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Key Points:

- The current state of knowledge on subsurface hydrology and biogeochemical processes in wetlands are provided
- A holistic, interdisciplinary approach to wetland restoration research is required to optimize carbon sequestration
- The interaction of hydrogeological and biogeochemical processes should be considered to ensure the long-term success of restoration projects

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Coastal Wetland Restoration Effects on Carbon Dynamics: A Groundwater Perspective

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Abstract Coastal wetlands are critical components of the global carbon cycle, yet many have been degraded by drainage and land-use changes, releasing stored carbon and exacerbating climate change. Interest in wetland restoration is increasing globally due to their carbon storage capacity, ecosystem values, and biodiversity co-benefits. This is further supported by the United Nations' declaration that 2021–2030 is the "*Decade on Ecosystem Restoration*." Various techniques have been applied to achieve ecological wetland restoration. However, additional opportunities should be explored to extend the co-benefits of restoration beyond ecological aspects. For these restoration co-benefits to be fully optimized, the science of wetland functions and processes must be understood and quantified. Groundwater plays a crucial role in wetland carbon transport, storage, and transformation. However, its processes are often overlooked or poorly quantified. This paper (a) provides the current state of knowledge on subsurface hydrology and biogeochemical processes in wetlands, (b) describes how the hydrologic restoration of wetlands may affect the physical and chemical processes within wetlands, and (c) discusses how restoration can alter wetland function, with a specific focus on carbon and greenhouse gas dynamics. The paper highlights the necessity for a holistic, interdisciplinary approach to wetland restoration research, considering the interplay of hydrological, biogeochemical, and climatic factors to optimize carbon sequestration and ensure the long-term success of restoration projects.

Plain Language Summary Coastal wetlands like mangroves and saltmarshes play a significant role in storing carbon, filtering water, and protecting shorelines. However, many of these ecosystems have been drained or damaged, turning them from carbon sinks into sources of greenhouse gases such as carbon dioxide and methane. As climate change accelerates, restoring these wetlands has become a global priority. While restoration efforts often focus on vegetation and surface water, this paper explores a less understood but equally important factor, groundwater. We reviewed how groundwater influences the flow, storage, and transformation of carbon in wetlands, especially after tidal flows are reintroduced during restoration. Groundwater interacts with surface water and sediments, affecting salinity, acidity, and microbial activity, all of which control how much carbon is stored or released. Our research shows that changes in groundwater levels and chemistry can shift wetlands from emitting carbon to storing it, depending on how restoration is managed. This paper calls for a more holistic approach to wetland restoration, one that includes groundwater processes, to better predict carbon outcomes. Understanding these underground dynamics is critical for ensuring that restored wetlands truly help combat climate change by capturing and storing carbon over the long term.

1. Introduction

1.1. The Significance of Coastal Wetlands

Coastal wetlands are ecosystems located at the interface between terrestrial and marine environments, that are characterized by periodic or continuous inundation by saltwater or freshwater (Costanza et al., 1997; Mitsch et al., 2015). They cover a wide range of habitat types including saltmarshes, mangroves, intertidal flats, seagrass meadows, peat, supra tidal communities and freshwater wetlands. These ecosystems, at the interface between land and ocean, often feature gradients in the characteristics of both environments. Coastal wetlands provide numerous ecosystem services, including water purification (Kim et al., 2020), habitat provision (Mitsch et al., 2015), biodiversity support (H. Yang et al., 2017), coastal protection (F. Sun & Carson, 2020), wave attenuation and erosion control (Costanza et al., 2021), surface water runoff retention (Narayan et al., 2017) and carbon sequestration (X. Li et al., 2018). Coastal wetland sediments play a vital role in the biogeochemical cycling of

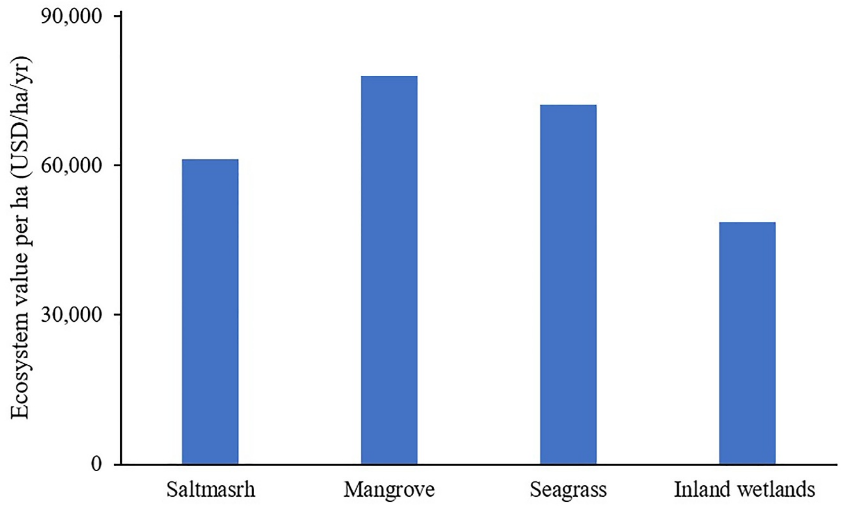

Figure 1. Coastal and inland wetland ecosystem service values. Data from De Groot et al. (2020a, 2020b).

metals, nutrients and carbon. These benefits contribute to the growing recognition of wetlands as valuable ecosystems on a global scale (Reddy & DeLaune, 2008).

Globally, coastal ecosystem services are estimated to be worth more than US $25,000 billion yr^{-1} (Nellemann & Corcoran, 2009) or an estimated mean value of US $194,000 ha^{-1} yr^{-1} (Costanza et al., 2014). Figure 1 shows the ecosystem service values for various coastal wetland ecosystems. The multi-dimensional significance of coastal wetlands also contributes to the fisheries and tourism sectors. For example, coastal wetlands in south-eastern Australia, contribute an estimated $35.5 million per year (US $33.1 million) to nearshore fisheries and recreational activities (Carnell et al., 2019). On a global scale, the estimated economic value of coastal wetlands for provisioning (e.g., fisheries) and cultural (e.g., tourism) services amounts to $2,998 and $2,193 (2007 US) ha^{-1} yr^{-1}, respectively (De Groot et al., 2012). In addition to their monetary value, coastal wetlands provide non-monetary cultural services that are deeply embedded in the traditions and identities of Indigenous communities. These include customary practices such as harvesting plant materials (e.g., tule reeds for basket weaving), spiritual connections to ancestral lands, and the maintenance of traditional ecological knowledge systems (Clarke et al., 2021; Martin et al., 2016).

Over the past century, coastal wetlands have been extensively modified and degraded for flood mitigation purposes (construction of levees), as well as land reclamation for agricultural and urban development (Ouyang & Lee, 2020). These activities typically enhance drainage and disconnect the floodplain from the estuary, which has resulted in the loss of between 64% and 71% of wetlands globally since 1900 (Davidson, 2014). These changes have caused (but are not limited to) habitat loss for commercial and recreational fisheries and wildlife, poor water quality, harmful algal blooms, acidification, and fish kills (Worm et al., 2006). Reduced biodiversity and ecosystem services of coastal wetlands can lead to changes in vegetation type and spatial distribution, reducing the ability of these systems to prevent coastal erosion (Lotze et al., 2006; Valiela et al., 2009; Worm et al., 2006).

In addition, increased drainage of coastal wetlands has released stored carbon, as CO_2, back into the atmosphere via the oxidation of organic matter and contributed to accelerated global warming (Pendleton et al., 2012). Estimates indicate that every year approximately 1.8% of the area covered by coastal wetlands is lost worldwide, resulting in the release of 450 million tons of CO_2, equivalent to the annual emission of 118 coal fired power plants (Herr et al., 2015). Predictions warn that a continuation of the current loss rate will result in an additional 30% of tidal wetlands being lost by the year 2100, with a global economic impact of $2.6 billion USD per year due to the additional carbon emitted to the atmosphere (Gulliver et al., 2020; Pendleton et al., 2012).

1.2. The Role of Coastal Wetlands in the Global Carbon Budget

Coastal wetlands store about 40% of the carbon buried in marine sediments, while only occupying around 1% of the ocean surface area (Duarte, 2017). As such, coastal wetlands play an important role in the carbon budget of

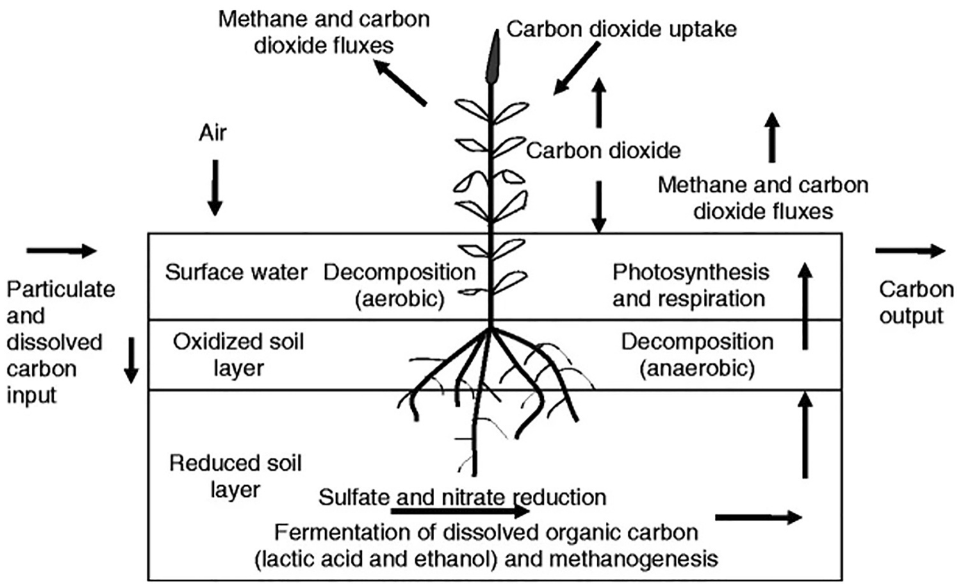

Figure 2. Carbon cycling and major components of the carbon cycle in coastal wetlands (Kayranli et al., 2010). Note: the photosynthesis and respiration process will be in the opposite directions (e.g., photosynthesis binding carbon and respiration releasing it).

aquatic and marine ecosystems (Webb et al., 2019). The magnitude of the net carbon fluxes depends on environmental factors, such as soil type and composition, vegetation type, nutrient availability, temperature, climate and hydrology (Reddy & DeLaune, 2008). Under natural conditions, high rates of carbon fixation through photosynthesis and low rates of carbon losses through microbial decomposition result in a net storage of organic carbon in wetland sediments (Reddy & DeLaune, 2008). Carbon can also be transported laterally in the form of dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), particulate organic carbon (POC), and as dissolved gases, such as carbon dioxide (CO_2) and methane (CH_4) (Figure 2).

Greenhouse gas production and consumption in coastal wetlands is highly variable (Mitsch et al., 2013; Nahlik & Mitsch, 2010; Rosentreter et al., 2021). Saturated soils typically reduce oxygen diffusion into the sediments, which, in turn, generates anaerobic (i.e., reducing) conditions. These reducing conditions slow microbial respiration, the rate of organic matter decomposition and the carbon dioxide production, which promotes organic matter accumulation in the sediments (Bridgham et al., 2013). However, reducing conditions at low redox potential typically lead to methanogenesis (Whiting & Chanton, 2001). As such, wetlands can contribute to CH_4 emissions when inundated and specific anaerobic conditions exist. On the other hand, wetlands can remove CH_4 from the atmosphere when their sediment becomes exposed to the atmosphere and CH_4 oxidation occurs (Whalen, 2005).

Saline waters (i.e., typically salinities higher than 18 ppt) can reduce CH_4 production in sediments compared to freshwater, due to differences in chemical composition (e.g., higher sulfate concentrations in seawater) and hence, tidal wetlands can have lower CH_4 emission rates (Poffenbarger et al., 2011). However, regardless of the salinity, once the dissolved sulfate is fully consumed (reduced) along longer and slower subsurface flow paths, CH_4 may still be produced (Capone & Kiene, 1988; McGenity & Sorokin, 2010). Hence, for subsurface processes, the starting salinity becomes less important than the combination of the organic matter availability, its degradation rate and the subsurface water residence time. Therefore, high degradation rates or long water residence times will favor methanogenesis.

Wetlands can also capture drifting and suspended sediments containing particulate organic carbon and laterally (horizontally) transport it from the land to the ocean via hydrological connectivity (Santos et al., 2019). Overall, carbon cycling in coastal wetlands is complex and wetlands can be a source or a sink of carbon based on environmental conditions and external perturbations (Kayranli et al., 2010).

1.3. Restoration of Coastal Wetlands

In recognition of historical wetland destruction and associated environmental impacts, wetland restoration has become a management priority worldwide, with the United Nations declaring 2021–2030 the "*Decade on Ecosystem Restoration*." Consequently, wetland restoration has emerged as a priority research discipline, with 60 countries pledging to restore a total of 3.5 million km^2 of degraded land (equivalent to the size of India) by 2030 (Waltham et al., 2020). This has led to the inclusion of natural and restored coastal wetlands in national greenhouse gas inventories following the recommendation by the 2013 supplement to the 2006 IPCC Guidelines for National GHG Inventories: Wetlands (Hiraishi et al., 2014). However, our understanding of carbon dynamics in wetlands after their restoration is limited and remains a priority objective of global greenhouse gas emissions reduction strategies (Villa & Bernal, 2018).

In many cases, the primary aim of ecological restoration is to re-establish, initiate, or accelerate the recovery of degraded and disrupted ecosystems (Hobbs & Harris, 2001). Wetland restoration comprises various on-ground activities that can be categorized into three main groups, including active, passive or creative methods (Q. Zhao et al., 2016). In active restoration, the topography and surface water hydrology of a wetland may be reshaped, vegetation planted and/or soil treatment carried out. This approach can also involve restoring hydrological connectivity through dam or levee removal and/or constructing breakwaters. Passive restoration approaches involve removing the elements that cause the loss or deterioration of wetlands and allowing the wetland to reach a healthy state by natural regeneration. Creative methods for wetland restoration refer to the construction of a wetland where a wetland did not exist beforehand (Q. Zhao et al., 2016). Regardless of the approach, restoring the correct hydrological conditions to a site is a key consideration to stimulate the target ecological response (Zedler & Kercher, 2005). Therefore, developing an optimized hydroperiod (i.e., duration of flooding in wetlands) that promotes the growth and establishment of a desired vegetation is a significant element in determining the success of a coastal wetland restoration project (Glamore et al., 2021; Sadat-Noori et al., 2021).

Effective wetland restoration for carbon storage requires a quantitative understanding of carbon sinks, sources and processes within restored sites (Hemes et al., 2018). Currently, little is known about carbon storage, cycling and transformation within restored wetlands and whether these processes differ between natural wetlands, wetlands undergoing restoration, and fully restored ecosystems (Hemes et al., 2019). For instance, it is not clear over what time scale, a restored wetland will sequester similar amounts of carbon compared to natural wetlands. For example, 18 years after restoration (rewetting) of a drained peat wetland in NW Germany, the site remains a GHG source, although GHG emissions have reduced (Schaller et al., 2022). In contrast, seagrasses restoration in Virginia USA, converted the site into a GHG sink, 15 years post restoration (Oreska et al., 2020). A study on sediment carbon accumulation post restoration of a saltmarsh wetland in southeastern Australia, reported higher carbon stock than reference sites, 9 years post restoration (Lovelock et al., 2021). In nontidal fresh and brackish saltmarsh wetlands in California's San Francisco Bay, CH_4 emissions countered atmospheric CO_2 removed for the first 2–8 years after restoration (Arias-Ortiz et al., 2021). As CH_4 is a stronger greenhouse gas, this example emphasizes that it is important to consider biochemical processes and the forms of carbon rather than just total carbon stocks and fluxes.

Further studies on carbon dynamics and the ability of restored wetlands to mitigate climate change effects (i.e., the radiative forcing effect of restored saltmarsh and mangroves) with a focus on "age post restoration" is required as the time after restoration until the wetland becomes a net sink of carbon can vary significantly. This can be an important factor for predicting carbon removal benefits for coastal wetlands restoration projects (Carnell et al., 2022; Williamson & Gattuso, 2022). Further, there is no comprehensive quantification regarding the role of groundwater in the transport, storage and transformation of carbon in restored coastal wetlands. Without a proper quantitative understanding of the mechanisms, the carbon budget in these ecosystems cannot be accurately constrained (Kelleway et al., 2020).

Knowledge of the processes that govern the cycling of coastal carbon in groundwater has benefits that go well beyond constraining the carbon budget. Carbon availability, chemical speciation (organic and inorganic), and the transformations between different forms of carbon control geochemical, ecological, and environmental processes. Quantifying the carbon cycle and transformation processes helps address a range of important issues in the coastal zone. These are drivers of ecological processes, including those leading to hypoxia, nutrient attenuation, contamination transformation, recycling carbon for food webs and the carbon sequestration potential of vegetated coastal ecosystems (Cai, 2011; Diaz & Rosenberg, 2008; Serrano et al., 2019). Therefore, understanding how coastal wetlands function (i.e., the physical and biogeochemical processes within them that affects the storage, transformation, and transport of carbon), requires urgent attention due to the rapid development of restoration as a natural climate mitigation solution and the need to better quantify the drivers that control whether coastal ecosystems are sources or sinks of carbon. Finally, this improved understanding of coastal wetland function will underpin restoration principles and technologies to systematically optimize carbon sequestration as one of the principal aims of wetland restoration.

To summarize, many coastal wetlands around the globe are degraded and there is an appetite for restoring wetlands as their importance and value have increased. As such, there is an opportunity to consider the benefits of restoration beyond ecological aspects including the potential carbon storage. The aim of this paper is to detail the current knowledge of subsurface hydrology and geochemical processes in coastal wetlands and to describe how restoration (i.e., reconnection with the tide) may affect physical and chemical processes within wetlands and alter wetland function. The paper has a specific focus on groundwater, carbon and greenhouse gas dynamics after restoration. In doing so, we highlight how this may inform systematic approaches to wetland restoration and overcome knowledge gaps that are currently hampering the full realization of coastal wetland restoration projects.

2. Subsurface Flow Dynamics in Wetlands

Groundwater hydrology refers to the recharge, storage and discharge of water below the land surface and is of critical importance in understanding the function of dependent ecosystems and in developing restoration plans for coastal wetlands (Tiner, 2018; Xin et al., 2022). The relationship between surface and groundwater should be understood under various conditions, including the re-establishment of wetland inundation with salt water, so that potential impacts on groundwater resources can be managed. This is particularly relevant in situations where freshwater sources have the potential to be affected by saline water after tidal flushing (del Pilar Alvarez et al., 2015; Glamore & Indraratna, 2009; Yabusaki et al., 2020). It is also important to understand how groundwater levels, quality and carbon dynamics may change under restoration and the reversion to natural hydrological conditions (Cui et al., 2021; Q. Zhao et al., 2017). As coastal wetlands can have a range of salinities from fresh to saline or even hypersaline, both coastal (marine) and freshwater hydrological processes will be discussed.

2.1. Groundwater Hydrology of Coastal Wetlands

The movement and flow of water below the surface of the earth and through porous sediments and fractured rocks is referred to as subsurface flow (Rosenberry et al., 2020; WMO, 1974). The flow direction can be horizontal or vertical and can occur for saturated or unsaturated conditions. Surface and subsurface hydrological processes are closely linked in coastal wetlands with the sources of water to a wetland being direct precipitation, infiltration after precipitation and subsequent groundwater discharge and/or surface water runoff (Xin et al., 2022). Natural processes that can remove water from wetlands are evaporation, transpiration and subsurface lateral or vertical flow (Guimond & Tamborski, 2021).

In coastal wetlands, the hydrology can transition from upland freshwater environments, where river overbank flow plays a major role in the wetland hydrology, to saline marine environments where tides and seawater inundation dominate the water balance (G. Sun et al., 2002; Y. Zhang et al., 2018). Factors that influence the hydrology of coastal wetlands are catchment size, topography, geomorphology, drainage, vegetation type and distribution, tidal regime, seasonality, rainfall, evapotranspiration and changes in groundwater level (Hughes et al., 1998; Rosenberry & Hayashi, 2013). Figure 3 illustrates the major processes driving coastal wetland subsurface hydrology.

2.2. Surface Water—Groundwater Interaction

Surface water—groundwater interactions occur by infiltration through the unsaturated sediment and by exfiltration from and infiltration into the saturated zone (Sophocleous, 2002; Xin et al., 2022). Coastal freshwater wetlands may experience gaining or losing conditions depending on forcing conditions (tidal cycles, storm surges, episodic catchment recharge, etc.) and a variety of environmental factors. A gaining condition is where groundwater discharges into the wetland while a losing condition is when groundwater is recharged by the wetland (Fleckenstein et al., 2010; Rosenberry et al., 2020; Winter, 1999).

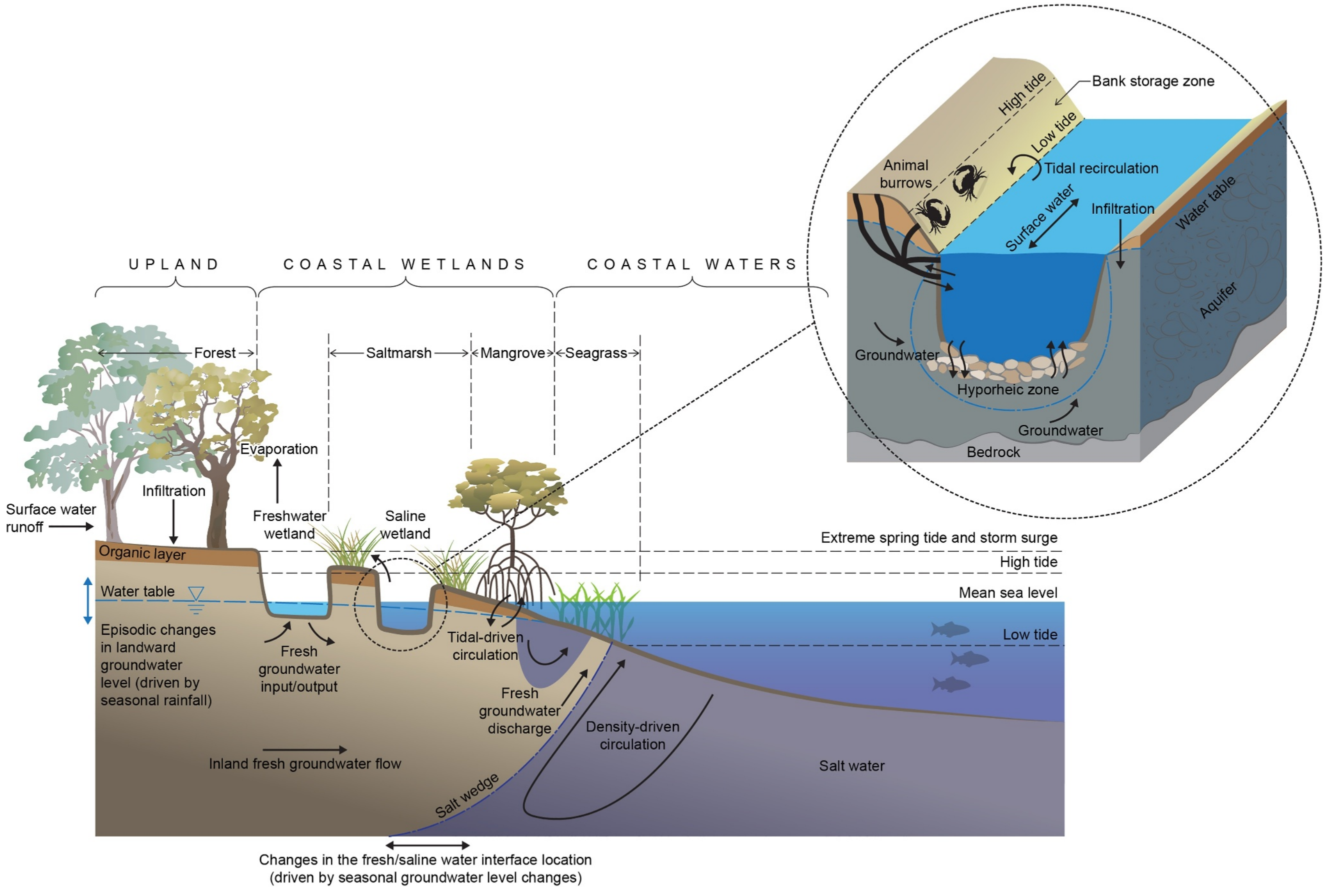

Figure 3. Conceptual diagram of the subsurface hydrology of coastal wetlands and subsurface flow dynamics at a tidal creek scale (adapted from Sophocleous (2002)).

The area of sediment beneath and around the river/creek where surface water and groundwater interact is defined as the hyporheic zone (Winter, 1999). Hyporheic flow can be separated from groundwater discharge, as hyporheic flow comprises surface water that flows into the sediment and returns to the same surface water body over relatively short spatial scales (e.g., cm to m), while groundwater discharge is usually one-directional and represents larger length scales (e.g., ~km) (Boano et al., 2014). It is often the case that the hyporheic zone water has a distinctly different chemical composition than the regional groundwater surrounding it. The mixing of surface water and groundwater in the hyporheic zone can therefore be a substantial driver of biogeochemical processes (Brunke & Gonser, 1997; van Dijk et al., 2017; Wörman et al., 2002). The location of hyporheic exchange is controlled by variation in bedforms that control up-welling and down-welling of water through the streambed of channels when there is surface flow caused by falling or rising tides, or run-off events (Boano et al., 2014).

In marine environments, subsurface flow exiting the sediments and entering the surface waters is termed submarine groundwater discharge (or SGD) (Burnett et al., 2006; Church, 1996). SGD comprises any water (groundwater and/or porewater) regardless of its composition (fresh, brackish, or saline) or driving force (Burnett et al., 2003; Santos et al., 2012; Taniguchi et al., 2019). SGD serves as a critical pathway linking surface and subsurface environments, facilitating biogeochemical and microbial processes through groundwater-surface water interactions (Adyasari et al., 2020; Moore, 2006; Ruiz-González et al., 2021). This mechanism plays a significant role in transporting dissolved constituents, such as nutrients and carbon, influencing carbon cycling in aquatic environments (Luijendijk et al., 2020; Santos, Chen, et al., 2021; S. J. Wilson et al., 2024).

Bank storage occurs when surface water levels are increased due to a large flow event and surface water propagates into the unsaturated zone of a phreatic aquifer. As surface water levels recede, this water then subsequently slowly discharges back into the stream as base flow (Winter, 1999). The amount of time the water remains in the aquifer, and is in contact with sediments or a stream bank slope, plays a significant role in driving geochemical processes that may influence the water quality (Sophocleous, 2002).

2.3. Drivers of Subsurface Flow

The driving force of the surface water—groundwater interaction is the pressure gradient between two areas. This is typically expressed by the hydraulic head (i.e., hydraulic head over a length scale). In the intertidal zone, subsurface flow is driven by the pressure gradient between the inland aquifer (groundwater level) and the coastal ocean (sea level) (Burnett et al., 2003; Xin et al., 2022). Besides the general seaward movement and discharge of groundwater in coastal catchments, different mechanisms of varying periodicity (with temporal scales ranging from minutes to decades and spatial scales spanning from mm to km) may perturb the water exchange at the surface-sub-surface interface (Boano et al., 2014; Santos et al., 2012; Xin et al., 2022).

At short timescales of sub-minutes, wave run-up causes surface water infiltration and local mounding of the groundwater table, which drives subsurface circulation (C. Robinson et al., 2014; Sawyer et al., 2013; Smith et al., 2009; Turner et al., 2016). At daily timescales, tidal fluctuation can cause surface water infiltration similar to wave run-up, but usually over larger areas, while spring tidal conditions cause the same infiltration processes but on a larger time scale (e.g., typically twice a month) (A. M. Wilson et al., 2015). Seasonal changes in the groundwater level can also shift the fresh and saline water interface landwards or seaward and substantially alter the subsurface flow and discharge magnitude (Dai & Trenberth, 2002; Michael et al., 2005; M. Robinson et al., 1998; Sadat-Noori et al., 2015). Episodic storm surges can also significantly affect subsurface flow dynamics and chemistry in coastal wetlands, which can cause extremely high surface water levels in creeks and wetland inundation due to runoff and increased groundwater recharge (Rupprecht et al., 2017; A. M. Wilson et al., 2011; J. Yang et al., 2015). On interannual scales, climate related processes such as El-Nino Southern Oscillation (ENSO) can create variable groundwater levels due to variations in rainfall recharge and therefore variable discharge conditions (Anderson & Emanuel, 2010).

In tidal wetlands, surface water channels usually connect the wetland to the estuarine intertidal zone (Xin et al., 2022). In these wetlands, the tidal regime significantly affects the wetland subsurface hydrology such that at high tide, surface water levels increase in the tidal channels and exceed the groundwater table (Harvey et al., 1987; A. M. Wilson & Gardner, 2006). This increase in the surface water pressure head creates a hydraulic gradient toward the groundwater and can lead to water infiltrating into the channel banks and the subsurface (Gardner, 2005; Montalto et al., 2006). At low tide, the hydraulic gradient reverses and water can discharge into the tidal creek (Howes & Goehringer, 1994). This process results in a periodic fluctuating groundwater level close to the tidal creek (M. Cao et al., 2012; Montalto & Steenhuis, 2004) and is known as tidal pumping (Stieglitz et al., 2013), which may be responsible for significant solute transport on a local scale.

The saturated hydraulic conductivity (K_{sat}) of wetland sediments is significant in determining the rate of subsurface flow and solute transport. Factors such as sediment texture, compaction and the presence of macropores influence K_{sat}, with higher values in sandy sediments or areas with animal burrows facilitating faster water infiltration and circulation (Guimond & Tamborski, 2021). Animal burrows (predominantly burrowing invertebrates) are a common feature of tidal wetlands and can act as preferential flow paths for subsurface flow (Guimond et al., 2020; McCraith et al., 2003). These flow paths can significantly increase the hydraulic conductivity of coastal wetlands, especially in sediments of saltmarsh and mangrove wetlands, where subsurface flow would be restricted due to the low permeability of clays and muds (Sadat-Noori et al., 2017; Stieglitz et al., 2013; Susilo & Ridd, 2005). In tidal environments, these macropores will increase the volume of tidal pumping (Xin et al., 2009; Xu, Xin, et al., 2021) and have been shown to enhance mixing between groundwater and surface water, leading to a significant increase in solute transport (Aller, 1980; Glamore & Indraratna, 2009; Shull et al., 2009; Xiao et al., 2019).

3. Carbon Chemistry in Coastal Wetlands

Coastal wetlands play an important role in global carbon storage, with a carbon storage rate per square meter that is an order of magnitude higher than that of terrestrial forests (Millero, 2007). Carbon stored in coastal wetlands can be in the form of solid (e.g., decaying vegetation and captured particulate carbon) or dissolved carbon (e.g., DIC and DOC). The dissolved fractions are small on a mass basis, however, they are potentially important as they are more mobile. Solid and dissolved carbon are subject to different geochemical processes in coastal wetland sediments, which are briefly described below.

3.1. Organic Carbon in Wetlands and Groundwater

Physically, organic matter (OM) is classified according to its size/filtration, with material that goes through a nominal 0.45 μm filter being referred to as dissolved organic matter (DOM) and that trapped by the filter referred to as particulate organic matter (POM) (Thurman, 2012). The different forms of OM have implications for biogeochemical processes. While POM can be suspended in the surface water column and mobile, it may settle to the sediment or be filtered out along subsurface flow paths. DOM can be transported through both surface and subsurface flows, however, field and laboratory studies have shown that a significant fraction of surface water-derived DOM can adsorb to mineral surfaces during subsurface transport (McDonough, O'Carroll, et al., 2020; Oudone et al., 2019). Although, mineral surfaces can become saturated with DOC, and thus sorption is not a sustained or continuous removal mechanism over time. The mineralization of OM is mediated by various types of microbes and controlled by the availability of electron acceptors (Megonigal & Neubauer, 2019). Organic matter is degraded partly to smaller molecules, some of which are dissolved and partly mineralized into DIC (Lønborg et al., 2020).

The dissolved organic carbon content tends to be lower in groundwater than in surface water, which indicates significant processing occurring in the subsurface (McDonough, Santos, et al., 2020). Chemical sorption (as mentioned above) and mineralization by microbes are processes that are proposed to be responsible for lower groundwater concentrations (Chapelle et al., 2012, 2016). Higher groundwater DOC values are associated with wetlands compared to other terrestrial environments, due to the presence of organic-rich sediment layers and surface water infiltration (Kolka et al., 2008; Meredith et al., 2020; Rutlidge et al., 2021). The characteristics of groundwater DOC in wetlands can be more aromatic and have a higher amount of hydrophobic DOC. In deeper groundwater, the content of aromatic and hydrophobic molecules is lower as their mobility is reduced due to adsorption (Hosen et al., 2018; Meredith et al., 2020; Rutlidge et al., 2021).

3.2. Inorganic Carbonate Chemistry

Dissolved inorganic carbon is defined as the sum of H_2CO_3 (carbonic acid, including dissolved CO_2), HCO_3^- (bicarbonate), and CO_3^{2-} (carbonate). The main source of CO_2 in groundwater is vegetation root respiration and oxidation of organic carbon in the sediment (Wood & Petraitis, 1984). Dissolved CO_2 concentrations are strongly dependant on the pH of water, where a very small change in pH (0.02 pH units) can change the partial pressure of CO_2 by up to 5% (Macpherson, 2009). Groundwater recharge is typically in equilibrium with soil gas CO_2, which is several orders of magnitude higher than that of atmospheric values (Wood & Hyndman, 2017). Infiltrating water rapidly equilibrates with this high concentration by forming carbonic acid, which may dissociate to hydrogen and bicarbonate ions (HCO_3^-). This DIC-enriched groundwater is eventually discharged into surface waters.

Coastal wetland sediments may contain large amounts of buried carbonate minerals, either as disseminated fine fragments of shells and corals in marine or aeolian deposited sands or as buried shell-lags. As carbonate minerals are known to dissolve quickly to reach saturation, on time scales shorter than most groundwater residence times, their dissolution may be described by equilibrium chemistry. In sediments containing carbonate minerals the concentration of DIC can, to a first approximation, be described by their mineral solubility according to the reaction:

$$CaCO_3(s) + H_2O + CO_2\ (aq) = Ca^{2+} + 2HCO_3^- \qquad \text{Log K} = -5.8 \tag{1}$$

CO_2 driving the dissolution typically originates from respiration in the rootzone or microbial processes in the soil horizon and deeper in the subsurface, such as organic carbon mineralization (see further below) (Cai et al., 2003; Y. Liu et al., 2021). The released bicarbonate (HCO_3^-) can subsequently convert to carbonate (CO_3^{2-}) or CO_2 depending on the ambient pH values of the water or due to other biogeochemical processes affecting the pH (Q. Liu et al., 2017; Saderne et al., 2019; Weathers et al., 2012). However, for most natural non-acidified waters, H_2CO_3 is dominant below a pH of 6.3 and HCO_3^- is dominant above a pH of 6.3 (in tidal wetlands, seawater is a major source of HCO_3^-). The released DIC can be taken up by wetland vegetation for photosynthesis either from

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Table 1
Microbially Mediated Redox Reactions Involving Carbon That Are Ubiquitous in Aquatic Systems

Equation	Reduction	Reaction with organic matter	$\Delta G°$ (kJ eq^{-1})
1	Oxygen reduction	$1/4\ O_2(g) + 1/4\ CH_2O \rightarrow 1/4\ CO_2(g) + 1/4\ H_2O$	−120
2	Denitrification	$1/5\ NO_3^- + 1/4\ CH_2O \rightarrow 1/4\ HCO_3^- + 1/10\ N_2(g) + 1/10\ H_2O + 1/20\ H^+$	−109
3	Manganese oxide reduction	$1/2\ MnO_2(s) + 3/4\ H^+ + 1/4\ CH_2O \rightarrow 1/4\ HCO_3^- + 1/2\ Mn^{2+} + 1/2\ H_2O$	−107
4	Iron hydroxide reduction	$FeOOH(s) + 7/4\ H^+ + 1/4\ CH_2O \rightarrow 1/4\ HCO_3^- + Fe^{2+} + 3/2\ H_2O$	−57
5	Sulfate reduction	$1/8\ SO_4^{2-} + 1/4\ CH_2O \rightarrow 1/4\ HCO_3^- + 1/8\ HS^- + 1/8\ H^+$	−13
6	Methanogenesis	$1/4\ CH_2O \rightarrow 1/8\ CO_2(g) + 1/8\ CH_4(g)$	−18

Note. Organic matter here is represented with the simple stochiometric formula of CH_2O. Gibbs free energy ($\Delta G°$) is based on 25°C. Values are from Morel and Hering (1993).

the porewater or the open water column when porewater (groundwater) is discharged and converted to organic carbon (Burrows et al., 2018) or partly degassed to the atmosphere as CO_2. DIC can also be transported to the ocean where it may enter deep storage (Santos, Burdige, et al., 2021).

3.3. Redox Processes and Patterns in Wetlands

Surface waters, especially in flowing streams and shallow wave exposed waterbodies, tend to be aerobic, as they are turbulently mixed and in contact with the atmosphere. Conversely, groundwater typically has a lower oxygen content or anoxic conditions, especially in wetlands, due to the increased sediment OM content or DOC driving redox processes (Chmura et al., 2003; Guan et al., 2021; Odum et al., 1995). In coastal wetlands, the subsurface is often particularly anoxic because the sediment tends to be saturated from the ground surface and hence, there is no or a minimal unsaturated zone present (Sima et al., 2009). In contrast, drained wetlands will have greater oxygen diffusion due to the presence of an unsaturated zone, which impacts the redox sequence and mineralization pathways for OM.

Microbial activity is the driving force of the mineralization of OM via redox processes in the subsurface for a range of environments, where OM is generally oxidized to DIC, in conjunction with the reduction of various electron acceptors (Lovley & Chapelle, 1995). The oxidation of OM leads to a transition from oxic to anoxic conditions along flow paths influencing the transformation of nutrients (e.g., nitrate), contaminants (e.g., heavy metals) and carbon speciation (Appelo & Postma, 2004). These processes contribute to natural wetland services by efficiently removing organics, sulfates, and nutrients (Sima et al., 2009; Szogi et al., 2004). The electron acceptors that are responsible for OM oxidation provide varying energy yields for the micro-organisms that mediate them. Therefore, the order in which the electron acceptors are depleted occurs in a sequence from high to low energy yields (as shown in Table 1 and Figure 4).

The common microbially mediated redox sequence is oxygen reduction, followed by nitrate, manganese- and iron-oxides reduction, sulfate reduction and finally methanogenesis (Appelo & Postma, 2004). This sequence leads to a transition from oxic to anoxic conditions with a decrease in nitrate and sulfate concentrations and an increase in dissolved reduced manganese, and ferrous iron, sulfide and eventually methane for very anoxic conditions, as shown in Figure 4. These redox reactions are a source of DIC (with oxygen reduction and acetogenic methanogenesis releasing CO_2 and the remaining redox-processes producing HCO_3^-) due to the mineralization of organic matter. In some cases, this DIC release may lead to supersaturation and precipitation of secondary carbonate minerals, such as calcite ($CaCO_3$) or siderite ($FeCO_3$). While microbial denitrification may permanently remove nitrate, the mineralization of OM under anoxic conditions generally releases nitrogen (as ammonium) and phosphate, nutrients which may fertilize vegetation or algae in the surface water and is a feedback mechanism that may capture carbon.

Further attention to CH_4 formation is important as it is a potent greenhouse gas. In wetlands, it is produced microbially either by acetoclastic (e.g., acetic acid fermentation) (Equation 2) or hydrogenotrophic (e.g., by CO_2 reduction) (Equation 3) pathways:

$$CH_3COOH \rightarrow CO_2 + CH_4 \tag{2}$$

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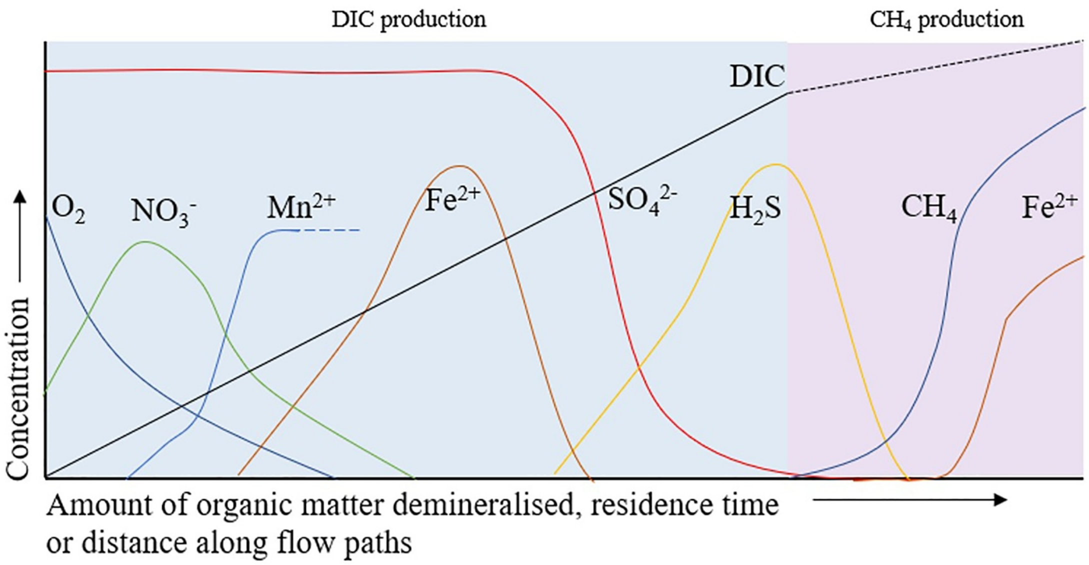

Figure 4. The redox sequence in aquatic environments. The *x*-axis may represent carbon consumed, distance along a flow path or time (Berner, 1981).

$$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O \tag{3}$$

In most coastal tidal wetlands with a neutral pH, acetoclastic methanogenesis is the dominant pathway for CH_4 production, while in freshwater peat wetlands that are acidic (pH < 5), CH_4 is predominately produced by hydrogen driven methanogenesis and the reduction of CO_2 (Conrad, 1999; Kotsyurbenko et al., 2019). However, this produced methane can be reoxidized back to CO_2 if it comes in contact with oxygen or other terminal electron acceptors like sulfate (Laanbroek, 2010):

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O \tag{4}$$

$$CH_4 + SO_4^{2-} \rightarrow HS^- + HCO_3^- + H_2O \tag{5}$$

Such methane reoxidation could be especially important for locations with recirculating flow, for example, induced by wave run-up or tidal pumping, where anoxic groundwater is mixed with oxic or sulfate containing surface water (Andersen et al., 2005). This could also be a process happening in the oxic surface water layer, or mediated by plants (Segarra et al., 2013).

Salinity differences between freshwater and seawater, and hence differences in chemical composition have implications for OM mineralization pathways. For example, sulfate concentrations are significantly higher in seawater. As methanogenesis occurs after sulfate reduction in the redox sequence, wetlands that contain seawater will only have methanogenesis commence after this additional sulfate is consumed (as shown in Figure 4). Relative to fresh groundwater, with its much lower sulfate content, the shift to methanogenesis happens only after significantly longer residence time in saline groundwater or may not occur at all. This means that groundwater from freshwater wetlands is more likely to undergo methanogenesis relative to those containing recent marine or brackish water. As such, fresh groundwater systems are likely to be a greater source of methane where they discharge to surface waters (Schutte et al., 2020).

Microbial reactions also differ between fresh and saline water as salinity can affect the microbial composition in the sediment (Chambers et al., 2013; Edmonds et al., 2009; Neubauer et al., 2013; She et al., 2016). Microbial organic matter mineralization can accelerate when freshwater ecosystems become open to saltwater with high SO_4^{2-}, leading to increased organic carbon loss from sediments (Herbert et al., 2015; Weston et al., 2011). Hence, inundating a previously freshwater wetland with saltwater, can lead to enhanced CO_2 emissions through larger sediment respiration (Dang et al., 2019). While methanogenesis is expected to reduce with the introduction of saltwater (containing sulfate), different studies have shown a variety of responses in regard to methanogenesis:

CH_4 production can decrease, recover after 1 month of saltwater intrusion, not significantly decrease, or even increase (Dang et al., 2019; Neubauer, 2013; Vizza et al., 2017; Weston et al., 2011). Clearly, more systematic studies are needed to clarify these mechanisms.

4. Impacts of Degradation on Coastal Wetlands

Coastal wetlands are increasingly threatened by a range of degradation processes, including drainage, land conversion for agriculture and development, tidal flow restrictions, and vegetation clearance (Murray et al., 2019). These alterations disrupt the natural hydrology, biogeochemistry, and ecological function of wetland systems. Among these stressors, drainage represents one of the most widespread and impactful forms of degradation, particularly in low-lying coastal areas (Glamore et al., 2021). Given its central role in reshaping hydrological and biogeochemical processes, this section focuses on the effects of wetland drainage on water flow and carbon dynamics.

4.1. Impact of Drainage on Water Flow

In drained wetlands, hydrological changes significantly impact both surface water and groundwater systems. Changes to the land surface, such as drainage, land conversion, and tidal restrictions, can lower the groundwater table and modify surface water levels. In natural wetlands, surface water typically remains for extended periods, allowing for saturated conditions. However, in drained systems, the reduced surface water retention leads to rapid runoff and reduces infiltration, resulting in less groundwater recharge. Simultaneously, the lowering of the water table shifts groundwater flow patterns, often causing a thicker unsaturated zone. This shift exacerbates soil drying and can lead to further degradation by reducing wetland connectivity with adjacent ecosystems and affecting water availability for both the wetland and nearby landscapes (Zedler & Kercher, 2005).

4.2. Impact of Drainage on Wetland Chemistry

The chemistry of drained wetlands, particularly carbon chemistry, is greatly affected by alterations in hydrology and organic matter decomposition. In a natural wetland, waterlogged conditions inhibit aerobic processes, allowing for the slow accumulation of organic matter. Drained wetlands, however, experience lower groundwater tables, causing aerobic oxidation of organic matter in the unsaturated zone, which may increase CO_2 emissions and deplete the organic matter pool (R. Cao et al., 2017; Juszczak et al., 2013; M. Zhao et al., 2020). Further, the oxidation of organic carbon, leads to moderate acidification of soils and a decrease in pH (Kayranli et al., 2010). As organic carbon is oxidized aerobically, the balance between carbonic acid, bicarbonate, and carbonate shifts toward CO_2, leading to further dissolution of carbonate minerals. The pH decline can impact microbial communities and nutrient cycling of the wetland (Allison & Treseder, 2008; Jaatinen et al., 2008; Laiho et al., 2003; Zhong et al., 2020).

In drained wetlands, the carbon dynamics are altered, particularly in terms of organic matter degradation, lateral carbon export, sediment emissions, and air-water carbon fluxes. The supply of atmospheric oxygen in the drained section above the water table creates aerobic conditions that stimulate microbial activity, leading to the release of stored carbon as CO_2, potentially resulting in increased CO_2 emissions from the unsaturated zone (Le Mer & Roger, 2001; Mustafa et al., 2024). In contrast, the drainage reduces CH_4 emissions from the unsaturated zone because methanogenesis, which requires anaerobic conditions, is suppressed (Couwenberg, 2011). However, this reduction in CH_4 emissions is often offset by a greater increase in CO_2 emissions, leading to a net rise in the overall emissions from the wetland. In addition, the disruption of the wetland's hydrological regime may remove tidal flushing and the supply of sulfate to the saturated zone, which without the sulfate inhibition on methanogenesis may increase the CH_4 emissions from the deeper saturated zones. The altered hydrology (drainage) thus impacts carbon dynamics accelerating organic matter oxidation, depleting soil carbon stocks and shifting the balance from long-term carbon sequestration to increased greenhouse gas emissions (Hu et al., 2017; Le Mer & Roger, 2001; Mustafa et al., 2024). Moreover, the altered chemistry and hydrology can enhance the release of carbon from sediments into the water column, contributing to subsequent increase in air-water CO_2 exchanges and further reducing the wetland's capacity to act as a carbon sink (M. Zhao et al., 2020).

Draining wetlands containing acid sulfate soils (common in coastal areas) exposes previously saturated sediments to oxygen, initiating the oxidation of sulfide minerals, such as pyrite, and generating sulfuric acid (Figure 5). This acidification can lower soil and water pH to below 3, leading to the mobilization of toxic metals, including

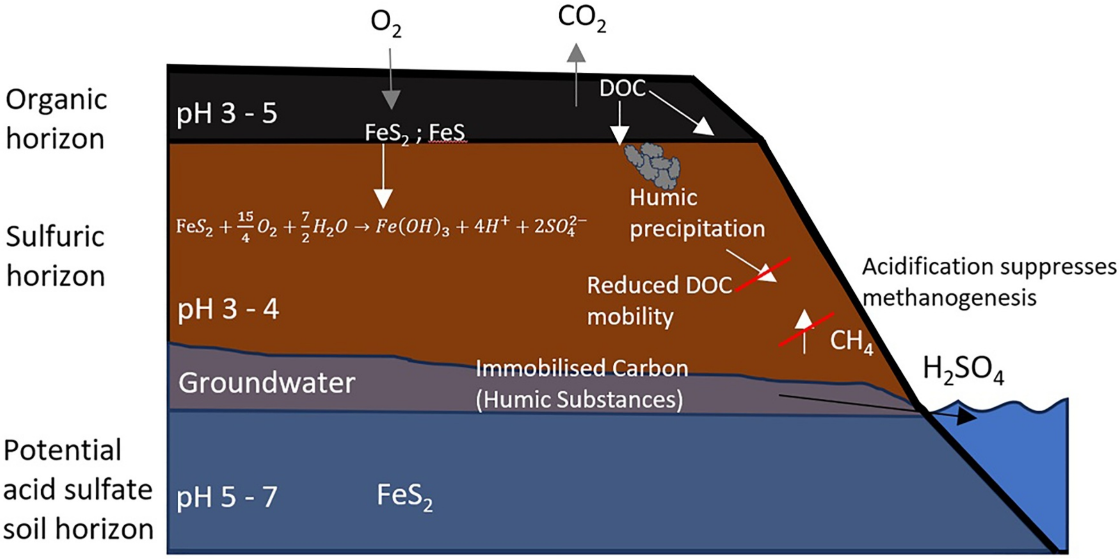

Figure 5. Conceptual diagram of a drained wetland acid sulfate soil profile showing the chemical processes within its horizons (adapted from Johnston et al. (2014)).

aluminum and iron (Johnston et al., 2005; Shand et al., 2018). These low pH levels inhibit microbial activity essential for organic matter decomposition, indirectly affecting carbon fluxes (Wu et al., 2024). Microbial processes typically function optimally between pH 6 and 7.5, but can decline by an order of magnitude when pH drops to 3 (Paul & Clark, 1996; Rousk et al., 2009).

In addition, acidification affects the mobility and potential lateral transport of DOC, as humic substances (typically the dominant fraction of DOC (Collins et al., 1986; Marhaba & Pu, 2000; Thurman, 2012)) may precipitate and adsorb to sediments at low pH (Ekström et al., 2011; Evans et al., 2012). Below pH 4, up to 50% of DOC may adsorb to mineral surfaces or precipitate (Kaiser & Guggenberger, 2000; Kalbitz et al., 2000). However, this sorption process is not unlimited as mineral surfaces can become saturated, and sorption should be considered a finite and potentially reversible mechanism rather than a long-term sink, particularly in dynamic wetland environments (Groeneveld et al., 2020; Morrissette et al., 2024). Managing water levels to maintain anoxic conditions is critical for mitigating the oxidation of sulfide minerals and aerobic OM degradation and CO_2 release.

4.3. Impact of Drainage on Subsidence

Draining wetlands can lead to subsidence as the organic-rich soils, which were formed under anaerobic and saturated conditions, become exposed to oxygen. This exposure accelerates the decomposition of organic matter, resulting in soil volume loss and elevation declines (Windham–Myers et al., 2023; Xu, Lu, et al., 2021). Sedimentation typically supports accretion in natural tidal ecosystems experiencing natural subsidence, but is often absent within drained areas, which may further intensify elevation loss (Cahoon, 2015; Cahoon et al., 2006). For instance, a study of tidal wetlands affected by hydrologic modifications, such as flap-gates, weirs, dikes, and culverts, found subsidence rates ranging from 0.10 to 1.67 cm yr^{-1} persisting for over 100 years after the changes began (Eugene Turner, 2004). These rates, while lower than those in drained agricultural freshwater wetlands, highlight the long-term impact of drainage on elevation in tidal wetlands.

5. Hydrological and Biogeochemical Responses to Tidal Wetland Restoration

Hydrological processes are potentially the most important drivers of change when drained wetlands are restored to tidal systems (Kroeger et al., 2017). While tidal restoration practices are increasingly popular worldwide (1 million km^2 of degraded land has been restored up to 2020) (Waltham et al., 2020), there is extremely limited research on how groundwater processes influence geochemical transformations, dissolved carbon export, and carbon sequestration and evasion post-restoration. As such, many important questions are unanswered regarding the impact of wetland restoration such as (a) the potential changes to subsurface geochemical processes as a site

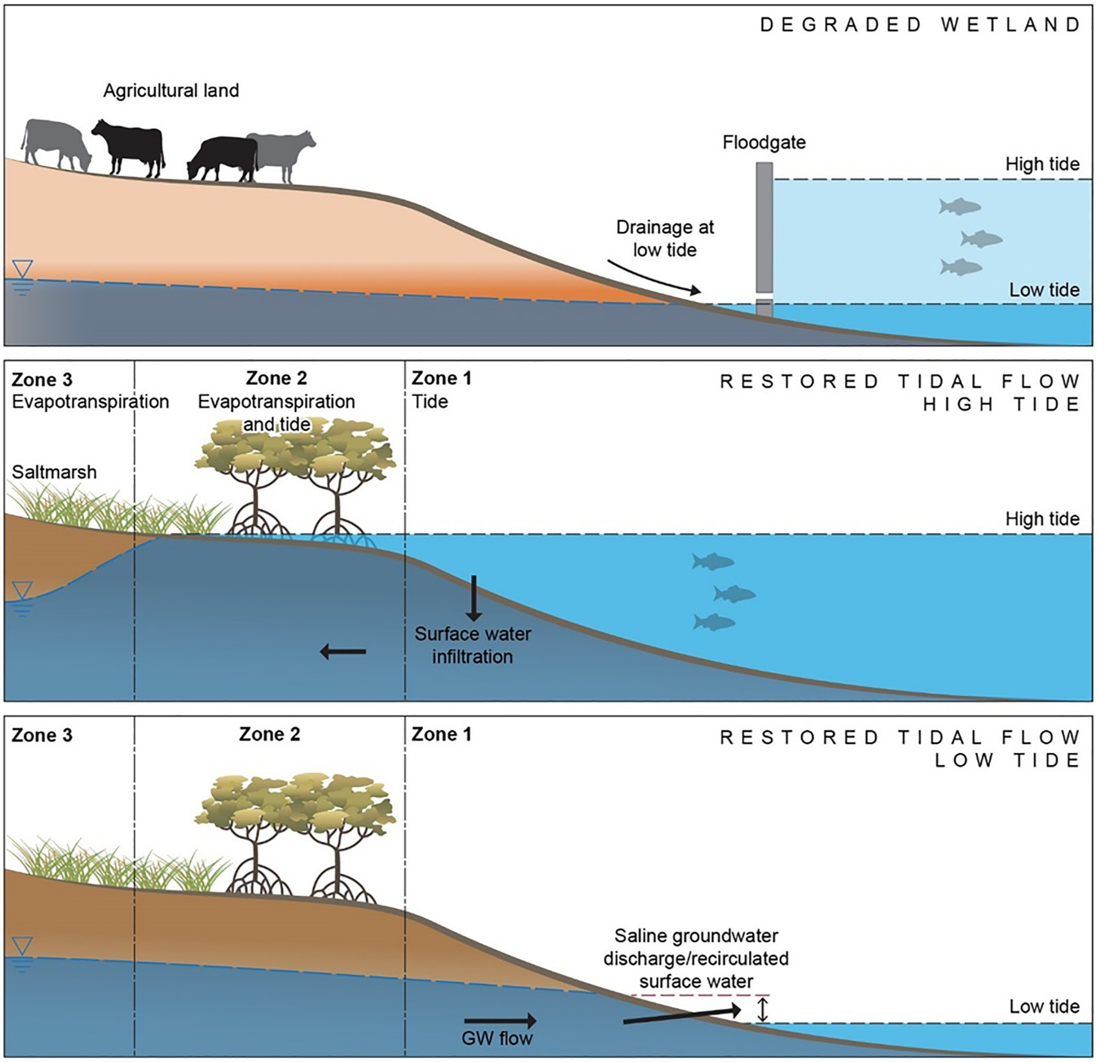

Figure 6. The influence of hydrological restoration on groundwater at a degraded wetland (top panel); restored wetland at high tide (middle panel); and restored wetland at low tide (bottom panel). Groundwater level is driven by tides in Zone 1, evapotranspiration and tides in Zone 2, and evapotranspiration in Zone 3 (adopted from A. M. Wilson and Gardner (2006)).

transition to a tidal regime, (b) groundwater-derived fluxes of carbon and greenhouse gases, and (c) carbon dynamics during different stages of restoration that is, the temporal evolution of carbon processes and carbon speciation.

5.1. Effects of Tidal Restoration on Groundwater Hydrology

Tidal restoration significantly influences groundwater hydrology and quality in coastal wetlands by modifying the interactions between surface water and groundwater, sediment water content, and chemical processes (Carol et al., 2012; Gardner, 2005; H. Li et al., 2005). Established mangrove and saltmarsh wetlands usually contain creeks or distributed drainage systems. These features play a significant role in the wetland's groundwater dynamics as the net groundwater flow is toward these features (Cabral et al., 2021; Gleeson et al., 2013; H. Zhang et al., 2014). As shown in Figure 6, the groundwater flow in restored wetlands can be categorized into three distinct zones based on dominant hydrological drivers.

Figure 6 shows groundwater dynamics in a degraded and restored wetland. In Zone 1 (Figure 6), located near the tidal boundary, restoration reintroduces tidal flow, leading to an elevated water table and enhanced groundwater–surface water interactions, such as hyporheic exchange due to tidal pumping. The tidal regime drives periodic surface water infiltration and saline groundwater discharge, contributing to increased salinity, bicarbonate, and sulfate concentrations in this zone (A. M. Wilson & Gardner, 2006; Xin et al., 2011). Zone 2 (Figure 6) represents the transitional area between the tidal boundary and the landward zone. Here, both tidal action and evapotranspiration jointly influence groundwater flow by altering hydraulic gradients. The presence of established saltmarsh and mangrove vegetation increases water circulation by increasing surface roughness and promoting localized flow pathways that facilitate mixing (Guimond & Tamborski, 2021; Xin et al., 2022). In the landward Zone 3 (Figure 6), where tidal effects are minimal, evapotranspiration and rainfall-induced recharge become the primary controls of groundwater levels. Restoration in this area stabilizes the groundwater table, as illustrated in Figure 6. These zones collectively highlight the dynamic interaction of tidal and landward groundwater processes in restored wetlands, emphasizing the importance of tidal restoration in re-establishing hydrological connectivity.

5.2. Effects of Tidal Restoration on Groundwater Salinity

In wetlands where freshwater inputs are low and evapotranspiration is high, restoring tidal flows can increase the porewater salinity of the wetland. Buchsbaum et al. (2006) reported a three-fold increase in shallow porewater salinity after tidal introduction to a restored coastal wetland in Ipswich Massachusetts, USA. Similarly, porewater salinity increased in the upper 40 cm of sediments after restoration of the natural hydrology at Pillmouth Managed Retreat site in Devon, UK (Blackwell et al., 2004). In a saltmarsh wetland restored through floodgate manipulation in south-eastern Australia, shallow groundwater (30 cm) in the wetland became hypersaline (EC > 55,000 μS/cm) 2 years post restoration due to evaporation (Sadat-Noori & Glamore, 2019). In a saltmarsh wetland on the Dutch Wadden Sea coast restored through de-embankment, salinity in groundwater increased gradually in the first years before leveling off after 4 years, with shallow groundwater (30 cm) salinization occurring more rapidly compared to deeper groundwater (120 cm) (Veenklaas et al., 2015).

5.3. Effects of Tidal Restoration on Carbon Dynamics

Degraded coastal wetlands often function as carbon sources due to the aerobic decomposition of organic matter, which releases significant amounts of CO_2. For example, studies demonstrate that restoring tidal inundation can reduce CO_2 emissions by up to 46% in saltmarshes and suppress CH_4 emissions by 30%–50% in saline wetlands (Kathilankal et al., 2008; Poffenbarger et al., 2011). Groundwater plays a critical role in this transformation (Audet et al., 2013), as the porewater salinity and water table depth of a wetland can control chemical processes and affect carbon storage and mineralization (Morrissey et al., 2014).

Figure 7 illustrates the trajectories of chemical evolution in restored wetlands as OM is mineralized. For freshwater restored wetlands the early depletion of sulfate and other electron acceptors leads to an earlier onset of methanogenesis (Figures 7a and 7b) (Koebsch et al., 2019). This corresponds to a higher potential for CH_4 emissions relative to CO_2. CH_4 production is increased due to the absence or scarcity of competing electron acceptors like sulfate, which is more prevalent in saline conditions (Segarra et al., 2013). The dominance of methanogenesis for freshwater flooding is supported by studies indicating that freshwater wetlands are typically sources of CH_4 due to limited sulfate availability (Poffenbarger et al., 2011). For a drained wetland, the gaseous oxygen supply to the unsaturated zone should be unlimited, with O_2 gas diffusion being approximately 10,000 times faster than aqueous diffusion (Appelo & Postma, 2004) and a redox sequence should therefore not form in the unsaturated zone. Hence, for the drained zone, OM mineralization would primarily be the aerobic pathway to CO_2.

Saline water intrusion into wetlands accelerates the mineralization of organic carbon, leading to a reduction in the substrate carbon pool (Kostka et al., 2002). This response is primarily driven by increased sulfate concentrations and conditions that promote the reproduction of sulfate-reducing microbes with strong carbon-degrading capabilities (Luo et al., 2019). In the short term (days to weeks), saltwater addition can stimulate microbial activity and increase CO_2 production due to the availability of sulfate as a favorable electron acceptor (Chambers et al., 2013; Weston et al., 2011). However, over longer timescales (months to years), this process may lead to substrate depletion and enzyme inhibition, ultimately resulting in lower greenhouse gas emissions (Neubauer et al., 2013). Notably, studies show that CO_2 production tends to increase with moderate salinity levels (~5–7.5 ppt), but often declines under higher salinity conditions (≥15 ppt) due to physiological stress and shifts in microbial community structure (C. Wang et al., 2017). While data from salinizing freshwater wetlands indicate overall reductions in both CO_2 and CH_4 emissions (Dinsmore et al., 2009; J. Yang et al., 2013), responses in brackish wetlands remain

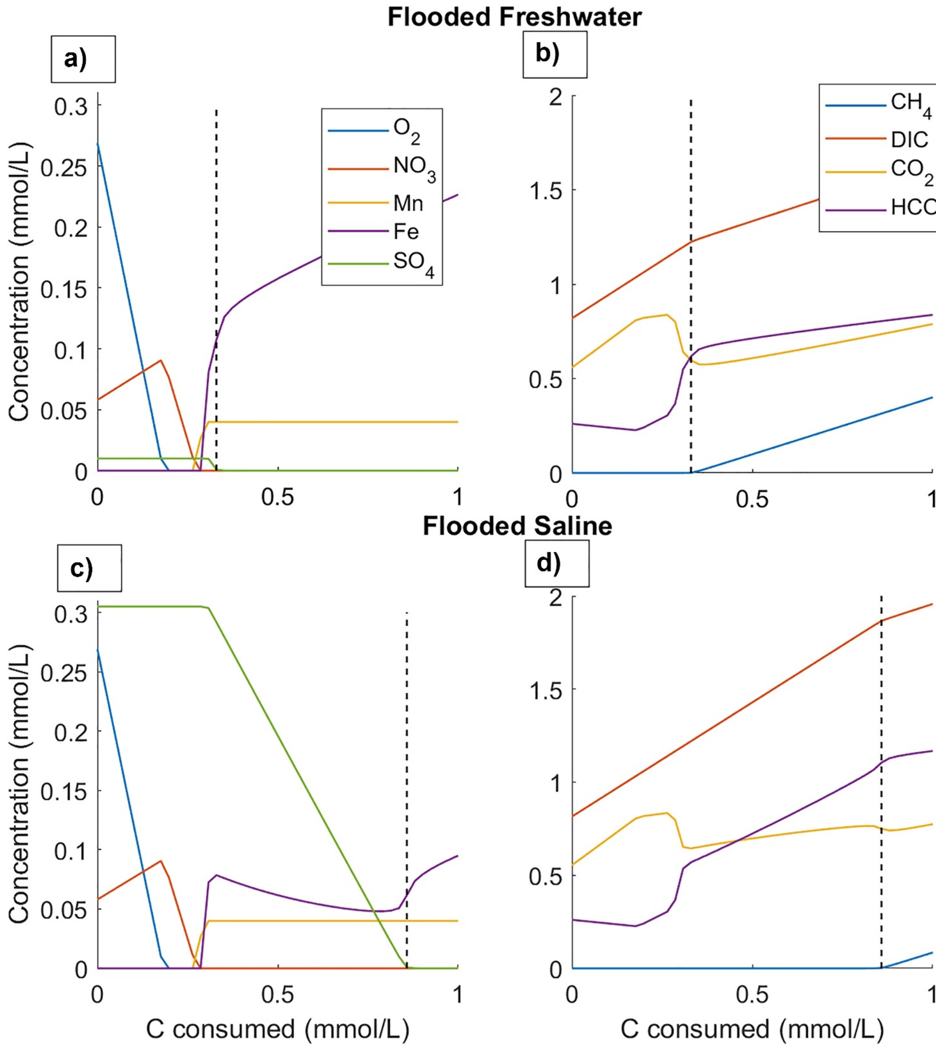

Figure 7. Redox sequences calculated for sediments flooded with freshwater and saline water, for 1 mmol/L of organic carbon mineralized, using the code PHREEQC Version 2 (Parkhurst & Appelo, 2003). The top panel (a and b) show the chemical evolution due to organic matter mineralization for a flooded freshwater wetland, while the bottom panel (c and d) show the evolution for a wetland flooded with surface seawater. The vertical dashed line indicates the onset of methanogenesis for each scenario. The scenario of sediments flooded with freshwater has an earlier onset of methanogenesis and therefore the highest potential ratio of CH_4/CO_2 emission.

less predictable and depend on initial soil chemistry, hydrology, and microbial composition (Luo et al., 2019). Nevertheless, it is well established that increasing salinity initially increases sulfate availability (particularly if it is associated with tidal restoration), stimulating sulfate-reducing pathways that dominate organic carbon mineralization in early phases of salt exposure (Keller & Bridgham, 2007; Kostka et al., 2002).

Figure 8 presents a conceptual model illustrating carbon dynamics in fresh and saline restored wetlands. In restored wetlands, inundation promotes vegetation growth and algae photosynthesis, driving the formation of biomass (Mitsch et al., 2013). The inundation generally reduces the aerobic zone near the surface where microbial OM degradation produces CO_2. CO_2 exchange with the atmosphere is limited to aqueous diffusion and gas ebullition from the sediments due to the inundation, as compared to gas phase diffusion in unsaturated soils of degraded wetlands (Craft et al., 1999). Below the aerobic zone, the inundation reestablishes a sequence of redox processes such as nitrate, manganese-oxide, iron-oxide, and sulfate reduction. The extent of each of these zones depends on the availability of electron acceptors and varies between fresh and saline wetlands (Reddy & DeLaune, 2003).

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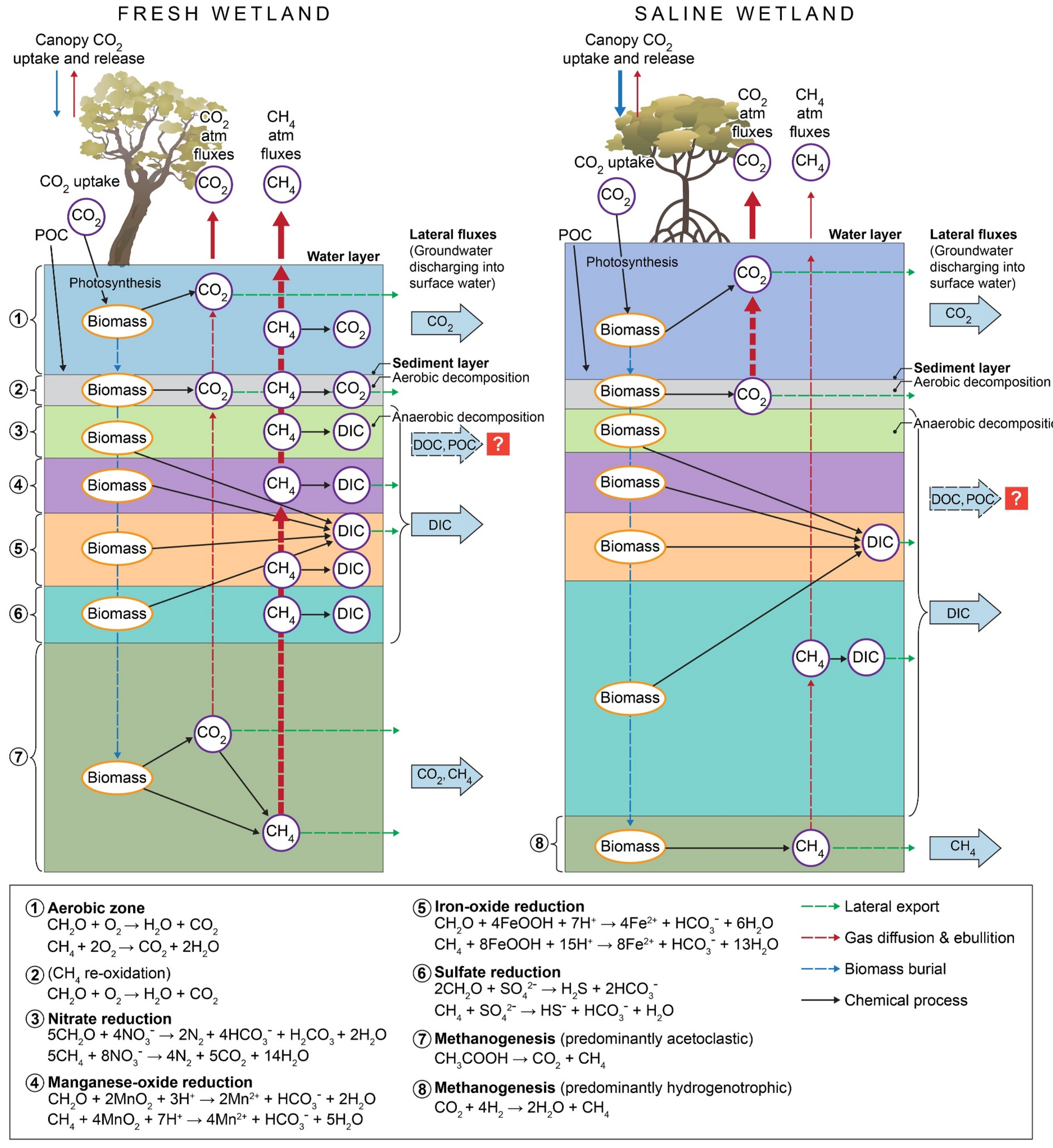


Figure 8. Conceptual model illustrating carbon cycling, redox processes, and lateral fluxes in fresh and saline tidal wetlands. The diagram highlights the interactions between photosynthesis, aerobic and anaerobic decomposition, biomass formation, and the release of carbon gases such as CO_2 and CH_4. It shows the different zones (aerobic and anaerobic), pathways of nitrate, manganese-oxide, iron-oxide, sulfate reduction, and methanogenesis. The comparison of fresh and saline tidal wetlands includes the effect of salinity on redox processes, carbon burial, and gas emissions, with an emphasis on lateral fluxes of DOC, POC, and DIC into groundwater and surface water.

In saline tidal wetlands, elevated sulfate concentrations expand the sulfate reduction zone, a dominant anaerobic process, suppressing methanogenesis relative to freshwater wetlands (Ardón et al., 2018; Poffenbarger et al., 2011). By contrast, in freshwater wetlands, the lower availability of sulfate favors methanogenesis (both acetoclastic and hydrogenotrophic), resulting in higher methane emissions. The restored tidal regime also facilitates lateral fluxes of DOC, POC, and DIC from groundwater to surface water (Bianchi et al., 2013). Some of this organic matter is buried in sediments, contributing to long-term carbon sequestration (Bauer & Bianchi, 2011).

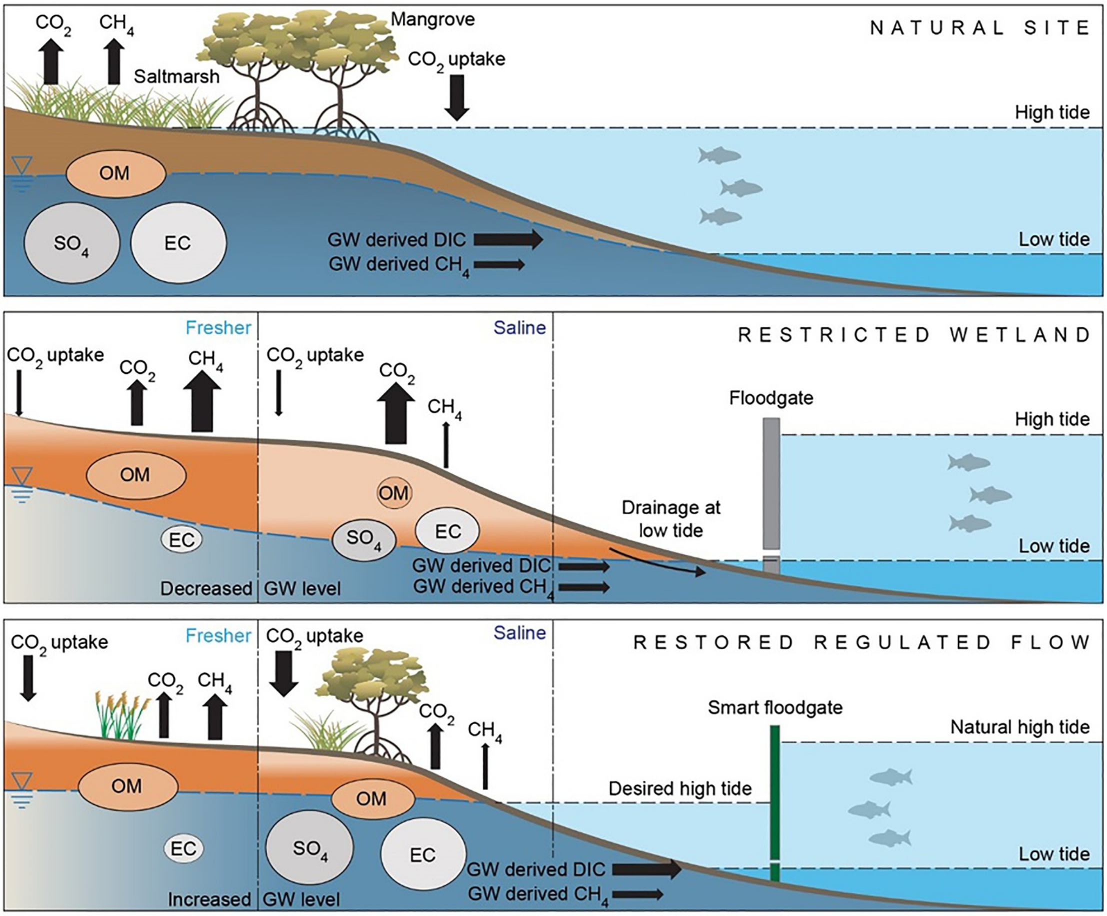


Figure 9. Conceptual illustration highlighting organic matter, groundwater/porewater salinity and sulfate in freshwater and saline tidal wetland under natural (top panel), restricted tidal flow (middle panel), and restored conditions (bottom panel). OM and EC refer to organic matter and electrical conductivity (Andersen et al., 2005; C. E. Robinson et al., 2018; Sadat-Noori et al., 2024).

In tidal systems, the wetland is generally a carbon sink due to high primary productivity (sequestration by plants). In these systems organic matter, sulfate, and salinity are present in the sediment and the system can be a limited, in relative terms, source of CO_2 and a small source of CH_4 (Olsson et al., 2015). For scenarios where the supply of sulfate may be limited (e.g., diffusion controlled transport), CH_4 emissions could still be small due to the aerobic conditions near the surface, and CH_4 reoxidation in shallow layers (Figure 9, Restricted Wetland—"Saline" panel) (Koh et al., 2009). However, it should be noted that only the part of the methane pool that rises to the surface directly by ebullition or diffusion will be re-oxidized. Methane transported by groundwater advection and discharged to surface waters, such as via a tidal channel, could largely avoid reoxidation as the contact time in the oxic environment could be short.

In wetlands with restricted tidal flow that receive freshwater from upstream catchment inflows, sulfate may decrease and if groundwater levels are high, anaerobic conditions can increase CH_4 production near the surface (Chmura et al., 2012). Concurrently, reduced salinity would lower ionic stress on plant growth and allow for higher primary productivity, increasing the sediment carbon pool through higher photosynthesis (Figure 9, Restricted Wetland—"Fresher" panel).

When tidal flows are restored to wetlands, groundwater levels, sulfate and salinity would increase. Sediment organic matter accumulates as vegetation establishes, with surface soil carbon (in upper 0–30 cm) approaching natural marsh levels within 4–15 years under optimal conditions, such as high sediment supply and rapid plant growth, although full carbon stock equivalence may take 50–100 years (Arias-Ortiz et al., 2021; Burden et al., 2013; Craft et al., 1999, 2003; Zou et al., 2022). In coastal tidal wetlands, saltmarsh and mangrove (blue carbon habitats) recruitment would increase CO_2 sequestration, while the higher water levels and lower organic matter degradation would reduce CO_2 emissions (H. Yang et al., 2020; M. Zhao et al., 2020). Kathilankal et al. (2008) reported that introducing tidal inundation into a saltmarsh in Virginia, USA caused a 46% reduction in atmospheric CO_2 fluxes compared to non-tidal conditions. In freshwater wetlands, as a limited sulfate pool from rainfall and dry deposition is consumed relatively quickly, CH_4 emissions would increase and, while the high organic matter pool can cause high CO_2 emissions in the short term, increased plant CO_2 uptake can counter this process over the long-term. Hence, tidal wetland restoration can transform a drained methane emitting freshwater system, into a carbon sequestering saltwater system, by enhancing vegetation carbon uptake and reducing microbial methane production and emission (Arias-Ortiz et al., 2021; H. Yang et al., 2020).

Reintroducing tidal inundation can increase fluid fluxes across the sediment-water interface, potentially mobilizing stored carbon in sediments, including CO_2 and other forms such as bicarbonate and dissolved organic carbon. These carbon forms may be transported to surface waters through processes like tidal pumping and subsequently exported to adjacent waterways (Z. A. Wang et al., 2016). However, a managed, restricted tidal range could reduce porewater exchange while still maintaining high water levels and porewater salinity (sulfate), which are essential for minimizing methanogenesis (Emery & Fulweiler, 2017). Optimizing the tidal range can provide a means to balance porewater exchange and water chemistry to minimize carbon exports. Nonetheless, the extent and significance of such mechanisms remain uncertain and warrant further investigation to align with broader findings in the field. To our knowledge, there are currently no documented studies that explicitly trial different tidal ranges with the objective of optimizing carbon outcomes (e.g., minimizing DOC or DIC export while maintaining high water levels and salinity to suppress methanogenesis). Most tidal wetland restoration projects focus on re-establishing full or partial tidal connectivity to restore ecological function (Sadat-Noori et al., 2021).

The rate of carbon sequestration after restoring coastal tidal wetlands (mangrove and saltmarsh) varies based on factors like the restoration method, site conditions, and sedimentation rates. For mangroves, carbon sequestration rates typically range from 4 to 10 metric tons of CO_2 equivalent per hectare per year, as shown through the large-scale restoration projects in the Mississippi River Delta, USA (Chmura et al., 2003; Duarte et al., 2013). Restored mangrove wetlands can reach carbon sequestration rates comparable to natural mangroves over time, especially when restoration improves soil organic carbon through sedimentation and plant establishment.

For saltmarshes, carbon accumulation rates are estimated at 2–6 metric tons of CO_2 equivalent per hectare per year, with higher rates observed in restored sites with active sedimentation (Chmura et al., 2003; Duarte et al., 2013). Early restoration phases often see rapid accumulation, but rates depend on factors like sediment organic content and restoration techniques. Expanding restoration efforts globally could result in the removal of approximately 841 (621–1,064) Tg CO_2 equivalent annually by 2030 (Macreadie et al., 2021). This would be achieved through a combination of preventing further emissions and increasing carbon storage via vegetation growth and soil carbon accumulation.

5.4. Effects of Tidal Restoration on Hydrological Carbon Exports

The link between the terrestrial and aquatic carbon cycle is important but remains one of the least constrained components of the global carbon cycle (Alongi, 2020). This uncertainty is primarily due to the poorly quantified flux of carbon transported by groundwater across the terrestrial-aquatic interface. These lateral subsurface flows are also among the least certain elements of the global hydrologic cycle, with significantly higher uncertainties compared to better-constrained components such as precipitation or evapotranspiration (Douville et al., 2021). Most previous research on wetland carbon has focused on quantifying sediment emissions and/or sediment carbon accumulation (carbon burial) (Atwood et al., 2017; Macreadie et al., 2017; Nahlik & Fennessy, 2016). These are also referred to as vertical fluxes. The wetland's capacity to sequester and store carbon is used as a basis to valuate wetland ecosystem services in regard to carbon (Duarte et al., 2013; Mcleod et al., 2011). However, recent research showed that coastal wetlands can export carbon horizontally (i.e., lateral surface water and groundwater fluxes), and that this carbon export, mainly driven by the wetland hydrology and tides (controlling hydraulic conductivity and gradients), can be large compared with carbon burial (Bogard et al., 2020; Maher et al., 2018; Sadat-Noori & Glamore, 2019).

Studies and estimates of lateral fluxes are limited compared to vertical fluxes, however, studies are now indicating that lateral fluxes of DIC, DOC, and POC from coastal wetlands can be high (Sadat-Noori et al., 2016; Santos et al., 2019). Some laterally (surface or groundwater) exported DIC can be a long-term carbon storage mechanism as alkalinity can stay dissolved in the ocean for hundreds of years, whilst some of the laterally exported organic carbon can subsequently be mineralized and then cycled to the atmosphere through water to air fluxes in the form of CO_2, and CH_4 (Middelburg et al., 2020; Sippo et al., 2016). This can partially offset the CO_2 sequestration benefit of wetland restoration, particularly in the case of CH_4 emissions due to its higher global warming potential compared to CO_2 (Rosentreter et al., 2018).

Groundwater discharge and porewater exchange play an important role in linking sediments to surface waters (i.e., coastal and oceanic) and the atmosphere (Guimond et al., 2025). Previous studies have reported that groundwater discharge and/or porewater exchange can be a major driver of DIC (Bouillon et al., 2008; Maher et al., 2013, 2018) and DOC exports in wetlands (Akhand et al., 2021; Cai, 2011; Dittmar & Lara, 2001; Z.-L. Wang et al., 2022). For example, a recent global study estimated that groundwater carbon export from coastal wetlands with mangrove vegetation may be the equivalent of 30% of the global river carbon inputs to the oceans (Chen et al., 2018). Re-introducing tidal flows to coastal wetlands can allow for tidal pumping, where groundwater flushes part of the sediments on every tide. Currently, as far as we are aware, no study in the literature reports lateral carbon export from restored coastal wetlands nor reports quantitative comparisons of pre- and post-restoration carbon fluxes, particularly lateral fluxes. Overall, our limited understanding of the fate of carbon following wetland restoration and the role hydrology, especially groundwater flow, plays in the processing and fate of carbon in restored coastal wetlands, hinders our ability to quantify the carbon sequestration potential of restored coastal wetlands.

6. Practical Wetland Restoration Strategies

Ecological restoration of coastal wetlands refers to activities that assist in establishing the function and structure of a site toward a functioning ecosystem complex that provides ecosystem services (Aradottir & Hagen, 2013; Han et al., 2021; McDonald et al., 2016). The three main components of a wetland comprise water, sediment and biota, and hence, restoration activities are mainly focused around these elements (Zedler & Kercher, 2005). Restoring the natural hydrology of a site ultimately drives its evolution, vegetation species composition and abundance, recovery, and functioning (Glamore et al., 2021; Montalto & Steenhuis, 2004).

The hydrology of coastal floodplains can be improved through several common remediation strategies, including drain infilling or reshaping, land raising, floodgate manipulation, decentralizing infrastructure, or full tidal wetland restoration. One of the most common strategies to restore coastal wetlands is through the effective and safe introduction of tidal flows (Burdick et al., 1996; Roman, 2012). Partial or full restoration of tidal flows can be used to establish blue carbon ecosystems, such as saltmarsh and mangroves (Elphick et al., 2015; F. Wang et al., 2021). This is typically achieved by modifying or completely removing physical water diversion and control structures or barriers restricting tidal flows at sites (Burden et al., 2013; Esteves & Williams, 2017). The hydrological processes restored or modified through these strategies directly influence groundwater levels, flow patterns, salinity gradients, and carbon dynamics, which are fundamental for maintaining wetland health and ecosystem services. Information and quantitative data on how restoration affects groundwater flow and transport is scarce. However, below we discuss various practical restoration strategies with their hydrological and biogeochemical implications. A conceptual design of each restoration method is presented in Table 2.

6.1. Drain Infilling

Drain infilling restores the natural hydrology by reducing the number of drains (Table 2—a, b), raising groundwater levels, and enhancing water retention across the floodplain (Armstrong et al., 2009). By extending the surface water residence period, the local groundwater system will have increased saturation and decreased lateral drainage. As such, groundwater flow paths are likely to change. By minimizing the surface drainage, this strategy slows groundwater discharge, allowing the water table to remain closer to the surface (Crooks et al., 2002). This may influence the wetting/drying regime of the soil matrix and influence the redox potential with depth, with implications for carbon processing and fluxes. It is also important to assess how drainage modifications may affect the existing land use and adjacent properties as water tolerant vegetation will colonize areas where the groundwater table remains high or closer to the surface for long periods of time (Cooper et al., 2014; Price, 1997). Conversely, non-water-tolerant vegetation may decline or die, potentially leading to temporary setbacks in carbon sequestration.

10.1029/2025RG000895

Table 2
Typical Remediation Strategies for Restoring the Natural Hydrology of Drained Coastal Floodplains

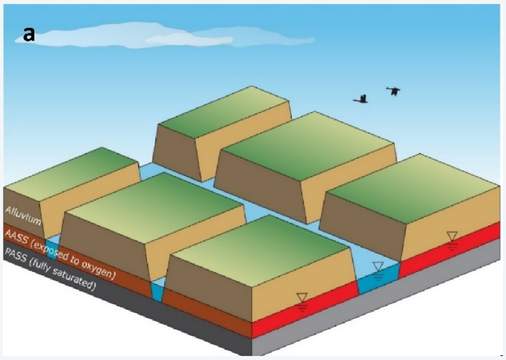
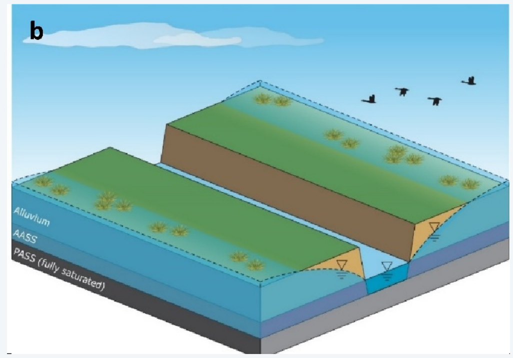
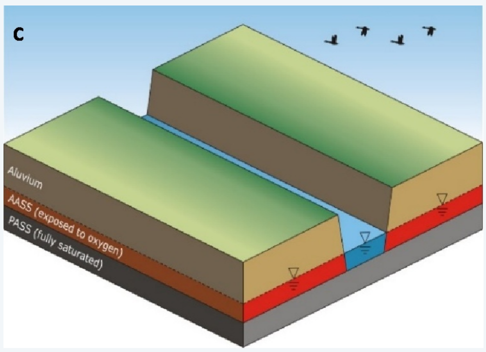
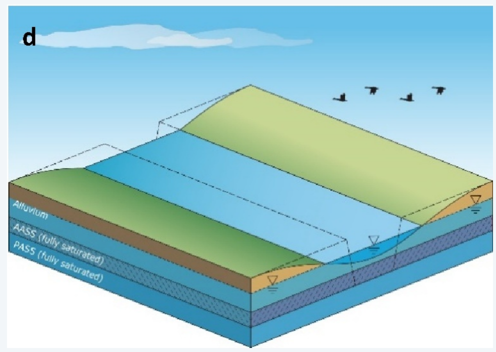
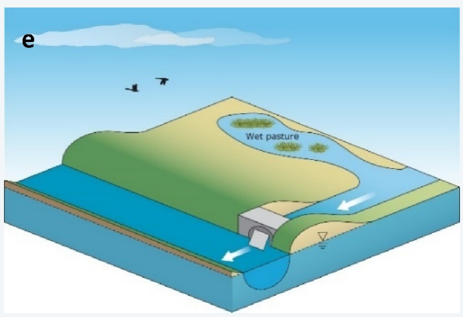
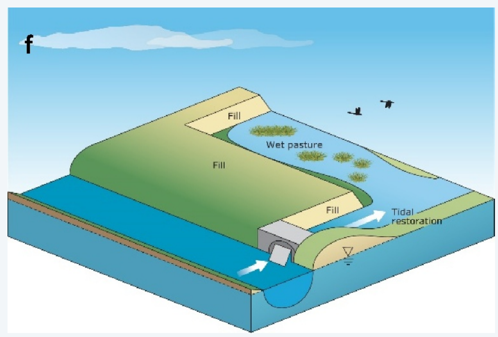
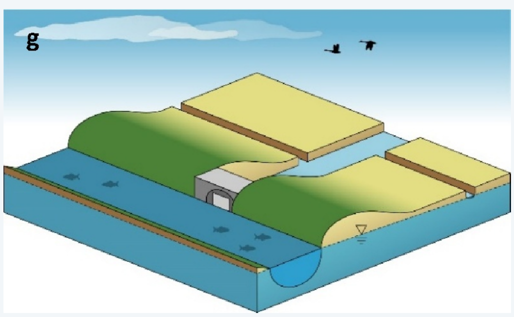
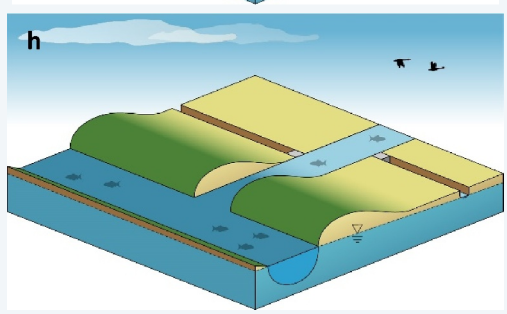
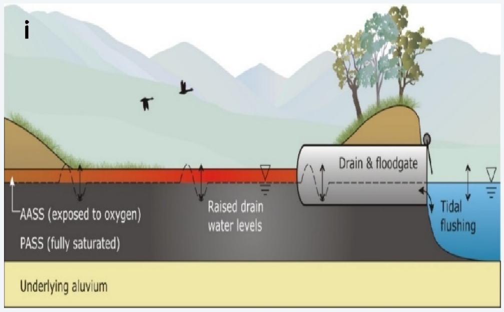
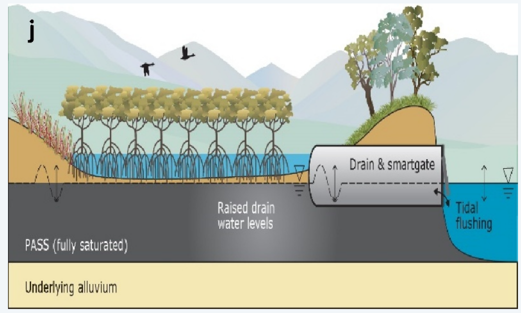
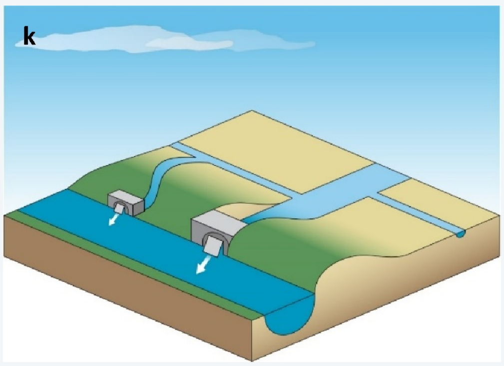
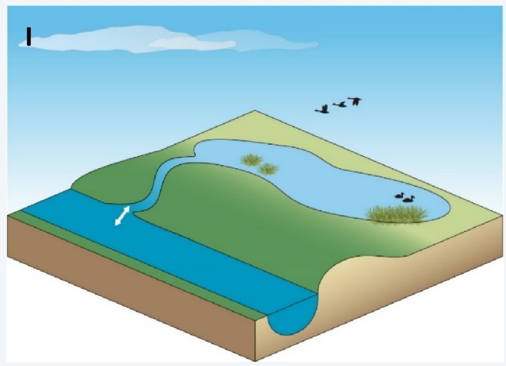
	Conceptual design	
Remediation strategy	Before	After
Drain Infilling		
Drain Reshaping		
Land Raising		
Decentralizing Infrastructure		

Table 2
Continued

Remediation strategy	Conceptual design: Before	Conceptual design: After
Floodgate Manipulation		
Full Tidal Wetland Recreation		

Note. PASS and AASS stand for potential and actual acid sulfate soils.

A well-documented example of this type of restoration comes from a Danish riparian wetland (Audet et al., 2013). Using a robust before–after–control–impact design, the researchers evaluated environmental changes following hydrological restoration involving the removal of drainage infrastructure (infilling) and rewetting of approximately 100 ha of formerly drained organic soils. Following restoration, the mean groundwater table rose by 20–30 cm, resulting in a shift from predominantly aerobic to anaerobic soil conditions and more prolonged surface saturation. This hydrological change led to a marked increase in CH_4 emissions, particularly during the growing season. CH_4 fluxes increased from near net zero (-0.04 g C m^{-2} d^{-1}) to peak values exceeding 0.5 g C m^{-2} d^{-1}, corresponding to seasonal emissions of over 31 g C m^{-2}. In contrast, CO_2 emissions from soil respiration decreased from ~1.5 g C m^{-2} d^{-1} to below 0.7 g C m^{-2} d^{-1}, reflecting reduced aerobic decomposition of soil organic matter. Concurrently, mean nitrate concentrations in shallow groundwater declined from ~5 mg NO_3^--N L^{-1} pre-restoration to <1 mg NO_3^--N L^{-1} post-restoration, attributed to higher denitrification under saturated conditions. These results highlight the complex trade-offs inherent in wetland rewetting, where reductions in CO_2 emissions and improvements in water quality may be offset by increased CH_4 fluxes, at least in the short term. The study highlights the need for long-term, integrated monitoring to fully evaluate the net climate and ecological benefits of wetland restoration.

6.2. Drain Reshaping

Shallowing and widening (or reshaping) drains (Table 2—c, d) can create a more natural hydrological gradient from floodplain to estuary while still maintaining efficient drainage from the target restoration area to the estuary (M. Zhang et al., 2009). This approach can stabilize groundwater levels, especially in acidic floodplains, and limit acid discharge into adjacent waterways (Johnston et al., 2003). By reducing groundwater drawdown, reshaped drains promote the persistence of wetland vegetation and associated carbon sequestration. The slower flow velocity also enhances sediment deposition, contributing to the accretion of carbon-rich soils. This strategy can be implemented without changes to existing land uses.

In contrast to drain infilling, drain reshaping can still influence vertical drainage of the subsoils and groundwater flow toward the drain, particularly at low tide. Ideally, drain reshaping is designed to facilitate the drainage of topmost organic soil strata while minimizing drainage of deeper acidic zones. By reducing the overall drawdown from a shallow drain compared to a deeper traditional drain, this approach reduces the drain's influence on the groundwater system. The primary objective of this restoration strategy is to limit pyrite oxidation and acidification while maintaining conditions that support carbon sequestration. Unlike conventional drainage, which can lead to prolonged exposure of carbon-rich topsoil to oxygen, drain reshaping aims to sustain periodic saturation, thereby reducing aerobic decomposition and CO_2 emissions from this zone.

An example of this type of restoration is the Tuckean Swamp floodplain on the Richmond River, NSW, Australia, which underwent drain reshaping to address acid sulfate soil impacts from extensive drainage works since the 1880s. By shallowing and widening drains, the restoration project created a gentler hydrological gradient, stabilizing groundwater levels and reducing acid discharge (previously pH < 3) into waterways (Johnston et al., 2003). Slower flow velocities promoted sediment deposition, enhancing carbon-rich soil accretion and supporting wetland vegetation recovery, such as sedges and saltmarsh species. Water quality improved with reduced trace metal discharges, while tidal flushing in reshaped drains aided restoration. The approach allowed continued agricultural use, like sugarcane farming, with minimal disruption. Challenges included managing monosulfidic black ooze and optimizing drain designs through numerical modeling, emphasizing the need for site-specific planning and ongoing monitoring (Johnston et al., 2003). Changes in carbon fluxes (CO_2 and CH_4) from the site were not reported.

6.3. Land Raising

Land raising refers to the targeted elevation of upland or adjacent transitional areas around inundated marshes through the addition of fill or reshaping of the ground surface (e.g., construction of levees or berms; Table 2—e, f). Unlike large-scale sediment addition projects aimed at reversing subsidence across entire wetland basins, this approach targets specific zones to improve local hydrological function. The primary motivations include improving water retention in priority areas, maintaining elevated groundwater levels, limiting the encroachment of saline or brackish water, and supporting diverse vegetation assemblages (Jing et al., 2008; Tweedy & Evans, 2001; M. Zhang et al., 2009). The engineering design will typically be dependent on various site-specific requirements, including existing water flow paths, inundation extent, property boundaries, etc., as well as the desired ecological outcomes (Jarzemsky et al., 2013). Elevated areas are better suited to supporting diverse vegetation. However, the addition of sediment must be carefully managed to avoid disrupting existing hydrological flow paths and nutrient dynamics. This approach is most common in locations where rivers are starved of sediment influx and subsiding such as in the lower Mississippi River delta. As the influence of saline or brackish water inundation is prevented or diminished by such a restoration strategy, methanogenic organic matter decomposition may be favored under low redox conditions.

A notable example of land raising for wetland restoration is the UW Bothell Wetland Restoration Project in Washington State, USA (Allard, 2019). The project involved adding up to 24 inches of fill to elevate specific upland areas and create a gentle slope across the site. This topographic modification aimed to improve hydrological function by increasing water retention and reconnecting the local creek (North Creek) with its floodplain. Although quantitative data on post-restoration groundwater levels are limited, the design was intended to promote saturated soil conditions and maintain higher groundwater levels supportive of diverse wetland vegetation (Allard, 2019). Information on carbon dynamics from the site was not provided in this study.

6.4. Decentralizing Infrastructure

Replacement of large flow control structures (e.g., floodgates and weirs) with smaller structures at strategic locations upstream (Table 2—g, h) can open creeks and drainage channels to tidal flushing, fish passage and create a semi-natural flow regime (J. Zhang & Sun, 2008). This approach restores hydrological connectivity, increasing groundwater-surface water exchange and facilitating the management of saline inflows across the floodplain. Decentralizing infrastructure can influence groundwater in a similar way to the installation of shallow drains, including minimizing the thickness of the unsaturated zone. The reintroduction of saline or brackish tidal water promotes the formation of salt-tolerant (blue carbon) vegetation. Deeper saline groundwater flow paths can be created by processes such as tidal pumping and saltwater intrusion (by density-driven flow).

An example of this type of restoration is the Oosterschelde Estuary, in the Netherlands, where a few large floodgates were replaced with several smaller tidal control structures to restore hydrological connectivity in a reclaimed agricultural wetland (Nienhuis & Smaal, 1994). These decentralized smaller structures enabled tidal flushing and fish passage, reducing the unsaturated zone thickness and promoting saline groundwater flow. This fostered salt-tolerant vegetation, potentially increasing blue carbon sequestration. The restoration resulted in water quality improvement and ecological benefits were achieved through restored creek connectivity. A particular challenge included managing saline intrusion to protect adjacent farmland (Ysebaert et al., 2016). The study did not report carbon fluxes from the restored site.

6.5. Floodgate Manipulation

One-way floodgates prohibit tidal inundation, maximize drainage, and maintain drain water levels at low tide elevations (Table 2—i, j). Floodgate manipulation to allow tidal flushing is typically undertaken to allow tidal inflows while limiting the tidal amplitude. This approach limits impacts on upstream land use. The installation of automatic tidal gates may assist in controlling tidal flushing to a predetermined flushing regime based on design criteria (Sadat-Noori et al., 2021). Maximum inundation elevations are usually dependent on the topography of the backswamp or the lowest elevation of assets of importance (Glamore et al., 2021; Rayner & Glamore, 2010).

Manipulating floodgates allows some tidal inflows to achieve hydrological restoration by increasing soil water saturation, decreasing soil organic matter oxidation and restricting in-drain acidity by buffering with tidal seawater exchange (Glamore, 2012). Restoration of tidal inundation via floodgate manipulation can have a large impact on the groundwater regime but strongly depends on the driving hydraulic head and the saturated hydraulic conductivity of the soil. Very low hydraulic conductivity values may result in no appreciable impact from tidal restoration, with salt transport restricted to diffusion only, whereas higher hydraulic conductivity values may result in tidal flushing of groundwater and consequently geochemical changes at considerable distances from the drains (Glamore & Indraratna, 2009). It should be noted that insufficient tidal exchange can reduce sulfate availability, favoring methanogenesis under prolonged anaerobic conditions.

An example of this type of restoration is Tomago Wetlands, NSW, Australia, where temporal floodgate manipulation was used to restore tidal flushing in a drained coastal wetland. Automatic tidal gates were installed to allow controlled tidal inflows while limiting impacts on upstream agricultural land (Glamore et al., 2021). The temporal control of the flushing facilitated groundwater-surface water exchange, introducing saline tidal water and reducing the unsaturated zone thickness (Sadat-Noori & Glamore, 2019). Additionally, salt-tolerant vegetation, such as mangroves, established, supporting blue carbon ecosystems (Rankin et al., 2023). A recent study at Tomago Wetlands revealed that surface water and groundwater carbon exports from the restored wetland were lower than those reported for natural wetlands, 4 years post-restoration (Sadat-Noori et al., 2024).

6.6. Full Tidal Wetland Creation or Restoration

Full tidal wetland restoration (Table 2—k, l) represents a comprehensive approach, where drainage networks are redesigned to allow regular tidal inundation (Goodwin et al., 2001). This strategy significantly raises groundwater levels (toward mean sea level at the boundary vs. low tide level prior to restoration—Figure 6a) and strongly alters the groundwater flow paths, with implications for the oxidation/reduction potential of the subsoils. It can also contribute to maintaining saturated conditions that favor water- and salt-tolerant vegetation, such as mangroves and saltmarsh (Lewis, 2005). Restored tidal wetlands can produce near net-zero greenhouse gas emissions if water levels are kept near the surface of the land (Zou et al., 2022).

An example of this type of restoration is the Nisqually River Delta, Washington, USA, which underwent full tidal wetland restoration by removing 8 km of dikes and redesigning drainage networks to allow regular tidal inundation across 370 ha (Woo et al., 2021). This raised groundwater levels to near mean sea level, resulting in increased saturation that supported saltmarsh vegetation. The restoration site had sediment carbon accumulation rates (81–210 g C/m^2/year) similar to a reference marsh site (115–168 g C/m^2/year), driven by allochthonous carbon contributions from nearby marshes and riverine inputs (Ellings et al., 2016).

10.1029/2025RG000895

7. Recommendations and Future Work

Despite considerable research efforts aimed at understanding groundwater physical and geochemical processes in wetlands, there is a need to investigate the role of groundwater in carbon transformation and transport in restored wetlands. Therefore, we recommend that future research into restoration projects focus on the following research topics:

1. *Integrated groundwater and surface water monitoring strategies*. Many studies on wetland restoration narrowly focus on either surface water or groundwater aspects of the wetland. However, groundwater and surface water are interconnected, and an understanding of their interaction is required to develop better insights into their combined effect on carbon movement in wetlands post restoration. Future studies should be designed to consider both surface water and groundwater hydrology and their interactions through different stages of restoration projects.
2. *Groundwater level and salinity monitoring*. Wetland studies reporting long-term groundwater level fluctuations and salinity post restoration are scarce in the literature. The water table and salinity in a wetland control some of the key wetland processes that affect vegetation growth and sediment carbon storage, respiration and solute transport. Future studies should consider long-term monitoring programs to document the temporal evolution of restoring wetlands.
3. *Biogeochemical trajectories of restoring wetlands*. Changes in surface and subsurface flow paths and residence times, as well as sources of water (fresh vs. saline) and their chemical composition, can be a significant driver of changes to subsurface biogeochemical reactions, altering the rate and pathways of carbon transformation. Further knowledge is required on how particular restoration designs can influence biogeochemical change, but also how such changes may vary over time as a restoring wetland matures and adjusts to changes. As vegetation development and groundwater flow and solute transport are slow processes, changes from a degraded to a restored state may take several decades. Some beneficial carbon sequestration processes or unwanted side-effects may be transient and not reflect the behavior of a fully restored wetland. Consequently, further knowledge is needed about how carbon fluxes and carbon speciation change over the entire restoration trajectory.
4. *Quantifying groundwater driven carbon and greenhouse gas fluxes*. As wetland restoration projects scale up worldwide, more studies need to focus on determining the fate of carbon in wetland sediments post restoration. Estimates of dissolved and particulate carbon export from restored coastal wetlands after restoration activities are scarce. Additional studies focusing on quantifying the role of groundwater flow as a pathway for transporting carbon from land to ocean are required. The combination of surface water and groundwater carbon exports would help develop a complete carbon budget for restored wetlands.
5. *Geographical bias*. Most studies related to groundwater and wetland processes are geographically focused on the east coast of the US, with China having the second most studies, both located in the northern hemisphere and largely without mangroves. Wetlands around the globe vary significantly in stratigraphy, tidal regimes, geomorphology, plant species and fauna. Additional studies on groundwater dynamics and carbon movement from various areas, climates and latitudes, especially in the southern hemisphere, are essential to develop more robust global carbon budgets and refine the understanding of hydrogeological drivers of carbon dynamics in restored wetlands.
6. *Climate change and sea level rise*. Quantifying the effects of rising sea levels and a changing climate on groundwater flow in wetlands, solute transport and biochemical cycles are required to identify climate driven feedback mechanisms and to understand their effect on carbon in restored wetlands.
7. *Microbial drivers and influences on carbon cycling in restoring coastal wetlands*. Limited research has been conducted to assess how microbially controlled organic matter reactions may alter as restoring coastal wetlands transform from drained and acidic to inundated and sulfur-rich. Further research could be undertaken to track these processes with depth, over time, and along restoration trajectories.
8. *Development of systematic, mechanistic and quantitative understanding of wetland processes*. Future studies should consider developing models that couple physical (hydrological and solute transport) and biogeochemical processes (carbon transformations) which can be used to aid the design of wetland restoration schemes and to predict or optimize their carbon sequestration.

Data Availability Statement

The data on which this article is based are available in De Groot et al. (2020a, 2020b), Kayranli et al. (2010), Berner (1981), Stumm and Morgan (2012), Johnston et al. (2014), A. M. Wilson and Gardner (2006), Parkhurst and Appelo (1999), Andersen et al. (2005), C. E. Robinson et al. (2018), and Sadat-Noori et al. (2024).

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