



Vulnerability of diked marsh ecosystems under climate change

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Received: 1 August 2022 / Accepted: 10 February 2023
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Abstract

Diked marsh soils are natural laboratories where soil-forming processes take place over a short period of time, such as the aeration of previously water-saturated soil environments along with desalinization. These manmade ecosystems are threatened by climate change in multiple ways. Since long-term data to evaluate the vulnerability of these settings is scarce, we merged hydrological (water table, WT; electrical conductivity, EC; sea level rise), pedological (redox potential, E_H ; air-filled porosity, AFP), and meteorological variables (evapotranspiration, ET_0 ; climatic water balance, CWB), and discussed the holistic relationship between these under future climate scenarios. Our multifactorial data identified ET_0 as the strongest driver of WT development with a causal dependency on AFP and subsequently on E_H . Within 11 years of intense monitoring, we encountered an extension of the soils' aeration windows ($E_H > 300$ mV) due to an enhanced seasonal WT component; i.e., the difference between winter and summer WT positions increased. This process has an impact on capillary rise from groundwaters and EC patterns due to increased seasonal variations. Desalinization stabilized two decades after diking, and the present EC does not indicate any saltwater intrusion to these near-coastal settings at present. However, sea level rise and a reduced CWB in the future will foster capillary rise from potentially salt-enriched groundwaters into the topsoils of these highly productive ecosystems. These mechanisms need to be evaluated to account for climate change-driven impacts on coastal-diked marsh soils. Indeed, a holistic view of pedological, meteorological, and hydrological variables is urgently needed.

Keywords Marsh soil · Monitoring · Electrical conductivity · Pedogenesis · Sea level rise · Climate change

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1 Introduction

The Wadden Sea is an intertidal shallow inshore zone which extends from the Netherlands to Denmark (de Jonge et al. 1993). Under conditions of rising sea level during the Holocene, the Wadden Sea ecosystem has evolved over the last 8000 years and, thus, is a very young ecosystem both from a geomorphological and an evolutionary perspective (CWSS 2017). The management of the coastal shoreline has been enhanced since AD 1600 with many areas being separated from the sea due to coastal engineering approaches (de Jonge et al. 1993), whereby the oldest and initial embankments along the coast of Schleswig–Holstein date back to the middle of the eleventh century (Goeldner 1999; Pons and Van Der Molen 1973). Soils within diked and drained areas are natural laboratories to study soil-forming processes that take place within years, as soon as seawater is prevented from flooding the soil during storm surges. Characteristic processes that take place feature at a relatively short time scale from years to decades (i) rapid changes in soil chemistry including oxidation of sulfidic material due to the lowering of the water table and (ii) changes of soil physical properties due to sediment subsidence (Portnoy 1999). Especially with respect to the latter point, they are fundamentally different in comparison with non-diked salt marshes that are regularly flooded by seawater and build up vertically by suspended sediment supply. Salt marsh accretion rates with 0.3–5 up to $> 30 \text{ mm year}^{-1}$ were reported globally (Coleman et al. 2022). Overall, embanked marsh ecosystems are of utmost a product of human activity (Joyce 2014). Generally, the sediments along the Wadden Sea are finely stratified, saline, calcareous, and sulfidic, whereby diking and drainage foster soil formation from a saline marsh towards a calcareous marsh (Blume and Schlichting 1985). The chronosequence of soils features a transition from *Salzmarsch* towards *Kalkmarsch* according to the German Soil Classification (AG Boden 2005), equivalent to a transition of *hydraquents* to *fluvaquents* according to US Soil Taxonomy (Soil Survey Staff 2014). The global extent of salt marsh area (pan, brackish, and saline wetlands) is not well inventoried with roughly 0.3% of the total surface area and 5% of total wetland area (Zedler et al. 2008). To the best of our knowledge, we have not found spatial information about the spatial coverage and distribution of embanked areas on an international scale. However, considering the fact that these soil types (e.g., *Calcaric Gleysols*) constitute the most fertile arable soil in Germany with a (fiscal) soil quality assessment value up to 90 from 100 numbers, their relevance for food supply if used agriculturally is of utmost importance.

Under humid conditions, desalinization is an important soil-forming process in diked salt marshes. Following polder construction and drainage, the so-called ripening is initiated and causes air to penetrate the unaerated soil material, and the artificial lowering of the water table (WT) depth enhances the leaching of salts from the topsoil (Pons and Van Der Molen 1973; Schroeder and Brümmer 1969). Thus, soil salinity progressively changes from saltwater with $> 25 \text{ g L}^{-1}$, towards brackish conditions with 1 to 25 g L^{-1} , and finally into freshwater with $< 1 \text{ g L}^{-1}$ total dissolved solids, respectively (Seim 1990). Salinity also affects seed germination rates and nutrient uptake in native and invasive marsh plants and is critical to achieve the re-establishment of plant communities depending on the wetland restoration goals (Baldwin et al. 1996; Bradley and Morris 1991; Moore et al. 2011). Therefore, knowledge of the salinity is important to assess potential threats when saltwater infiltrates freshwater aquifers and potentially rises the groundwater, as is the case for diked marsh soils prone to sea level rise. Besides desalinization, diked marsh soils feature oscillating redox conditions as evident by the continuous monitoring of the redox potential (E_H) using permanently installed platinum electrodes (Mansfeldt 1993). Enhanced

evapotranspiration with accompanying decreased soil moisture during summers certainly affects the redox conditions in the coastal vegetated soils of salt marshes (Blume and Müller-Thomsen 2007) but should also be relevant for embanked marsh soils at a distance from the diked shoreline. A deepening of oxygen penetration depths not only increases the decomposition of soil organic matter (SOM) via aerobic respiration and alters greenhouse gas emissions (Kroeger et al. 2017) but is also expected to drive the oxidation of reduced minerals (e.g., iron monosulfide) (Blume and Müller-Thomsen 2007; Spivak et al. 2019). These impacts on diked marsh soils are typically not assessed on the long-term.

Future scenarios considering all available projections suggest that the North Sea region, including the Meldorf Bay as part of the German Bight, is experiencing a changing climate (Quante and Colijn 2016). Anthropogenic climate change is a threat to coastal areas in general (IPCC 2019) with multifaceted impacts on diked marsh soils. A mean warming of 1.7–3.2 °C for the period from 2071 to 2100 under the Representative Concentration Pathways (RCP) 4.5 and 8.5 scenarios, a reduction of precipitation during warm seasons, and accelerated sea level rise by 2100 of up to 1 m are only some of the expected consequences (Quante and Colijn 2016). The individual climate change–induced risks for soils are in most cases expressed in a qualitative manner or based on mathematical simulations (Pfeiffer et al. 2017). However, determining a quantitative relationship would be advantageous, e.g., as recently shown for the relationship between the climatic water balance (CWB) and the redox status of soils featuring a perched water table (Dorau et al. 2020).

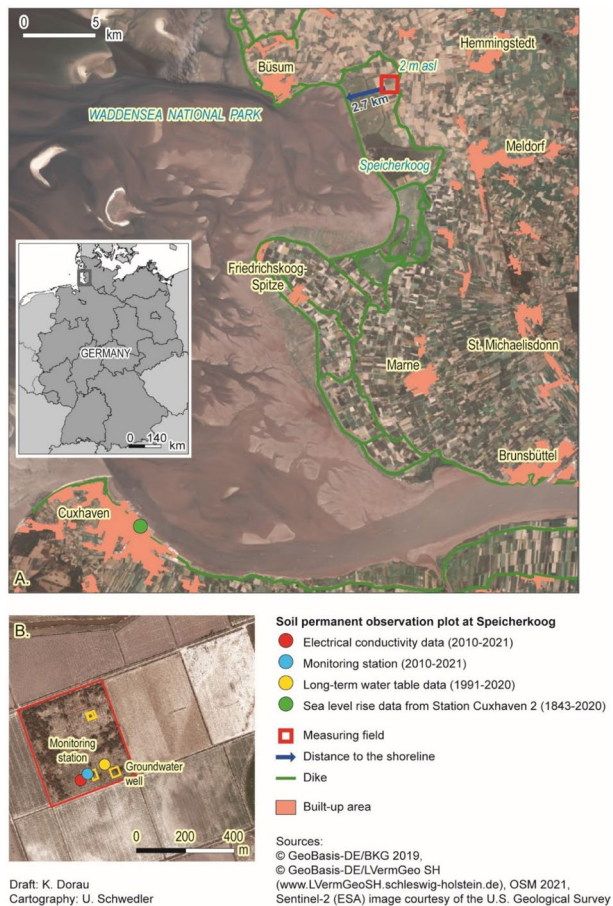
More than 10% of the world's population live in coastal areas, and this habitat is important for food production with fertile soils. These highly productive environments are particularly vulnerable to multifaceted climate change, e.g., due to salinization during rising sea levels (Chmura et al. 2003; Ratliff et al. 2015). The value of diked marshes to provide ecosystem services such as food production, habitat and refugia, and disturbance regulation cannot be exaggerated (Gedan et al. 2009). Most of the studies on marsh migration into uplands focused on forested land, while agricultural land was not in the focus yet. In this study, we merged meteorological, hydrological, and pedological data with the overall goal of (i) discussing the relationship and dependency of meteorological forcing on soil hydrology and pedological features such as the soil aeration status and (ii) addressing the vulnerability of diked marsh soils under future climate and sea level rise scenarios embedded within an agrarian landscape. By doing so, we tackle some important questions and influencing factors of climate change for a diked marsh soil in Northern Germany.

2 Materials and methods

2.1 Study site

Polder Speicherkoog is situated 30 km north of the Elbe River in Schleswig–Holstein, Northern Germany (54°8'1"N, 8°58'28" E; 2.1 m asl). It is part of the Meldorf Bay and was diked in 1978, making the area one of the youngest diked polders along the German North Sea coastline (Ricklefs and Asp Neto 2005). The soil developed from calcareous marine sediments and is classified as a Calcaric Gleysol (Eutric) according to FAO (IUSS Working Group WRB 2022). As a special feature located 3 km away from the shoreline (Fig. 1), the study site is embedded within an agrarian landscape but has not undergone any cultivation practices. Thus, besides diking and the construction of ditches and drainage systems

Fig. 1 Overview of the study site (A) with a close-up of the permanent observation plot at Polder Speicherkoog (B)



around the study site, the soil properties and the climate-driven impact on the redox conditions of the soil itself have not been influenced by anthropogenic factors.

2.2 Data collection

All measurements were performed on the same field and were obtained < 50 m apart from each other (Fig. 1). The longest consecutive and ongoing monitoring is based on biweekly water table readings from May 1991 until December 2020 (Klüver et al. 2020). Manual readings of the electrical conductivity (EC) were independently conducted within a 200 cm (50 mm Ø) perforated polyvinyl chloride (PVC) pipe by a conductivity meter from January 1990 to December 1991, and this measurement was automated by a stand-alone monitoring station operating from April 2010 until present. All data at the monitoring station were collected on an hourly basis and featured the following parameters: the water table (WT) was measured by a PDL-CTD probe (ecoTech, Bonn, Germany) and the soil matric potential (MP) was measured with a pF-meter® until April 2016 followed by Tensiomark® measurements, both featuring the same measurement principle of the soils' molar heating capacity (ecoTech, Bonn, Germany). Thus, soil temperature (ST) was simultaneously measured at depths of 10, 20, 30, 60, 100, and 150 cm. Additionally, the soil redox potential (E_H) was measured by

permanently installed platinum electrodes in triplicate repetition per depth and placed in stellar configuration around a silver-silver chloride reference electrode (Ag–AgCl; 3 M internal KCl electrolyte). Long-term measurements of E_H have been proven to be reliable even in sulfidic soil horizons (Mansfeldt 1993) and, thus, are a valid tool to derive trends when the redox electrodes are placed continuously in the soil (Dorau and Mansfeldt 2016). Meteorological parameters (2 m above the soil surface) were measured by a multisensory weather transmitter (WXT520, Vaisala, Finland) and comprised the air temperature (AT; °C), solar radiation (SR; $W m^{-2}$), relative humidity (RH; %), wind speed (WS; $m s^{-1}$), and precipitation (PP; mm). The data derived by the automated measurements on an hourly basis were converted, when needed, to daily, weekly, monthly, and annual averages or sums. In order to visualize the data, contour plots to represent a 3D surface, e.g., to display time (x), soil depth (y), and the variable (z), were utilized by using linear interpolation between discrete soil depths.

2.3 Soil properties

For a detailed description of the soil properties from the study site, the reader is referred to Dorau and Mansfeldt (2016). Undisturbed soil samples were briefly obtained from the sensor depths by digging a soil pit close to the monitoring station in August 2013. Soil samples were obtained in triplicate per depth using a 250-cm³ steel cylinder and water retention curves were measured using the evaporation method with the HYPROP device (Meter Group AG, Munich, Germany). The samples were water saturated prior to the measurement procedure and the retention data were fitted with the Durner equation (Durner 1994) because it resulted in the lowest root mean square error for our retention data ($\theta_{RMSE} < 0.01$):

$$S_e = w_1(1 + (a_1 h)^{n_1})^{-m_1} + w_2(1 + (a_2 h)^{n_2})^{-m_2} \quad (1)$$

where S_e is the effective water content (-), w_1 and w_2 are weighting factors, and a_1 , a_2 , n_1 , n_2 , m_1 , and m_2 are empirical fitting parameters for the hydraulic function.

2.4 Post-processing of the data

The redox data were corrected by adding +207 mV to correct for the deviation of the standard hydrogen electrode against the Ag–AgCl reference electrode, and in addition, the E_H was corrected by incorporating the ST as suggested by Dorau et al. (2021) to minimize the diel E_H pattern. The soil water contents (θ ; $cm^3 cm^{-3}$) were calculated by implementing the matric potential into Eq. [1] and using the depth-specific fitting parameters of the hydraulic function. Thereupon, we aimed to calculate the air-filled porosity (AFP; $cm^3 cm^{-3}$) as:

$$AFP = \theta_s - \theta, \quad (2)$$

with θ_s being the volumetric water content at saturation ($cm^3 cm^{-3}$). Meteorological data served as an input to calculate the crop reference evapotranspiration (ET_0) according to the FAO-56 Penman–Monteith model using the “Evapotranspiration” package (Guo et al. 2016) within the RStudio environment (RStudio Team 2021). Based on ET_0 and PP, we calculated the climatic water balance (CWB; $mm d^{-1}$) as:

$$CWB = PP - ET_0 \quad (3)$$

2.5 Future scenarios until 2100

Data for the climate scenarios were obtained from the Cuxhaven synoptic meteorological station located 20 km south of the study area. Daily data comprised PP, AT_{\max} , AT_{\min} , RH_{\max} , RH_{\min} , SR, and WS from which the ET_0 was calculated from the present day up to the year 2100. The future scenarios were derived from the regional climate model STARS and featured the RCPs 2.6 and 8.5 with a q50 realization (German Meteorological Service 2009; Orłowski et al. 2008; Potsdam Institute for Climate Impact Research 2013). The ET_0 and CWB were calculated as previously described. In addition, long-term records of sea level data from 1843 until present were taken from the Cuxhaven 2 station ($53^{\circ}52'00.6''N$, $8^{\circ}42'59.7''E$; 8.8 cm mean tidal level above mean sea level). These data were extended by merging future projections of sea level rise from the IPCC Special Report on the Ocean and Cryosphere in a Changing Climate (IPCC SROCC 2019) for both the RCP 2.6 and 8.5 scenarios. While this approach seems generally valid to predict near future sea level change, uncertainties remain due to the accelerated non-linear ice-sheet contributions of the late twenty-first century (Wang et al. 2021).

2.6 Statistical analysis

We applied a trend analysis using the “Seasonal-Trend decomposition procedure based on Loess (STL)” (Cleveland et al. 1990) to decompose a time series into a trend, seasonal, and residual component. Input data on a monthly basis with a periodicity of the seasonal component of 12 (i.e., months per year), an s.window value of 25, and a high t.window value to smoothen the trend of 100 were used. We performed the analysis with the support of the “stlplus” package (Hafen 2016) within RStudio. Because some of the variables are interrelated, we reduced the dimensionality of the data and conducted a principle component analysis (PCA) using the “FactoMineR” package (Le et al. 2008) and the Spearman correlation coefficients were calculated and displayed as a correlogram using the “ggcorrplot” package (Kassambara 2019). Only data with a p value < 0.05 were considered as significant and therefore displayed.

3 Results and discussion

3.1 Current state of climatic conditions and future projections

For the study site featuring mild maritime climate, ET_0 was on average 440 mm and precipitation 720 mm resulting in a positive CWB of 240 mm year^{-1} (Fig. 2A to C). Among the years under investigation, the year 2018 was particularly warm with below average amounts of precipitation that cumulated into a negative CWB (Fig. 2A to C, Fig. S1A to S1C; Table 1). 2018 was the warmest and sunniest, and was one of the driest since 1881 with extreme droughts from February to November (DWD 2018). Generally, it has been shown that more warm and fewer cold extremes are being observed for the entire North Sea region (Quante and Colijn 2016) with 6.9 ice days and 2.2 hot days per year at present (Fig. S2A and S2B). The spring season received the fewest precipitation (Fig. S1B) and largely contributed to a negative CWB during the months of April and May (Fig. S1C). Under progressing climate change, ET_0 is expected to remain relatively stable under the RCP2.6 scenario but increases under the RCP8.5 scenario until 2100 (Fig. 2C bottom panel). This will certainly curtail water availability and thus groundwater storage, whereby evapotranspiration was identified as the main driver of climate-driven impacts on groundwater supply rather

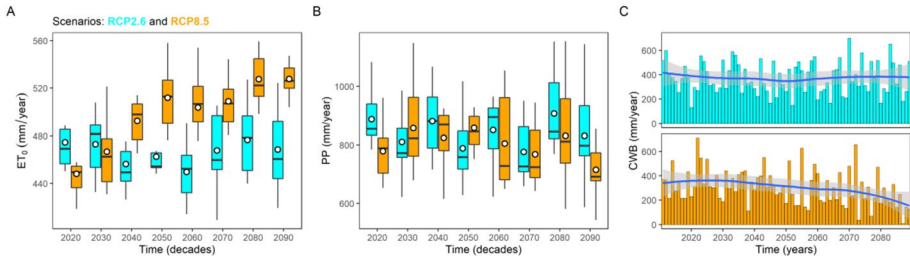


Fig. 2 The long-term projections for the RCP2.6 and RCP8.5 scenarios until 2100 shown for reference crop evapotranspiration according to the Penman–Monteith formulation (ET_0 ; **A**), precipitation (PP; **B**), and the climatic water balance (CWB; **C**)

than trends in precipitation throughout the twenty-first century in key mid-latitude aquifers (Wu et al. 2020). Up to now, summers have become drier and winters wetter with enhanced and more extreme precipitation events (Quante and Colijn 2016). The year 2018 is contrary to this trend because the winter season featured below average precipitation compared with the total study period (Fig. S6). With respect to the forecast until 2100, no clear trends can be deciphered considering the total amount of precipitation (Fig. 2E). Nevertheless, the likelihood for years with a CWB similar to 2018 increases under the RCP8.5 scenario (Fig. 2F).

3.2 Water table depth and electrical conductivity

Shallow groundwater prevails in the wintertime followed by a steep drawdown in the summertime up to -200 cm below ground level (Fig. 3A). The anti-cyclical behavior of WT and EC was accompanied with the lowest EC measured during March (0.49 ± 0.08 mS cm^{-1}) compared with

Table 1 Summary statistics of selected meteorological variables from the study site of Polder Speicherkoog for the observation period from 2011 to 2021. The year 2010 is not included since data acquisition started in April 2010

Year	ET_0^A (mm year ⁻¹)	Precipitation	CWB ^B
2011	471	794	323
2012	427	755	328
2013	423	666	243
2014	480	843	363
2015	396	944	548
2016	424	614	189
2017	401	863	462
2018	475	461	-14
2019	493	639	146
2020	521	861	340
2021	429	560	131
Average	—	449	278
Standard deviation	—	41	160

^AReference crop evapotranspiration according to the Penman–Monteith formulation

^BClimatic water balance

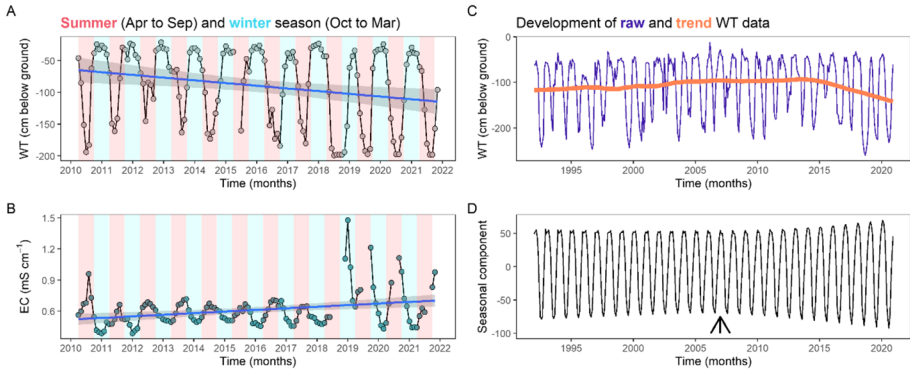


Fig. 3 Development of monthly water table (WT; **A**) and electrical conductivity (EC; **B**) for Polder Speicherkoog. No EC could be measured when the WT was below 1.8 m, e.g., in the summer of 2018. The gray fields in **A** and **B** highlight the 95% confidence interval for predictions from a linear model represented by the blue line. Long-term WT development with the trend line is displayed in **C** along with the seasonal component derived by the seasonal and trend decomposition using Loess (STL) procedure (**D**)

October ($0.73 \pm 0.24 \text{ mS cm}^{-1}$) (Fig. 3B, Fig. S1E; $r_s -0.66, p < 0.001$). Typically, the WT rises during late September of each year, but due to the exceptionally dry conditions in 2018, the WT was even in December at -150 cm below ground level. Shortly after the re-wetting of the soil was established due to enhanced precipitation and water intake into the catchment, the EC rose to 1.5 mS cm^{-1} in January 2019, which was the highest value observed during the recent monitoring campaign. It seems reasonable that evapotranspiration processes influenced the chemical composition of shallow groundwaters due to an increase of ions, which was also shown by Kozłowski and Komisarek (2016). Long-term WT data from the study site indicated an increase from -117 cm in 1990 up to -93 cm in 2013, but the above-average warm years from 2018 to 2020 had an impact on the trend development and fostered a decrease of the WT (Fig. 3C). The absolute sea level has risen by $2.11 \text{ mm year}^{-1}$ over the past 170 years at the Cuxhaven station (Fig. 4A), which is slightly more than the global mean sea level (GMSL) rise of 21 cm from 1900 to 2020 at rates of 1.7 mm year^{-1} (Palmer et al. 2021). At present, the process of sea level rise is not reflected by WT data for the diked coastal marsh soil studied here because the trend actually indicates a drop in WT rather than increasing levels (Fig. 3C). This is mainly due to the seasonal component, which gained impact from 2007 onwards and underlines an enhanced

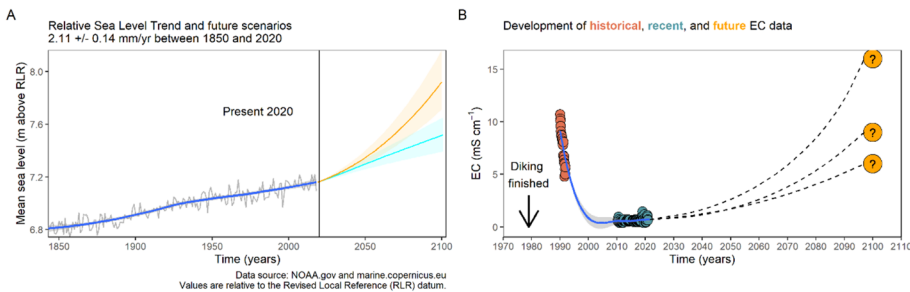


Fig. 4 Local data of sea level rise until present (2020) merged with modelled global sea level projections up to the year 2100 according to the latest IPCC Special Report on the Ocean and Cryosphere (SROCC) (A). Historical and recent electrical conductivity (EC) data were measured at Polder Speicherkoog with hypothesized development due to sea level rise in the future (B)

contrast between winter and summer WT depths (Fig. 3D), as exemplified for the year 2018. Overall, it is expected that summer WTs would drop more strongly, and this goes hand in hand with the cumulative number of aeration days, e.g., the number of days that the WT was below the 60-cm soil depth for a given year (Fig. S2E).

Typically, a rapid decrease of EC and the exchangeable sodium (Na) percentage occurs after the soil reclamation of diked marsh soils as shown under artificial conditions in the laboratory (Portnoy and Valiela 1997) or under field conditions along a chronosequence (Iost et al. 2007). To the best of our knowledge, long-term information under natural conditions is not available in the literature. Figure 4B displays a steady decrease in EC ten years after diking from 10.7 mS cm^{-1} in January 1990 up to $0.61 \pm 0.18 \text{ mS cm}^{-1}$ on average for the recent monitoring campaign. We neither have evidence that the position of the WT shifted due to sea level rise, nor that EC increased considerably due to the leakage of salts into the aquifer.

3.3 Soil aeration window

Reference evapotranspiration peaks in the summer season and the annual pattern have a striking effect on the WT development (Fig. S3A). Accompanied by a drop in the WT, the soil profile successively drained and the period when the soil pores have an AFP $> 0.02 \text{ cm}^3 \text{ cm}^{-3}$ agrees well with a switch from reducing towards oxidizing soil conditions in the temporarily water-saturated soil at depths of 30, 60, and 100 cm (Fig. 5A and B, Fig. S3C). Whereas the topsoil at 10- and 20-cm depths was consecutively aerated with $E_H > 300 \text{ mV}$ throughout the study period, the E_H at the 30-, 60-, and 100-cm depths fluctuated between strongly reducing ($E_H < -100 \text{ mV}$) and oxidizing ($E_H > 300 \text{ mV}$) conditions (Fig. 5B). This occurred because of the annual pattern in ET_0 and the resulting variable position of the WT (Fig. S1A, S1D, S1H, and S1I). The WT was the strongest explanatory variable to explain the spatiotemporal patterns in AFP and E_H in the temporarily water-saturated soil depths (Fig. 6A and B). However, the soil was strongly reducing at a depth of 150 cm throughout the study period. Currently, the soil features oxidizing conditions 200 days per year on average at the 60-cm depth but

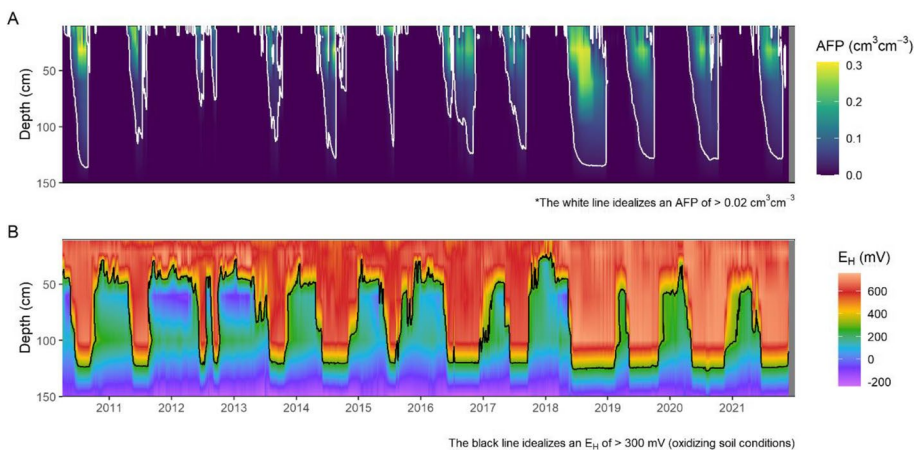


Fig. 5 Contour plots of air-filled porosity (AFP; **A**) and redox potential (E_H ; **B**) for the study site of Polder Speicherkoog

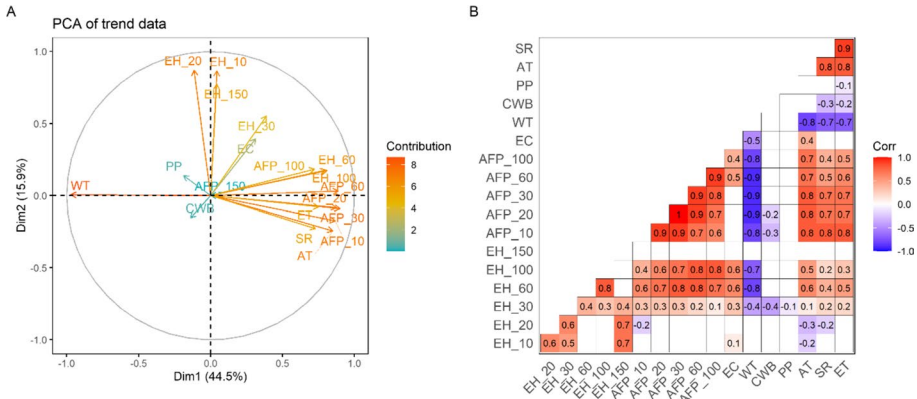


Fig. 6 Principal component analysis (A) and correlogram with Spearman correlation coefficients for selected variables from Polder Speicherkoog (B)

the pedological indicators of oxidizing days ($E_H > 300$ mV at pH 7) were above average for the dry years of 2018–2020 (Fig. S2D). An extension of the soil aeration window—which we define as the period when the soil E_H is > 300 mV at pH 7 at a particular depth, typically present above the water table and depending on the capillary fringe—in temporarily water-saturated soils impacts the biogeochemical processes and is therefore critical in various ways. Since wetland soils show large rates of organic accumulation due to anoxic conditions, lower soil moisture as was the case in the summer of 2018 (Fig. S4), and thereupon, the prevalence of oxidizing conditions enhances decomposition and SOM turnover (Schlesinger and Emily 2013), and this might occur more often under the RCP8.5 scenario (Fig. 2). The enhanced seasonality of the WT and extending the aeration window produces very large changes in decomposition rates—which render diked-marsh soils particularly vulnerable to enhance climate change through positive feedback mechanisms—compared to changes in temperature and moisture in other regions of the world (Sierra et al. 2015).

3.4 Vulnerability of diked marsh ecosystems

Coastal engineering approaches, such as the construction of dikes, change the soil chemistry of salt marshes (Portnoy 1999) and more generally soil formation mechanisms. As a result, the chemical composition of groundwater changes, whereby the EC is referred to as a master variable in addition to temperature, pH, and redox potential (Michalski 1989). Monitoring the EC reveals important information regarding saltwater leaking into an aquifer (Michalski 1989). In the present study, desalinization occurred soon after diking was initiated but at present, we have no evidence of tidal impacts by storm surges, nor did we observe an increase in EC over time for the period from 2010 to 2021 (Fig. 4B). However, sites located closer to the shoreline might already exhibit progressive salinization due to saltwater intrusion. For instance, Nguyen et al. (2014) studied the spatiotemporal variability of salinity of diked marsh soils and found a higher EC at low elevations. Even subtle altitude differences (cm to dm) had pronounced impact and study sites closer to the adjacent river connected to the lagoon or sites closer to the dike had higher EC. However, study

sites in 2 km distance from the dike—similar to the distance within our study—did not indicate any intrusion of seawater.

Capillary rise can draw saline water up to the topsoil even when the position of the WT is as deep as 2 m below ground (Eamus et al. 2006), as is the case for the study site (Fig. 1). Three mechanisms are likely to contribute to salinization in the near future: (i) enhanced ET_0 with prolonged periods of WT drawdown and enhanced WT seasonality, (ii) a prolonged period per year when capillary rise from groundwater exceeds drainage, and (iii) sea level rise with accompanied saltwater intrusion. Regarding the first point, ET_0 significantly rises from 448 ± 19 for the decade from 2020 to 2030 towards 535 ± 30 mm year⁻¹ until 2100 in the worst-case RCP8.5 scenario (Fig. 2D). Since precipitation is expected to be relatively constant (Fig. 2E), this has implications for the CWB that tends towards more neutral values and enhances the likelihood of a negative CWB, as was the case during the extreme summer of 2018 (Fig. 2C) and will happen on rare occasions in the future (Fig. 2F). However, ET_0 development in the future is uncertain to predict because of man-made-associated changes in vegetation cover by the conversion from marshland towards cultivated land (Shen et al. 2020). Also natural shifts of the marsh-upland boundary due to sea level rise, e.g., by changing upland vegetation replaced by halophyte marsh plants (Fagherazzi et al. 2019), will certainly alters the water budget due to variable ET_0 . For instance, plant transpiration is the dominant pathway of atmospheric water flux in tidal freshwater marsh but not relevant in salt marsh (Hussey and Odum 1992). Changes in vegetation cover are potentially slowed down in diked marsh ecosystems prevented from overflow of saline water by storm surges over non-diked areas. Second, an enhanced contrast between winter and summer WT depths (Fig. 4B) and changes of the hydraulic gradient with prolonged periods of capillary rise during the summer season (Fig. S4B) enhance the translocation of salts from groundwater. Hydraulic gradients verified drainage conditions during winter for 95% of the time (Fig. S5A), whereas capillary rise prevailed for 40% of the time during the summer (Fig. S5B). The subsoil depths were particularly vulnerable during the above average dry years and would potentially contribute to salinization during the summer season in the future. In addition, since the study site is embedded within an agrarian landscape, the magnitude and variability of variable PP patterns along with global population growth is likely to increase irrigation, which exacerbates irrigation-driven salinization (Herbert et al. 2015). Third and finally, global sea level rise is occurring at an unprecedented rate and is expected to rise by 0.435 m (0.285 to 0.589 uncertainty range) or by 0.842 m (0.609 to 1.105) in the RCP2.6 and 8.5 scenarios, respectively (Fig. 4A). Consequently, salinization is a widespread threat to the functioning of inland and coastal wetlands with manifold impacts on their biogeochemistry (Herbert et al. 2015).

In addition to problems associated with salinization, SOM decomposes slower under anaerobic conditions than with aerobic conditions (Schlesinger and Emily 2013) and, thus, an extension of the soils' aeration window in the future represents a major positive feedback mechanism for climate change. To which extent this process is counterbalanced by sea level rise remains unclear and should be evaluated consecutively by national and international monitoring programs. What has been reported so far is that methane (CH₄) emission due to water logging in tidal saline wetlands appears to decrease with salinity and increase in soil temperature. Marsh sites have abundant SO₄²⁻ for SOM decomposition through sulfate reduction rather than producing CH₄ as it is the case in peatlands (discussed in Chmura et al. 2003). Thereupon, subsoil warming up to +2.3 °C even in 100-cm depth since the mid-twentieth century (Dorau et al. 2022) would certainly further reduce CH₄ emission under these particular cases. Wetland soils capture the highest contents of SOM in the biosphere (Nahlik and Fennessy 2016) with a more complex composition

as previously thought having modelled turnover times between 640 to 9951 years (Luk et al. 2021). Blume and Müller-Thomsen (2007) discussed possible soil processes for non-diked salt marshes under climate change and mentioned, in addition to diked marsh soils, enhanced bioturbation, the formation of soil structures, redoximorphism, acidification, and thereupon silicate weathering, as further. Contrary to diked marsh soils, sea level rise might be counteracted by sedimentation for non-diked areas (Blume and Müller-Thomsen 2007) and, thus, the impact on soil formation or transformation is very site specific depending on diked and non-diked areas.

3.5 Mechanisms of future soil formation and study limitations

Soil formation directly after embankment of salt marshes and creation of manmade anthropogenic drainage proceeds relatively fast within years to decades (i.e., marsh ripening; Fig. 7A and B). Rising sea levels along with re-salinization is by far more difficult to verify, because these processes occur more likely over decades to centuries. Diked marsh soils are both impacted by above (i.e., meteorological conditions) and belowground drivers (i.e., hydrological conditions). The mechanisms and status quo of relevant processes highlighted within this study are summarized in Fig. 7C. Climatic forcing is a strong driver and forces soil water shortage in the summer season, extends the soil aeration window, and prolongs the period by capillary rise where salts might potentially be translocated upwards in the soil profile from a saline groundwater. As mentioned earlier, the outcome of each of these processes is very site dependent since sea level rise is not consistent throughout the world and dike systems, adequate drainage facilities, and agrarian irrigation systems are very individual (Nguyen et al. 2014). Therefore, the results and identified processes from this study are certainly not a blueprint for all diked marsh soils. Thereupon, monitoring pedological, hydrological, and meteorological data at high temporal resolution over decade-long periods is important, but the results must be carefully portrayed when observed for a single site, as it is the case for the given study. However, long-term areal soil monitoring, if coming from a single site or from a multitude of sites, is urgently needed as a diagnosis and prognosis tool to unravel the multifaceted threat of climate change on diked marsh soils.

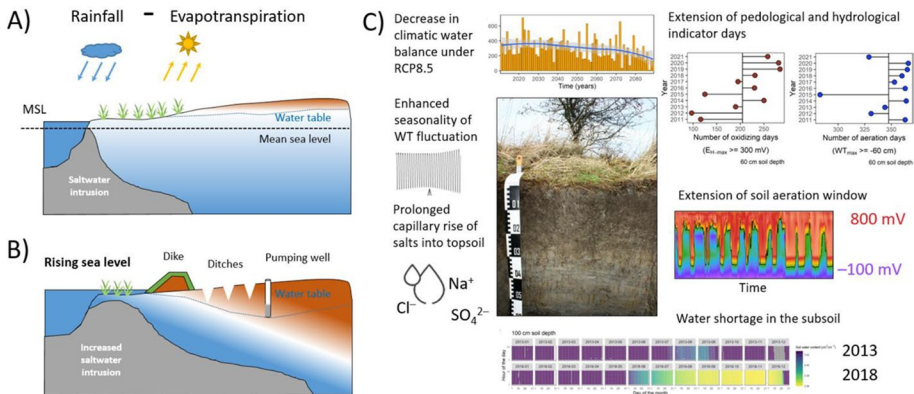


Fig. 7 Schematic diagram of non-diked salt marshes (A), and the current status for diked marsh ecosystems under progressing climate change, e.g., due to sea level rise (B). The vulnerability towards soils within these ecosystems (exemplarily for a Calcaric Gleysol) is summarized and discussed for long-term data and future forecasts derived within this study (C)

4 Conclusions

Real long-term data are rarely measured to evaluate consequences for marsh soils and diked marsh soils received less attention in comparison with salt marshes. This highlights the need for these particular ecosystems that besides meteorological parameters, soil-related monitoring is mandatory to investigate the footprint of climate change. Therefore, we identified two key processes that render agrarian used diked marsh soils vulnerable to climate change. First, it is only a matter of time until saltwater intrusion impacts the soils of diked ecosystems due to sea level rise. We identified the enhanced seasonality of WT depths (differences between summer and winter WT) and increased evapotranspiration. Both factors extend the period of capillary rise into the topsoil by alterations of hydraulic gradients, especially under the RCP8.5 scenario and during the summer season. Second, the position of the WT, drainage of pores, and successive aeration of the soil during each year have a tightly coupled mechanistic relationship. The soil aeration window, as evident by an increase of pedological indicator days (e.g., days per year that the E_H is > 300 mV at a specific depth) and the above-average dry year of 2018, can be considered as a harbinger of what will be expected to occur frequently in the future if we reach the RCP8.5 scenario.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s10584-023-03498-0>.

Acknowledgements We greatly appreciate the support of collaborators (Karen Kluever, Marek Filipinski and Eckhard Cordsen) from the State Agency for Agriculture, Environment, and Rural Areas Schleswig-Holstein (LLUR), Ms. Ulrike Schwedler for creating the study site map, and Ms. Alexandra Jahn for providing the aerial image.

Funding Open Access funding enabled and organized by Projekt DEAL.

Data availability The datasets generated and/or analyzed during the current study are available from the corresponding author on reasonable request.

Materials availability Not applicable.

Code availability The R script will be made available upon request from the corresponding author.

Declarations

Competing interests The authors declare no competing interests.

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