Losses and destabilization of soil organic carbon stocks incoastal wetlands converted into aquaculture ponds

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Abstract

Coastal-wetlands play a crucial role as carbon (C) reservoirs on Earth due to their C pool composition and functional sink, making them significant for mitigating global climate change. However, due to the development and utilization of wetland resources, many wetlands have been transformed into other land-use types. The current study focuses on the alterations in soil organic-C (SOC) in coastal-wetlands following reclamation into aquaculture ponds. We conducted sampling at 11 different coastal-wetlands along the tropical to temperate regions of the China coast. Each site included two community types, one with solely native species (*Suaeda salsa*, *Phragmites australis* and Mangroves) and the other with an adjacent reclaimed aquaculture pond. Across these 11 locations we compared SOC stock, active OC fractions, and soil physicochemical properties between coastal wetlands and aquaculture ponds. We observed that different soil uses, sampling sites, and their interaction had significant effects on SOC and its stock ($p<0.05$). Reclamation significantly declined SOC concentration at depths of 0-15cm and 15-30cm by 35.5% and 30.3%, respectively, and also decreased SOC stock at 0-15cm and 15-30cm depths by 29.1% and 37.9%, respectively**.** Similar trends were evident for SOC stock, labile organic-C (LOC), dissolved organic-C (DOC) and microbial biomass organic-C (MBC) concentrations ($p<0.05$), indicating soil C-destabilization and losses from soil following conversion. Soils in aquaculture ponds exhibited higher bulk density (BD) (11.3%) and lower levels of salinity (61.0%), soil water content (SWC) (11.7%), total nitrogen (TN) concentration (23.8%) and available-nitrogen concentration (37.7%) ($p<0.05$) than coastal-wetlands. Redundancy-analysis revealed that pH, BD and TN concentration were the key variables most linked with temporal variations in SOC fractions and stock between two land use types. This study provides a theoretical basis for the rational utilization and management of wetland resources, the achievement of an environment-friendly society, and the preservation of multiple service functions within wetland ecosystems.

Keywords: soil organic carbon (SOC), coastal wetland, reclaimed aquaculture pond, active OC fractions

1. Introduction

Coastal wetlands, serving as crucial transitional ecosystems between terrestrial and marine ecosystems (Hao et al., 2024), play a vital role in promoting siltation and tidal prevention, protecting dykes, and providing habitats for saline plants and animals (Osland et al., 2022; Xia et al., 2022). Despite covering only 0.3% of the oceans, they contribute approximately 47% of the total organic carbon burial in marine sediments, with the global rate of organic carbon (C) accumulation estimated to be between 0.07-0.22 Pg C yr^{-1} (Hopkinson et al., 2012; Duarte et al., 2013). Consequently, compared to inland wetlands, coastal wetlands exhibit higher rates of C deposition and possess stronger C sequestration capacity (Mao et al., 2015; Yang et al., 2022a), storing more C per unit area (Zhu et al., 2020; Zhang et al., 2021), and offering greater potential for mitigating the greenhouse effect (Hu et al., 2020; Tan et al., 2020; Gong et al., 2023).

Due to a combination of factors such as the geographical advantages of wetlands and the delayed recognition of their ecological functions, wetlands have historically been prioritized for development and utilization (Xu et al., 2020; Hong et al., 2021; Tan et al., 2022). Unfortunately, global wetlands have experienced 50% decline within a century, primarily due to their conversion for agriculture and other purposes (Bridgham et al., 2006; Davidson & Finlayson, 2019; Wang et al., 2021). In China, the coastal aquaculture industry has flourished since the 1990s (Yang et al., 2022a). Over the past few decades, growing demand for food and economic benefits has fueled the rapid expansion of coastal aquaculture, leading to intense competition for land use (Yao, 2013; Troell et al., 2016). Consequently, a substantial portion of China's coastal wetlands, estimated to be 15,632.64 km^2 in 2017, accounting for 41.10% of the total area of the low-lying coastal zone (below 17 m), has been widely reclaimed for aquaculture ponds (Duan et al., 2020). This reclamation process involves removing existing vegetation and converting dikes into steep slopes (Yang et al., 2022b), altering the original habitat conditions and transforming natural wetlands into artificial wetlands with increased anthropogenic disturbances (Kashaigili et al., 2008; Hu et al., 2009). These changes inevitably affect the spatial and temporal distribution of runoff, wetland vegetation, and various biogeochemical properties such as soil cation exchange capacity (CEC), pH, water status, bulk density, and nutrient concentration (Tan et al., 2022; Yang et al., 2022a; Zhang et al., 2023).

To date, most research has focused on analyzing soil physicochemical properties and nutrients, including carbon (C), nitrogen (N), and phosphorus (P) concentrations (Bai et al., 2013; Macreadie et al., 2013; Wang et al., 2014; Bu et al., 2015; Wan et al., 2018), as well as carbon emissions (Yang et al., 2017; Yang et al., 2022a), in individual estuarine or coastal wetlands. Additionally, previous studies focusing on broader scales have relied on datasets compiled from published literature with integrated meta-analysis, rather than field measurements (Xiao et al., 2019; Xia et al., 2022). For instance, Tan et al. (2020) conducted a global meta-analysis exploring the effects of various land use and land cover change (LULCC) types, including aquaculture ponds, on CO_2, CH_4, and N_2O emissions from natural coastal wetlands, riparian wetlands, and peatlands. Notably, due to sensitivity to plants and microbial activities, the active OC from the total SOC pool, which includes labile organic carbon (LOC), microbial biomass organic carbon (MBC) and dissolved organic carbon (DOC), is highly susceptible to oxidation and decomposition (Wang et al., 2019; Lin et al., 2022). Therefore, distinguishing the active OC fraction from the total SOC pool is crucial for assessing the impact of coastal wetland reclamation on soil C dynamics. Furthermore, since active OC provides a substrate for microbial activity and is a primary source of CO_2 and CH_4 produced by microbes, proper management of this active OC pool is critical for mitigating global climate change and increasing SOC stock (Wang et al., 2019). However, there is still limited research on exploring SOC pools and how OC stock respond to reclamation for aquaculture across broader geographic and environmental gradients. In this study, we aimed to investigate (i) changes in soil total OC stock and active OC fractions (i.e., LOC, MBC, and DOC), and (ii) the relationships between these changes in SOC pools and other key soil physicochemical traits by comparing 22 pairs of wetland sites with nearby reclaimed ponds across 11 coastal wetland areas in China.

2. Materials and methods

2.1 Study region

We selected 11 locations with a total of 22 paired sampling plots, each consisting of one wetland plot and one nearby reclaimed pond, across 11 independent coastal wetlands spanning tropical to temperate climate zones in China (**Fig.** 1). These locations include the Liaohe River Estuary (S1), Yellow River Estuary (S2), Yancheng (S3), Yangtze River Estuary (S4), Hangzhou Bay (S5), Minjiang River Estuary (S6), Jiulong River Estuary (S7), Zhangjiang River Estuary (S8), Zhenzhu Bay (S9), Zhanjiang (S10), and Dongzhai port (S11). Relevant sampling site information is outlined in Table 1. At each site, we collected samples from two different community types: the native species community, consisting of *Suaeda salsa*, *Phragmites australis*, and Mangroves (*Aegiceras corniculatum*, *Kandelia candel*, *Bruguiera gymnorhiza*), and the adjacent reclaimed aquaculture pond. The native species *Suaeda salsa* is native to S1 and S2; *Phragmites australis* is native to S3~S6; and Mangrove is native to S7~S11. Historical records were used to confirm the native species composition of each site and to identify the ponds from which the native species were converted.

2.2 Experimental design and sampling

We randomly selected five vegetation plot (10 m × 10 m square) in each site and land-use, and then chose one small subplot (1 m × 0.6 m) in the middle of the vegetation plot. Soil samples (0-15 cm and 15-30 cm depth layers) were collected from each of the 11 selected sites at five replicated quadrats from October to November 2018 using a soil drilling sampler with a 9 cm diameter. A total of 220 samples (11 sampling sites × 2 community types × 2 soil layers × 5 replicate quadrats) were collected and immediately stored in sterile bags in ice coolers before transportation to the laboratory. Soil samples were subsequently homogenized, sieved through a 2 mm mesh to remove gravel, organic debris, and plant residues, and then separated into two equal proportions. One proportion was air-dried at room temperature and passed through a 0.25 mm sieve for physical and chemical analysis. The second proportion was stored at 4 °C for SOC fraction analyses.

2.3 Sample analysis

Soil pH was measured with a pH meter (Starter 300, Parsippany, USA), and salinity was measured using a 2265FS EC Meter (Spectrum Technologies Inc., Paxinos, USA). Bulk density (BD) was measured from three additional 5 × 3 cm cores from each site. Soil water content (SWC) was measured using the gravimetric method (Lu, 2000). Soil total N (TN) concentration was determined using an elemental analyzer (Elementar Vario MAX, Elementar Scientific Instruments, Hanau, Germany). Soil total P (TP) concentration was determined after digestion of the soil with $HClO_4$-H_2SO_4 acid (Lu, 2000). Soil available N (AN) concentration was determined by the alkaline hydrolysis diffusion method and extracted using 40 mL of 2.0 mol·L^{-1} KCl. Soil available P (AP) concentration was determined by the diffusion method with the Mehlich III using a San++ continuous flow analyzer (Lu, 2000). The ratios of C:N, C:P, N:P, and AN:AP (in mass basis) in soil were also calculated.

We analyzed the concentration of SOC and labile organic C (LOC) of the air-dried, finely ground soils, and that of microbial biomass organic C (MBC) and dissolved organic C (DOC) of fresh soils. SOC concentration was determined using an elemental analyzer (Elementar Vario MAX, Elementar Scientific Instruments, Hanau, Germany) and, soil LOC concentration was determined using the 333 mM $KMnO_4$ digestion method (Chung et al., 2010; Xu et al., 2011), in which soil slurry (soil to distilled water w/v ratio of 1:5) was shaken at 300 rpm and 25 °C for 30 min, centrifuged at 10,000 rpm for 20 min, and then filtered through a decarbonized 0.45-µm filter. MBC concentration was determined using the fumigation–extraction method (Vance et al., 1987; Fang et al., 2020). Briefly, 10 g fresh soil samples were extracted by 0.5 M K_2SO_4 (1 h shaking). A paired 10 g fresh-soil sample was fumigated with alcohol-free chloroform in a desiccator for 24 h in the dark at 22 °C, and the fumigated soils were extracted by the 0.5 M K_2SO_4 solution (1 h shake). The extracts from non-fumigated and fumigated soils were filtered through glass-fiber filter papers. Soil

DOC concentration was determined using a total organic C analyzer (TOC-V, Shimadzu Scientific Instruments, Kyoto, Japan), following extraction using deionized water (1:5 w/v). The SOC stock at the depths of 0-15 cm or 15-30 cm was estimated following the approach of Mishra et al. (2010):

$$SOC_{stock} = SOC_{concentration} \times BD \times 15 \times 10$$

where SOC_{stock} is the soil organic carbon stock at depths of 0-15 cm or 15-30 cm (t C km^{-2}), $SOC_{concentration}$ is soil organic carbon concentration (g kg^{-1}), BD is the bulk density (g cm^{-3}).

2.4 Statistical analyses

SPSS version 22.0 (IBM, Armonk, NY, USA) statistical analysis software were used to carry out the measured data, and Origin version 2022 (OriginLab Corp., Northampton, MA, USA) software was used for plotting. The data were reported as the mean ± standard error (SE) for five replicates. Before the analysis of variance (ANOVA), all datasets were tested for normality (Shapiro-Wilk's test) and homoscedasticity (Bartlett-test). Unless stated otherwise, the main effects of treatment were compared using the least significant differences at 5%. Two-way analysis of variance (ANOVA) was used to analyze the effects of treatment and site in SOC and active OC fractions, and SOC stock across 11 sampling sites. We choosen to use a two-way ANOVA with sites as fixed factor and non a mixed model with site as random factor to can detect possible interactions between the land use (treatment) and site given the great gradient of sites with different native vegetation. We performed an independent sample *t* tests to test the differences in other soil studied variables between coastal wetlands and aquaculture ponds using $p < 0.05$ as the cutoff for statistical significance. The associations between SOC fractions, stock and soil physicochemical properties between the coastal wetlands and aquaculture ponds were tested using Pearson's correlation coefficients in the "corrplot" R packages (version 4.0.3; R Core Team, 2020). Redundancy analysis (RDA) was used to analyze the potential physicochemical factors affecting SOC fractions and stock between the coastal wetlands and aquaculture ponds.

3. Results

3.1 *Soil physicochemical variables*

Significant variations were observed in mean soil BD (Fig. 2a), salinity (Fig. 2c), SWC (Fig. 2d), TN (Fig. 2e), AN (Fig. 2f), C:N (Fig. 2i), C:P (Fig. 2j), N:P (Fig. 2k) and AN:AP (Fig. 2l) between coastal wetlands and aquaculture ponds, while no significant variations were found in mean soil pH (Fig. 2b), TP (Fig. 2g) and AP (Fig. 2h) at 0-30 cm depth (Table S1). Compared to the coastal wetlands, soil BD and pH were higher in the aquaculture ponds by 11.3% ($p < 0.05$) and 3.28% ($p > 0.05$), respectively. Additionally, significant decreases in soil salinity, SWC, TN and AN were observed in the aquaculture ponds, by 61.01%, 11.7%, 23.8% and 37.7%, respectively ($p < 0.05$). Similar trends were evident for C:N, C:P, N:P and AN:AP levels after coastal wetlands reclamation ($p < 0.05$).

3.2 Variation in SOC concentration across coastal wetlands and aquaculture ponds

There were significant effects of treatment and site on SOC concentration at 0-15 cm layer ($p < 0.001$) and 15-30 cm layer ($p < 0.001$), and a treatment × site interaction for SOC concentration at 0-15 cm layer ($p < 0.01$) and 15-30 cm layer ($p < 0.05$) (Fig. 3a). Across 11 sampling sites, SOC concentration ranged from 6.40-31.8 g kg^{-1} at the 0-15 cm depth and 4.90-26.5 g kg^{-1} at the 15-30 cm depth coastal wetlands. In aquaculture ponds, SOC concentration ranged from 5.14-17.59 g kg^{-1} at the 0-15 cm depth and 3.55-18.1 g kg^{-1} at the 15-30 cm depth. In general, reclamation of coastal wetlands into aquaculture ponds significantly declined SOC concentration at 0-15 cm and 15-30 cm depth by 35.5% and 30.3%, respectively ($p < 0.05$). Across 11 sampling sites, SOC concentration of coastal wetlands was the highest in S10 (29.1 ± 1.57 g kg^{-1}) and the lowest in S4 (5.77 ± 0.32 g kg^{-1}), while SOC concentration of aquaculture ponds was the highest in S8 (17.7 ± 0.49 g kg^{-1}) and the lowest in S1 (4.34 ± 0.38 g kg^{-1}) (Fig. 3b).

3.3 Variation in SOC stock across coastal wetlands and aquaculture ponds

Overall, there were main effects of treatment and site on SOC stock at the 0-15 cm layer ($p < 0.001$) and the 15-30 cm layer ($p < 0.001$). Moreover, there was a treatment × site interaction for SOC stock at the 0-15 cm layer ($p < 0.01$) and the 15-30 cm layer ($p < 0.05$) (Fig. 4). Across 11 sampling sites, SOC stock at the 0-15 cm depth ranged from 1230.50 to 3357.07 t C km^{-2} (mean = 2367 t C km^{-2}) in coastal wetlands and from 10898.98 to 22654.61 t C km-2 (mean = 1754 t C km^{-2}) in aquaculture ponds. At the 15-30 cm depth, SOC stock ranged from 927.95 to 3013.33 t C km^{-2} (mean = 2107 t C km^{-2}) in coastal wetlands and from 8076.87 to 23832.64 t C km^{-2} (mean = 1738 t C km^{-2}) in aquaculture ponds. In summary, the conversion of coastal wetlands to aquaculture ponds significantly decreased the SOC stock by 29.13% at the 0-15 cm depth and by 37.89% at the 15-30 cm depth ($p < 0.05$).

3.4 Variation in active OC fractions concentrations across coastal wetlands and aquaculture ponds

There were significant differences in concentrations of soil LOC (Fig. 5a, Fig. S1), MBC (Fig. 5b, Fig. S2), and DOC (Fig. 5c, Fig. S3) between coastal wetlands and aquaculture ponds. Compared to the coastal wetlands, soil LOC concentration was found to be 33.49% lower in aquaculture ponds ($p < 0.05$). Similarly, a significantly lower (25.2%) soil MBC concentration was observed in aquaculture ponds compared to coastal wetlands ($p < 0.05$). However, soil DOC concentration was 15.8% higher in the aquaculture ponds than in coastal wetlands ($p < 0.05$).

3.5 Environmental drivers of SOC stock and fractions

The SOC concentration had a positive correlation with SOC stock ($r = 0.82$), LOC ($r = 0.84$), SWC ($r = 0.68$), C:N ($r = 0.67$) and C:P ($r = 0.61$) ($p < 0.01$), and a negative correlation with pH ($r = –0.57$), BD ($r = –0.73$) and DOC ($r = –0.39$) ($p < 0.01$) (Fig. 6). A negative correlation between soil MBC concentration and pH ($r = 0.17$, $p < 0.05$) was also evident. Moreover, DOC concentration was negatively correlated with AP ($r = –0.25$) and N:P ($r = –0.18$) ($p < 0.05$).

Correlations between environmental variables and SOC fractions, stock were also examined using redundancy analysis (RDA) (Fig. 7). Based on the results of RDA, the first two RDA axes explained 56.5% of the variance in total cumulative species. Notably, pH (explaining 51.7% of variations, $p < 0.01$), BD (15.1%, $p < 0.01$) and TN (11.4%, $p < 0.01$) were the variables that explained most of the temporal variations in SOC fractions concentration and stock. In addition, variances were significantly explained by C:N (10.6%, $p < 0.01$), AN:AP (2.8, $p < 0.01$), salinity (2.2%, $p < 0.01$), N:P (2%, $p < 0.01$), SWC (1.4%, $p < 0.01$), AN (1.1%, $p < 0.05$), and C:P (0.9%, $p < 0.05$).

3.6 *Collectively changes in SOC pools and driving factors*

After reclamation, the SOC concentration and stock showed a decreasing trend (Fig. 8). Additionally, compared to the coastal wetlands, soil LOC and MBC decreased, while DOC increased in the aquaculture ponds. Notably, soil pH and BD were identified as key factors controlling SOC stock and fractions. Higher soil pH and BD contributed to a decrease in SOC concentration, whereas soil DOC concentration tended to increase.

4. Discussion

4.1 Effects of land-use change on soil physical and chemical properties

Changes in land use alter the soil microenvironment, inevitably affecting soil physical and chemical properties such as, soil water content, bulk density, electrical conductivity, and pH (Spohn et al., 2013; Tan et al., 2022). Previous studies have found that unsuitable utilization, irrational scale of development in economical use, and unscientific management methods can lead to soil fertility decline, land degradation, and soil erosion (Tan et al., 2022).

We found that the bulk density of the soil both above and below the cultivated wetland pond increased significantly ($p < 0.05$). Natural vegetation, that is very dense in these natural wetlands, maintains soil pore and soil aggregate formation, improving soil texture and maintaining a low bulk density (Li et al., 2023a). Thus, its removal and change into an aquaculture pond were affected by human disturbance, management measures, and cover removal. In these new conditions, the mechanical structure of the soil was partially disrupted, resulting in an increase in bulk density. Following wetland reclamation for aquaculture ponds, there was a variable decrease in soil salinity. This decrease can be attributed to the higher initial levels of soil salinization in coastal areas, the cultivation of salt-tolerant vegetation, and irregular freshwater recharge during aquaculture operations (Jin et al., 2012; Yang et al., 2015). In general, the conversion of Phragmites australis and mangrove wetlands into aquaculture ponds led to an increase in soil pH. Lin & Lin (2022) revealed that bait input during the aquaculture process could significantly increase soil sulfide, resulting from sulfate (SO_4^{2-}) reduction with high organic loading (Canfield et al., 1993; Lv et al., 2018). This sulfate reduction may be the main reason for the increase in pH in the aquaculture ponds after reclamation. The change from a *Suaeda salsa* natural wetland to a pond did not imply a change in soil pH, thus, the change of soil pH with conversion depends on the type of original natural wetland vegetation type (Zhang et al., 2008). This study observed a consistent decrease in soil pH from the wetlands of the Liaohe River Estuary to Dongzhai Port, following a north-to-south gradient. This decrease corresponds to a decline in soil alkalinity and an increase in acidity, aligning with the typical pattern of changes in soil acidity and alkalinity along latitudinal gradients.

As an important characterization parameter of soil fertility level and property status, soil nutrients can directly or indirectly affect ecological indicators such as SOC, C fractions, and C pool stability. Previous studies have found that diverse factors influence soil nutrient concentrations, including vegetation type, soil depth, climatic conditions, and management practices (Tu et al., 2017). In this study, soil TN concentration of the whole profile (0-30 cm) significantly decreased ($p < 0.05$) after mangrove wetlands reclamation, and the study showed that SOC and TN concentration were high in areas with high biomass (Tian et al., 2010). Mangroves are usually tall and densely covered, grow vigorously, and accumulate abundant biomass in subtropical and tropical climates. However, after being transformed into aquaculture ponds, soil TN was lost due to vegetation removal, human turning, water change, and other activities. The mean soil AN concentration in the upper and lower layers showed that aquaculture ponds had higher AN levels than *Suaeda salsa* wetlands, while *Phragmites australis* and mangrove wetlands had higher AN levels than aquaculture ponds.This was mainly due to the fact that *Suaeda salsa* were shorter and had less biomass than *Phragmites australis* and mangrove, with this low biomass being associated with low soil AN concentration (Tian et al., 2010). Thus, the AN concentration increased in aquaculture ponds when they are replaced by a farming activity due to the presence of artificial baiting, animal residues, and feces during the farming activity (Jobbágy & Jackson, 2001). There was no significant change in soil TP concentration before and after wetlands reclamation ($p < 0.05$), possibly because phosphorus bait feed can artificially supplement soil P in aquaculture ponds.

4.2 Effects of land-use change on soil organic carbon

SOC is of great importance in coastal wetland ecosystems, and small fluctuations in its concentration or composition can profoundly impact wetland cycling of C, potential sequestration of C, and stability of C pools (DeGryze et al., 2004). Land use change, as a direct and influential form of human intervention, has long-term effects on the biogeochemical cycle of soil. It drives changes in physical and chemical properties and nutrient conditions of the soil by altering above-ground vegetation types, affecting the decomposition and transformation processes of SOC and ultimately impacting the stock of wetland SOC (Blair et al., 1995; Chmura et al., 2003).

The results of this study showed a decrease in the average SOC concentration of *Suaeda salsa*, *Phragmites australis*, and mangrove wetlands after reclamation, consistent with previous observations. This decline can be attributed to the decomposition of litter and residues from above-ground vegetation, root rot of underground plants, and secretion discharge, which serve as the primary sources of SOC input in wetland soil (Zhang et al., 2010), and all of which are greatly reduced after wetland reclamation. Moreover, frequent river and tidal effects on natural coastal wetlands can bring a large amount of suspended organic matter, which also contributes to increases in SOC input to natural wetland soil to a certain extent (Li et al., 2023b). Nonetheless, the conversion of these wetlands into aquaculture ponds resulted in a decrease in SOC input due to activities like cover removal and anthropogenic disturbance. Additionally, irregular water cover, aeration, and drainage in aquaculture ponds contribute to SOC loss. The within-site comparative analysis revealed no significant alteration in SOC concentration before and after the reclamation of *Suaeda salsa* wetlands ($p > 0.05$). This lack of change can be attributed to the shorter height, lower biomass, reduced apoptosis, and plant residues of Suaeda salsa compared to *Phragmites australis* and mangrove species, resulting in lower SOC concentration and stock in the soil.

The effect of wetland reclamation on SOC concentration and stock in aquaculture ponds is not only related to the drop of litter and root exudates inputs due to changes in cover condition, but is also closely related to changes of soil physicochemical properties and nutrient conditions associated with the wetland reclamation (Spaccini & Piccolo, 2013). In this study, SOC was positively correlated with SWC ($p < 0.01$). Soil water status will change the turnover process of SOC by affecting life activities such as soil microbial enzyme activity and metabolism, thus leading to changes in SOC concentration and stock (Laiho et al., 2004). When the ambient water is sufficient but allows air penetration, the microbial activity and metabolic rate increase, leading to enhanced decomposition and mineralization of SOC and a subsequent decrease in its concentration and stock. However, as water levels increase, the oxygen supply in soil diminishes, which stresses the growth and reproduction of soil microorganisms. This results in a slower decomposition and transformation rate of SOC, leading to an increase in its concentration and stock (Xia et al., 2022). In addition, soil SOC was significantly negatively correlated with bulk density and pH ($p < 0.01$) (Fig. 8). The RDA results also confirmed that one of the main reasons for the SOC decrease after *Phragmites australis* and mangrove wetlands reclamation was the increase of soil bulk density and pH. Soil bulk weight is the most effective parameter for predicting changes in soil structure and function; generally, the richer the organic matter, the more granular structure, the lower the degree of soil compactness, the greater the porosity, and the smaller the bulk weight. Soil pH stimulates or inhibits microbial growth and reproduction by influencing enzyme activity and community composition of soil microorganisms, thereby accelerating or attenuating the rate of SOC degradation to a certain degree, and altering the SOC concentration and stock (Li et al., 2023a). Moreover, SOC was significantly and positively correlated with TN, AN, and AP concentration ($p < 0.05$). Soil N and P are vital nutrients in wetland ecosystems, and soil microorganisms will stimulate a variety of complex metabolic activities to obtain their optimal C/N/P ratios for growth and reproduction, thus affecting the input and output pathways of SOC concentration and stock (Li et al., 2023b; Zhou et al., 2023). Notably, in this study, there was no consistent pattern in the influence of different climatic zones or geospatial distributions on SOC stock, similar to the findings of Xia et al. (2022). Although climate can regulate the magnitude or rate of SOC transfer from one status to another, the SOC stock is usually controlled by soil physico-chemical properties (Gao et al., 2019; Luo et al., 2021; Xia et al., 2022). In other words, soil variables were completely beyond the control of climate on SOC stock.

4.3 Effects of land-use change on soil active OC fractions

Soil LOC is a very active part of SOC with simple structure, poor stability and fast turnover rate, which can provide energy for the life activities of soil microorganisms in a short time (Zhang et al., 2011). Our findings revealed decreased soil LOC concentration after reclamation of mangrove wetlands, while no significant changes were observed after reclamation of *Suaeda salsa* and *Phragmites australis* wetlands. This can be attributed to variations in the quantity and quality of organic matter introduced into the soil through different vegetation, affecting the population size and functional structure of soil microorganisms to varying degrees (Thorburn et al., 2012). However, the factors influencing soil LOC are not solely responsible, as they represent the combined effects of multiple factors. Additionally, these factors may have promoting or counteracting impacts, resulting in generally insignificant differences in soil LOC concentration after the reclamation of *Suaeda salsa* and *Phragmites australis* wetlands. The correlation analysis results showed that soil LOC was significantly positively correlated with SWC and SOC ($p < 0.01$). Plant residues, particularly lignin fragments, were identified as the primary source of soil LOC and were sensitive to changes in the surrounding environment (Datta et al., 2010). Moreover, SOC and LOC were often in dynamic equilibrium and could be transformed into each other to a certain extent (Hagedorn et al., 2000; Prommer et al., 2020). Therefore, habitats with high SOC concentration and stock typically exhibit higher soil LOC concentration.

Soil MBC plays an essential role in wetland ecosystems' active OC components which can reflect soil microbial activity and characterize soil structure and function (Lin et al., 2022). This study found no significant changes in soil MBC concentration in the whole profile after the reclamation of the *Suaeda salsa* and mangrove wetlands, while there was a significant overall decrease in soil MBC concentration after the reclamation of *Phragmites australis* wetlands. This can be related to the differences in the amount and type of deadfall entering the soil on soil MBC concentration. After reclamation, although the aboveground vegetation was removed, the presence of farm biomass residues, excreta, and artificially added bait fertilizers provided microbial sources to the soil, resulting in the replenishment of the soil MBC concentration (Jangid et al., 2008). This study observed a vertical decrease in average soil MBC concentration with increasing soil layer depth. This can be attributed to the higher accumulation of organic matter, such as decaying plant material and residues, in the upper layer compared to the lower layer. Additionally, influenced by root distribution, the upper layer exhibits greater root aggregation and secretion, providing richer nutrients sources for soil microorganisms to decompose and utilize. Moreover, the upper layer benefits from better hydrothermal conditions and air permeability, creating a more favorable environment for the growth and reproduction of soil microorganisms.

Soil DOC concentration is highly active and readily soluble in water (Ibrahim et al., 2021). It is produced through the decomposition of organic matter by soil microorganisms, and can in turn act on soil microorganisms to provide energy for their life activities (Lin et al., 2012; Han et al., 2012). The results showed that, in general, the average concentration of soil DOC in both the upper and lower soil layers increased after reclamation of the *Suaeda salsa* and *Phragmites australis* wetlands. Moreover, the reclamation of wetlands for aquaculture ponds had a greater impact on the DOC concentration and stock in the lower soil layer than the upper soil layer. Previous studies have found that aboveground vegetation litter, underground root exudates and humus are the primary input sources of soil DOC. In contrast, soil microbial mineralization into CO_2 or leaching with water is the main forms of output. Dead branches, rotten leaves, plant roots, and other substances introduce DOC to natural wetlands. Perennial tides and rivers can partially wash away the soil, resulting in DOC loss. Furthermore, we observed that soil DOC was more sensitive to precipitation and showed a decreasing trend with increasing rainfall along the studied climate gradient, consistent with previous studies (Borken et al., 2008). This was also one of the reasons for the low DOC concentration in mangrove wetland soil in areas with abundant precipitation. However, after reclamation into aquaculture ponds, a relatively closed environment was established, reducing water flow and diminishing erosion. In addition, the sampling time in this study was in the late stage of aquaculture, when farmers would take intermittent drainage measures to facilitate fishing. Meanwhile, it has been shown that structural and qualitative variations in organic matter input to the soil can also lead to large differences in the chemical composition and properties of DOC (Kalbitz et al., 2007). The chemical composition of DOC greatly influences its adsorption and retention capacity within soils. In general, substances with larger volume, larger specific surface area, and lower mobility are more prone to being trapped by soils and binding to mineral surface adsorption sites (Gu et al., 1994). Correlation analysis showed that soil DOC concentration was positively correlated with pH and BD ($p < 0.05$). Soil pH primarily influences the composition of microbial communities and enzyme activity within soils. It can promote or inhibit the metabolic activities of different bacteria, subsequently affecting soil organic matter degradation rates and transformation processes of soil DOC. As the soil BD increases, the soil becomes more compact and more likely to trap DOC, making it less susceptible to loss and leading to an increase in concentration (Wang et al., 2017). In summary the reduction in SOC stock following the conversion of coastal wetland to aquaculture ponds can be attributed to several factors. Among them to:

-Vegetation removal, which disrupts the natural carbon sequestration processes and reduces the input of organic matter to soil.

-The construction of aquaculture ponds involves significant soil disturbance, including the alteration of soil structure and the removal of the topsoil layer, which contains a high concentration of organic carbon,, and morover the change in soil physicochemical properties such as: soilpH, bulk density, wàter status and bulk density.

-Anthropogenic disturbances such as the addition of feed and chemicals, that futher disrupt soi microbial activity, affecting carbon cycling processes

-Hydrological Changes: The conversion alters the natural hydrology of wetlands, affecting the water table and soil moisture content, which are critical factors for maintaining high SOC levels in wetland soils.

4.4 Implications for wetland C management

Coastal wetlands provide a wide range of economic and ecosystem services and play a crucial role in carbon sequestration (Macreadie et al., 2013; Hao et al., 2024). Our study clearly shows the significant loss of carbon caused by the conversion of coastal wetlands to aquaculture ponds, thereby affecting the carbon-sink capacity of wetlands. Duan et al. (2020) reported, by analyzing Landsat 8 images of 2017, that human-reclaimed ponds in China's coastal area covered a total surface of 15,633 km^2. Given our mean estimation of a loss of 983 t C km^2 in the 0-30 cm of soil by conversion, we can estimate that approximately 15,352t of carbon has been lost in this process. This is equivalent to 0.56 Gt of CO_2. Considering that in 2022 the total CO_2 emitted by human activities globally was 36.8 Gt (International Energy Agency), the wetland to pond conversion in China over the last decades is equivalent to 1.52% of current annual human CO_2 emissions.

There is an urgent need for specific initiatives and actions that encourage the sustainable management and conservation of coastal wetlands to maintain the stability of soil carbon pools and long-term carbon sequestration potential, and to minimize the continuation of reclamation for agricultural needs and economic benefits (Bu et al., 2015). Additionally, the targets of the Kunming-Montreal Global Biodiversity Framework (KM-GBF) call for the reduction of losses in areas of high biodiversity importance to near zero by 2030 (Joly, 2022). Therefore, there is an urgent need to find a balance between economic development and ecological protection while safeguarding the livelihoods of coastal residents. Encouraging the sustainable management and protection of coastal wetlands is critical (Bu et al., 2015; Fu et al., 2024). Assessing the restoration potential of these lands requires a better understanding of current land use, the objectives of the communities that own and manage the land, and acceptable alternative livelihood options.

5. Conclusions

Wetland conversion, sampling sites, and their interaction significantly affected SOC concentration and its stock. Specifically, the reclamation of coastal wetlands into aquaculture ponds resulted in a significant mean decrease in SOC concentration at depths of 0-15 cm and 15-30 cm by 35.5% and 30.3%, respectively ($p < 0.05$), as well as a decrease in SOC stock, LOC, and MBC concentrations ($p < 0.05$). In contrast, soil DOC concentration was found to be 15.8% higher in aquaculture ponds compared to coastal wetlands ($p < 0.05$). Furthermore, soils in aquaculture ponds had significantly higher bulk density (11.3%) and lower salinity (61.0%), SWC (11.7%), total nitrogen concentration (23.8%), and available nitrogen concentration (37.7%) compared to coastal wetlands ($p < 0.05$). Redundancy analysis indicated that pH, bulk density, and total nitrogen concentration explained most of the temporal variations in SOC fractions and stock between two community types. Our results clearly showed that the conversion from wetland to ponds destabilizes SOC and, by reducing organic matter inputs, strongly decreases soil C stock. But these effects significantly depends on the natural vegetation removed during wetland conversion. The extent of SOC loss is closly related to the biomass of natural vegetation; the greater the biomass removed, the more substantial the SOC loss after conversion.

Declaration of competing interest The authors declare no competing interests.

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Table 1. Sampling sites description. The reclamation years of aquaculture ponds at each sampling site were obtained by remote sensing image interpretation, literature review and field investigation. Soil samples were collected at depths of 0-15 cm and 15-30 cm.

Sampling sites abbreviation	Location of sampling sites	Geographic location	Wetland types before reclamation	Years of breeding ponds
Liaohe River Estuary (S1)	Lujia Town, Shuangtaizi District, Panjin City, Liaoning Province	N40°41'~41°27' E121°30'~122°30'	*Suaeda salsa*	Around 40 years
Yellow River Estuary (S2)	Kenli District, Dongying City, Shandong Province	N37°40′~38°10′ E118°41′~119°16′	*Suaeda salsa*	Around 15 years
Yancheng (S3)	Sheyang County, Yancheng City, Jiangsu Province	N32°48′~34°29′ E119°53′~121°18′	*Phragmites australis*	Around 20 years
Yangtze River Estuary (S4)	Chongming Island, Chongming District, Shanghai City	N31°25′~31°38′ E121°50′~122°5′	*Phragmites australis*	Around 30 years
Hangzhou Bay (S5)	Cixi City, Zhejiang Province	N30°17'~30°20' E121°5'~121°10'	*Phragmites australis*	Around 30 years
Minjiang River Estuary (S6)	Tantou Town, Changle District, Fuzhou City, Fujian Province	N26°00'~26°03' E119°34'~119°40'	*Phragmites australis*	Around 30 years
Jiulong River Estuary (S7)	Fugong Town, Longhai District, Zhangjiang City, Fujian Province	N24°23′~24°27′ E117°54′~117°56′	Mangroves (*Aegiceras corniculatum*)	Around 20 years
Zhangjiang River Estuary (S8)	Dongxia Town, Yunxiao County, Zhangjiang City, Fujian Province	N24°23′33″~24°27′ E117°23′~117°32′	Mangroves (*Kandelia candel*)	Around 40 years
Zhenzhu Bay (S9)	Fangcheng District, Fangchenggang City, Guangxi Province	N21°31'~21°37' E108°00'~108°16'	Mangroves (*Aegiceras corniculatum*)	Around 20 years
Zhanjiang (S10)	Gaoqiao Town, Lianjiang City, Guangdong Province	N21°9'~21°34' E109°44'~109°56'	Mangroves (*Bruguiera	Around 20 years

			gymnorhiza)	
Dongzhai port (S11)	Yanfeng Town, Haikou City, Hainan Province	N19°51′~20°1′ E110°32′~110°37′	Mangroves (*Bruguiera gymnorhiza*)	Around 20 years

Figure captions

Figure 1. Sampling sites and plant communities of tropical to temperate regions of the China coast were sampled at 11 dependent wetland types and 11 sampling sites, including Liaohe River Estuary (S1), Yellow River Estuary (S2), Yancheng (S3), Yangtze River Estuary (S4), Hangzhou Bay (S5), Minjiang River Estuary (S6), Jiulong River Estuary (S7), Zhangjiang River Estuary (S8), Zhenzhu Bay (S9), Zhanjiang (S10), and Dongzhai Port (S11), with a total of 220 samples. Each site had two community types, one with only native species (*Suaeda salsa*, *Phragmites australis*, and Mangroves) and the other with reclaimed aquaculture ponds. Soil samples were collected at depths of 0-15 cm and 15-30 cm.

Figure 2. Physicochemical characterization at 0-30 cm depth for the coastal wetlands and aquaculture ponds across the 11 sampling sites. Properties measured include soil bulk density (BD; a), pH (b), salinity (c), soil water content (SWC; d), total nitrogen (TN; e), available nitrogen (AN; f), total phosphorus (TP; g), available phosphorus (AP; h), C:N ratio (i), C:P ratio (j), N:P ratio (k), and AN:AP ratio (l). Data represent means and standard errors (n=110). Different letters indicate significant differences between coastal wetlands and aquaculture ponds ($p < 0.05$).

Figure 3. (a) Variations at 0-15 cm and 15-30 cm depth of SOC concentration after converting wetlands into aquaculture ponds across 11 sampling sites, shown in the left-side histogram, and corresponding boxplots on the right. Two-way ANOVA of the effects of land use (treatment) and site on SOC concentration are shown in the upper left corner of figure. (b) Distribution at 0-30 cm depth of SOC after coastal wetlands reclamation into aquaculture ponds across 11 sampling sites. Data represent means and standard errors (n=55). Different letters indicate significant differences between coastal wetlands and aquaculture ponds ($p < 0.05$).

Figure 4. Variations at 0-15 cm and 15-30 cm depth of SOC stock after converting wetlands into aquaculture ponds across the 11 sampling sites, shown in the left-side histogram, and corresponding boxplots on the right. Two-way ANOVA of the effects of land use (treatment) and site on SOC stock are shown in the upper left corner of figure. Data represent means and standard errors (n=55). Different letters indicate significant differences between coastal wetlands and aquaculture ponds ($p < 0.05$).

Figure 5. Boxplots of soil labile organic carbon (LOC; a), microbial biomass organic carbon (MBC; b), dissolved organic carbon (DOC; c) at 0-30 cm depth across coastal wetlands and aquaculture ponds. Data represent means and standard errors (n=110). Different letters indicate significant differences between coastal wetlands and aquaculture ponds ($p < 0.05$).

Figure 6. Pearson correlation analysis between SOC fractions and stock, and environmental variables at 0-30 cm depth across coastal wetlands and aquaculture ponds (n=110). SOC, soil organic carbon; LOC, labile organic carbon; MBC, microbial biomass organic carbon; DOC, dissolved organic carbon; BD, bulk density; SWC, soil water content; TN, total nitrogen; AN, available nitrogen; TP, total phosphorus; AP, available phosphorus. * and ** indicate significant correlations at $p < 0.05$ and 0.01, respectively.

Figure 7. Results of redundancy analysis (RDA), based on SOC fractions and stock, and environmental variables, using a Monte Carlo permutation test ($p < 0.05$). When the angles between the response and explanatory variables are acute, it represents a positive correlation between the

two variables; while obtuse angles represent a negative correlation. The pie chart shows the percentage of the variance of SOC fractions and stock explained by the different variables. SOC, soil organic carbon; LOC, labile organic carbon; MBC, microbial biomass organic carbon; DOC, dissolved organic carbon; BD, bulk density; SWC, soil water content; TN, total nitrogen; AN, available nitrogen; TP, total phosphorus; AP, available phosphorus.

Figure 8. Conceptual diagram illustrating the effects of reclamation into aquaculture ponds on SOC and active OC fractions along the tropical to temperate regions of the China coast. SOC, soil organic carbon; LOC, labile organic carbon; MBC, microbial biomass organic carbon; DOC, dissolved organic carbon; BD, bulk density.