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**Soil organic carbon stocks in a tidal marsh landscape are dominated by human marsh
embankment and subsequent marsh progradation**

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Running title: *Dynamics of SOC during stepwise embankment*

Abstract

Tidal marshes are coastal and estuarine ecosystems that store large amounts of sedimentary organic carbon (OC). Despite the valuable ecosystem services they deliver, tidal marshes have been converted to other land use types at high rates over the past centuries. Although previous studies have reported large decreases in soil organic carbon (SOC) stocks after tidal marsh embankment, knowledge on the magnitude and rate of OC losses is still limited. Here, we studied the effect of stepwise embankments of brackish and salt marshes and subsequent marsh progradation on SOC stocks in the Scheldt estuary (The Netherlands). We collected samples from soil profiles along tidal marsh–reclaimed tidal marsh chronosequences and determined total OC stocks and the stable carbon signature of the OC. Our results showed that large losses of previously sequestered SOC occur on a decadal timescale with the embankment of brackish ($-8.7 \pm 1.8 \text{ kg OC m}^{-2}$) and salt marshes ($-6.8 \pm 3.1 \text{ kg OC m}^{-2}$). The (incomplete) replacement of tidal marsh OC by agricultural OC is substantially faster in topsoils (ca. a century) compared to subsoils (multiple centuries). Simulations with a coupled land use–SOC model showed that large rates of marsh progradation following embankment construction resulted in a substantial increase in landscape-scale SOC storage, whereas large SOC losses occurred in landscapes dominated by embanked tidal marshes. The findings of our study might help to assess not only how these management practices affect regional SOC stocks, but can also be used to design embankment schemes that minimize SOC losses.

Keywords: $^{13}\text{C}/^{12}\text{C}$ isotopic ratio, tidal marsh, soil organic carbon, agricultural soil, land use change

37

38 **Highlights:**

- 39 • We studied how OC stocks in tidal marsh sediments are affected by embankments
- 40 • OC stocks of both brackish and salt marsh sediments were reduced by ca. 60 % with
- 41 embankment
- 42 • Gains in landscape-scale SOC storage occurred with?? marsh progradation after
- 43 embankment
- 44 • If no marsh progradation occurs, embankments always lead to SOC loss at the landscape
- 45 scale

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1. Introduction

Tidal marshes are highly productive ecosystems located along coastlines and in estuaries, delivering a wide range of important ecosystem services (Barbier *et al.*, 2011). Furthermore, it has been well-established that vegetated coastal ecosystems, including tidal marshes, mangroves and seagrasses, store large amounts of organic carbon (OC) in their sediments (Duarte *et al.*, 2013).

Despite the social, economic and environmental importance of vegetated coastal ecosystems, they are under immense pressure. Apart from a rise in sea level, which can cause loss of tidal marsh from submergence and erosion, the most imminent threat to vegetated coastal wetlands is by direct human transformation (Kirwan & Megonigal, 2013). Globally, up to half of the area covered by vegetated coastal ecosystems has been lost over the past century with anthropogenic conversion of these ecosystems for mainly agricultural, aquacultural, industrial and residential purposes (Huang *et al.*, 2010; Mcleod *et al.*, 2011). This process is referred to as ‘marsh embankment’, referring to the construction of flood protection structures to convert tidal marshes into agricultural land. Embanked agricultural land is described by the Dutch word ‘polder’. Over the past centuries, tidal marsh embankment was often carried out in countries that were developing economically, such as The Netherlands (Hoeksema, 2007), Germany (Goeldner, 1999) and the United Kingdom (Ellis & Atherton, 2003). Over the past decades, however, coastal wetland embankment has shifted to countries where economic development is more recent. For example, in east Asia large areas of intertidal wetlands have been embanked since the 1950s, e.g. in China (Huang *et al.*, 2010; Ma *et al.*, 2014), the Korean Peninsula (Murray *et al.*, 2014) and Japan (Suzuki, 2003). Moreover, many more embankments of intertidal wetlands are planned in east Asia in the years to come (Wang *et al.*, 2014).

The destruction of natural coastal wetlands has important consequences for the health of coastal ecosystems (Hood, 2004; Ma *et al.*, 2014) and can cause considerable economic losses in regions that are affected (Zhu *et al.*, 2017). This, combined with the large rates at which coastal wetlands have been reduced over the past decades (globally 0.7–3 % year⁻¹) (Pendleton *et al.*, 2012; Duarte *et al.*, 2013), explains the strong need to assess correctly how wetland embankment affects the properties of these ecosystems and the ecosystem services they deliver. An example of such an ecosystem service is OC storage. Several studies have shown that the embankment and drainage of OC-rich coastal sediments leads to a loss of previously stored OC (Bai *et al.*, 2013; Bu *et al.*, 2015; Deng *et al.*, 2016). In a study summarizing available data on this process, Pendleton *et al.* (2012) calculated that the conversion and degradation of vegetated coastal ecosystems might result in an average annual emission of ca. 0.45 Pg CO₂, equivalent to 3–19 % of annual CO₂ emissions from global deforestation.

Despite acknowledgement that the embankment of vegetated coastal ecosystems affects regional and global OC budgets, research concerning this topic has been limited until now (Pendleton *et al.*, 2012). Moreover, OC losses in these systems are generally studied using a space-for-time conversion approach along chronosequences of embanked tidal marshes whereby losses assessed by comparing the SOC stocks of current marshes with those of arable land formed by marsh embankment at a particular point in time. Although this approach enables the calculation of SOC losses from embanked sediments, it disregards the fact that at the seaward side of the embankment a new tidal marsh is often initiated (Jongepier *et al.*, 2015), which can store additional OC. To the best of our knowledge, the effects of new marsh formation have not been included until now in assessments of the effect of tidal marsh embankment on regional SOC budgets. Evidently, this

might lead to an overestimate of calculated SOC losses because newly formed marshes could be important locations of organic carbon sequestration.

To address these issues, we studied temporal changes in regional SOC budgets in landscapes created through stepwise embankment of tidal marshes in the Scheldt estuary (The Netherlands). The Scheldt estuary is particularly interesting because embankment started several centuries ago and continued until the late twentieth century. Furthermore, embankment dates are well documented. We aim to answer the following research questions: (i) how much and at which time scales do OC stocks in tidal marsh sediments change after conversion to an agricultural land use, and is there a difference between sand and brackish marshes in this respect? and (ii) how are landscape-scale SOC stocks affected by the stepwise embankment and subsequent progradation of tidal marshes?

2. Materials and methods

2.1 Study area

The polder regions studied here are in the saline (salinity > 18 , unitless according to UNESCO (1985)) and brackish (salinity ca. 17) parts of the Scheldt estuary (southern Netherlands) (Figure 1). The embankment of tidal marshes was a stepwise process. The embankment of tidal marshes typically initiated the formation of a new tidal marsh on the initially bare tidal flats at the seaward side of the embankment. Subsequently, this new tidal marsh was in its turn embanked (Jongepier *et al.*, 2015) (Figure S1, Supporting Information). This process was repeated several times through history

The sampled polder landscape in the brackish part of the Scheldt estuary (region 1, Figure 1) initially consisted of extensive intertidal sand flats on which brackish marshes became established

(van der Spek, 1997). In this region, 10 polders which were embanked between the 1571 and 1973 were sampled. Throughout this study we refer to these polders as brackish polders. The sampled polder landscapes in the saline portion of the Scheldt estuary are located in four distinct regions (Figure 1). Here, 15 polders reclaimed between the 1542 and 1952 were sampled and are referred to as salt polders. A detailed description of the embankment history of the brackish and saline polder regions is provided in Wilderom (1968) and Wilderom (1973), respectively (in Dutch). We used depth profiles of OC and stable carbon isotopes obtained from an active tidal marsh that was close to the sampled polders as reference data, representing the situation of the polders before embankment. (Figure 1 and Figures S2–S6, Supporting Information). At every tidal marsh, both a low- and high-lying location was sampled. A detailed description of these tidal marshes and the sampling procedures can be found in Van de Broek et al. (2018). An overview of all the sampled locations is provided in Table 1. Detailed maps of the sampled polders landscapes are given in Figures S2– 6, and the coordinates of sampling locations are provided in Table S3 (Supporting Information).

2.2 Field sampling

Soil samples were collected in August 2015 in the brackish polders (at 10 locations) and in August and September 2016 in the salt polders (at 15 locations). Sampling locations were chosen carefully to ensure that they were in former tidal marsh sediments with a silt-dominated particle size (Van de Broek *et al.*, 2018). This is important because embanked tidal creeks have a sand-dominated particle size (Jongepier *et al.*, 2015). The sampling locations were chosen (i) using historical topographic maps (see Figure S1), (ii) using soil particle-size maps and (iii) by visual assessment of the particle size during sample collection.

At every sampled location in the polders, two sets of three undisturbed soil profiles were collected using a gouge auger (0.06-m diameter). The distance between sampling locations was two metres. The first set was used for the analyses of physical and chemical soil properties (depth increments of 0.03 m), whereas the second set was used to determine soil bulk density (depth increments of 0.05 m). When the boundary between silty (former tidal marsh sediments) and sandy (former tidal sand flat below the marsh) sediments was encountered in the upper metre, a continuous depth profile down to the upper sand layer was collected. If this boundary was deeper than one metre, sample collection was generally limited to this depth because compactness of the soil impeded the collection of samples below this depth with a manual auger. However, the thickness of previous tidal marsh sediments was less than 1 m at most sampling locations (Table 1). The tidal marshes were sampled using identical procedures (Van de Broek *et al.*, 2016, 2018).

2.3 Laboratory analyses

Before laboratory analyses for of general soil characteristics the samples were oven-dried at 50 °C, crushed until they passed through a 2-mm sieve and macro-vegetation residues and roots were removed manually using tweezers. Particle size was determined with a laser diffraction particle-size analyser (LTSM 13 320, Beckman Coulter, GIVE CITY and COUNTRY) for one replicate soil core at every location at 0.18-m depth intervals, and classified into clay ($< 2 \mu\text{m}$), silt (2–63 μm) and sand ($> 63 \mu\text{m}$) size fractions. Before particle-size analysis, organic matter was removed using hydrogen peroxide (35 %) and aggregates were destroyed by ultrasonic dispersion. The bulk density was determined after drying the soil samples collected for this purpose at 105 °C. The weight of these samples was combined with the inner volume of the gouge auger for a depth section of 0.05 m to calculate the bulk density. The inorganic carbon content of the soil samples collected in the brackish region was analysed by the pressured calcimeter principle (Horváth *et al.*, 2005).

Organic carbon concentrations and $\delta^{13}\text{C}$ values were analysed for soil samples from one of the replicate soil profiles at every location. Every other soil sample collected along the depth profile was excluded from analysis, therefore, samples from the following depths were analysed: 0–0.03, 0.06–0.09, 0.12–0.15 m, and so on. Analyses were performed with an elemental analyser–isotope ratio mass spectrometer (FlashEA 1112 HT, Thermo Scientific–DELTA V Advantage, city and country). Before analysis, soil samples were weighed in pre-combusted silver cups and carbonates were removed using a 10 % HCl solution. For soil samples from the other two replicate profiles, OC concentrations were determined by mid-infrared (m-IR) spectroscopy. Soil samples for m-IR spectroscopy analyses were finely ground and five replicates of each soil sample were scanned at wavenumbers between 400 and 4000 cm^{-1} at a spectral resolution of 4 cm^{-1} with a PerkinElmer® FT-IR spectrometer which was coupled to a PIKE® XY Autosampler (give city and country of instruments). Spectra obtained from a dried and finely ground KBr powder were used as a background. Organic carbon concentrations were subsequently calculated separately for the salt and brackish polders in R by partial least squares regression (PLSR) (Stevens & Ramirez Lopez, 2014) to minimize variation within the datasets. The average absolute deviation between measured and predicted OC concentration was 0.15 % OC for the brackish polders and 0.12 % OC for the salt polders. The procedures applied are described in detail in the Supporting Information.

To account for soil compaction with the embankment and drainage of tidal marshes, OC and inorganic carbon stocks are calculated for a constant mass of bulk soil (top 400 kg) for a surface area of 1 m^2 . The average depth to which this mass of sediments was present for the different regions is provided in Table 2. The $\delta^{13}\text{C}$ value of these sediments was calculated by weighting the $\delta^{13}\text{C}$ value of each soil layer by the amount of OC present in that layer. Bulk density is reported as the average bulk density in the top 0.3 m of the profile.

2.4 Landscape-scale soil organic carbon model

The effect of stepwise embankments of tidal marshes on regional SOC stocks was assessed using a coupled land use–SOC model. The processes simulated in the model are (i) the vertical and lateral progradation of tidal marshes and the accumulation of OC in these sediments, (ii) the embankment of tidal marshes and potentially a part of the sand flat in front of the marsh, depending on the rate of lateral marsh progradation and (iii) soil compaction and loss of OC from the embanked sediments. We did not aim to reconstruct historical changes accurately in SOC stocks because the necessary data for model calibration and validation are lacking for this purpose. Rather, the aim of this modelling exercise was to assess how the embankment of tidal marshes affects landscape-scale SOC stocks over time, depending on the area covered by tidal marshes and polder soils in the modelled regions, and the spatial distribution in the thickness of tidal marsh sediments.

Model simulations were performed for the brackish polder region 1 and the salt polder region 2 (Figure 1, Figure S2 and S3, Supporting Information) at a spatial resolution of 10 m. The model assumes an initial situation in which only sand flats are present (van der Spek, 1997) on which tidal marshes subsequently develop at the seaward side of constructed embankments. The timing of embankment constructions was based on historical data. For the brackish region, simulations were performed for the period 1500–2100 AD, for the salt region from 1700 to 2100 AD, assuming the landscape structure remained unchanged after the construction of the last embankment. The simulated accumulation of OC in tidal marsh sediments was based on measured depth profiles of OC in a tidal marsh in the simulated polder regions (Figure 1). The rates at which SOC stocks decline and bulk densities increase with embankment were calculated using empirical functions derived from the measured values of SOC stocks and bulk density as a function of time since

embankment (see results section). By tracking the amount of OC present in every simulated grid cell for every time step, both the spatial variation in OC stocks and the total amount of SOC present in the simulated region could be calculated for every simulated time step. The model results are presented as the SOC stock for the upper 400 kg m⁻² of sediments (kg OC m⁻²) (i) for every grid cell and (ii) as the average landscape-scale SOC stock for the simulated regions. A detailed description of the model, together with parameter values, is provided in the Supporting Information. The model was coded in Python 3.4; the model code is available in the Supplementary Data.

2.5 Statistical analysis

Statistical significance of the differences between sedimentary OC stocks and $\delta^{13}\text{C}$ values were assessed by a one-way analysis of variance (ANOVA), after checking for homogeneity of variances (Levene's test) and normality (Shapiro–Wilk test). These analyses were performed in Matlab®. We used a confidence level of 0.05.

3. Results

3.1 Particle size, pH, inorganic carbon content and bulk density

An overview of the average general soil characteristics per study site are provided in Table S2 (Supporting Information). In general, particle size was similar between polders and tidal marshes in the same region. Variation in average particle-size fractions with depth is limited in the former marsh sediments, with average fractions were 16.6 ± 4.0 % clay, 61.5 ± 9.3 % silt and 21.6 ± 12.5 % sand for the brackish polders and 9.2 ± 2.7 % clay, 58.5 ± 12.4 % silt and 32.2 ± 14.7 % sand for the salt polders. The inorganic carbon content (IOC) for the brackish chronosequence was less for some polders ($4.1\text{--}2.9$ kg IOC m⁻²) than for the present-day tidal marsh ($4.4\text{--}4.9$ kg IOC m⁻²),

whereas other polders contained similar amounts of IOC as the active tidal marsh. As no clear pattern was found for the brackish marshes, IOC contents were not assessed for the salt polders. In the brackish region, pH values decreased after reclamation, from 7.5–7.7 on active tidal marshes to steady-state values of ca. 7.1 in the polder soil. In the salt region, in contrast, pH values showed no decrease after the conversion of tidal marshes (pH 7.4–7.9). The average soil bulk density in the top 0.3 m increased substantially after embankment and drainage (Figure S7, Supporting Information). Average bulk density in the topsoil of the brackish marshes and brackish polders was $0.81 \pm 0.11 \text{ g cm}^{-3}$ and $1.25 \pm 0.25 \text{ g cm}^{-3}$, respectively. In the salt region of the estuary, the average topsoil bulk densities for the tidal marshes and polders were $0.61 \pm 0.18 \text{ g cm}^{-3}$ and $1.35 \pm 0.26 \text{ g cm}^{-3}$, respectively.

3.2 Organic carbon and $\delta^{13}\text{C}$ depth profiles

Depth profiles of OC concentration for selected polder soils and tidal marshes in the brackish (region 1) and salt polder region (region 2) are shown in Figure 2 (a) and (c), respectively. Depth profiles of OC and $\delta^{13}\text{C}$ of all sampled locations are provided in Figures S9–S12; the data are available in the Supplementary Data. In both the brackish and salt polder regions, SOC concentrations declined after the embankment of tidal marshes. In the brackish region, OC concentrations in the tidal marshes ranged from 14 % OC in the topsoil to ca. 2 % OC below 0.4-m depth. The polder that was embanked 41 years before sample collection had an OC concentration of ca. 3 % OC, whereas OC concentrations were smallest for polders that had been embanked for more than 60 years (ca. 1.4 % OC in the top 0.4 m). This suggests that new equilibrium OC concentrations are reached ca. 60 years after the reclamation of brackish marshes. The depth profiles of OC concentration of the youngest polders in the salt portion of the estuary, embanked ca. 60 years prior to sampling, do not show substantial differences from older polders (ca. 1.1 %

OC), indicating that here new equilibrium topsoil OC concentrations are reached < 60 years after reclamation.

The depth profiles of $\delta^{13}\text{C}$ of the same marshes and polders are shown in Figure 2 (b) and (d). For the polders in both the brackish and salt regions, variation in $\delta^{13}\text{C}$ depth profiles among polders is larger than that in profiles of OC concentration. In the brackish polders, $\delta^{13}\text{C}$ values decreased consistently with polder age in the top 0.4 m, from ca. -25.3‰ for the polder embanked 62 years prior to sampling to ca. -26.7‰ for the polder embanked 111 years before sampling. This pattern of decreasing $\delta^{13}\text{C}$ values over time was, however, not consistent with the $\delta^{13}\text{C}$ depth profile of the brackish marsh, which has a value of -28.1‰ at the soil surface and increased to -25.5‰ from a depth of 0.3-m downwards (Figure 2b). For the polders in the salt region (region 2), there was a consistent change in $\delta^{13}\text{C}$ values along the salt marsh–polders chronosequence. The $\delta^{13}\text{C}$ values for the salt marsh varied between -21 and -22‰ in the top 0.5 m and were on average -26.3 and -27.1‰ in the plough layer of the 64- and 116-year old polders, respectively. Moreover, the decrease in $\delta^{13}\text{C}$ values through time occurred at a faster rate in the plough layer than the subsoil.

3.3 Changes in organic carbon stocks and $\delta^{13}\text{C}$ values

The time for OC stocks in embanked tidal marshes to reach a new equilibrium is ca. 100 years in the brackish polders, whereas it is <64 years in the salt polders (Figure 3, Table 2). This contrasts with the time required for OC concentrations to reach equilibrium, which was 60 years or less in both regions. For the brackish region, the average OC stock in tidal marsh sediments was $14.76\text{ kg OC m}^{-2}$ and this decreased to an average equilibrium stock of $6.10 \pm 1.83\text{ kg OC m}^{-2}$ in embanked polders. For the salt regions, the average amount of OC in the salt marsh sediments showed considerable variation with an average of $11.82 \pm 2.90\text{ kg OC m}^{-2}$ and decreased to an average of

5.14 $\pm$ 1.12 kg OC m⁻² after embankment (Table 2). The equilibrium OC stocks of brackish and salt polders were not significantly different ($P = 0.16$), with the homogeneity of variance ($P = 0.27$) and normality of the distribution of the data ($P > 0.05$) confirmed. Calculated rates of absolute OC losses over the time periods that losses were observed for (100 and 60 years for the brackish and salt polders, respectively) showed that embanked brackish marshes on average released OC at a somewhat smaller rate (86.6 $\pm$ 18.3 g OC m⁻² year⁻¹) than for embanked salt marshes (111.3 $\pm$ 51.8 g OC m⁻² yr⁻¹). The relative loss of OC after conversion was, however, similar between polders in both regions (59 $\pm$ 12 3 g OC m⁻² yr⁻¹ and 57 $\pm$ 30 % for brackish and salt polders, respectively).

The average $\delta^{13}\text{C}$ values of SOC in the top 400 kg m⁻² of sediments showed, on the one hand, that for the brackish region an initial increase in $\delta^{13}\text{C}$ values was observed along the chronosequence (Figure 4), as the $\delta^{13}\text{C}$ value of currently active tidal marshes (-26.7 ‰) was more negative than for the youngest polder soils (-25.2 to -26.1 ‰) (Figure 2). The $\delta^{13}\text{C}$ values of older polder soils ranged between -25.5 and -28.11 ‰. For the salt region, on the other hand, a consistent decline in $\delta^{13}\text{C}$ values was observed in the first decades after marsh conversion; active tidal marsh sediment had $\delta^{13}\text{C}$ values between -22.1 and -25.8 ‰ and polder soils had $\delta^{13}\text{C}$ values between -25.6 and -27.7 ‰. The $\delta^{13}\text{C}$ values of OC, after equilibrium was reached, were not significantly different ($P = 0.69$) between brackish and salt polders, with homogeneity of variance ($P = 0.16$) and the normality of the distribution of the data ($P > 0.05$) confirmed.

3.4 Temporal changes in landscape-scale organic carbon stocks

The results of the landscape simulations for regions 1 (brackish) and 2 (salt) are shown in Figure 5, whereas temporal changes in the simulated landscape structure are shown in movies M1–M4. In the brackish region (Figure 5a,c), the initial estimated average SOC stock was 2.80 kg OC m⁻²

when the landscape consisted solely of tidal sand flats, as calculated for the uppermost 400 kg m⁻² of sediments. Subsequent tidal marsh progradation and embankment led to increasing average SOC stocks to a maximum of 6.10 kg OC m⁻² following a period of intensive embankment construction. This resulted in large rates of marsh progradation and a relatively large area occupied by tidal marshes. After the last embankment (1973), the average estimated SOC stock declined and was calculated to reach an equilibrium of 4.70 kg OC m⁻² around the year 2100.

For the salt polder region (Figure 5b,d), the estimated average initial SOC stock was 0.59 kg OC m⁻². During the first 150 years simulated the average estimated SOC stock increased to 2.30 kg OC m⁻² with tidal marsh progradation. Subsequent embankments and growth of new tidal marshes during the next century resulted in limited variation in the total area of tidal marshes and a small increase in the average estimated SOC stock to a maximum of 3.11 kg OC m⁻². After the last embankment (1952), no new tidal marshes were initiated in this region leading to a steep decline in the average estimated SOC stock to 1.72 kg OC m⁻² at the end of the twenty-first century.

4. Discussion

4.1 Organic carbon losses from embanked tidal marsh sediments

4.1.1 Changes in soil organic carbon stocks

Tidal marshes located at different degrees of salinity along an estuary have been shown to differ substantially with regard to (i) the amount of OC that is present in their sediments and (ii) the sources from which this OC originates (Van de Broek *et al.*, 2016, 2018). In general, salt marshes trap OC from mainly marine sources and contain less SOC than brackish marshes, which trap a mixture of OC originating from marine and terrestrial sources. The larger SOC stocks in brackish marshes than salt marshes resulted in larger OC losses in the former (8.7 kg OC m⁻²; 86.6 g OC m⁻² yr⁻¹) than the latter (6.68 kg OC m⁻²; 111.3 g OC m⁻² yr⁻¹). However, the smaller equilibrium

318 SOC stock in polders originating from salt marshes resulted in similar relative OC losses in both
 319 systems (ca. 58 % of initial stocks).

320 The relative decline in OC stocks along the tidal marsh–polder chronosequences studied (58 %)
 321 was substantially larger than reported declines in SOC stock after the conversion of forest to arable
 322 land in temperate (-32 ± 20 %) (Poeplau *et al.*, 2011) and tropical ecosystems (-25 %) (Don *et al.*,
 323 2011). This is not surprising because tidal marsh sediments generally contain more SOC than
 324 terrestrial forests (Duarte *et al.*, 2013; De Vos *et al.*, 2015), however, the equilibrium SOC stocks
 325 in the polder soils studied was similar to that in the top 0.3 m of non-polder farmland in western
 326 Europe (Lettens *et al.*, 2004).

327 Comparing these results with those from previous studies in intertidal landscapes was not
 328 straightforward because most studies reported only changes in OC concentrations without
 329 reporting changes in OC stocks (Ellis & Atherton, 2003; Iost *et al.*, 2007; Cui *et al.*, 2012). Studies
 330 that report SOC stocks, however, usually do so by comparing SOC stocks to a common depth
 331 between different sites. For example, Bu *et al.* (2015) reported a the loss of 21 % of initial OC in
 332 the top metre after embankment, whereas Bai *et al.* (2013) and Deng *et al.* (2016) reported declines
 333 in SOC stocks of 44 % (0–0.2-m depth) and 30 % (0–0.5-m depth), respectively. These studies did
 334 not account for soil compaction after the embankment of coastal wetlands, which has been shown
 335 to be considerable not only in our study area (section 3.1) but also elsewhere, with relative
 336 increases in bulk density of 25–50 % (Ellis & Atherton, 2003; Bai *et al.*, 2013; Cui *et al.*, 2014).
 337 Neglecting soil compaction when calculating changes in SOC stocks can therefore lead to a
 338 substantial underestimate of OC losses (Don *et al.*, 2011), and might explain why the losses
 339 reported here were larger than those reported in previous studies. Therefore, we strongly suggest

that future studies should account explicitly for changes in soil bulk density when calculating SOC losses along chronosequences.

4.1.2 Changes in organic carbon characteristics along the depth profile

The $\delta^{13}\text{C}$ value of allochthonous OC inputs to the salt marshes (-22 - -24 ‰ (Van de Broek *et al.*, 2018)) was substantially different from the $\delta^{13}\text{C}$ value of terrestrial vegetation (-25 - -30 ‰, with an average of -27 ‰ (Kohn, 2010)). This was reflected in the $\delta^{13}\text{C}$ depth profiles of the salt polders (Figure 2d) where the $\delta^{13}\text{C}$ value of OC in the saltmarsh sediments increased to values associated with agricultural OC after conversion (-26 - -27 ‰ in the top 0.4 m), indicating replacement of the initial tidal marsh OC by inputs from agricultural crops. In the brackish marsh (Figure 2b), the $\delta^{13}\text{C}$ of topsoil sediments deviates from the pattern observed along the chronosequence. This can be attributed potentially to intensive dredging in this part of the estuary from the 1970s onwards, which have been shown to have a large effect on the sediment dynamics in the estuary (van Kessel *et al.*, 2015). Dredging is done to deepen the navigation channel through which large ships can reach the port of Antwerp. Typically, the dredged bedload material is redeposited in other parts of the Scheldt estuary, thereby disturbing the natural sediment dynamics there (van Kessel *et al.*, 2015). Consequently, the characteristics of sediments deposited on tidal marshes close to dredging sites might be altered substantially. This could explain the deviating pattern in $\delta^{13}\text{C}$ values in the top 0.3 m of the brackish marsh (Figure 2b). The pattern of decreasing $\delta^{13}\text{C}$ values with increasing age in the brackish polder sediments indicates that after tidal marsh embankment the original sedimentary OC is being replaced by OC originating from agricultural inputs.

The rate at which tidal marsh OC is replaced by OC from agricultural practices was substantially different between the top 0.4 m and deeper soil layers (Figure 2). In the top 0.4 m, which is directly affected by ploughing and rooting, the $\delta^{13}\text{C}$ values shifted to terrestrial values (-27 ‰) on a

timescale of ca. one century in all sampled chronosequences (see Figures S8–S12 for all depth profiles). This indicates that, although the total amount of OC did not change substantially from ca. 60 years after embankment onwards, there was still a substantial replacement of OC for about a century after conversion. In the subsoil (> 0.4 -m depth), the shift in $\delta^{13}\text{C}$ values towards terrestrial values takes substantially more time. Brackish polders to ages of ca. 250 years have subsoil $\delta^{13}\text{C}$ values of -25 – -26 ‰, indicating the presence of a substantial fraction of tidal marsh OC. Only after ca. 320 years a $\delta^{13}\text{C}$ value of -27 ‰ was reached in the subsoil (Figure S8). In the salt regions, polders reclaimed < 167 years ago had subsoil $\delta^{13}\text{C}$ values between -26 and -24 ‰. Only in the oldest polder, with an age of 416 years, did subsoil $\delta^{13}\text{C}$ values reach ca. -27 ‰ in the top 0.6 m (Figure S11, Supporting Information). These data showed that the time needed for complete replacement of subsoil tidal marsh OC with OC from agricultural crops was in the order of centuries. It is likely that different mechanisms of OC inputs and soil disturbance along the soil profile of polder soils control the rates of OC replacement. In topsoils, on the one hand, soil drainage and aeration (through ploughing) are expected to stimulate the decomposition of marsh OC, and direct inputs of OC from crop residues and roots are likely to replace OC rapidly. In subsoils, on the other hand, leaching of DOC from the topsoil and subsequent interactions with the mineral phase of the soil are likely to control the slower rate of SOC replacement (Kaiser & Kalbitz, 2012; Balesdent *et al.*, 2018).

4.2 Landscape-scale changes in soil organic carbon storage

Our findings that tidal marsh sediments lose a substantial amount of OC on embankment accord with previous research, which has similarly reported large SOC losses with the conversion of coastal wetlands (Pendleton *et al.*, 2012). However, previous analyses did not account for geographical variation in losses and gains in SOC during stepwise embankments, thereby

disregarding additional SOC storage in newly created wetlands. The results of the landscape-scale SOC model (Figure 5) show that average landscape-scale SOC stocks vary considerably at decadal to centennial timescales during the course of land reclamation by stepwise embankment. The new equilibrium landscape-scale SOC stocks increased by ca. 2 and 1.1 kg OC m⁻² in brackish and salt polders, respectively, compared to the initial sand flat. Similarly, Bai *et al.* (2013) reported increasing regional SOC stocks after the conversion of tidal marshes from additional OC storage in ditches in the reclaimed region. In addition, Huang *et al.* (2010) found that the conversion of terrestrial marshland to farm land in a large region in northeast China (Sanjiang plain, 11 Mha) resulted in reduced regional greenhouse gas emissions, despite large losses of SOC with the conversion of the marshes. The observed reduction in greenhouse gas emissions was mainly a consequence of reduced methane (CH₄) emissions because of a reduction in the area occupied by marshland, a large source of CH₄. These results stress the need to account for multiple processes when assessing the greenhouse gas balance of marshland, whether terrestrial or estuarine, after conversion to agricultural land use types.

The simulated landscape-scale equilibrium SOC stocks were significantly larger for the brackish (4.70 kg OC m⁻²) than the salt polders (1.72 kg OC m⁻²). This is a consequence of (i) larger sedimentation rates of OC-rich sediments at the surface of brackish marshes (Temmerman *et al.*, 2004), (ii) larger equilibrium SOC stocks in the brackish polders (Table 2) and (iii) a substantial number of tidal marshes in the brackish region after construction of the last embankment, which was not the case in the salt region. In addition, the model results showed that the rate of increase of simulated landscape-scale SOC stock was largest when the area covered by tidal marshes was the largest (Figure 5). This also explains why the landscape-scale SOC stocks were largest during the period of active embankment when tidal marshes were growing both vertically and laterally.

409 In the model, the assumption is made that the construction of an embankment leads to the growth
410 of a new tidal marsh. It can be hypothesized that, under natural conditions, the entire initial sand
411 flat would eventually become covered by tidal marshes as a consequence of natural marsh
412 progradation. In that case, the area would store more sedimentary OC than in the current situation.
413 Previous studies in other estuaries have found that the construction of embankments led to a
414 reduction of the tidal prism, i.e. the volume of ebb discharge during a tidal cycle. This reduction
415 led to lower water velocities and increasing rates of sediment deposition after embankment
416 construction (Gregory Hood, 2004; Cuvilliez *et al.*, 2009; Xie *et al.*, 2018). In the Scheldt estuary,
417 in contrast, no reduction in the volume of the tidal prism was observed over the past centuries
418 because the reduction of intertidal surface area was compensated for by an increase in the celerity
419 and amplitude of the tidal wave (Van den Berg *et al.*, 1996). The modelled polder regions were in
420 small intertidal basins fringing the estuary (Figure 1), therefore, it is possible that the volume of
421 the local tidal prism was reduced after the construction of each embankment, promoting sediment
422 deposition. In addition, there has been a net import of sand into the estuary over the past centuries
423 (Van den Berg *et al.*, 1996). As a consequence of the increase in embanked areas over time, this
424 increase in amount of sediment had to be deposited on a smaller surface area. This led to potentially
425 increasing rates of sediment deposition and marsh progradation. Jongepier *et al.* (2015) showed
426 that the rate at which marshes next to an embankment expand laterally was largest in the first
427 decade after embankment construction (40–200 m year⁻¹). The average rate of lateral expansion
428 for marshes older than ca. 20 years was much less (< 10 m year⁻¹). This suggests that the
429 construction of an embankment can induce the formation of a new, fast-growing tidal marsh.
430 Although these observations do not provide direct evidence, they do suggest that the process of

stepwise embankments has positively affected marsh progradation in the regions studied over the past centuries.

5. Conclusion

In this study, the effect of the stepwise embankment of tidal marshes on sedimentary OC stocks in the Scheldt estuary was studied (The Netherlands). Our results showed that with embankment, the absolute amount of sedimentary OC lost from brackish marsh sediments ($8.7 \pm 1.8 \text{ kg OC m}^{-2}$ at a rate of $86.6 \pm 18.3 \text{ g OC m}^{-2} \text{ yr}^{-1}$) was larger than SOC losses from salt marsh sediments ($6.8 \pm 3.1 \text{ kg OC m}^{-2}$ at a rate of $111.3 \pm 51.8 \text{ g OC m}^{-2} \text{ yr}^{-1}$). In addition, the time to reach a new equilibrium for SOC stocks was considerably shorter in polders originating from salt marsh sediments (< 60 years) than those originating from brackish marsh sediments (ca. 100 years). However, $\delta^{13}\text{C}$ depth profiles showed that the time over which tidal marsh OC is replaced by terrestrial OC was substantially longer than that required for total OC stocks to reach equilibrium; it might take over 300 years to replace most of the tidal marsh OC in the subsoil. Reconstruction of historical embankments and their effects on regional SOC stocks showed that landscape-scale SOC stock increased substantially during periods when tidal marshes were expanding at large rates, whereas these landscapes lost large amounts of SOC once they became dominated by embanked and drained marshes. These results showed that management strategies aiming to maximize SOC storage in intertidal landscapes, while at the same time benefiting from many other ecosystem services provided by tidal marshes, should prevent complete embankment of these vegetated coastal ecosystems. This is of great importance because more embankments of coastal wetlands are planned for the near future in many economically developing countries.

Supporting Information

Please list this

Supplementary Data

List

Conflict of interest statement

The authors confirm that there is no conflict of interest.

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Figure 1 Map of the sampled salt and brackish polders along the Scheldt estuary and associated tidal marshes (dark grey) and sand flats (light grey). Polder regions in dashed polygons have been used for modelling. The numbers refer to the different polder regions (Table 1). Detailed maps of the different regions are provided in Figures S2–S6. Source of the background digital elevation model: EU-DEM (<https://www.eea.europa.eu/data-and-maps/data/eu-dem>).

592

593 **Figure 2** Depth profiles of organic carbon concentration (%) (a ,c) and $\delta^{13}\text{C}$ values (‰) (b, d) of
 594 selected polder soils in the brackish (a, b) and salt (c, d) region (regions 1 and 2 in Figure 1,
 595 respectively) with a different time since embankment (age). The blue dashed lines represent a
 596 tidal marsh in the same region. The legend of (b) is similar to (a), the legend of (d) is similar to

(c). Error bars represent the standard deviation based on three replicates, shown for selected OC points to improve readability (every 0.18 m) and were not available for $\delta^{13}\text{C}$.

Figure 3 Mass of organic carbon (kg OC m⁻²) in the uppermost 400 kg m⁻² of sediments in both present-day tidal marsh sediments and embanked salt and brackish polder soils. Error bars represent the standard deviation based on 3 replicates.

Figure 4 Average $\delta^{13}\text{C}$ values (‰) of OC in the uppermost 400 kg m⁻² of sediments in both present-day tidal marsh sediments and embanked polder soils.

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608

609 **Figure 5** Results of the landscape-scale SOC model for the brackish (region 1; a, c) and salt
 610 (region 2; b, d) polder landscapes. Panels (a) and (b) show the temporal variation in the surface
 611 area (km²) covered by tidal marshes, polders and sand flats, panels (c) and (d) show the
 612 temporal change in simulated average SOC stock for the simulated regions for the upper 400 kg
 613 m⁻² of sediments (kg OC m⁻²), calculated for the entire simulated region.

Table 1 Characteristics of the sampled polders and tidal marshes per region (s denotes the present-day average salinity of the estuarine water at the respective locations).

Name and embankment year	Sediment depth /m ^a	Name and embankment year	Sediment depth /m ^a
<i>Region 1 (brackish, s = 17)</i>		Van Dunnépolder (1907)	1.8
Waarde marsh		Van Wuijckhuisepolder (1912)	0.55
Waardepolder (1571)	0.3	Dijckmeesterpolder (1920)	0.6
Mairepolder (1694)	0.4	Braakmanpolder (1952)	0.55
Reigersbergsche polder (1773)	0.7		
Tweede Bathpolder (1862)	0.5	<i>Region 3 (salt, s = 30)</i>	
Emmanuëlpolder (1864)	> 0.9	Sloehaven marsh	
Damespolder (1884)	0.4	Nieuwe West-Kraaijertpolder (1676)	0.5
Völckerpolder (1904)	0.5	Jacobpolder (1856)	0.55
Kreekrakpolder (1923)	0.35	Quarlespolder (1949)	1.05
Unknown (1953)	> 0.9		
Paviljoenpolder (1973)	> 0.7	<i>Region 4 (salt, s = 24)</i>	
		Zuidgors marsh	
<i>Region 2 (salt, s = 27)</i>		Hattumpolder (1957)	0.45
Paulina marsh			
Koudenpolder (1542)	0.75	<i>Region 5 (salt, s = 22)</i>	
		Hellegat marsh	
Kleine Stellepolder (1866)	0.55	Aan- en Genderdijkepolder (1600)	1.1
Koninginnepolder (1893)	0.35	Van Lijndenpolder (1877)	0.55
Mosselpolder (1900)	0.55	Hellegatpolder (1926)	0.55

^aDepth of former tidal marsh sediments at the sampling location.

Table 2 Depths to which SOC stocks were calculated (for a mass of 400 kg m⁻² of bulk soil) and the average SOC stocks of soils in the different regions. Standard deviations represent the spatial variation within the sampled polders based on three replicate soil profiles. The rate of OC loss for the brackish polders was calculated for a time span of 100 years, for the salt polders a time span of 60 years was used. For tidal marshes for which only two locations were sampled, the standard deviation was not calculated, as indicated with *.

	Average marsh sediment depth /m	Average polder soil depth /m	Time to OC equilibri um /years	Average marsh OC /kg m ⁻²	Average polder equilibriu m OC /kg m ⁻²	Absolute average OC loss /g OC m ⁻² yr ⁻¹	Relative OC loss /%
Region 1	0.56	0.35	Ca. 100	14.76*	6.10 ± 1.83	86.6 ± 18.3	59 ± 12 %
Region 2	0.52	0.28	< 64	9.23*	4.41 ± 0.54	80.5 ± 9	52 ± 6 %
Region 3	0.59	0.29	< 66	14.41*	5.45 ± 0.78	149.3 ± 13	62 ± 5 %
Region 5	0.63	0.32	< 89	12.54*	6.34 ± 1.37	103.3 ± 22.8	49 ± 11 %
All salt regions combined	0.59	0.29	< 64	11.82 ± 2.90	5.14 ± 1.12	111.3 ± 51.8	57 ± 30 %