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Wetland topography drives salinity resilience in freshwater tidal ecosystems

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ABSTRACT

The restoration and creation of tidal freshwater wetlands is increasingly becoming important, yet the success of these efforts is limited by salt intrusion, a growing concern due to climate change and human activities. Key topographical features, such as (re)constructed channel network, might help mitigate salt intrusion in these areas. Using a hydrodynamic model and idealized topographies based on real-world data from natural marshes and various constructed wetlands, we analysed how topographies respond to saltwater intrusion events. Our findings reveal that, although wetland topographies based on natural marshes experience faster salinity increases at the onset of an event, they also achieve quicker salinity reductions at its conclusion, resulting in shorter overall periods of salinization compared to artificial wetland designs (e.g. up to 8.10 % in the drought simulations and 48.72 % in the storm surge simulations). The rapid reduction in salinity is driven by the distinct topography of natural marshes, particularly the creek system, which amplifies salt fluxes. Compared to the reference topography, the natural marsh topography exhibited 6.50 % higher salt fluxes in drought (S + W) simulations and up to 41.02 % higher in storm surge (S + W) simulations. These findings emphasize the importance of incorporating natural marsh characteristics, such as slope and channel network design, into tidal freshwater wetland restoration and creation projects to improve resilience against salt intrusion and ensure their long-term sustainability in the face of climate change.

1. Introduction

The restoration and creation of estuarine wetlands is gaining momentum as these wetland ecosystems have proven resilient to sea level rise and offer a wide array of ecosystem services (Barbier et al., 2011; Costanza et al., 1997; Van Coppenolle and Temmerman, 2019). One particular type of estuarine wetland is tidal freshwater wetlands, located in the freshwater part of the estuary that is still influenced by ocean tides (Barendregt and Swarth, 2013; Odum, 1988). These wetlands are unique because, although they contain freshwater, their hydrology and ecological processes are shaped by tidal movements rather than saline conditions. Often situated near highly populated areas, these wetlands face significant anthropogenic pressures, leading to their degradation and loss (Barendregt and Swarth, 2013; Little et al., 2022; Lotze et al., 2006). Despite their rarity, tidal freshwater wetlands remain rich in biodiversity and provide essential ecosystem services (Barendregt and Swarth, 2013; Odum, 1988). For example, estuarine wetlands can reduce storm surges, improve water quality, and support high biodiversity (Havrdova et al., 2023; Temmerman et al., 2013; Willemsen et al., 2020). They are also cost-effective flood defence solutions compared to grey infrastructure, providing additional benefits such as carbon sequestration (Narayan et al., 2016; Turkelboom et al., 2021; Turner et al., 2007; Vuik et al., 2019). These areas are resilient to climate change and can adapt with sea level rise, provided there is sufficient sediment supply (French, 2006; Liu et al., 2021; Temmerman et al., 2013). Restoring these habitats is essential to counteract climate change

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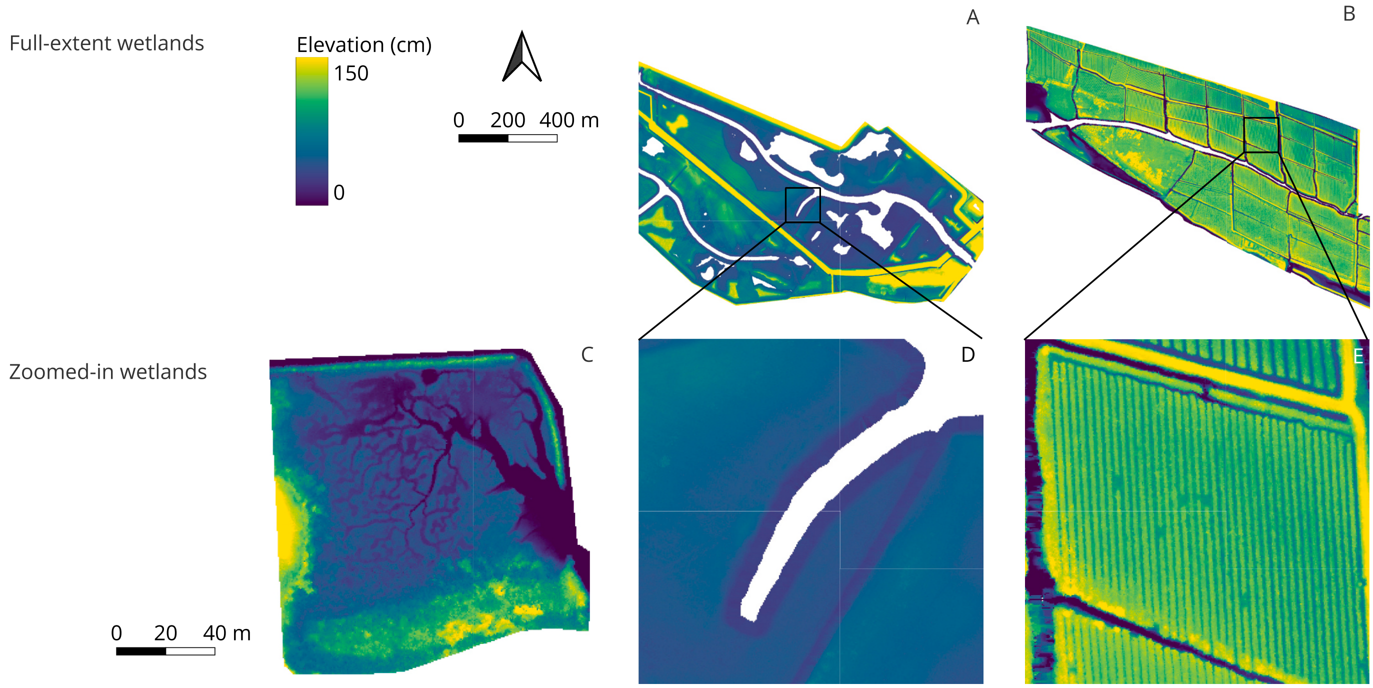

Fig. 1. Elevation of three of the selected wetlands based on the Digital Elevation Model. A, B: full extent of the de-embanked area (A) and the willow plantation (B). C, D, E: natural freshwater marsh in its full extent (C) and a zoom-in of the de-embanked area (D) and the willow plantation (E). White areas are areas that are too deep to have accurate results in the DEM.

and biodiversity loss (Zhao et al., 2016).

An emerging threat to these tidal freshwater wetlands, arising from climate change and human activities, is salt intrusion. Salt intrusion is a growing global concern, advancing upstream along estuaries due to a combination of climate change and human activities (Costa et al., 2023; Lee et al., 2024). Climate change contributes to salt intrusion by increasing sea levels, the frequency and intensity of droughts - which reduce freshwater river discharge - and storm surges – which push seawater further upstream into estuaries (Intergovernmental Panel on Climate, 2023; Jones et al., 2024; Lee et al., 2024). Human alteration of rivers, such as dredging, groundwater extraction, and river diversion, further exacerbate the problem (Herbert et al., 2015; Siemes, 2024; Tully et al., 2019). These factors collectively reduce freshwater input and thereby increase the pressure from the sea, pushing saltwater further upstream (Geyer and MacCready, 2014; Gong and Shen, 2011; Hendrickx et al., 2023; Monismith et al., 2002). This imposes an additional danger to the already threatened tidal freshwater wetlands, possibly leading to increased mortality of freshwater organisms and setting the stage for shifts in functionality and biodiversity (Ardon et al., 2013; Craft et al., 2009; Herbert et al., 2015; White and Kaplan, 2017).

A critical aspect of the impact of salt intrusion on organisms is its duration, which determines the resistance and recovery of wetland communities. The length of saltwater exposure is crucial, as some species can withstand and recover from short salinity exposure, but may die if exposed for longer periods (Flynn et al., 1995; Hootsmans and Wiegman, 1998; Li and Pennings, 2018b). Recovery of wetlands after temporary salt intrusion is possible, with rates depending on post-intrusion salinity and inundation levels (Flynn et al., 1995). As salt intrusion progressively affects freshwater wetlands, increased saltwater exposure could have a negative impact on the already threatened tidal freshwater wetlands. New strategies are therefore needed for the conservation of restored and created tidal freshwater wetland ecosystems. Instead of striving to completely exclude saltwater from wetlands, a more cost-effective strategy might involve acting on the duration of salt intrusion within the wetlands.

One potential approach to mitigating salt intrusion in tidal freshwater wetlands is through topographical modifications. Topographic characteristics, such as the initial creek network and site elevation, are essential for the success of wetland restoration, as they dictate vegetation colonization, water flow, flood attenuation, and the morphological evolution of the wetland (Adam, 2019; Brooks et al., 2015; Chirol et al., 2024; Lawrence et al., 2018; Mossman et al., 2012). For example, narrower and shallower channels lead to higher storm attenuation rates (Stark et al., 2015). Many authors suggest digging channels with morphological properties similar to those of natural mature system to help these wetlands reach a mature state more quickly and achieve the same level of ecosystem services and biodiversity as natural wetlands (Adam, 2019; Chirol et al., 2024; Lawrence et al., 2018; Zeff, 1999). While proper channel design is essential in restoring or creating wetlands (Adam, 2019; Zeff, 1999), its impact on the duration of salt intrusion and, consequently, on vegetation communities, is not yet fully understood.

This study examines how topographical features, particularly channel network design, influence the duration and intensity of salt intrusion in tidal freshwater wetlands and their impact on freshwater ecosystems' resilience. Using the hydrodynamic model Delft3D Flexible Mesh (Kernkamp et al., 2011), we simulated saltwater intrusion across four distinct wetland topographies under different scenarios. The selected topographies represent a range of wetland types commonly found along Belgian, Dutch, and British estuaries, including i) natural marshes, ii) de-embanked areas, and iii) willow plantations, as well as iv) a reference area without topographical features. We modelled two contrasting salt intrusion scenarios: one characterized by drought/low river discharge, leading to a prolonged event with lower salt concentrations, and another driven by a storm surge, marked by a shorter duration but higher salt concentrations. While previous studies have compared the development and functioning of restored wetlands with natural wetlands (Chirol et al., 2024; Gourgue et al., 2022) or examined the flood attenuation capacity of different engineering approaches (Kiesel et al., 2020; Stark et al., 2016), research on the effects of various restoration and design strategies on wetland salinization, as well as their comparison with a natural marsh, remains largely lacking. Our goal was to identify which

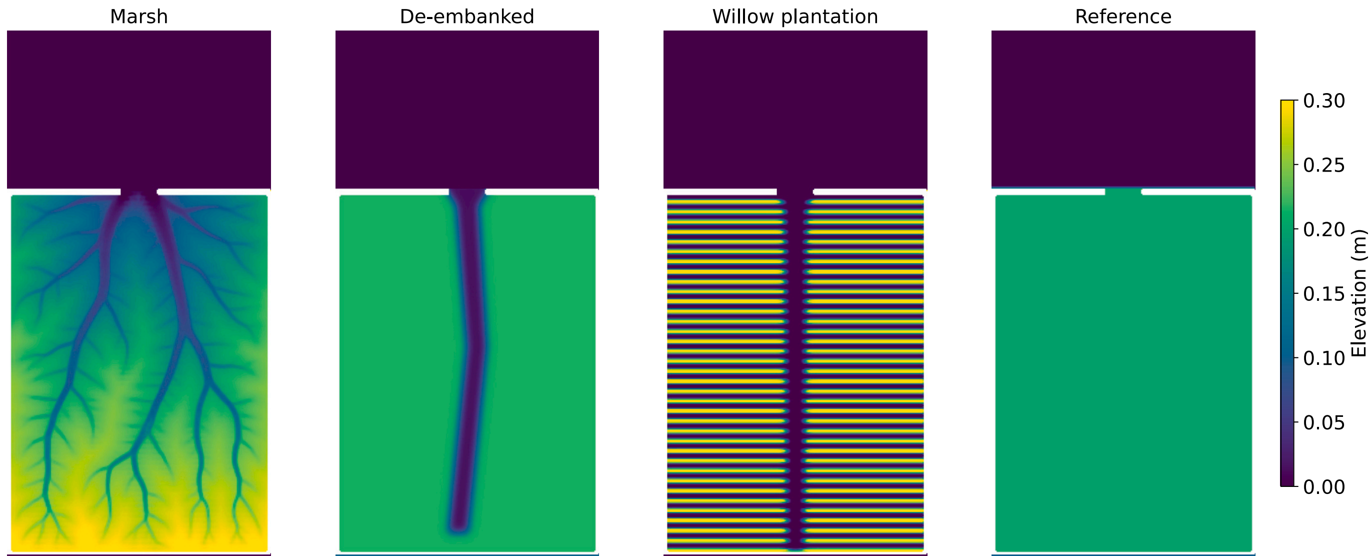

Fig. 2. Simulated wetlands. From left to right: natural tidal marsh, de-embanked, low willow plantation, and reference case. The high willow plantation is exactly the same as the low willow plantation but 0.60 m higher. The dikes have been removed from the map to help visualize the different topographies better.

topographies best minimize the duration and intensity of salt intrusion, reducing stress on freshwater organisms. The findings provide practical guidance for designing and managing wetland restoration projects, helping to create wetlands that are more resilient to climate change and capable of maintaining long-term ecological functioning.

2. Methods

2.1. Wetland types used as case study

For our modelled landscape topographies, we used average wetland characteristics extracted from a sample of real-world natural and artificial wetlands to create generic topographies representative of freshwater estuarine landscapes. To achieve this, we selected eight wetlands along the Rhine-Meuse estuary (The Netherlands) and extracted their general characteristics, i.e. area, elevation and channel dimensions, from a Digital Elevation Model (as explained below). These values were then used to design the idealized wetlands implemented in the hydrodynamic model.

We categorized the eight wetlands along the Rhine-Meuse estuary into three topographies: one naturally developed freshwater marsh, three de-embanked restoration areas, and four former willow plantations (Fig. 1). Due to the highly anthropized environment, we found only one small natural marsh in the whole area, which was significantly smaller in extent compared to the other artificial wetlands (Fig. 1). Natural marshes develop a self-organized creek network as they accrete vertically from mudflats. This process involves the creation of creeks through sediment accumulation and erosion, mediated by vegetation (Chirol et al., 2024; Temmerman et al., 2007; van Wesenbeeck et al., 2007). In contrast, restored or created de-embanked areas often involve breached dikes and excavated channels to facilitate creek network development (Chirol et al., 2024). Many of these areas are created from former agricultural land and typically have flat elevations with only one or few central channels dug (Brooks et al., 2015; Chirol et al., 2024). Willow plantations arose in Belgium and The Netherlands in the 14th century when humans started replacing pre-existing forests along estuaries with willows for basketry and their use in land reclamation (Paalvast and van der Velde, 2014; Struyf et al., 2009). Willow plantations have distinctive topographies, with one or more main ditches connected to a series of perpendicular smaller ditches, and willows planted in rows along the ridges between these ditches. This system, designed to drain riparian areas and flood forests along the river, is now primarily protected as a nature reserve, and in some occasions willow plantations are maintained for conservation and cultural heritage purposes. However, these areas typically show low biodiversity and are no longer economically profitable (Paalvast and van der Velde, 2014; van den Bergh et al., 2009).

The selected wetlands were manually delimited using QGIS (QGIS.org 2023). Digital elevation models (DEM) for these areas were downloaded from the Actueel Hoogtebestand Nederland website (AHN3, https://ahn.arcgisonline.nl/ahnviewer/), with no-data values within a 10-pixel radius filled in and then clipped to the wetlands' extents. For each area, we calculated the extent and mean elevation (Fig. 1 and Table S1). To create simple rectangular wetlands for the model, we extracted small rectangular subsections of the willow plantations and the restored wetlands. The natural marsh was significantly smaller in extent compared to the other wetlands; therefore, we did not consider this wetland when calculating the extent to be used for the idealized wetland topographies, but developed a simulated natural wetland topography. For the willow plantations, we calculated the average distance between ditches and used this value in the simulated wetlands. For the de-embanked areas, we calculated the width of the central channel. By dividing the channel width by the wetland width and averaging, we obtained an average channel:wetland ratio to design our idealized de-embanked wetland.

While the DEM employed an elevation in NAP (Normaal Amsterdams Peil), the model used the local mean water level as boundary conditions. For this reason, we transformed all elevation data from NAP to mean water level, using water level data recorded by Rijkswaterstaat (https://waterinfo.rws.nl/). We selected three measuring points closest to the selected wetlands: Spijkenisse (51°52′16"N 4°19′51″E), Goidschalxoord (51°49′49"N 4°27′06″E), and Krimpen a/d Lek (51°53′25"N 4°37′39″E). Data from these locations, recorded from 01/02/2008 to 21/03/2023 at 10-min intervals, were used to determine the local mean water level. This local mean water level was then applied to convert the DEM values from NAP to mean water level, facilitating comparisons between sites with varying mean water levels (Table S1).

2.2. Simulated wetlands

The observed wetland characteristic values were used to create the idealized wetland topographies. As this study focussed on the influence of different topographies on salt intrusion, including the effect of contrasting vegetation communities present in the selected wetlands could

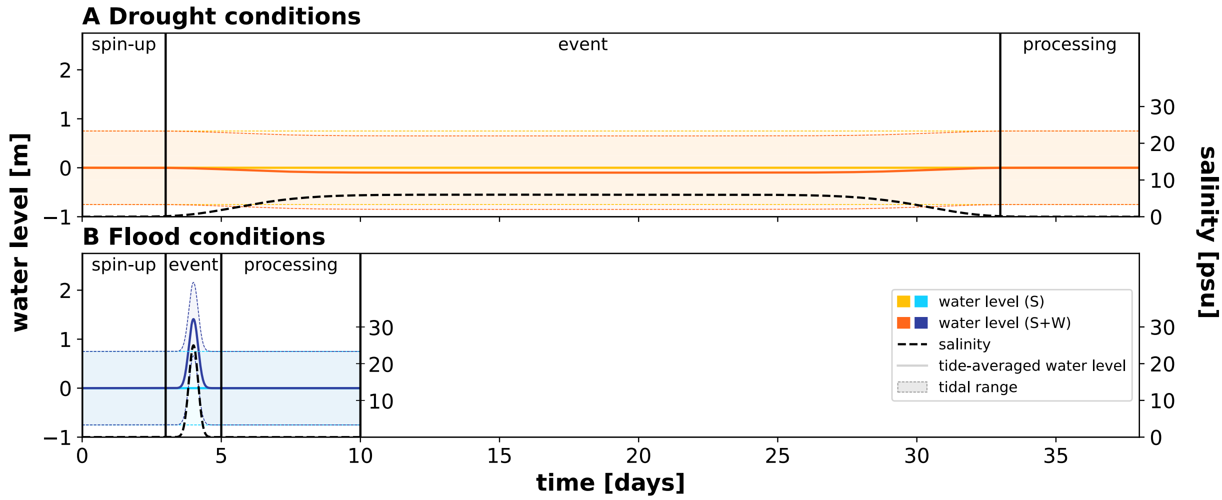

Fig. 3. Water level and salinity at the top boundary for the four simulated scenarios. A) drought where only salinity was modified S and where both the mean water level and salinity were modified S + W; B) storm surge where only salinity was modified S and where both the mean water level and salinity were modified S + W. Dashed lines represent salinity, continuous lines represent mean water level and the dotted area represent tidal range.

have created a confounding effect on the resulting salinity. For this reason we did not include above-ground biomass in the model runs, and was treated as being of negligible height and roughness. In order to obtain comparable results, all topographies have a rectangular shape of 1000 × 500 meters, which corresponds to the average size of willow plantation and de-embankments. These wetlands are surrounded by dikes on four sides, with an opening on the upper side to allow water to enter and leave the system (Fig. 2). In front of the open dike, we placed a slope of 3 m long and 10 m deep (with an angle of 73.3°). This slope simulates the river feeding water to the riparian wetlands, avoids numerical instabilities and shallow water effects, and ensures a gradual change from the river boundary.

Five topographies were created: natural tidal marsh, de-embanked, two willow plantations (higher and lower), and a reference case (Fig. 2). To allow comparison between different wetlands, all the wetlands have an average elevation of 0.2 m, which is similar to the average elevation of the marsh and de-embanked areas along the Rhine-Meuse river. However, willow plantations are generally higher than the other wetlands, so we added an additional topography for the willow plantation with a more realistic elevation of 0.8 m. This allowed comparison between wetlands with the same topography (i.e. willow plantation) and different elevation.

The marsh topography was created using the model proposed by Cornacchia et al. (2024), which is based on the model developed by van de Vijsel et al. (2023). This biogeomorphological model studies the interactions between water flow, sedimentation, and vegetation in forming a self-organizing wetland, i.e., a tidal saltmarsh. Although this model can develop a plausible-looking wetland, we modified some parameters to obtain a more realistic representation (Table S2). By doing so, we were able to obtain an idealized marsh as close as possible to the elevation and creek network of the natural marsh topography (Table S1 and Fig. 1). A more detailed description of the model and the modifications can be found in the Supplementary Materials.

The de-embanked topography is similar to the reference case but includes a channel dug at the centre of the wetland, styled similarly to real-world de-embanked areas. The willow plantation topographies feature a central straight channel reaching the dike on the opposite side. From this central channel, secondary smaller ditches extend perpendicularly to the opposite dikes. The channel in the de-embanked area has an elevation of 0.02 m, while the central and lateral channels in the willow plantation have an elevation of 0.03 m in the lower willow plantation and of 0.57 m in the higher willow plantation. The reference case is a flat topography created to serve as a basic landscape without topographic features that could influence water flow.

Table 1
Top boundary values implemented in the four hydrodynamic simulations.

Scenario	Mean water level (m)	Change in mean water level (m)	Tidal range (m)	Salinity (PSU)	Event duration (days)
Drought (S)	0	0	1.50	6	30
Drought (S + W)	0	−0.10	1.50	6	30
Storm surge (S)	0	0	1.50	25	1
Storm surge (S + W)	0	+1.41	1.50	25	1

2.3. Hydrodynamic model description

For the hydrodynamic model simulations, we used the software Delft3D Flexible Mesh (Kernkamp et al., 2011). Due to the shallow nature of the topographies, we assumed the water column to be well mixed, and thus executed the simulations in two dimensions instead of three (which is generally required for salinity-focused studies). We excluded the effect of vegetation on hydrology, and any morphological processes from the simulations to focus only on the effect of wetland topography on salt intrusion. The horizontal domain was discretised into square cells of 2.5 × 2.5 m. The forcing consisted of a tidal signal at the upper edge of the domain. Together with this tidal signal we simulated either a drought or a storm surge event (Fig. 3). Additionally, a minimal alongshore current was added to prevent the accumulation of fresh and saline water in front of the wetland opening. The forcing conditions were based on data recorded by Rijkswaterstaat (https://waterinfo.rws.nl/) in Spijkenisse (51°52′16.7"N 4°19′51.5″E) from 01/02/2008 to 21/03/2023 at 10-min intervals. We used this data to calculate mean salinity in Practical Salinity Unit (PSU), mean water level in m above NAP, and average tidal range (in m). Based on the used time-series, mean water level is 0.25 m above NAP. Subsequently, we have used mean water level as a reference level in the model runs. Additionally, we set the salinity to 0 PSU instead of 1.7 PSU obtained from the Rijkswaterstaat data to simplify the simulations. Two salt intrusion events were selected as forcing conditions for our simulations: a drought/low discharge occurring from 26/07/2022 to 26/08/2022 and a storm surge

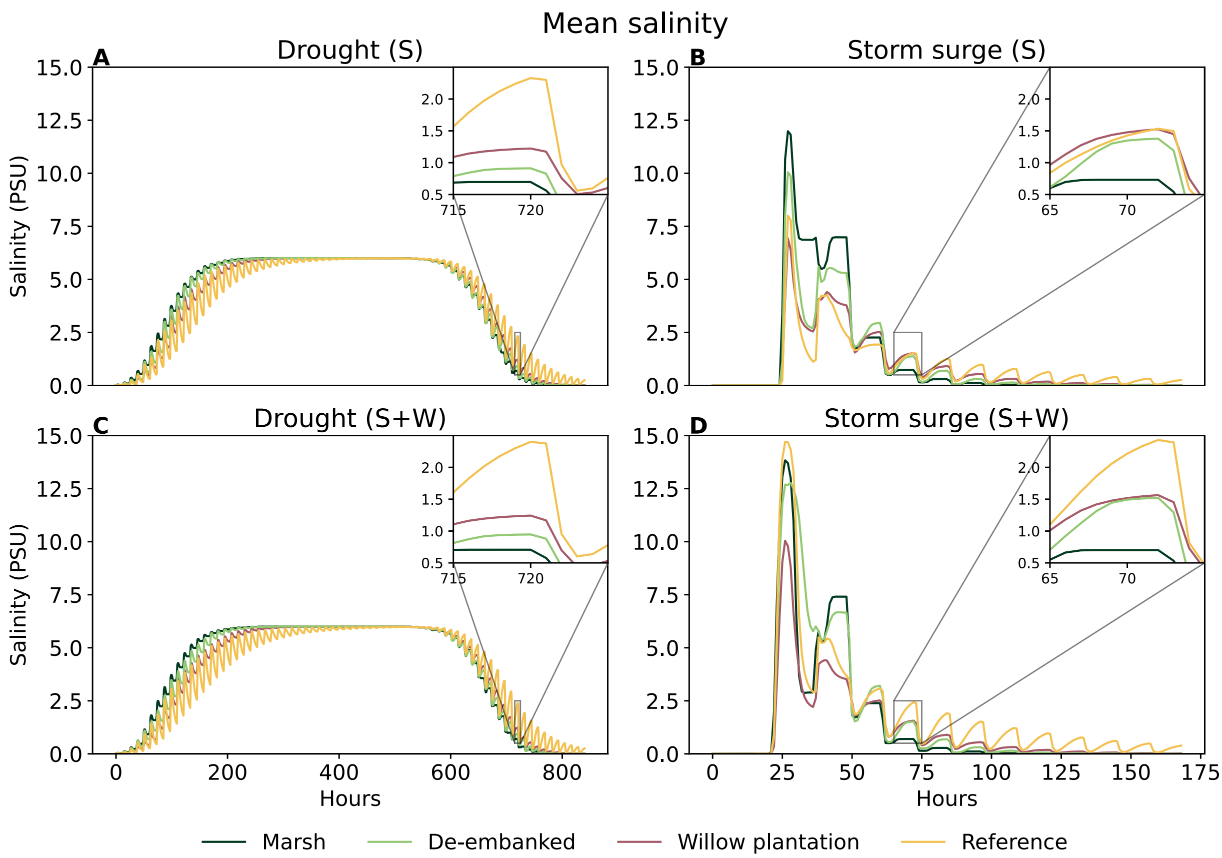

Fig. 4. Mean wetland salinity during the simulations. A) drought S scenario; B) storm surge S scenario; C) drought with reduced water height (S + W); D) storm surge with increased water height (S + W). Different colours indicate different wetlands.

occurring on 06/12/2013. Based on this data we simulated four scenarios: two for the drought case and two for the storm surge case. Both the drought and storm surge cases included one scenario where only salinity was modified (S) and a second scenario where both the mean water level and salinity were modified (S + W, Table 1). This approach allowed us to distinguish the effect of a variation in salinity (simulations S) from the combined effect of changes in water level and salinity (simulations S + W), which usually occur during salt intrusion events. These scenarios were implemented on the top boundary of the wetlands. The simulated salt intrusion event started after a spin-up period of 3 days. The drought scenario lasted 30 days, while the storm surge scenario lasted 1 day. Following the event, we reduced salinity and simulated a recovery period of 5 days for all scenarios. Both the salt intrusion event and the recovery period were used to analyse the differences between wetlands in increasing and decreasing salinity during salt intrusion events.

2.4. Model output analysis

To understand whether there are differences between wetlands during the salt intrusion event, we calculated descriptive metrics of wetland salinity from the model output. Firstly, we removed all data relating to the frontal slope and the dikes from the model outputs (salinity, water depth, flow velocities along and across shore), as these areas are outside the scope of this study. Secondly, we calculated critical events in the salinity profiles of the wetlands, namely:

1. **Salinity buildup time**: this metric helps to understand how long it takes for salinity to reach the highest salinity levels at the onset of the salt intrusion event. Due to the different characteristics of the drought and storm surge simulations, this value was calculated differently between the two:
   a. In the drought simulations, since the wetlands never reach the maximum boundary salinity of 6 PSU but oscillate between 5.9 and 6, we calculated how many hours were needed to reach 5.9 PSU.
   b. In the flood scenarios, since the boundary conditions had only one peak salinity, we calculated how many hours were needed to reach the maximum salinity.
2. **Start of salinity reduction after the drought**: the start of the reduction phase is defined as the moment at which salinity drops to levels below 5.9 PSU and it's calculated only in the drought scenarios.
3. **Salinity builddown time**: we calculated the time it takes for salinity to return to 0.1 PSU after the salt intrusion event. We chose 0.1 PSU since not all wetlands return to 0 PSU.
4. **Difference between salinity buildup and builddown time**: we calculated the difference between the hours needed to reach high salinities in the wetlands (buildup) and to reduce them again after the event (builddown).
   a. For the drought scenarios, where salinity remains high for a prolonged period, we excluded this plateau period from the calculations by determining the difference as: *Salinity builddown time* − *Start of salinity reduction after the drought* − *Salinity buildup time*.
   b. For the flood scenarios, the difference was calculated as *Salinity builddown time* − 2**Salinity buildup time*.

We calculated these critical events for each cell of the model output, this allowed us to also analyse the spatial variability of salt intrusion within the wetlands. Additionally, to assess if certain wetlands retained

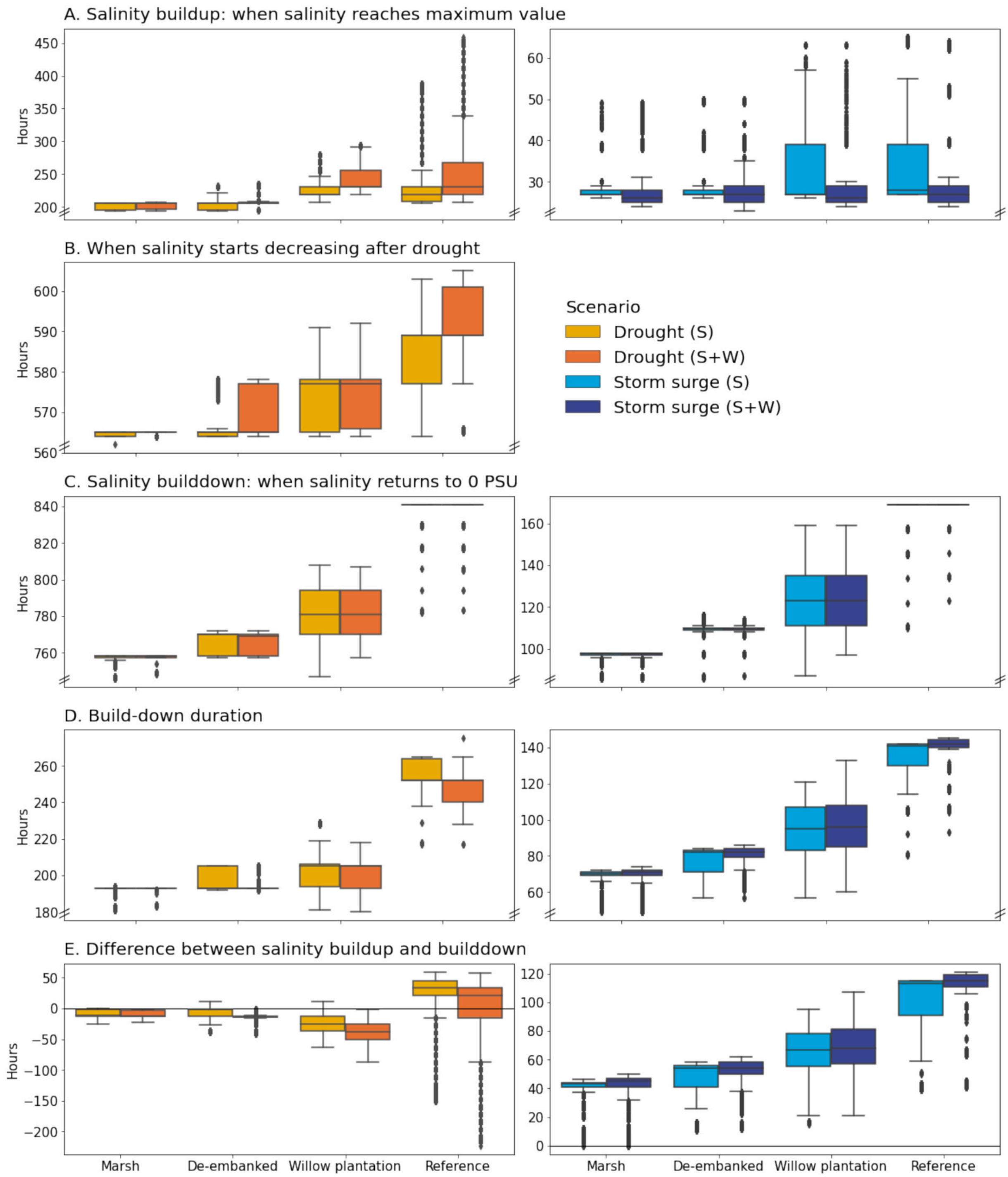

Fig. 5. Boxplots showing the hours needed for: A) salinity build up; B) reduction after the drought (not relevant for the storm surge scenarios); C) salinity builddown; D) builddown duration: hours needed to reduce salinity until 0.1 PSU; E) difference between builddown and buildup. S indicates only salinity is modified, while S + W indicates both salinity and water level are modified. The broken axis symbol on the y-axis indicates that the axis doesn't start from 0.

high salinities longer than others, we calculated the number of hours each wetland (i.e., at least one cell) stayed above 2 PSU. This salinity threshold was chosen based on previous studies suggesting this level does not damage freshwater trees (Markus-Michalczyk et al., 2014). These metrics provide insights into how different wetlands respond to salt intrusion events, helping us to understand their resilience and ability to recover from high salinity conditions.

To investigate the reasons behind the observed differences in salinity between wetlands, we calculated the flow magnitude between the cross-shore and along-shore flow velocities and the flux of salt entering the wetland through the dike opening, F_{gap} as:

$$F_{gap} = hwvs \tag{1}$$

where h is the water depth (i.e., difference between the fluctuating water level and the bottom level); w is the width of the gap; v the flow velocity perpendicular to the cross-section; and s is the salinity at the gap. The salt flux, F_{gap}, is based on the general definition of flux (concentration multiplied by area and flow velocity) and allowed us to analyse salt transport, specifically the input and output of salt in the wetland. By

Table 2
Summary of Critical Metrics for Salinity Resilience Across Different Wetland Types.

Wetland type	Simulation	Buildup time (hrs, mean ± SD)	Builddown time (hrs, mean ± SD)	Peak salinity (PSU, max)
Natural Marsh	Storm surge (S + W)	28.33 ± 5.27	68.54 ± 5.76	23.13
De-embanked	Storm surge (S + W)	27.71 ± 3.95	79.90 ± 5.91	22.75
Willow Plantation	Storm surge (S + W)	29.02 ± 6.70	97.32 ± 14.40	22.22
Reference	Storm surge (S + W)	28.77 ± 7.65	140.13 ± 7.75	23.32
Natural Marsh	Drought (S + W)	203.07 ± 4.83	192.78 ± 0.45	6
De-embanked	Drought (S + W)	208.85 ± 6.00	195.71 ± 5.14	6
Willow Plantation	Drought (S + W)	242.94 ± 17.94	201.86 ± 8.37	6
Reference	Drought (S + W)	267.32 ± 75.17	250.67 ± 8.60	6

examining the salt flux, flow magnitude, and water depth, we aimed to understand the factors contributing to the observed differences in salinity across the different wetland types. We did not perform any statistical analysis on the model output, as the models used in this study are deterministic, meaning the output is entirely dependent on the initial topography.

3. Results

Clear differences in salinity concentration and duration can be seen between wetland topographies (Fig. 4). In the drought scenarios, these differences are evident both at the onset, when saltwater levels gradually increase upon entering the wetland, and at the end, when salinity at the boundary decreases as saltwater exits the wetland. In the storm surge scenarios, differences are noticeable both during the peak of the storm surge and afterwards, as saltwater starts to recede. Overall, salinity increases faster in the natural marsh topography than in other wetland topographies but also decreases more quickly. One exception is the storm surge scenario with increased water level (S + W), where the reference case reaches a higher maximum salinity compared to other wetland topographies. All events show a recurring influx and outflux of saltwater after the event has passed, as the external water has not been fully flushed and re-enters the wetland during flood tide. This effect is minimal for the marsh topography and most pronounced for the reference topography. By comparing the high willow plantation with the low willow plantation we can study the effect of elevation on wetland salinization (Fig. S3): in the high willow plantation, salinity remains low in simulations with regular or reduced water levels (drought S, drought S + W, and storm surge S), but increases when water levels rise (storm surge S + W). In the latter simulation, the high willow plantation topography cannot be completely flushed after a storm surge event, leaving residual saltwater in the wetland.

3.1. *Salinization duration*

When analysing the hours needed to increase salinity at the beginning of the event and to decrease salinity after the event (Fig. 5, Table 2), we observe a constant trend: salinity always increases and decreases faster in the marsh topography compared to the other wetland topographies, followed by the de-embanked, the willow plantation and the reference topography. The main differences between the S and S + W simulations are present in the willow plantation and the reference topographies, while for the other wetlands this difference is less clear. These results show that water level fluctuations are generally irrelevant for the duration of salt intrusion in wetlands. By looking at the builddown duration (Fig. 5D), we notice that even if salinity is lower in the drought scenario than in the storm surge scenario, it takes more than four days to decrease to values lower than 0.1 PSU compared to the storm surge scenarios. The difference between builddown and buildup (Fig. 5E) gives insights into the relative duration of the two phases, with the positive values in the flood scenarios indicating a longer builddown than a buildup and vice-versa for the negative values in the drought scenarios. These differences might be due to the different boundary conditions used in the simulations.

The simulated wetland topographies showed clear differences in the duration of salinity exceeding the 2 PSU threshold. Consistent with previous trends, the natural marsh topography generally experienced shorter periods of high salinity, with a 8.10 % reduction compared to the reference topography in the drought (S + W) simulations and a 48.72 % reduction in the storm surge (S + W) simulations (Fig. 6). These results indicate that although salinity increases more rapidly in natural marsh topographies compared to artificial wetland topographies, it also

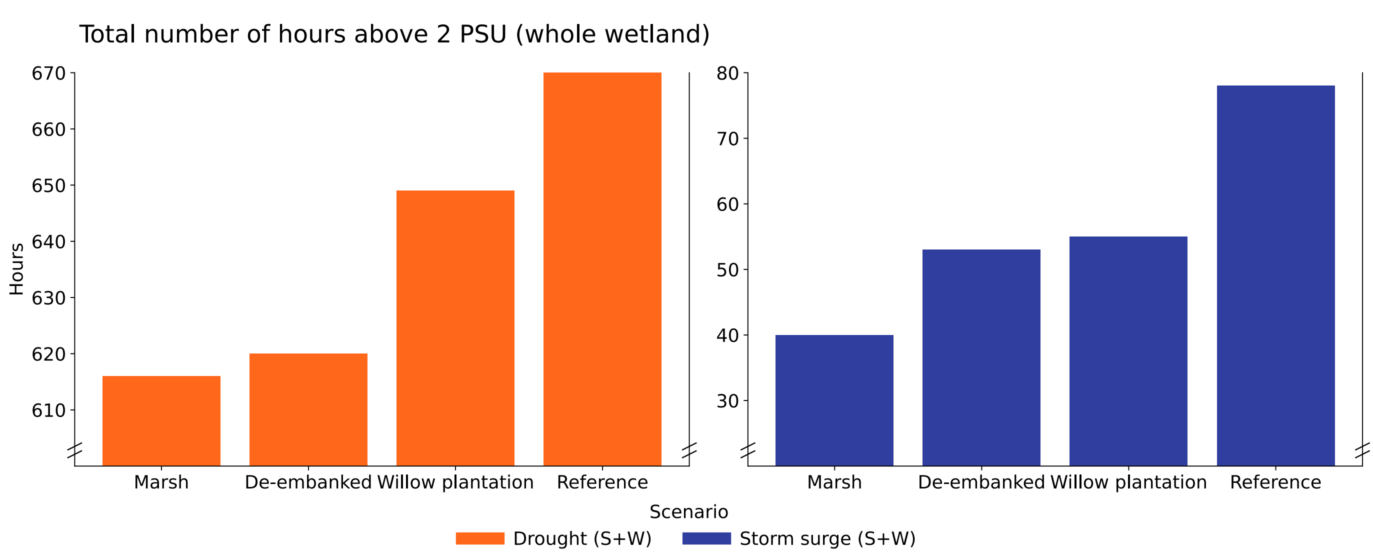

Fig. 6. Hours during the simulations when the maximum salinity inside the wetland is above 2 PSU. 1) Storm surge scenarios, 2) drought scenarios. The broken axis symbol on the y-axis indicates that the axis doesn't start from 0.

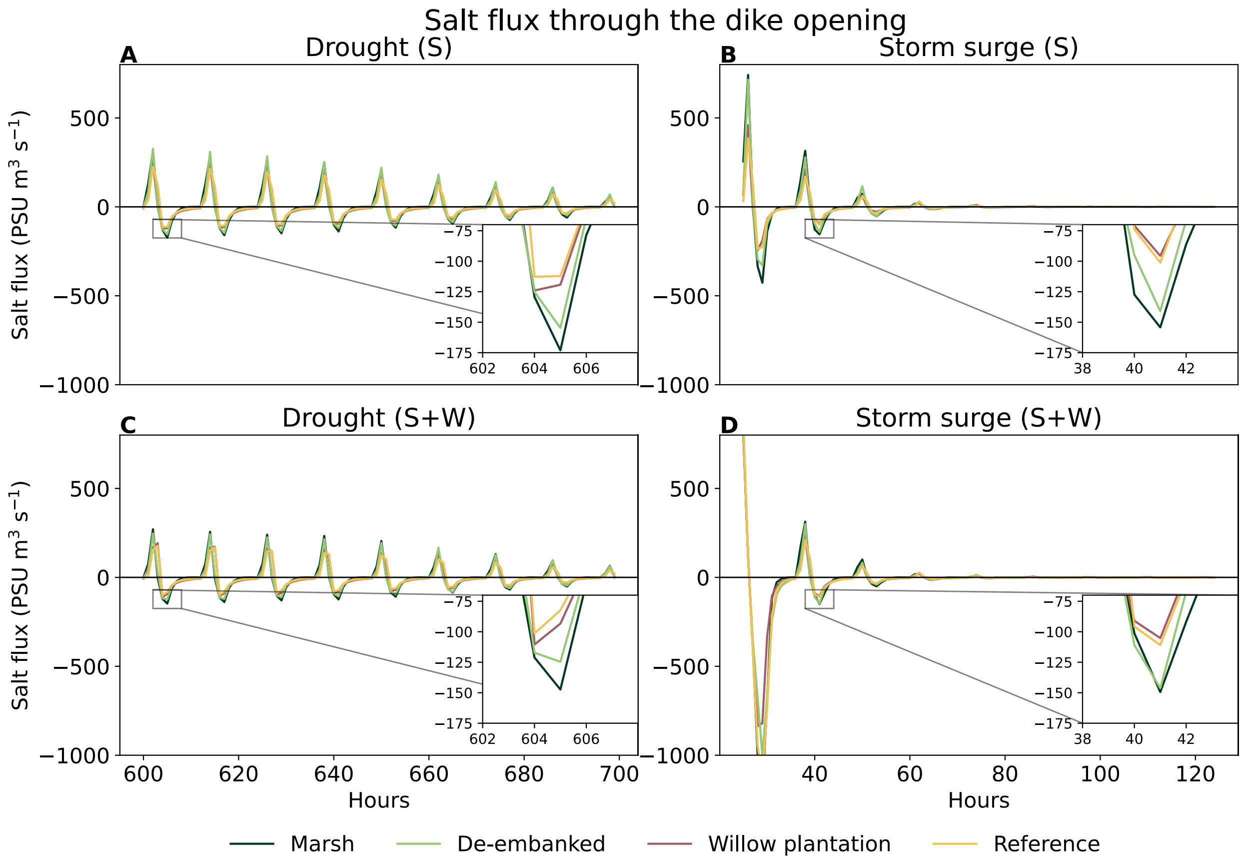

Fig. 7. Salt flux through the dike opening. A) Drought S scenario; B) storm surge S scenario; C) drought with reduced water height (S + W); D) storm surge with increased water height (S + W). Different colours indicate different wetlands.

decreases more quickly after the event, leading to an overall shorter period of high salinity.

3.2. Salt transport

The salt flux shows two peaks: a higher one during flood tide and a lower one during ebb tide (Fig. 7). This pattern is also evident in the flow magnitude (Fig. S4), indicating that these wetlands are flood-dominated. Notably, regardless of topographical differences, the willow plantation and the reference wetland behaved similarly, with almost overlapping curves (Fig. 7, Fig. S4 and Fig. S5). While differences between wetland topographies are minimal during flood tide, a clear division is observed during ebb tide. This suggests that although saltwater enters the wetlands equally, it drains differently based on topographical variations. The differences in salt flux during ebb tide are similar to the salinity (Fig. 7), with the natural marsh topography reaching fluxes up to 6.50 % faster than the reference topography in the drought (S + W) simulations and up to 41.02 % faster in the storm surge (S + W) simulations. These differences indicate that natural marsh topographies can drain saltwater more efficiently during ebb tide than artificial topographies, thereby reducing the duration of exposure to high salinity levels.

Notable differences in salt flux and in its related components – water depth and water velocity – were observed during low tide, as can be observed in the distribution of cross-shore velocity and water depth during low tide across the dike entrance (Fig. 8A and B). These differences followed the same trend as the salinity, with the natural marsh topography exhibiting the highest flow velocity and the lowest water depth. These seem to be the key factors contributing to the shorter duration of high salinity in the natural marsh topography compared to artificial topographies.

3.3. Spatial variability of desalinization

When examining the spatial variability of salt reduction after the event, we observe that the natural marsh topography drains salt homogeneously throughout its area, while the willow plantation topography has zones that struggle to refresh, and the reference topography retains higher salinities throughout the area (for example in the S simulations, where only salinity is modified, Fig. 9). This discrepancy may be attributed to differences in water velocity at ebb tide between wetland topographies (Fig. 10). In the natural marsh topography, the self-organized creek network generates higher water velocities by minimizing friction loss, efficiently reaching all points of the basin and leading to a shorter period of elevated salinity. In contrast, artificial topographies experience high water velocities near the dike opening, but energy dissipates quickly further away, resulting in lower velocities and insufficient flow to effectively reduce salinity across the entire basin. This effect is particularly evident in the willow plantation topography, where the extensive channel network fails to promote rapid salt reduction: although water velocity is high in the central channel, it drops sharply in the secondary creeks deeper into the wetland. This suggests that the structure of the channel network is crucial in determining water velocity and, consequently, the wetland's ability to reduce salinity after a salt intrusion event.

4. Discussion

Tidal freshwater wetland restoration and creation is gaining momentum due to the numerous services they provide, such as carbon storage and flood protection, as well as their ability to adapt to sea level rise by means of sedimentation (Temmerman et al., 2013; Van Coppenolle and Temmerman, 2019). Salt intrusion, which is increasing during

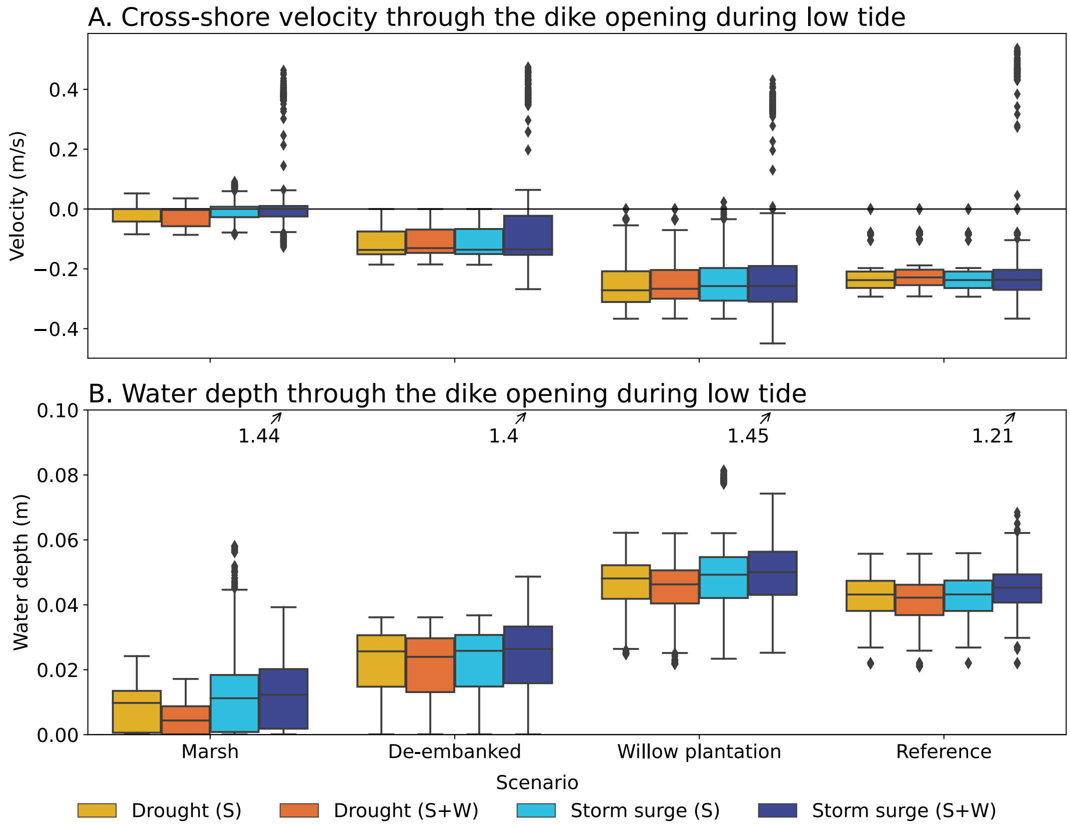

Fig. 8. Boxplot showing A) cross-shore water velocity through the dike opening at low tide; B) water depth through the dike opening at low tide. S indicates only salinity is modified, while S + W indicates both salinity and water level are modified. The arrows indicate the maximum value (outside the plot extent) in the storm surge (S + W) scenario.

droughts and storm surges as a result of climate change, could threaten the success of these measures by hampering both vegetation and topographic development. In this study, we analysed how different natural and artificial wetland topographic designs influence salt intrusion. Our results show that while salinity increases faster in natural marsh topographies compared to artificial wetland topographies, it also decreases faster at the event conclusion, leading to an overall lower duration of salinization. Natural marsh topographies, with their higher flow velocities and lower water depths at low tide, are more effective at draining saltwater, thus mitigating prolonged exposure to high salinities. In contrast, artificial wetland topographies are generally less efficient in this regard, with the exception of the de-embanked wetland topography during drought, likely due to the slow decrease in salinity imposed by the boundary conditions. Our research hence highlights that tidal freshwater wetlands with a natural marsh topography are least affected by salt intrusion, emphasizing the importance of incorporating natural marsh characteristics in wetland restoration and creation designs to enhance their resilience to climate change.

Previous research has shown that natural marshes self-organize their creek networks to efficiently reach all points of the basin in the shortest time, minimizing friction loss and effectively distributing water and sediment across the marsh (D'Alpaos et al., 2019; Kearney and Fagherazzi, 2016; Temmerman et al., 2005; Vandenbruwaene et al., 2013). This process enhances wetland functioning and resilience, supporting marsh accretion in response to sea level rise (Liu et al., 2021). Our study demonstrates that this naturally formed creek network is also effective in reducing saltwater exposure. In contrast, restored and created wetlands often feature artificial designs that do not mimic natural patterns, leading to lack of topographic heterogeneity, and lower channel density, which increases their potential for water accumulation (Adam, 2019; Brooks et al., 2015; Lawrence et al., 2018). These issues tend to persist years after wetland creation or restoration, as these systems generally struggle to develop the complex topography and vegetation patterns characteristic of naturally developing marshes (Lawrence et al., 2018; Mossman et al., 2012). Our results suggest that the lack of topographic diversity in many restored marshes not only reduces service provision and biodiversity (Brady and Boda, 2017; Esteves, 2013; Moreno-Mateos et al., 2012; Mossman et al., 2012; Schuster et al., 2024) but also increases their susceptibility to salinization. One way to enhance both the functionality of artificial wetlands and their capacity to regulate salinity levels is to encourage self-organizing creek development. This can be achieved, for instance, by excavating channels with morphological properties similar to those found in natural systems, as suggested by previous studies on both natural and restored systems (Adam, 2019; Chirol et al., 2024; Lawrence et al., 2018; Zeff, 1999). Furthermore, the morphology of natural marshes is shaped by site-specific factors, such as tidal amplitude, sediment availability and plant species composition (Friedrichs and Perry, 2001; Schwarz et al., 2018). The design of wetland restoration and creation projects should always be guided by local environmental parameters or, when necessary, by those from comparable settings to ensure that they evolve towards a more natural state, provide improved ecosystem services and are more resilient to climate change. Salt intrusion is a serious threat to freshwater wetlands, directly impacting the survival of organisms. The differences in salt intrusion between natural and artificial wetland topographies observed in this study can have significant consequences for the species living in these environments. In natural marsh topographies, organisms experience shorter periods of salt stress compared to those in artificial wetland topographies, as indicated by the reduced hours with salinity above 2 PSU, which improves their chances of survival. Although the natural marsh topography may reach higher salinity levels during storm surge peaks,

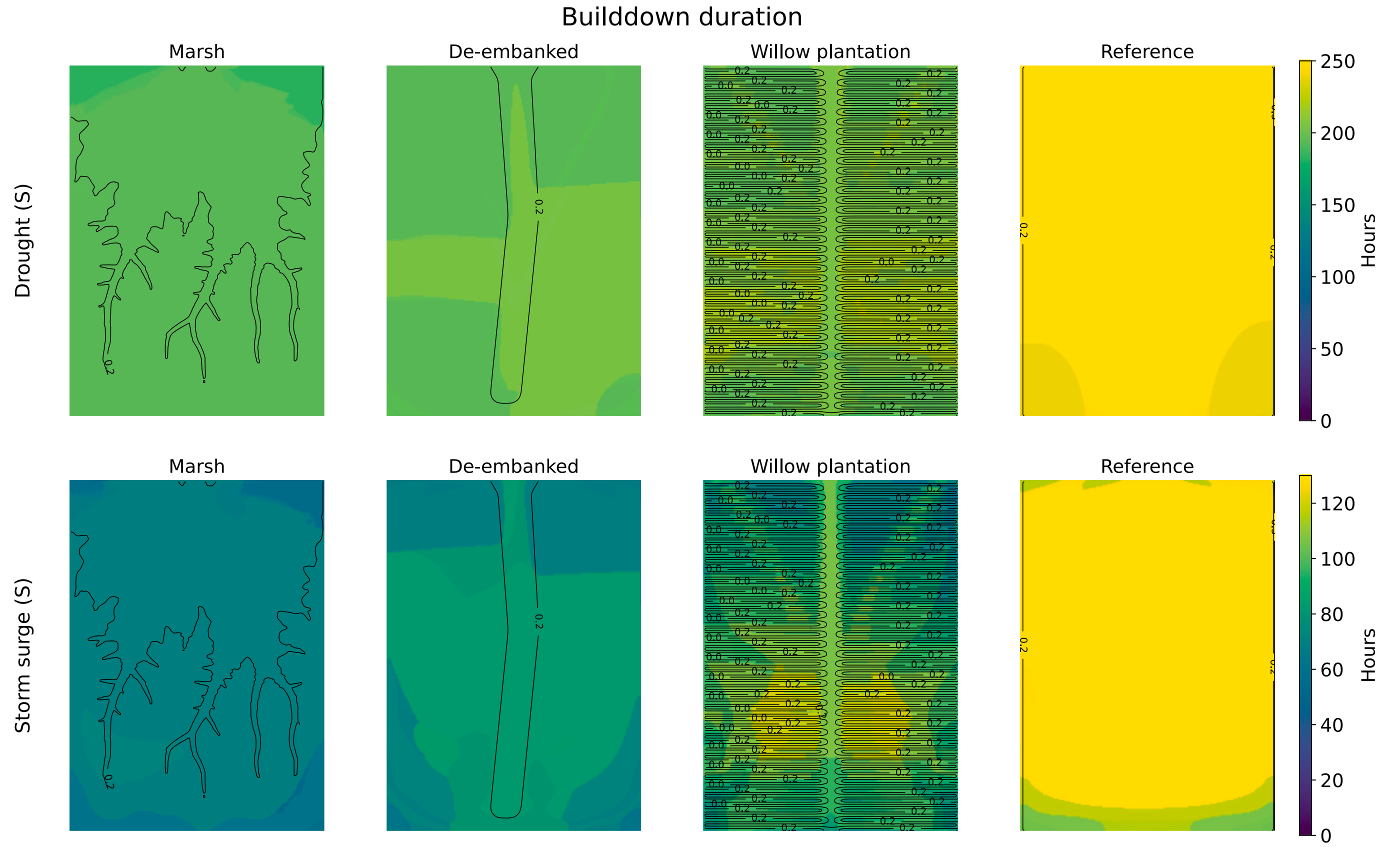

Fig. 9. Builddown duration: hours needed to reduce salinity until 0.1 PSU for the drought S (top) and storm surge S (bottom) simulations when only salinity is modified. From left to right: natural tidal marsh, de-embanked, low willow plantation, and reference case. Similar results are obtained in the S + W simulations, where both salinity and water level are modified.

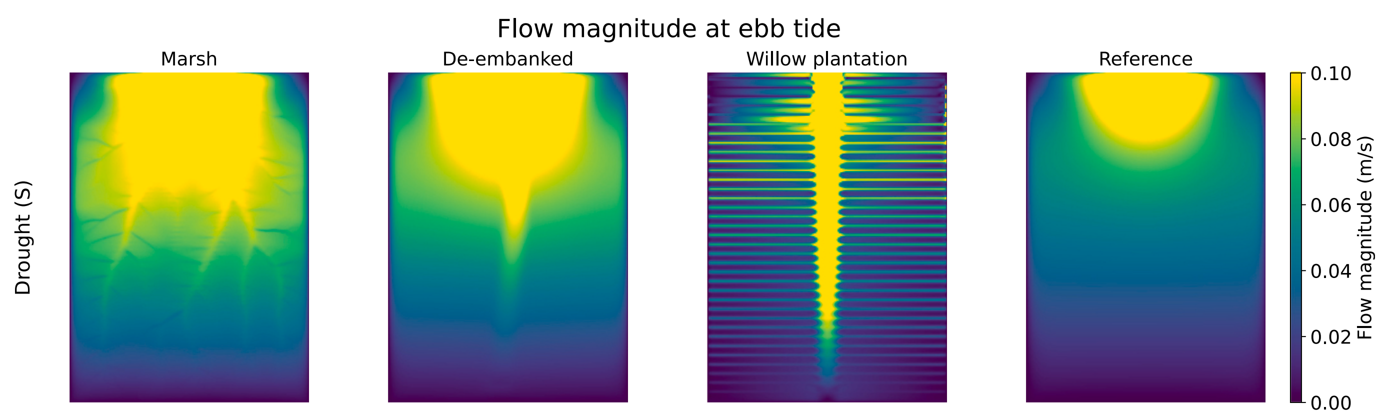

Fig. 10. Flow magnitude at ebb tide for the drought simulation S (at hour 605). From left to right: natural tidal marsh, de-embanked, low willow plantation, and reference case. The storm surge S simulation has the same values of water velocity.

these brief pulses of high salinity are generally not detrimental to organisms, which can either withstand the stress or recover quickly (Flynn et al., 1995; Hootsmans and Wiegman, 1998; Li and Pennings, 2018a; Mobilian et al., 2023; Nielsen et al., 2007). Conversely, in artificial wetland topographies, where saltwater can become trapped in channels and behind levees after a storm surge (Keim et al., 2019), the situation is different. The prolonged presence of saltwater allows it to infiltrate the soil, making it more difficult to reduce salinity. This prolonged exposure can lead to significant vegetation stress, high mortality rates, and slow recovery (Gardner et al., 1992; Keim et al., 2019; Yu et al., 2016). Therefore, the key difference lies in drainage efficiency. In natural marsh topographies, where saltwater is quickly drained and does not infiltrate the soil, the impact of salt intrusion on organisms is likely to be temporary and recoverable. However, in artificial wetlands, where saltwater persists and permeates the soil, the consequences can be severe and long-lasting. Without considering the influence of topography on salt intrusion, restoration efforts in tidal freshwater wetlands may fail, resulting in widespread mortality.

The spatial variability of salt intrusion in wetlands significantly affects organism survival and distribution. Our simulations revealed key differences: in natural marshes, salinity spread evenly, while in artificial wetlands, it varied, decreasing faster near the dike breach and more slowly farther away. This uneven salinity distribution was particularly pronounced in the willow plantation, where two central areas near the dikes showed slower salinity reduction. These differences appear to be driven by water velocity, which depends on the channel network's shape (Chirol et al., 2024; Stark et al., 2016), highlighting its importance on salt intrusion. In artificial wetlands, especially willow plantations, freshwater vegetation located far from the dike opening will experience prolonged salt stress compared to vegetation located closer to the dike opening, potentially leading to reduced growth and increased mortality if the stress persists for too long (Hootsmans and Wiegman, 1998; Li and Pennings, 2018b; Nielsen et al., 2007). After freshwater vegetation has died in these areas, new salt-resistant vegetation may struggle to colonize due to their distance from propagation sources such as the river. If salt-resistant vegetation manages to colonize these areas, the wetlands might develop two (or more) distinct vegetation communities: a freshwater one closer to the dike opening and a more salt-tolerant one further away near the dikes. Further research is needed to resolve the uncertainties on vegetation distribution highlighted by our results.

The use of idealized topographies, based on real-world data, allowed us to generalize our findings and to apply them to a broader scale, irrespective of location. Idealized topographies are a common approach to infer general characteristics of (ecological) systems (Best et al., 2018; Hendrickx et al., 2023; Schwarz et al., 2014). This approach is particularly valuable when field validation is not feasible, as in our case, due to the rarity of still persisting natural tidal freshwater wetlands in our study location and the difficulty in measuring unpredictable salt intrusion events. Moreover, by employing this modelling methodology, we can extend the trends observed in this study to other cases and deltas, providing valuable insights into the capacity of wetlands to respond to saltwater intrusion. Finally, this idealized setup enabled us to establish cause-and-effect relationships between topography and salinity, offering in depth understanding of the dynamics involved in these interactions. One potential limitation of this study is that, while it focused on the influence of different topographies on salt intrusion, we chose not to account for the potential effects of variability in vegetation type, which affects bottom friction. Different plants are naturally found in the various wetland types we studied, and contrasting vegetation characteristics could influence how water flow is attenuated within the vegetation (Bouma et al., 2013; van Wesenbeeck et al., 2007). The reason for this omission is that we wanted to concentrate on the differences in wetland design, rather than vegetation type, on salt intrusion. Additionally, we lacked information on the differences in friction imposed by the various plant types. Yet, differences in vegetation structure may further affect salinity distribution within the wetlands. Therefore, we would recommend further research to investigate the role of vegetation-characteristics in modulating salinity and the long-term evolution of wetlands in response to increasing salinity.

5. Conclusions

In the context of climate change and increasing salt intrusion, designing tidal freshwater wetlands for managed realignment offers a cost-effective, nature-based solution for coastal protection and biodiversity (Narayan et al., 2016). Our findings reveal that current artificial wetland topographies are more vulnerable to salt intrusion than natural wetland topographies. Therefore, promoting the development of a self-organizing creek system that emulates natural marsh channel geometry can significantly reduce organisms' exposure to saltwater. Such a system not only mitigates salt intrusion but also accelerates the wetland's topographical evolution, helping it reach maturity faster while enhancing biodiversity and ecosystem services like flood defence (Adam, 2019; Chirol et al., 2024; Hughes, 2012; Stark et al., 2016; Zeff, 1999). Incorporating natural marsh characteristics into tidal wetland restoration and creation is therefore essential for improving their resilience to salt intrusion and generating ecosystems capable of withstanding and adapting to climate change.

CRediT authorship contribution statement

Eleonora Saccon: Writing – original draft, Visualization, Methodology, Formal analysis, Conceptualization. **Gijs G. Hendrickx:** Writing – original draft, Visualization, Methodology, Formal analysis, Conceptualization. **Suzanne J.M.H. Hulscher:** Writing – review & editing, Supervision, Project administration, Funding acquisition. **Tjeerd J. Bouma:** Writing – review & editing, Supervision, Project administration, Funding acquisition. **Johan van de Koppel:** Writing – review & editing, Supervision, Project administration, Funding acquisition.

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Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2025.107650.

Data availability

The data of this study are openly available in the research repository 4TU: doi: 10.4121/93d8d223-e2ab-486e-a85b-da14789532d4.

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