounded (areas with vegetation loss and shallow standing water due to ditching, runneled (areas where runnels were installed 2–9 years prior), and reference (healthy vegetation). Compared to reference areas, impounded plots had 65.8% more bare area and 52.0% less vegetative cover. Runneled plots fared better, but they still had 45.7% more bare areas and 37.8% less vegetative cover compared to reference areas. In total, only six of the 11 runneled plot sites experienced vegetation recovery, suggesting that runnel effectiveness is likely dependent on site-specific characteristics.

Benthic macroinvertebrates

In estuarine salt marshes, benthic invertebrates are key indicators of ecosystem health, secondary production, and food web support after restoration or creation.

<u>Dike removal</u>

- Rubin et al. (2024) measured invertebrate responses to large-scale dike removal in the Nisqually River Delta, Washington. Tidal inundation was restored to 364 ha of former salt marsh habitat through phased dike removals starting in 1996. Sampling from 1 year before to 3 years after dike removal (2009–2012) showed that the recently restored salt marsh experienced a 15-fold increase in macroinvertebrate abundance and a 12-fold increase in biomass. Key taxa began colonizing in 2010, with peak abundance reached in 2012. However, while species composition recovered quickly within a year of restoration, density and biomass did not approach full recovery over the 3 years of the study.

Nekton

Nekton are free-swimming animals in aquatic ecosystems that play important roles in energy transfer, connecting benthic habitats, the water column, and terrestrial food webs through predation by birds and mammals.

<u>Culvert replacement</u>

- To study the effects of culvert replacement on nekton communities, Raposa (2008) collected data on nekton communities in Potter Pond, a 2.3-ha estuarine pond bordered by a narrow fringe of salt marsh vegetation located within the Narragansett Bay National Estuarine Research Reserve in Rhode Island. Prior to restoration, the pond’s connection to Narragansett Bay was severely restricted by two crushed concrete culverts and a third crushed plastic culvert. In April 2003, all the damaged culverts were replaced to restore tidal exchange between Potter Pond and Narragansett Bay. Nekton species were sampled in 2000 during tidal restriction, and in 2003 and 2004 following tidal reconnection. Before restoration, only 10 species of nekton were found in Potter Pond, and only a few of these were abundant. In contrast, a nearby reference salt marsh supported 14 nekton species, including several common salt marsh species that were absent from Potter Pond. Although species richness in Potter Pond generally increased each year after restoration, these changes were not statistically significant between year one and year two, and similar changes in species richness were also observed in the reference marsh. Notably, despite improved marsh access and better water quality following restoration, overall nekton density in Potter Pond remained significantly lower in both 2003 and 2004 compared to 2000, whereas nekton density in the reference marsh showed no significant change.

- Buchsbaum et al. (2006) observed that tidal restoration alters nekton communities in a species-specific manner. After tidal exchange increased following replacement of an undersized culvert with a larger one, nekton abundance in the recovering marsh shifted to more closely resemble that of the reference marsh, which also saw increases in certain species such as mummichog and sand shrimp. These changes differed across species: while some maintained similar relative abundances, others declined, disappeared, or exhibited contrasting trends between the restored and downstream marshes.

<u>Tidal restoration of diked areas</u>

- An analysis by Raposa and Talley (2012) using 69 datasets obtained from reference salt marshes (33), tidally restricted salt marshes (25), and salt marshes undergoing restoration (11), mostly from Rhode Island and Massachusetts, found that, although nekton community composition in reference salt marshes was significantly different from that in tidally restricted salt marshes, overall species richness did not differ significantly among the three salt marsh types, and only 17% of species showed significant differences in density. The authors attributed these variable responses to differences in the type and extent of tidal restrictions. Salt marshes that are diked and drained tend to hold less water during high tides, limiting nekton access to the vegetated surface. Limited evidence suggests that these marshes support degraded nekton communities, which then respond favorably to salt marsh restoration. Diked and impounded salt marshes may provide a more stable water body that supports higher nekton densities pre-restoration, potentially leading to declines after tidal restoration.

<u>Rotational Impoundment Management</u>

- A review of research on the effectiveness of Rotational Impoundment Management (RIM) along Florida’s Indian River Lagoon by Brockmeyer et al. (1997), as described above, categorized fish and crustaceans into 13 species that complete their life cycle entirely within the impounded salt marsh (resident) and 94 species that require periodic access to estuarine or ocean water (transient). Following RIM, resident species made up 94% of individuals and 46% of the total biomass, with individual resident species present in 0.6–43% of data collections. In contrast, although transients comprised only 6% of individuals, they accounted for 54% of the biomass, showed highly seasonal occurrences (0.4–24% by species), and were most negatively impacted by isolation from adjacent estuarine waters. The authors noted that over 80% of transient species were important for sport and commercial fisheries, indicating that impoundments without seasonal management can have significant economic repercussions for Florida's east coast fisheries.

Dike breaching and channel construction

- Able et al. (2000) evaluated the effects of a marsh restoration project in lower Delaware Bay on fish use patterns. The restored marsh had been used as a salt hay farm managed with dikes that were open in the fall/winter and closed in the spring/summer. Site restoration consisted of six dike breaches and construction of 5,500 m of channels inside the diked salt marsh area. Three years following restoration there were major changes in species richness, composition and abundance of fishes. Species richness increased from two species collected pre-restoration to 12 and 10 species collected with trawls and weirs respectively, post-restoration. The overall abundance of fishes was greater in both large and small creeks at the restored marsh than at the reference marsh. Species richness in the small creeks was highest in the restored creeks (18 species) and lower at the two reference sites (11 and 13 species). The authors concluded that the positive responses of fishes to the salt marsh restoration were due to the addition of large subtidal creeks and the resulting hydrological reconnection.

Ditch plugging

- In a study by Adamowicz and Roman (2002) assessing how plugging historic drainage ditches influenced three salt marshes at the Rachel Carson National Wildlife Refuge in Maine (as described above), the initial response of the nekton community (fishes and decapod crustaceans) was evaluated by monitoring the use of salt marsh pools. These responses differed among sites: the Moody and Granite Point sites showed no significant change in species richness, density, or community composition relative to control marshes. However, at Marshall Point, nekton species richness and density were significantly higher in the plugged marsh (11% open-water pools) than the control (< 2% open water pools). The authors cautioned that although their findings are initial responses and more long-term monitoring is necessary, ditch plugging could enhance nekton habitat.

Birds

Coastal wetlands serve as important habitat and breeding grounds for birds. Migratory birds rely on salt marshes, in particular, as important stop-over points on their migration routes. Other birds have evolved to live in the marsh environment exclusively (EPA 2020).

Culvert removal

- Raposa (2008) examined the response of the bird community following the removal of three failed culverts on the salt marsh at Potter Pond, Rhode Island, in 2003 (as described previously). Following the restoration, overall bird abundance increased by 1,400% during the first year. Although abundance decreased somewhat in the second year, it remained 1,000% higher than under the prior tide-restricted conditions. The most significant changes in the avian community occurred in the first year after restoration which was marked by a shift from open-water foragers to shorebirds. Importantly, seven additional bird species were observed at Potter Pond one year after restoration, and the total number of birds recorded rose from 6 to 85 per viewing effort, primarily due to large flocks of shorebirds taking advantage of the newly exposed mudflats created by the restoration. These findings underscore the rapid and substantial ecological responses that can follow the restoration of tidal connectivity in salt marshes, particularly in terms of biodiversity and habitat use by key indicator groups like birds.

<u>Culvert installation and creek construction</u>

- To study changes in bird occurrence in restored salt marshes over time, Warren et al. (2002) studied the relative abundance of salt marsh specialists and generalists in three salt marsh sites in Connecticut. Two sites were diked and impounded in 1947 and subsequently underwent tidal restoration 40-44 years later through dike breaching with culvert installation. A third site, diked and filled with dredge spoil in 1954, was restored through fill removal and the dike was breached by a 2-m restored creek in 1990. Along with a reference salt marsh, these sites were sampled in 1994-95 and 1999. The researchers found that the relative abundance of specialist bird species on the oldest restoration site was comparable with the reference site 15 years after restoration. Use of the restored sites by generalist bird species was initially high and remained at nearly twice that of the reference site after 20 years. The results indicate that tidal reconnection can result in the restoration of ecological functions of degraded marshes within two decades, although reduced tidal action can delay restoration of some functions.

<u>Culvert installation</u>

- Brawley et al. (1998) studied avian species' use of a 21-hectare diked and impounded salt marsh at Barn Island, Connecticut, where tidal flow had been restored 14 years prior to the project through installation of two culverts. Bird species were divided into generalist and specialist groups for analysis. Results showed that as the dominant vegetation changed from *Phragmites* to *Spartina*, the restored marsh provided adequate breeding habitat for specialist species like the Seaside Sparrow. The authors commented that restoration of tidal flow may initially eliminate breeding habitat for birds that nest on the marsh surface, but the re-establishment of *S. alterniflora* demonstrated that tidal restoration can eventually create suitable conditions for these specialized species.

<u>Ditch plugging</u>

- In the study by Adamowicz and Roman (2002) assessing how plugging historic drainage ditches influenced three salt marshes at the Rachel Carson National Wildlife Refuge in Maine (as described above), bird use varied: Granite Point exhibited a rise in species richness from approximately 15.4 pre-plug to 26.2 in 2000 and 38.7 in 2001, while Moody Marsh's low sample size precluded reliable comparisons; overall bird density (birds per hectare) remained statistically unchanged at Granite Point despite a non-significant upward trend. The authors conclude that ditch plugging could enhance bird richness.

Chemical and physical indicators

Restricting or blocking tidal flow alters the salinity of wetland soils, as well as the exchange of sediment and organic material. This can lead to loss of soil organic matter and resulting subsidence through soil compaction and can also affect redox and other soil properties such as bulk density. In contrast, tidal restoration can lead to rapid accumulation, accretion, and elevation change, especially in the short term, as the salt marsh responds to the higher water level (Anisfeld 2012).

Below are examples of how chemical and physical indicators have been evaluated as part of salt marsh hydrological restoration efforts.

Porewater Salinity

Small variations in salinity play a significant role in shaping the distribution of vegetation and organisms within salt marshes. Salinity directly affects plant zonation and animal habitat use, as each species has evolved to tolerate a specific salinity range and will occupy portions of the marsh that match its physiological needs (Merkey et al. 2005).

<u>Culvert removal</u>

- Eagle et al. (2022) studied five salt marshes in Cape Cod, Massachusetts where tidal exchange was restored by removal or enlargement of undersized culverts between 5 and 14 years prior to sample collection (as described above). Porewater salinity was lower at three of the restored sites compared to their reference salt marshes, whereas the other two sites were saltier than the natural marsh, potentially due to evapotranspiration and a longer water residence time at these locations.

<u>Dike removal</u>

- In the study by Rubin et al. (2024), described above, investigators measured invertebrate responses to large-scale dike removal in the Nisqually River Delta, Washington. At the Restored Refuge site (dike removed in 2009), porewater salinity increased from 5 in 2009 to 15 in 2010-2012, as would be expected from the return of tidal influence, whereas salinity at both the natural reference sites and the Restored Marsh site (restoration conducted in two phases in 2002 and 2006) unexpectedly decreased from 20 in 2009 to 5-10 in 2012.

Runnel installation

- Sullivan et al. (2025) examined the effect of runnels on porewater salinity in 11 ditched and impounded salt marshes across New England (as described above) by measuring soil cores collected from the three treatment types (impounded, runneled, and reference) before and after restoration. Porewater salinity in both runneled and impounded plots averaged 33.9, which was higher than that in reference plots (24.4).
  - To better understand the interaction between porewater salinity and the presence of vegetation, the marsh sites were divided into sites showing vegetation recovery following runnel treatment (n=6) and those with little to no recovery (n=5). Where vegetation had recovered there was little difference between average porewater salinity in impounded (26.1), runneled (25.4), and reference sites (24.0). Where vegetation had not recovered, average porewater salinity in impounded (41.5) and runneled sites (42.1) were similar, although each was significantly higher than the reference sites (25.5). The authors speculated that the high salinities in runneled sites with no vegetation recovery might be the result of higher bulk density and soil compaction limiting water filtration into the sediment, such that runneling was unable to restore the natural tidal flushing to allow for revegetation.

Sediment characteristics

Several characteristics of salt marsh sediment, such as organic matter content and grain size, are useful for determining habitat recovery following tidal restoration. Soil bulk density can be used as an indicator of soil compaction, defined as the mass of solid particles (e.g., sand, silt, and clay) in a given volume of soil (USDA, NRCS (2008) Soil Quality Indicators).

Dike breaching by flap gate removal

- Cotton (2004), as described above, sampled sediment grain size in a diked/impounded and a reference brackish marsh located along the Ogeechee River that had been restored through flap gate removal. Prior to flap gate removal, sediment at the impoundment site was composed predominantly of sands with little silt and clay (< 10%), whereas the reference site had greater, although variable, amounts of silt and clay. The lack of particulates in the grain size data before flap gate removal likely contributed to the marsh subsidence in the impoundment.

Dike removal

- Craft et al. (2001) evaluated nutrients in a salt marsh on Sapelo Island, Georgia, following restoration. The marsh, originally diked in 1948 for agriculture and later converted to an unvegetated salt pan, had its dike breached in 1956, allowing tidal waters to return. *Spartina alterniflora* re-established over the next five years, re-vegetating the area. In 1998, soil cores (0–10 cm) from both the restored and a reference marsh were analyzed for bulk density, organic carbon, total nitrogen, and phosphorus. Results indicated that the soil nutrient pools and elemental ratios (C:N and N:P) were similar in both marshes, except for phosphorus accumulation, which was 26% higher in the restored marsh (0.83 $g/m^2/yr$ compared to 0.66 $g/m^2/yr$ in the reference). The authors concluded that the 42-year-old restored marsh provides the same level of biogeochemical and water quality improvement functions as a natural marsh.

- In the study by Rubin et al. (2024) of large-scale dike removal in the Nisqually River Delta, Washington (as described above), change in grain size across years did not differ significantly among the salt marsh sample sites. Sediment percent organic matter was roughly 30% in the Restored Refuge site before dike removal and then decreased to <10% by 2012, whereas organic matter was consistently low (<=5%) across years in the Restored and Reference marsh sites.

Runnel installation

- Sullivan et al. (2025) investigated the effect of runnels on soil organic matter and bulk density in 11 ditched and impounded salt marshes across New England (as previously described). There was an interaction between organic matter content and vegetation recovery in runneled sites: runneled plots where vegetation recovered had an average organic matter content of 57.4%, which was in the range of that observed in reference plots and higher than runneled plots without recovery (43.6%). Impounded plots had lower organic matter content than reference sites, regardless of vegetation (47.8-50.3%). The authors commented that the increase in organic matter in recovered runneled sites could be a result of higher inputs of organic matter from recovering vegetation. They suggest that at sites where runnels lead to revegetation, they appear to drain sediments enough to promote primary productivity, while minimizing aerobic decomposition of organic matter.
    - Bulk density averaged 0.26 g cm-3 at impounded sites, 0.24 g cm-3 at runneled sites, and 0.20 g cm-3 at reference plots, regardless of vegetation recovery. The researchers speculated that higher bulk density and soil compaction in impounded areas might be leading to accumulation of hypersaline water and inhibiting plant growth.

Redox potential

Oxidation-reduction potential (redox) is a key indicator used to assess the intensity of anaerobic conditions in tidal marshes (Craft 2022). Redox data also provides restoration practitioners with an indirect measure of the duration and frequency of soil inundation, as well as a direct measure of which chemical forms (e.g., reduced or oxidized compounds) are likely present in the marsh soil (Merkey et al. 2005). Although redox indicators are not routinely reported in restoration monitoring programs, they are useful in broadly distinguishing aerobic and anaerobic soil conditions (Boon, Pollard and Ryder 2014).

Simulated Dike Removal

- In a greenhouse microcosm experiment, Portnoy and Giblin (1997) simulated salt marsh restoration by flooding sediment cores collected from diked-drained and diked-impounded marsh sites with seawater. Saltwater flooding led to decreases in redox potential in both sediment types. However, the declines were most dramatic in the diked-drained soils, especially in the surface layers, because these layers contained higher amounts of organic matter that rapidly depleted available electron acceptors.

Runnel installation

- The runnel study by Sullivan et al. (2025) (as previously described) found a significant interaction between vegetation recovery success and treatment on redox potential. Redox potential was negative in both impounded and runneled plots where vegetation did not recover, as opposed to positive values observed in vegetated plots. The authors concluded that the initial redox potential, not runnel treatment, was the dominant factor influencing vegetation regrowth, and that, while runnels can enhance recovery, their effectiveness depends heavily on pre-existing sediment conditions.

Accretion/Elevation

Marsh sediment is built from organic material and minerals deposited by the tides. Tidal restoration can lead to rapid accumulation, accretion, and elevation change, especially in the short term, as the salt marsh responds to the higher water level (Anisfeld 2012).

Culvert removal

- As described above, Eagle et al. (2022) evaluated soil vertical accretion in five coastal wetlands that had been impounded for over a century and then restored to tidal exchange 5 to 14 years prior to sampling. Soil core analysis revealed that impoundment suppressed elevation gain by 30–70%, which accounted for most of the elevation deficits observed in impacted sites relative to natural ones. Notably, only one site experienced substantial subsidence, likely from oxidation of soil organic matter. Following restoration, all sites exhibited increased rates of vertical accretion. At the site with subsidence, the vertical accretion rate was double that of the reference marsh, driven mainly by organic matter accumulation.

Dike breaching by flap gate removal

- Cotton (2004) measured sedimentation rates in a diked and impounded brackish marsh along the Ogeechee River that had been restored through flap gate removal, as described above. Long-term data collection (1999–2001) at the impounded site showed that sedimentation increased by nearly 2 cm per year. A subsequent sampling in 2001, which included both the impounded site and a nearby reference marsh, revealed that sediment was accumulating in the impoundment at a faster rate than in the reference marsh. This rapid sedimentation was promising, as soil elevations inside the impoundment had been lower than those of the reference marsh due to subsidence.

Dike removal

- Craft et al. (2001) compared sediment accumulation between the restored and natural marsh on Sapelo Island (as described above). They found that vertical accretion was higher in the restored marsh (5.0 mm/yr) than in the natural marsh (3.82 mm/yr), likely due to increased sediment deposition from renewed tidal inundation. Twenty-two years later, Craft et al. (2023) revisited the Sapelo Island marsh to assess its resilience to sea level rise by analyzing vertical accretion and sedimentation. Over 64 years, the restored marsh accumulated about 30 cm of soil. Its soil, comparable in bulk density and nutrient content to the natural marsh, accreted at a higher rate (4.8–5.1 mm/yr) than the reference, which is just keeping pace with the current sea level rise of 3.4 mm/yr. The authors concluded that the accelerated accretion is compensating for subsidence that occurred when it was drained and is driving higher C sequestration and N burial rates.

<u>Ditch remediation</u>

- Burdick et al. (2020) measured changes in salt marsh elevation for three years following ditch remediation through placement of a 15–20 cm layer of braided mowed hay onto nine ditched New England salt marshes (2015-2017), as described above. The increase in vegetation (see above) resulted in the capture of fine-grained sediment, and there was an 18 cm decrease in ditch depth in treated ditches by 2017.

Part 3. Conclusions

Hydrology is an important determinant of coastal wetland structure and function. Through both direct and indirect pathways, hydrological connectivity influences habitat conditions, plant zonation, and species interactions. Preserving and restoring hydrological connectivity is therefore essential for wetland conservation.

In this report, we analyzed the effects of tidal restrictions on salt marshes caused by culverts, ditching, and dikes and how these effects can be remediated through hydrological restoration. Our review focused on restoration projects located on the eastern coast of the U.S. In Georgia and South Carolina, tidal reconnection of historic agricultural diking and wildlife impoundments required dike breaching through tide gate or culvert removal/installation. This contrasts with New England states, which were more likely to employ ditch plugs and runnels to remedy salt marsh damage caused by past ditching projects. Remediation of mosquito ditching was common to all the states.

Success in salt marsh restoration is often subjective and depends on the criteria applied. Mitsch and Wilson (1996) define success as the ability of a system to sustain viable populations and natural processes, or to function comparably to nearby reference salt marshes. Common benchmarks include meeting project goals within a reasonable timeframe (Craft 2022); re-establishing ecological functions such as nutrient cycling and the maintenance of native communities (Hopkinson et al. 2019); and achieving outcomes at broader spatial scales where restored salt marshes become ecologically integrated into their watersheds (Cadier et al. 2020). In practice, restoration may be considered successful when parameters such as hydrology, soil properties, and biological communities in restored salt marshes approach those of reference sites, even if full convergence is not immediate (Anisfeld 2012). While some projects have used single metrics (e.g., 75% vegetative cover) to define success, such measures may neglect the ecological value and composition of the species present. Other studies stress comparisons of multiple ecological parameters, including water levels and fish usage, against reference marshes that serve as functional benchmarks (Burdick et al. 1997). Critics contend that restoration success must extend beyond surface metrics, allowing time for natural processes to re-establish and recognizing the inherent self-organizing capacity of ecosystems (Mitsch and Wilson 1996; Cotton 2004).

One conclusion from this review is that the effectiveness of tidal reconnection varies widely and depends strongly on the condition of the salt marsh prior to restoration. Comparisons across studies are further complicated by the use of different indicators and metrics to evaluate outcomes. In addition, most projects have lacked long-term monitoring, with the majority of studies reporting results only within 1–10 years after restoration. Despite these limitations, a general trajectory of salt marsh ecosystem recovery can be discerned (see Box 1).

Restoration success can be improved by taking a flexible approach. Restoration to a pristine, natural wetland is not always possible due to pressures such as climate change and human development. Rather than recreating historical marsh conditions, it may therefore be beneficial to consider a wide range of desirable outcomes that emphasize the creation of self-sustaining ecosystems that can thrive under future conditions (Greening et al. 2023; Reyes-Aldana 2024). It is also important to recognize that the approach used in a restoration project may need to change in response to unintended impacts or changes in reference wetlands (Crain, Gedan, and Dionne 2009).

Monitoring a restoration project can take significant effort. However, advances in AI, machine learning, and remote sensing, can help with data collection, analysis, and synthesis. Automation reduces the need for constant human intervention, lowers long-term costs, and provides timely, integrated information for decision-making (Ditria et al. 2022; Greening et al. 2023). New genetic tools, including genome-wide association studies, synthetic biology, and gene editing (e.g., CRISPR/CAS9), can also help match the adaptability of restored plant and animal populations to future environmental conditions. Strategies such as assisted adaptation, targeted translocations, and genetic rescue can bolster the resilience of populations, ensuring that restoration efforts prepare ecosystems to face ongoing habitat deterioration and climate change (Coleman et al. 2020).

Coastal areas have long supported human settlement due to their ecological abundance and productivity. As coastal populations expanded, however, salt marshes were frequently altered in ways that disconnected them from tidal waters. Many were drained or impounded with dikes, ditches, and culverts to create farmland, pasture for livestock, waterfowl habitat for hunting, transportation corridors, or to control mosquitoes. Because tidal hydrology underpins salt marsh health, alterations to inundation regimes degrade habitat, reduce native biodiversity and productivity, and diminish natural protections against erosion and flooding. Hydrological restoration seeks to reverse these impacts, enabling salt marshes to recover their ecological functions and productivity while continuing to provide unique and highly valuable benefits to human communities.

Box 1. Trajectory of salt marsh recovery following hydrological restoration

⇒ **Immediate to Short-Term Response (0–1 year)**

- *Hydrology*
  - Tidal exchange can resume almost immediately after culvert enlargement, dike breaching, or runnel installation (Eagle et al., 2022).
  - Porewater salinity may initially increase or fluctuate due to tidal flushing and altered evapotranspiration regimes (Sullivan et al., 2025; Rubin et al., 2024).
- *Soil Redox Conditions*
  - Soil begins shifting from anoxic to more oxidized states as tidal flushing returns (Sullivan et al., 2025).

⇒ **Initial Biological and Physical Change (Years 1–3)**

- *Vegetation*
  - Early colonization by salt-tolerant pioneer species (e.g., *Spartina alterniflora*) begins (Eagle et al., 2022; Buchsbaum et al., 2006; Burdick et al., 2020).
  - Coverage may increase rapidly but is patchy; species composition may differ from reference marshes (Brockmeyer et al., 1997; Adamowicz and Roman 2002).
  - Recovery is slower in areas with altered elevation or hypersaline soils (Sullivan et al., 2025; Cotton 2004).
- *Sediment characteristics*
  - Nutrient profiles (e.g., nitrogen, phosphorus) begin adjusting (Craft et al., 2001; Craft et al., 2023).
  - Organic matter accumulation may begin if vegetation establishes (Burdick et al., 2020; Sullivan et al., 2025).
  - Sediment accumulation begins in impounded areas (Cotton 2004; Burdick et al., 2020; Craft et al., 2001).
- *Faunal Communities*
  - Some benthic macroinvertebrates, birds, and small fish return quickly (months to 1 year) (Rubin et al., 2024; Raposa 2008; Adamowicz and Roman 2002).
  - Tidal connectivity improves, enabling fish passage and foraging (Buchsbaum et al., 2006).
  - Bird community abundance may increase due to vegetation and habitat recovery (Raposa 2008).

⇒ **Intermediate Recovery Phase (Years 4–10)**

- *Vegetation*
  - Species composition begins to resemble reference marshes but may still differ (Besterman et al., 2022).
- *Sediment characteristics*
  - There is a gradual increase in soil organic matter and decrease in bulk density (Sullivan et al., 2025).
  - Vertical soil accretion begins to increase and normalize (Eagle et al. 2022; Craft et al., 2023).
- *Nekton and Birds*
  - Nekton assemblages become more robust (Able et al. 2000).
  - Bird species composition may shift in accordance with habitat changes (Raposa 2008; Warren et al., 2002; Brawley et al., 1998).

⇒ **Long-term recovery trajectory (10–20+ years)**

- Drainage patterns
  - Artificial channels continue to affect drainage patterns 50-100 years after breaching (Cotton 2004).
- *Plant Community Structure*:
  - May fully resemble reference marsh in some cases; others remain distinct due to legacy effects such as altered elevation and salinity (2004; Sullivan et al., 2025).
- *Soil Properties*:
  - Soil organic matter, nutrients, bulk density, and salinity gradients may take 10–20 years or longer to match natural marshes (Craft et al., 2001; Sullivan et al., 2025).
- *Ecosystem Function*:
  - Primary productivity, denitrification, and carbon sequestration can approach reference levels (Craft 2022).
  - Trophic networks stabilize and support long-term resilience (Warren et al., 2002)

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