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Impacts of land-use change on carbon dynamics in China's coastal wetlands

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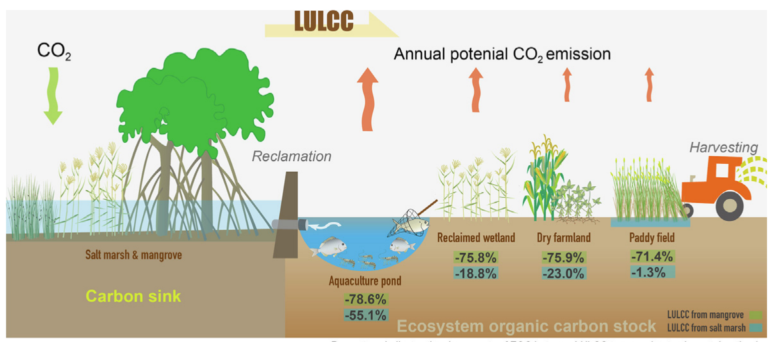
HIGHLIGHTS

- Land-use change in wetlands resulted in organic C loss, while increased inorganic C.
- Potential CO_2 emission caused by organic C loss was dependent on land use types.
- Change rate of ecosystem organic C was greater in mangroves than in salt marshes.
- Loss rates of C showed a significant increasing trend with decreasing latitude.
- Environmental variables that may drive the response of ecosystem C were identified.

GRAPHICAL ABSTRACT



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ABSTRACT

The impact of land-use and land-cover change (LULCC) on ecosystem carbon (C) dynamics has been previously documented at local and global scales, but uncertainty persists for coastal wetlands due to geographical variability and field data limitations. Field-based assessments of plant and soil C contents and stocks of various LULCC types were conducted in nine regions along the coastline of China (21°–40°N). These regions cover natural coastal wetlands (NWs, including salt marshes and mangroves) and former wetlands converted to different LULCC types, including reclaimed wetlands (RWs), dry farmlands (DFs), paddy fields (PFs) and aquaculture ponds (APs). The results showed that LULCC generally decreased the C contents and stocks of the plant–soil system by 29.6 % ± 2.5 % and 40.4 % ± 9.2 %, respectively, while it slightly increased the soil inorganic C contents and stocks. Wetlands converted to APs and RWs lost greater ecosystem organic C stocks (EOC, sum of plants and top 30 cm of soil organic C stocks) than other LULCC types. The annual potential CO_2 emissions estimated from EOC loss depended on the LULCC type, with an average emission of 7.92 ± 2.94 Mg CO_2-eq ha^{-1} yr^{-1}. The change rate of EOC in all LULCC types showed a significantly deceasing trend with increasing latitude ($p < 0.05$). The loss of EOC due to LULCC was larger in mangroves than in salt marshes. The results showed that the response of plant and soil C variables to LULCC was mainly related to differences in plant biomass, median grain size, soil water content and soil NH_4^+-N content. This study emphasized the importance of LULCC in triggering C loss in natural coastal wetlands, which strengthens the greenhouse effect. We suggest that the current land-based climate models and climate mitigation policies must account for specific land-use types and their associated land management practices to achieve more effective emission reduction.

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Abbreviations

Land-use types

NW	Natural coastal wetland
RW	Reclaimed wetland
DF	Dry farmland
PF	Paddy field
AP	Aquaculture pond

Carbon variables

STC	Soil total carbon
SOC	Soil organic carbon
SIC	Soil inorganic carbon
PBC	Plant biomass carbon
W_{XXX}	Content of carbon components in soil (STC/SOC/SIC)
S_{XXX}	Stock of carbon components in soil (STC/SOC/SIC)
EOC	Ecosystem organic carbon stock

Environmental variables

PB	Plant biomass
MGS	Soil median grain size
BD	Soil bulk density
SWC	Soil water content
EC	Soil electrical conductivity
NH_4^+-N	Soil ammoniacal nitrogen
NO_3^--N	Soil nitric nitrogen
TP	Soil total phosphorus content

1. Introduction

Large stocks of carbon (C) have been stored in global coastal wetlands such as mangroves, salt marshes and seagrass beds, accounting for approximately 50 % of the annual C burial in the ocean (Mcleod et al., 2011; Duarte et al., 2013). High C accumulation in coastal wetlands can be attributed to high levels of plant productivity and burial rates of allochthonous soil and preservation of organic matter in water-saturated and anaerobic soils (Mcleod et al., 2011; Neubauer and Megonigal, 2021). However, many coastal wetlands have been intensively developed by land-use and land-cover change (LULCC) for more than a century owing to the rising population and subsequent economic growth (Hong et al., 2021; Tan et al., 2022a, 2022b), with an estimated global annual wetland loss rate of 0.2 %–5.0 % (Davidson and Finlayson, 2019). Taking China as an example, intensive anthropogenic LULCC since the 1950s has resulted in the conversion of >50 % of natural coastal wetlands to agricultural lands and municipal construction (Chen et al., 2017; Don et al., 2011; Meng et al., 2017).

Conservation of natural wetlands is of growing global interest because it contributes to mitigating greenhouse gas emissions and associated climate warming (Hiraishi et al., 2014; IPCC, 2003; Kauffman et al., 2018; Tan et al., 2020). Many studies have demonstrated that LULCC in coastal wetlands may lead to notable CO_2 emissions into the atmosphere due to decomposition of organic C previously stored in the plant biomass and soils of coastal wetlands (Gong et al., 2023; Lovelock et al., 2017; Tan et al., 2020, 2021). However, previous studies have shown inconsistencies in the magnitudes of ecosystem C dynamics resulting from different LULCC types in coastal wetlands (e.g., various crop lands, aquafarms, and constructed wetlands). This is probably attributed to the impacts of varying field management practices of soil drainage, tillage, fertilization, biomass harvesting, animal feeding and dredging (Sasmito et al., 2018; Tan et al., 2020; Wang et al., 2022). Therefore, it is necessary to improve the assessments of the impact of LULCC types on plant and soil C dynamics and potential greenhouse gas emissions against those of natural wetlands.

Land management practices associated with different LULCC types can alter ecosystem C dynamics through shifts in vegetation cover and composition (Kauffman et al., 2018; Lovelock et al., 2017; Sasmito et al., 2020), nutrient supply (Chen et al., 2020; Wang et al., 2014), soil physicochemical properties, and microbial activities (Wallenius et al., 2011; Wang et al., 2014; Lovelock et al., 2017; Zhang et al., 2019; Zhang et al., 2021). For instance, when coastal wetlands are converted into agricultural lands, soil drainage and tillage reduce the soil water content and increase soil aeration, thus stimulating the activity of aerobic microbes and decomposition of soil C (Ruis et al., 2022; Wallenius et al., 2011). Nitrogen and carbonate fertilization and irrigation in farmlands increase soil inorganic carbon and nitrogen contents and thus enhance C turnover (Wang et al., 2014; Bughio et al., 2016). Additionally, harvesting practices or elimination of weeds in agricultural lands lower the net plant productivity and change the vegetation composition, leading to a frequent loss of plant biomass (Eid et al., 2019; Kauffman et al., 2018; Lovelock et al., 2017; Sasmito et al., 2020).

However, in some studies, it was found that the conversion of coastal wetlands to croplands and aquaculture ponds increased the soil organic carbon (SOC), which was attributed to the inhibition of tidal disturbance and the increase in organic matter due to fertilization and feeding (Wang et al., 2022; Xu et al., 2019; Zhang et al., 2019). On the other hand, soil inorganic carbon (SIC) is a significant component of the total soil C pool, especially in coastal saline-alkaline regions (Goddard et al., 2007; Zhang et al., 2019; Zhu et al., 2019), while the responses of SIC to LULCC are less concerning. Under natural conditions, atmospheric CO_2 can be sequestered in inorganic form in soils by various processes, such as the weathering of silicate minerals (Adams, 1993; Wu et al., 2009). A previous study suggested that the conversion of salt marshes to cropland led to irrigation- and NH_4HCO_3-induced SIC accumulation, which can almost offset and compensate for the loss of SOC (Zhu et al., 2019). Therefore, a full understanding of SOC and SIC dynamics is essential because of the crucial roles they play in the wetland C budget.

Several studies have previously assessed the impact of LULCC in natural wetlands on soil C contents and stocks on local to regional scales (Andreetta et al., 2016; Arifanti et al., 2019; Eid et al., 2019; Gulliver et al., 2020; Zhu et al., 2020). However, large-scale studies on the impact of various LULCC types on plant and soil C contents and stocks of natural coastal wetlands (e.g., salt marshes and mangroves) are still lacking, particularly across continental-scale climatic zones. To address this knowledge gap, we proposed to understand through this study the impact of various LULCC types on ecosystem C dynamics due to the conversion of natural coastal wetlands. Across the coastal regions of China, the variations in plant and soil C contents and stocks following four popular LULCC types from natural salt marshes and mangroves were investigated. We hypothesized that LULCC would lead to the loss of plant and soil C in natural coastal wetlands. Therefore, the main objectives of this study are (1) to quantify the effect of different types of LULCC on plant and soil C contents and stocks; (2) to identify the environmental variables that may drive the response of plant and soil C variables to LULCC; and (3) to test the latitudinal pattern of the response of ecosystem C dynamics to LULCC, reflecting the effects of climatic variations.

2. Materials and methods

2.1. Study area

The study was conducted in nine coastal sites from eight estuarine regions along the Chinese coastline, spanning nearly 20° of latitude and ranging from temperate to tropical climate zones (Fig. 1). The sampling sites included the Liaohe River Estuary (40°53′–40°54′N, 121°49′–121°53′E; salt marshes), Yellow River Estuary (37°39′–37°42′N, 118°54′–119°04′E; salt marshes), Yancheng River Estuary (32°52′–32°53′N, 120°52′–120°56′E; salt marshes), Yangtze River Estuary (31°27′–31°29′N, 121°53′–121°56′

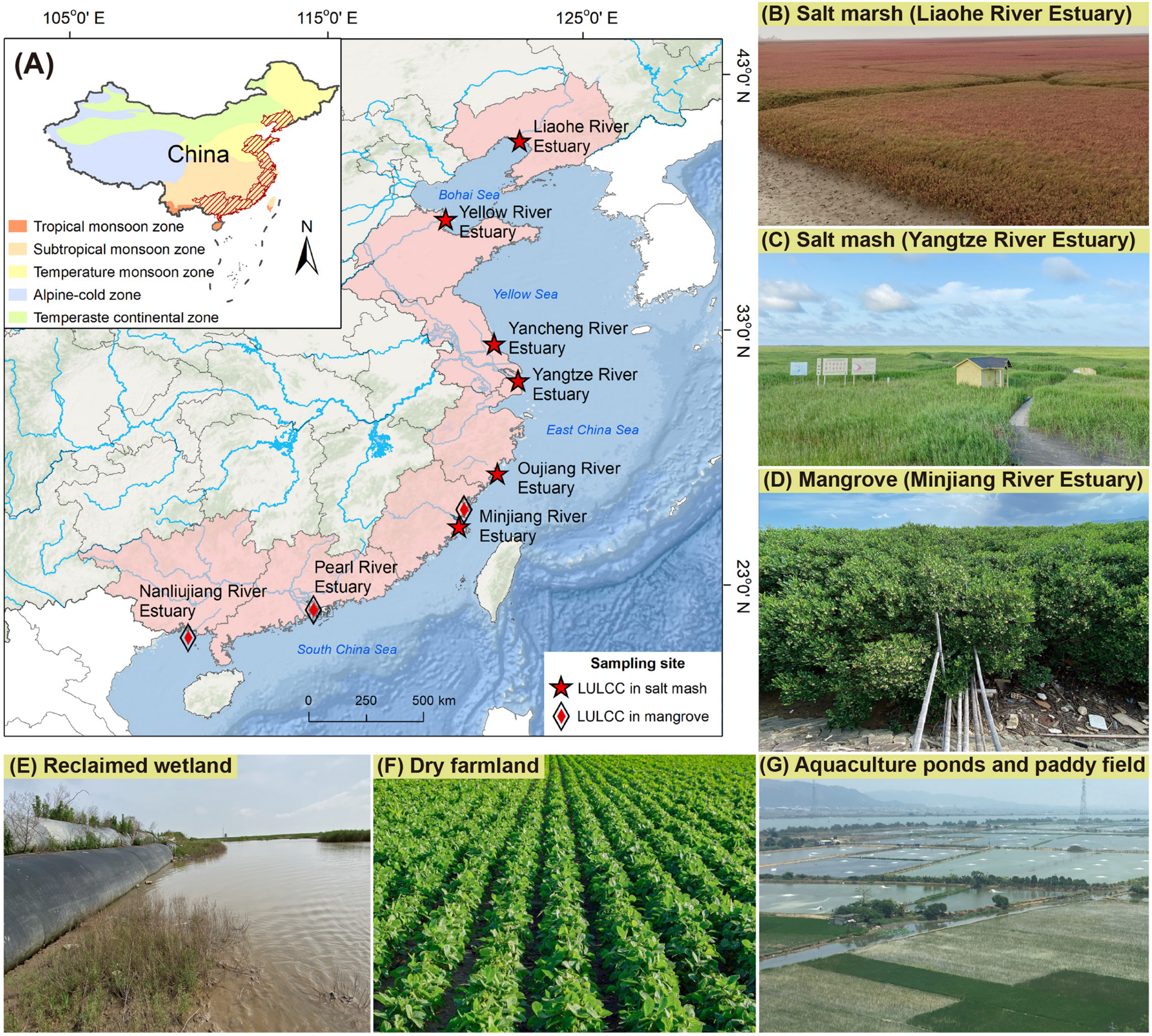

Fig. 1. Location of sampling sites across nine coastal regions of China. Each sampling site contained natural coastal wetlands (including mangroves and salt marshes, B–D) and the following LULCC types: reclaimed wetlands (E), dry farmlands (F), paddy fields (G) and aquaculture ponds (G).

E; salt marshes), Oujiang River Estuary (27°44′–27°44′N, 120°44′–120°45′ E; salt marshes), Minjiang River Estuary (26°01′–26°02′N, 119°37′–119°38′E; including both mangroves and salt marsh sites), Pearl River Estuary (22°37′–22°38′N, 113°34′–113°36′E; mangroves), and Nanliujiang River Estuary (21°36′–21°37′N, 109°02′–119°03′E; mangroves). The climate type, mean annual temperature and mean annual precipitation of each region are listed in Table S1. According to a prestudy of field surveys and satellite image interpretation in natural coastal wetlands (NWs) and adjacent developed lands, four popular LULCC types, namely, reclaimed wetlands (RWs), dry farmlands (DFs), paddy fields (PFs) and aquaculture ponds (APs), were identified (Fig. 1). Specifically, RWs here were the abandoned wetlands that had been enclosed and lightly dried and that were waiting for further development. DFs were relatively aerated land where agricultural plants such as vegetables, cereals, and flowers were cultivated. PFs were used to cultivate rice or Manchurian wild rice by flooding. APs were the earthen ponds used to breed fish, shrimp, and shellfish.

The selected NW sites were grown by the local dominant species community, and the planted (or cultured) species in different LULCC types were widely planted (or cultured) locally. The conversion age (years before present) of the LULCC types was determined through consultation of Google Earth Engine historical images and local residents' oral information on land-use history at each sampling site. The vegetation types (or animal types in the case of aquaculture) and conversion age of each land-use type are listed in Table S2.

2.2. Plant and soil sampling

From July to August (plant growing season) in 2020, plant and soil samplings were conducted at the nine estuarine sites (Fig. 1). During the sampling period, the majority of typical field management practices in each LULCC type were active. At each site, plant biomass and soil samples were collected in NWs and adjacent converted RWs, DFs, PFs and APs. Four replicate quadrats (1 × 1 m for NWs of all salt marsh sites and all RW, DF, PF and AP sites; 10 × 10 m for all NWs of mangrove sites) were randomly selected in well-established vegetation communities. In the salt marsh sites and all RW, DF, PF and AP sites, total plant biomasses (including aboveground biomass and belowground biomass) were harvested from a sub-quadrat (0.5 × 0.5 m). At the mangrove sites, the trunk diameter at breast height, trunk diameter at 30 cm above the highest prop root of the *Rhizophora* species, trunk diameter at the lowest living branch, tree height, and total number of trees in the quadrats were measured. Then, the plant biomass of mangrove species was determined according to the allometric equations established by Komiyama et al. (2005). In addition, some plant materials of mangrove organs were collected to determine the C content of the plant biomass.

Previous studies have suggested that soil properties in the top 30 cm of the soil profile are most sensitive to management practices associated with LULCC (Eid et al., 2019; Hennings et al., 2021; Hiraishi et al., 2014). Accordingly, the top 30 cm soil layer was sampled. In each quadrat, five soil cores out of the sub-quadrats for plant biomass harvesting were collected and mixed together at soil depth intervals of 0–10, 10–20, and 20–30 cm using a soil auger (length, 0.5 m; diameter, 0.1 m). A total of 540 soil samples (9 sampling sites × 5 land-use types × 4 replicates × 3 soil layers) and 120 plant samples (6 sampling sites × 5 land-use types × 4 replicates) were collected.

2.3. Plant and soil C content and stock

The plant samples were carefully washed with deionized water and divided into aboveground and belowground parts before being oven-dried at 60 °C to a constant weight to determine the plant aboveground and belowground biomass. The dried samples were ground to a powder and passed through a 0.149 mm sieve for determination of C content (% C) in the aboveground and belowground biomass. Plant biomass C content was measured with a Vario EL III elemental analyzer (Elementar Scientific Instruments, Germany). The stocks (Mg C ha^{-1}) of plant aboveground ($S_{PBC\text{-}AG}$) and belowground biomass C ($S_{PBC\text{-}BG}$) were obtained by multiplying the respective biomass (Mg ha^{-1}) and C contents (% C), and the total plant biomass C stock (S_{PBC}) was obtained by adding $S_{PBC\text{-}AG}$ and $S_{PBC\text{-}BG}$.

The fresh soil samples were divided into two parts after removal of roots and gravel. One part of the soil was air-dried, finely ground (<0.149 mm) and used for determination of the content of soil total C (W_{STC}), organic C (W_{SOC}), and inorganic C (W_{SIC}). To separate soil organic C from total C, diluted HCl (1 M) was added to the samples to remove inorganic C. W_{STC} and W_{SOC} were measured with a Vario EL III elemental analyzer (Elementar Scientific Instruments, Germany). The W_{SIC} was calculated as the difference between W_{STC} and W_{SOC}.

Soil C stocks (i.e., S_{STC}, S_{SOC} and S_{SIC}; Mg C ha^{-1}) were calculated as the product of soil bulk density (g cm^{-3}, see Section 2.4 below) multiplied by the C content (i.e., W_{STC}, W_{SOC} and W_{SIC}) scaled by the soil depth intervals (cm). The ecosystem organic C stock (EOC) of each land-use type was the sum of S_{PBC} and S_{SOC} (top 30 cm of soil).

2.4. Soil physicochemical properties

Soil bulk density (BD) and gravimetric soil water content (SWC) were determined based on the measurements of the wet and dried weight and the given soil volume. Soil salinity (as electrical conductivity, EC), pH and median grain size (MGS) were measured using a salinity meter (Spectrum 2265FS, USA; sediment-to-water ratio 1:2.5 w/v), pH meter (STARTER 300, USA; sediment-to-water ratio 1:2.5 w/v), and laser particle size analyzer (Malvern MS-2000, UK), respectively. Soil NH_4^+-N and NO_3^--N content were extracted with KCl (2 M) and measured using a flow injection analyzer (Skalar San^{++}, Netherlands). Soil total P (TP) content was measured by $HClO_4$-H_2SO_4 digestion followed by ammonium-molybdate colorimetry and measurement using an UV-2450 spectrophotometer (Shimadzu, Japan).

2.5. Potential CO_2 emissions estimated from loss of C stock

Potential C emissions (CO_2 equivalent) in the plant–soil system were estimated according to the organic C stock-change approach as described in the guidelines of the Intergovernmental Panel on Climate Change (IPCC), which assume that the loss of organic C is completely oxidated into CO_2 (Hiraishi et al., 2014; Kauffman et al., 2018). First, the difference in EOC between natural wetlands and converted LULCC types was estimated as follows:

$$\Delta EOC = EOC_{LULCC} - EOC_{NW} \quad (1)$$

where ΔEOC is the change in EOC due to LULCC (Mg C ha^{-1}); EOC_{LULCC} is the EOC (Mg C ha^{-1}) of each LULCC type (i.e., RW, DF, PF and AP); and EOC_{NW} is the EOC (Mg C ha^{-1}) of natural coastal wetlands. Then, the mean annual potential CO_2 equivalent emissions (CO_2e, Mg CO_2-eq ha^{-1} $year^{-1}$) were estimated using the following equation (Hiraishi et al., 2014; Kauffman et al., 2018):

$$CO_2e = \Delta EOC \times 3.67/T \quad (2)$$

where 3.67 is the molecular ratio of CO_2 to C and T is the conversion age (year) of LULCC types. Referring to the IPCC guidelines again (Hiraishi et al., 2014), the 95 % confidence interval was used to quantify the uncertainty of CO_2e estimation via Monte Carlo analysis. Total annual CO_2 emissions (Mg CO_2-eq $year^{-1}$) were then calculated based on the CO_2e and surface area of coastal wetlands lost between 1979 and 2020 (see Table S3).

2.6. Statistical analyses

First, the Kolmogorov–Smirnov and Levene's test showed that all the data groups met the assumptions of normality and homogeneity of variances, respectively. One-way analysis of variance (ANOVA) with Tukey's HSD multiple comparison test was conducted to test the differences in plant and soil C variables (incl. W_{STC}, S_{STC}, W_{SOC}, S_{SOC}, W_{SIC}, S_{SIC}, S_{PBC} and EOC) and environmental variables (incl. PB, MGS, BD, SWC, EC, pH, NH_4^+-N, NO_3^--N and TP) between the natural wetlands and the LULCC types.

The response ratio (RR) and weighted RR (RR_{++}) are commonly used variables to quantify the effects of environmental changes on ecosystems (Hedges et al., 1999). In this study, RR and RR_{++} were used to evaluate the responses of plant–soil C contents or stocks and relevant environmental variables to LULCC. RR is defined as the natural logarithm of the ratio of the C content or stock in the LULCC groups to that in the natural wetland groups. The values of RR_{++} were calculated from the individual RR pairwise comparison between the LULCC and natural wetland groups (Hedges et al., 1999). To explore whether there are latitudinal patterns in the response of C contents and stocks to LULCC, linear regression was used to examine the relationships between the RR of C variables and latitudes.

Redundancy analysis (RDA) was performed to test the relationships between the RR of the plant and soil C variables and the environmental variables to estimate the relative influence of the environmental variables. Moreover, a structural equation model (SEM) was constructed to further examine the direct and indirect effects of environmental variables on ecosystem organic C dynamics. Based on the results of the RDA, the environmental variables were successively entered into the SEM in descending order of relative influence of each environmental variable until the SEM no longer achieved significance. The SEM with different paths among the selected environmental variables was tested, and the final model with the minimum Akaike's information criterion (AIC) was selected. Maximum-likelihood estimation was used to obtain the path coefficients, and the χ^2 goodness-of-fit test, degrees of freedom, Chi-square and AIC values were used to evaluate the model.

One-way ANOVA was performed using the SPSS statistics 23.0 package (SPSS Inc., Chicago, IL, USA). RDA was performed using the CANOCO 5.0 software (Microcomputer Power, Ithaca, NY, USA). The SEM analysis was performed in R V3.5.3 (R Foundation for Statistical Computing, 2013) with lavaan package. In all the statistical tests, a significance level of $p < 0.05$ was used.

3. Results

3.1. Response of plant and soil C dynamics to LULCC

The plant and soil (top 30 cm) C contents and stocks of the natural wetlands and the associated four LULCC types are shown in Fig. 2. Compared to

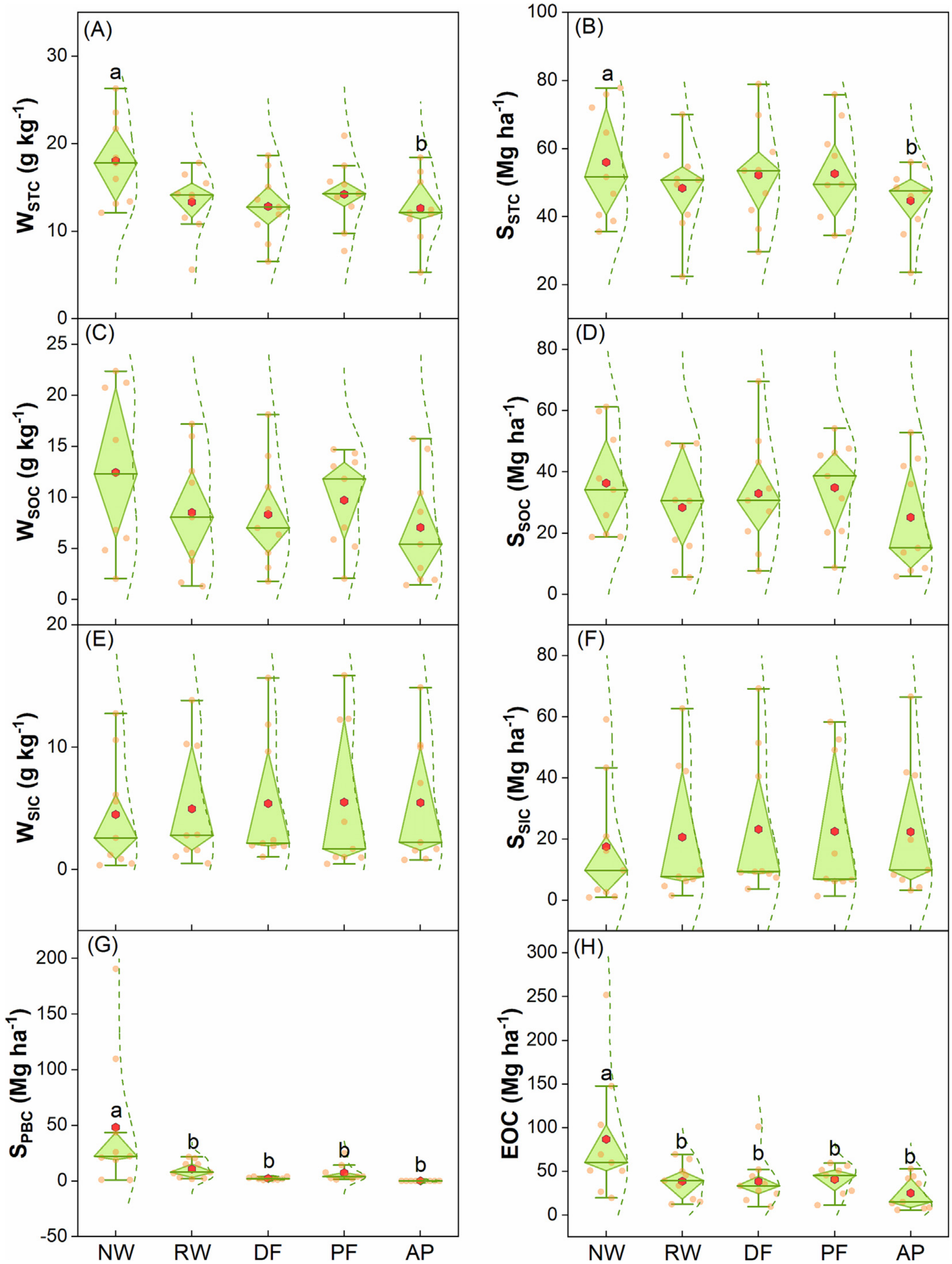

Fig. 2. Variations in C component contents (W_{STC}, W_{SOC} and W_{SIC}) (A, C, E) and stocks (S_{STC}, S_{SOC}, S_{SIC}, S_{PBC} and EOC) (B, D, F, G, H) between natural coastal wetlands (NWs) and converted LULCC types (RW, reclaimed wetland; DF, dry farmland; PF, paddy field; AP, aquaculture pond). The boxes, centerline, and whiskers represent the 25th–75th percentiles, median value, and 5th–95th percentiles, respectively. The red circles represent the area-weighted averages. Different letters indicate significant differences between LULCC types at $p < 0.05$.

natural coastal wetlands (NWs), LULCC significantly ($p < 0.05$) decreased the content of soil total C (W_{STC}), content of soil organic C (W_{SOC}), stock of soil total C (S_{STC}), stock of soil organic C (S_{SOC}), plant biomass C (S_{PBC}) and ecosystem organic C (EOC) by an average of 26.6 % ± 2.0 %, 32.6 % ± 4.4 %, 11.7 % ± 3.3 %, 16.5 % ± 6.1 %, 89.7 % ± 4.9 %, and 58.7 % ± 4.2 %, respectively. However, LULCC slightly ($p > 0.05$) increased the soil inorganic C content (W_{SIC}) by 10.16 % on average and significantly ($p < 0.05$) enhanced the SIC stock (S_{SIC}) by 32.65 % on average across the LULCC types.

The relative responses (regarding RR_{++} values) of plant and soil C variables (i.e., W_{STC}, W_{SOC}, W_{SIC}, S_{STC}, S_{SOC}, S_{SIC}, S_{PBC} and EOC) to LULCC conversion in salt marshes and mangroves are shown in Fig. 3. The change rates of W_{STC} and W_{SOC} (except RW and AQ in mangroves) between the LULCC conversion from salt marshes and mangroves were comparable (Fig. 3A and C). The change rates of S_{STC} and S_{SOC} due to LULCC in salt marshes were slightly higher than those in mangroves (Fig. 3B and D; $p > 0.05$), while the change rates of W_{SIC}, S_{SIC}, S_{PBC} and EOC showed the opposite trend (Fig. 3E–H; $p < 0.05$ for S_{PBC} and EOC).

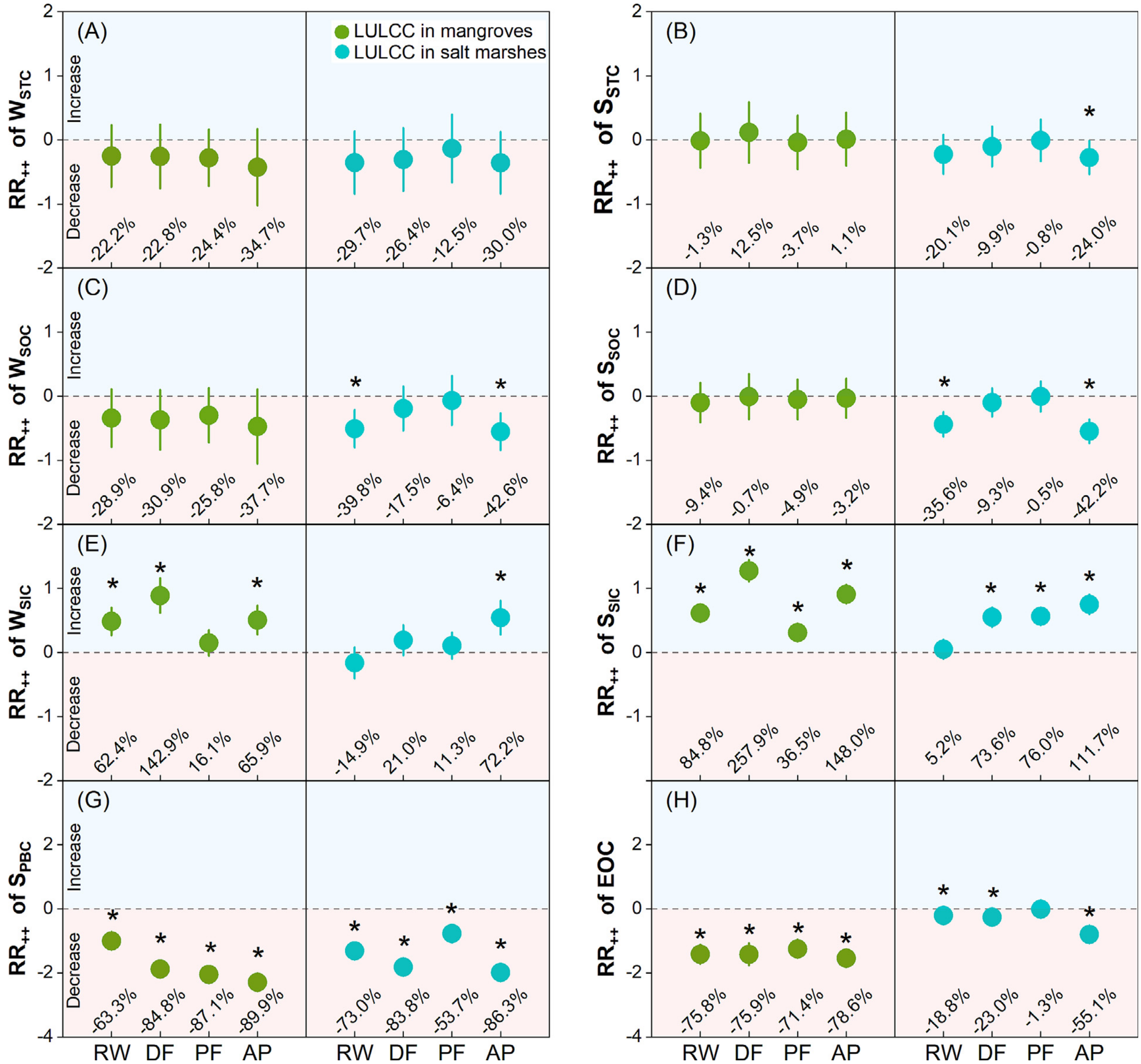

Fig. 3. Weighted response ratio (RR_{++}) of C contents (W_{STC}, W_{SOC} and W_{SIC}) (A, C, E) and stocks (S_{STC}, S_{SOC}, S_{SIC}, S_{PBC} and EOC) (B, D, F, G, H) due to conversion of natural coastal wetlands into LULCC types (RW, reclaimed wetland; DF, dry farmland; PF, paddy field; AP, aquaculture pond). Error bars represent the 95 % confidence interval of RR_{++}. If error bars did not overlap with the zero line (horizontally dashed lines drawn at $RR_{++} = 0$), the response to LULCC was considered significant. The change rates for each variable are shown below the circles and asterisks (*) denote significant changes ($p < 0.05$).

Conversion to aquaculture ponds (APs) from salt marshes resulted in a notable decrease in S_{STC} in salt marshes ($p < 0.05$, Fig. 3B), and conversion to reclaimed wetlands (RWs) and APs significantly decreased both W_{SOC} and S_{SOC} ($p < 0.05$, Fig. 3C and D). Conversions to RWs, dry farmlands (DFs) and APs from mangroves and APs from salt marshes significantly increased W_{SIC} ($p < 0.05$, Fig. 3E). For all LULCC types converted from both salt marshes and mangroves, S_{SIC} (except for RWs from salt marshes) increased significantly ($p < 0.05$, Fig. 3F), while S_{PBC} and EOC (except for paddy fields [PFs] from salt marshes) significantly decreased ($p < 0.05$, Fig. 3G and H).

3.2. Potential CO_2 emissions due to LULCC

Compared to NWs, the conversion to APs had the highest loss of EOC and potential CO_2 emissions following the conversion of both salt marshes and mangroves, followed by the conversion to RWs, PFs and DFs (Table 1). LULCC in mangroves resulted in significantly ($p < 0.05$) higher potential CO_2 emissions, approximately 3.5 times higher than those in salt marshes. Across the salt marshes and mangroves, the annual mean potential CO_2 emissions due to LULCC were 7.92 ± 2.94 Mg CO_2-eq ha^{-1} yr^{-1}. The total potential CO_2 emission due to LULCC of the nine regions was estimated as 123.42 ± 32.91 Tg CO_2-eq during 1979–2020 (see Fig. S1).

3.3. Latitudinal patterns of C variables

Fig. 4 shows the latitudinal variations in the response of plant and soil C contents and stocks to the different LULCC types. The response ratio (RR) of W_{STC} increased with increasing latitude in PFs ($p < 0.05$, Fig. 4A). The RR values of W_{SIC} and S_{SIC} in both RWs and DFs decreased with increasing latitude ($p < 0.05$, Fig. 4E and F). The RR values of S_{PBC} and EOC in RWs, DFs, PFs and APs showed a significantly increasing trend with increasing latitude ($p < 0.05$, Fig. 4G and H). In addition, Fig. S1a shows a significant decrease in potential CO_2 emissions due to LULCC with increasing latitude ($p < 0.05$).

3.4. Effects of environmental variables on C dynamics due to LULCC

Compared to NWs, all the LULCC types significantly ($p < 0.05$) decreased the plant biomass (PB) of salt marshes and mangroves (Fig. 5A).

Table 1
Ecosystem carbon stocks (EOC), loss of ecosystem carbon stocks and annual potential CO_2 emissions in natural coastal wetlands and land-use and land-cover change (LULCC) types. Mean ± S.E.

LULCC types	EOC (Mg ha^{-1})	Loss of EOC (Mg ha^{-1})	Annual potential CO_2 emission (Mg CO_2-eq ha^{-1} yr^{-1})
From salt marsh ecosystems			
Natural coastal wetland	37.58 ± 13.94		
Reclaimed wetland	23.57 ± 9.39	−14.00 ± 13.43	2.50 ± 2.42
Dry farmland	20.58 ± 6.39	−16.99 ± 13.55	3.06 ± 2.69
Paddy field	28.34 ± 9.19	−9.23 ± 14.62	2.19 ± 2.66
Aquaculture pond	11.58 ± 5.54	−26.00 ± 12.79	6.48 ± 2.73
		Average	**4.38 ± 2.02^{b}**
From mangrove ecosystems			
Natural coastal wetland	156.53 ± 52.68		
Reclaimed wetland	53.29 ± 5.59	−103.24 ± 54.71	17.70 ± 12.20
Dry farmland	60.35 ± 21.54	−96.18 ± 73.62	6.04 ± 9.79
Paddy field	50.32 ± 5.44	−106.21 ± 49.93	10.75 ± 4.72
Aquaculture pond	44.38 ± 4.86	−112.15 ± 54.95	25.18 ± 11.26
		Average	**14.92 ± 7.14^{a}**
Overall			
Natural coastal wetland	77.02 ± 24.76		
Reclaimed wetland	34.85 ± 6.94	−42.17 ± 23.11	8.12 ± 4.58^{AB}
Dry farmland	34.57 ± 8.95	−42.45 ± 26.22	4.74 ± 3.39^{B}
Paddy field	37.77 ± 5.66	−39.25 ± 23.62	5.53 ± 2.64^{B}
Aquaculture pond	22.58 ± 6.14	−54.44 ± 22.04	14.16 ± 4.58^{A}
		Average	**7.92 ± 2.94**

Positive values of annual potential CO_2 emissions indicate the loss of plant–soil C into the atmosphere. Different lowercase letters indicate significant differences between salt marshes and mangroves at $p < 0.05$. Different uppercase letters indicate significant differences between LULCC types at $p < 0.05$.

Median grain size (MGS) markedly increased in DFs ($p < 0.05$, Fig. 5B), and soil bulk density (BD) significantly increased in DFs and APs ($p < 0.05$, Fig. 5C) compared to NWs. Soil water content (SWC) and electrical conductivity (EC) significantly decreased in DFs, PFs and APs ($p < 0.05$, Fig. 5D and E), while NO_3^--N content significantly increased in DFs compared to NWs ($p < 0.05$, Fig. 5H). The differences in soil pH, NH_4^+-N and total phosphorus (TP) content between NWs and LULCC types were insignificant. Additionally, the magnitude of changes in environmental factors due to LULCC in mangroves was generally greater than that in salt marshes (Fig. S2).

RDA and the consequent SEM tests were performed to identify the key environmental variables affecting plant and soil C dynamics due to LULCC (Fig. S3 and Fig. 6). When NWs underwent LULCC, the changes in PB, NH_4^+-N, MGS and SWC together contributed >85 % of the relative influence on the C contents and stocks in both salt marshes and mangroves (Fig. S3A and B). PB, NH_4^+-N and SWC were positively correlated with W_{STC}, S_{STC}, W_{SOC}, S_{SOC}, S_{PBC} and EOC. MGS was positively correlated with W_{SIC} and S_{SIC}, while it was negatively correlated with W_{STC}, S_{STC}, W_{SOC}, S_{SOC}, S_{PBC} and EOC.

The SEM explained 83.4 %, 50.8 %, 47.6 % and 72.6 % of the variance in S_{PBC}, S_{SOC}, S_{SIC} and EOC, respectively (Fig. 6A). PB, SWC and EC had a significant positive effect on S_{PBC} ($p < 0.01$), while MGS had a significant negative effect on SPBC ($p < 0.01$). PB, NH_4^+-N and EC had a significant positive effect on S_{SOC} ($p < 0.05$), while the effects of MGS and SWC were insignificant. NH_4^+-N and MGS had a significant positive effect on S_{SIC} ($p < 0.05$), while SWC and EC had a significant negative effect on SSIC ($p < 0.05$). NH_4^+-N and SWC had a significant positive effect on EOC ($p < 0.05$). Furthermore, EOC was significantly positively correlated with S_{PBC} and S_{SOC}. MGS had the greatest total effect (negative) on EOC dynamics, followed by SWC, PB, and NH_4^+-N (positive), and EC had a relatively minor effect (Fig. 6B).

4. Discussion

4.1. Effect of LULCC on wetland C dynamics

High levels of primary productivity in coastal wetlands support a high organic C input into the ecosystem C pool. Meanwhile, the high soil water content and salinity hinder microbial activities and decomposition rates, which leads to a high C sequestration capacity (Alongi, 2014; Mcleod et al., 2011; Chambers et al., 2013). However, the conversion of coastal wetlands into human-disturbed land-use types may seriously compromise the C sequestration capacity and result in C loss (as summarized in Table 2). In this study, natural coastal wetlands (NWs, including salt marshes and mangroves) had a high mean S_{STC} (top 30 cm of soil) and S_{PBC}, with 55.95 ± 5.61 Mg ha^{-1} and 48.21 ± 20.86 Mg ha^{-1}, respectively (Fig. 2). However, the results showed that the conversion of NWs to reclaimed wetlands (RWs), dry farmlands (DFs), paddy fields (PFs), and aquaculture ponds (APs) decreased plant–soil C contents and stocks by 0.5 %–89.9 %, causing significant C loss (Fig. 3). These results validate our hypothesis. Overall, LULCC slightly increased W_{SIC} and S_{SIC} by 10.16 %–32.65 %.

Previous studies have suggested that general field management practices in LULCC types (e.g., soil drainage, tillage, irrigation, fertilization and crop harvesting) could profoundly alter plant and soil variables, further affecting plant–soil C input and decomposition processes (Kauffman et al., 2018; Lovelock et al., 2017; Sasmito et al., 2020; Wallenius et al., 2011; Wang et al., 2014). During the initial establishment or reclamation of NWs into RWs, DFs, PFs and APs, soil drainage dramatically decreased the soil water content (SWC) (Fig. 2). Lower SWC could increase the oxygen concentration in the soil and enhance the aerobic decomposition of SOC and plant litter (Lovelock et al., 2017; Wallenius et al., 2011), resulting in a decrease in W_{SOC} and S_{SOC}. Moreover, the input of particulate matter, including particulate C from marine environments, is restricted after reclamation. Rainfall and field management practices such as tillage and irrigation cause soil loss, especially for small particle-sized soil, and increase the median grain size (MGS) corresponding to LULCC types (O'Rourke et al., 2015; Tan et al., 2022a, 2022b). A coarse-texture soil, on the one hand, generally has a higher porosity and lower capacity to hold water, both enhancing the conditions for the oxidation of soil organic materials (Hassink, 1997; Spohn, 2020; Zhang et al., 2019). On the other hand, a higher sand fraction soil has a lower surface area or reactive minerals to sorb or retain organic matter, hence lowering the W_{STC} and W_{SOC} (Hassink, 1997; Spohn, 2020). The results of SEM analysis showed that MGS had the highest total influence on changes in S_{SOC} and EOC (Fig. 6).

When NWs are converted to RWs, the reclaimed area is enclosed and lightly dried for other land uses (Fig. 5D); thus, RWs are a transitional land-use type that can typically last for several years. Relatively dry environments reduce plant growth and the input of plant litter into the soil

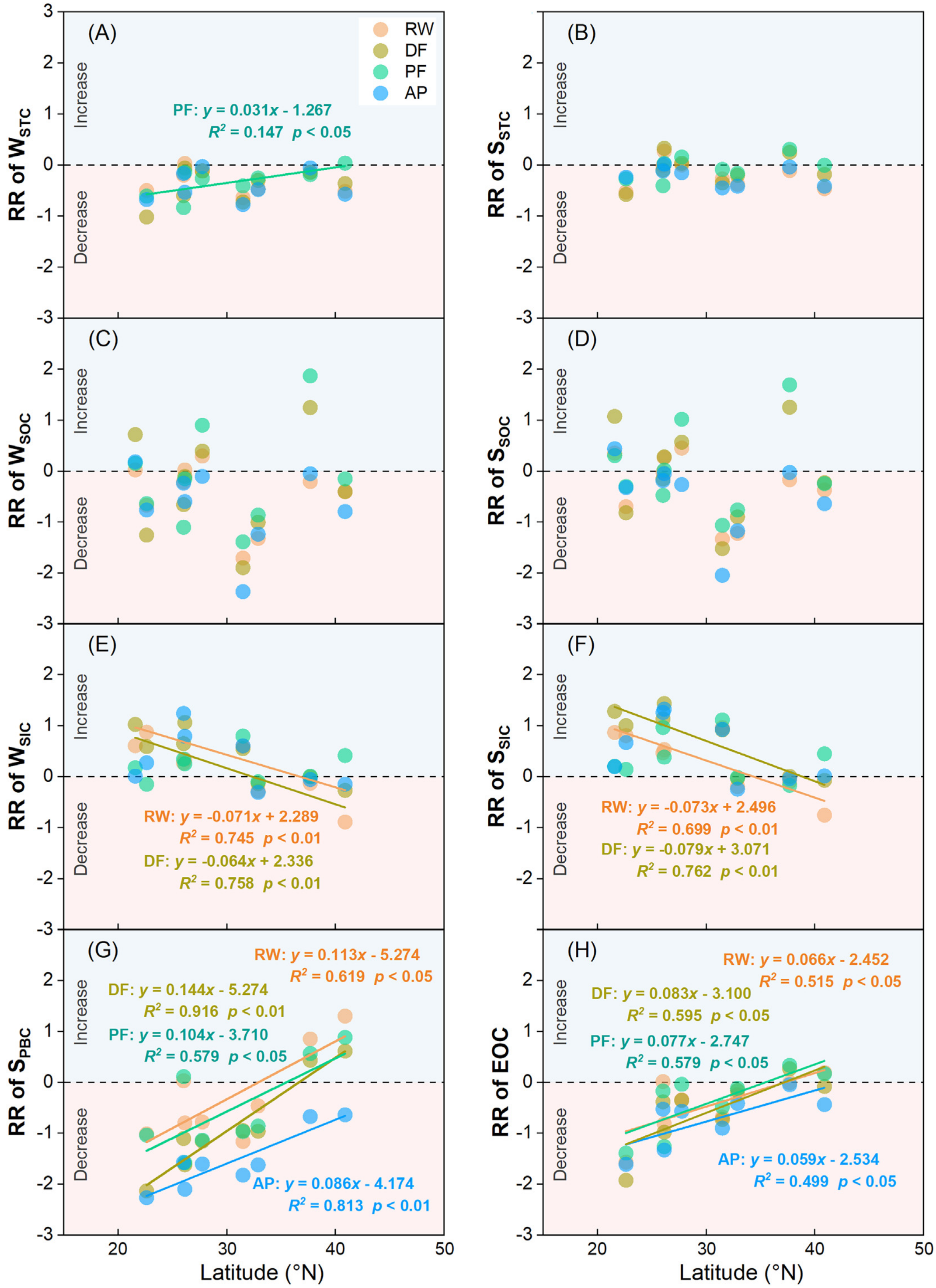

Fig. 4. Latitudinal variations in the response ratio (RR) of C contents (W_{STC}, W_{SOC} and W_{SIC}) (A, C, E) and stocks (S_{STC}, S_{SOC}, S_{SIC}, S_{PBC} and EOC) (B, D, F, G, H) for different LULCC types (indicated by different colors; RW, reclaimed wetland; DF, dry farmland; PF, paddy field; AP, aquaculture pond).

(Musarika et al., 2017; Sasmito et al., 2020), causing a decrease in W_{SOC}, S_{SOC}, S_{PBC} and EOC. Moreover, decreased electrical conductivity (EC) and increased MGS due to reclamation in RWs (Fig. 6) weaken the inhibitory effects of salt stress on microbes and promote aerobic microbial decomposition (e.g., Alongi, 2014; Chambers et al., 2013; Neubauer et al., 2013), leading to a decrease in W_{STC}, S_{STC}, W_{SOC} and S_{SOC} (Fig. 2). The enhanced soil CO_2 concentration produced by decomposition, combined with the presence of water in the soil pore space, could shift the carbonate equilibrium following the equation $CaCO_3 + H_2O + CO_2 \leftrightarrow Ca^{2+} + 2HCO^-$ toward the right, resulting in a higher soluble SIC content (Kindler et al., 2011).

Regarding the conversion to DFs, frequent agricultural activity alters the environmental variables and plant–soil C dynamics. The soil particle size in most of the DFs was sandy, with the highest MGS of all studied LULCC types (Fig. 5). High MGS combined with the effect of desalination and low intensity irrigation in DFs could decrease EC and SWC and promote the decomposition of C, as mentioned above. The use of agricultural machinery could also compact the soil and significantly increase the BD

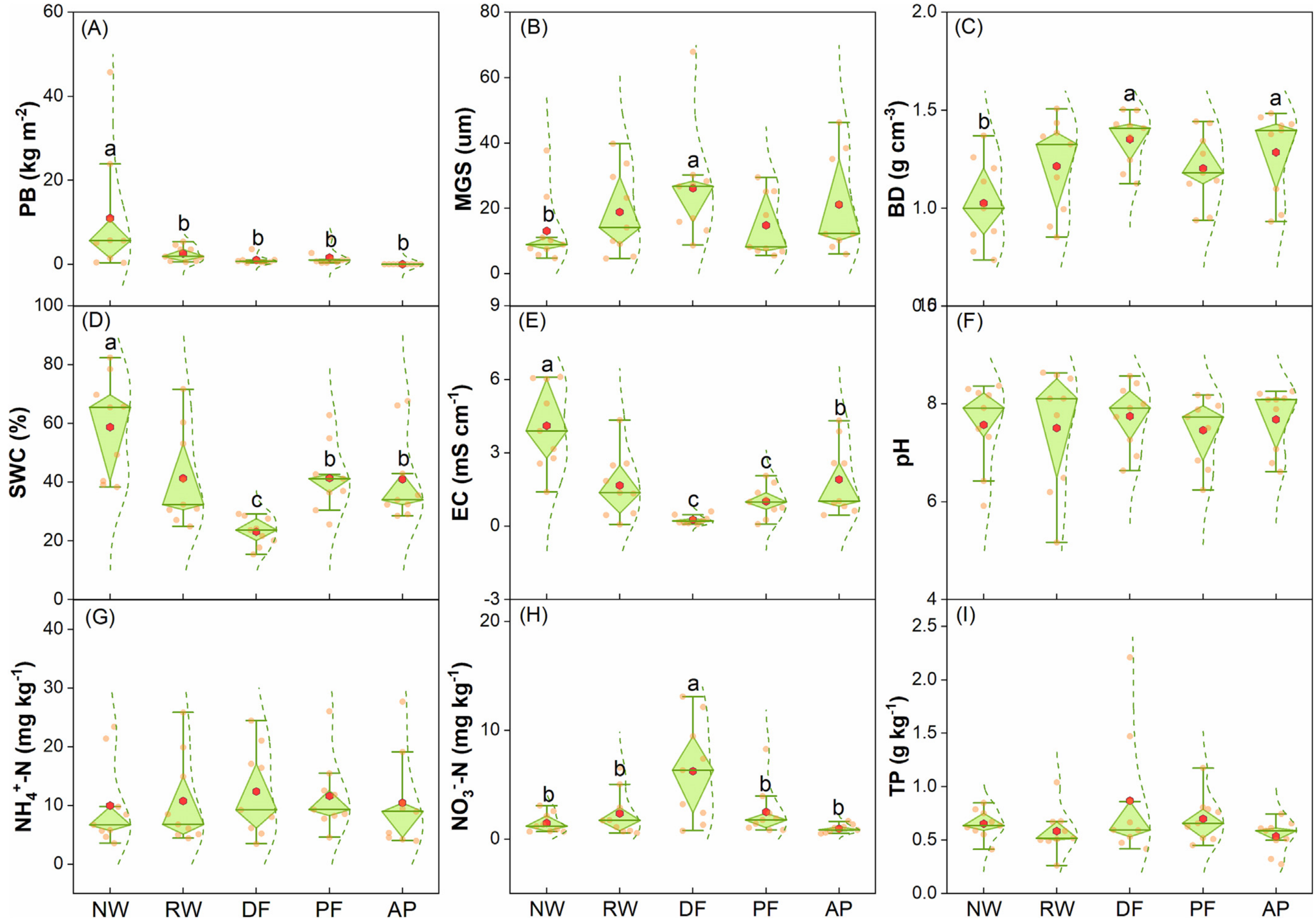

Fig. 5. Variations in plant biomass (PB, *A*), soil median grain size (MGS, *B*), soil bulk density (BD, *C*), soil water content (SWC, *D*), soil electrical conductivity (EC, *E*), soil pH (*F*), soil NH_4^+-N content (*G*), soil NO_3^--N content (*H*) and soil total phosphorus content (*I*) in natural coastal wetlands (NWs) and LULCC types (RW, reclaimed wetland; DF, dry farmland; PF, paddy field; AP, aquaculture pond). The boxes, centerline, and whiskers represent the 25th–75th percentiles, median value, and 5th–95th percentiles, respectively. The red circles represent the area-weighted averages. Different letters indicate significant differences between LULCC types at p < 0.05.

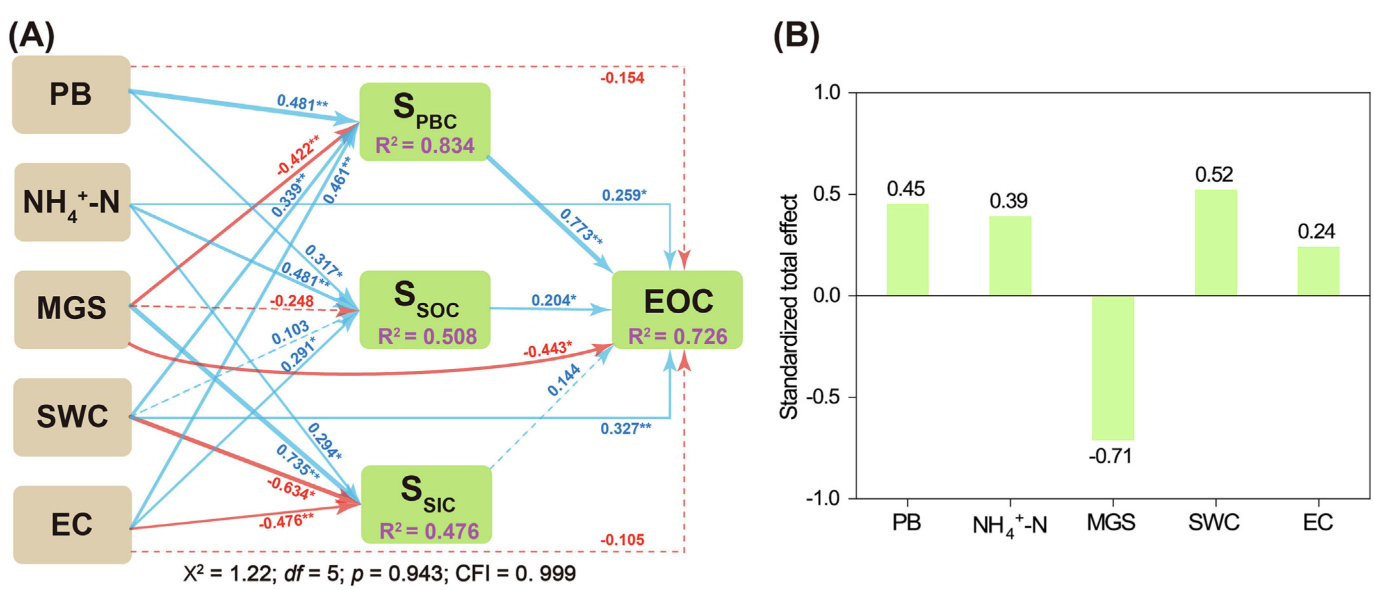

Fig. 6. Structure equation model (SEM) of the relationship between ecosystem organic carbon stocks and environmental factors (A) and the total effect of environmental factors (B) when natural coastal wetlands underwent LULCC (to RW, reclaimed wetland; DF, dry farmland; PF, paddy field; AP, aquaculture pond). One asterisk (*) indicates p < 0.05, and two asterisks (**) indicates p < 0.01.

Table 2
Summarise of percentage change of carbon variables under various LULCC types in the coastal wetlands[a]

Land-use type	Percentage change of carbon components (%)							Reference
	S_{PBC}	W_{STC}	S_{STC}	W_{SOC}	S_{SOC}	W_{SIC}	S_{SIC}	
LULCC from natural salt marsh								
Reclaimed wetland	49.9			−7.3–29.6				Literature data
Reclaimed wetland	−73.0	−29.7	−20.1	−39.8	−35.6	−14.9	5.2	**This study**
Dry farmland	13.6–85.7	−71.4–20.4		−92.0–73.0	−58.0–126.0	−11.7–124.9	66.5–151.9	Literature data
Dry farmland	−83.8	−26.5	−9.9	−17.5	−9.3	21.0	73.6	**This study**
Paddy field	86.4	−74.55 to −51.4	−51.6	−58.7–325.5		−14.5–35.7		Literature data
Paddy field	−53.8	−12.5	−0.8	−6.4	−0.5	11.3	76.0	**This study**
Aquaculture pond		−99.5 to −67.0	−78.4 to −33.1	−25.4–132.5	125.4	−8.2–7.9		Literature data
Aquaculture pond	−86.3	−30.0	−24.0	−42.6	−42.2	72.2	111.7	**This study**
LULCC from natural mangrove								
Reclaimed wetland	−100.0 to −8.0		−42.0 to −9.0					Literature data
Reclaimed wetland	−63.3	−22.2	−1.3	−28.9	−9.4	62.4	84.8	**This study**
Dry farmland	−84.8	−22.8	12.5	−30.9	−0.7	142.9	257.9	**This study**
Paddy field				7.2	−64.2 to −7.6			Literature data
Paddy field	−87.1	−24.5	−3.7	−25.8	−4.9	16.1	36.5	**This study**
Aquaculture pond	−91.7 to −85.0	−75.6 to −15.0	−60.5 to −7.5	−27.1 to −1.6	−31.9			Literature data
Aquaculture pond	−89.9	−34.7	1.1	−37.7	−3.2	65.9	148.0	**This study**

[a] The full table is showed in Table S4.

(Ellert and Bettany, 1995; Wallenius et al., 2011; Fig. 4), leading to a higher soil C stock (i.e., S_{STC}, S_{SOC} and S_{SIC}) relative to the other LULCC types (Fig. 2).

When NWs are converted to PFs, the decreased MGS and the increased SWC due to inundation limit oxygen availability and inhibit the oxidation of soil organic matter, resulting in higher W_{STC}, S_{STC}, W_{SOC} and S_{SOC} compared to DFs and RWs (Hassink, 1997; Zhang et al., 2019; Fig. 2). Although fertilization with organic manure in DFs and PFs probably increases the soil C contents and stocks (e.g., W_{STC}, S_{STC}, W_{SOC} and S_{SOC}), periodic harvesting in agricultural lands (e.g., DFs and PFs) would lead to a great loss of plant organic matter and a reduction in S_{PBC} and EOC (Figs. 2, 5 and Graphical abstract). Moreover, the groundwater used for irrigation in coastal areas commonly has a high carbonate concentration, which may contribute to the increase in SIC in irrigated DFs and PFs (Zamanian et al., 2016; Zhang et al., 2019).

Regarding the conversion to APs, the elimination of original vegetation in APs significantly decreased plant biomass (PB) and S_{PBC} by 95.5 %, causing a nearly zero input of plant organic C to the soil (Fig. 2 and Graphical abstract). The SEM analysis showed that PB had a high influence on S_{PBC} and EOC dynamics, which may be related to limited C input in the system (Fig. 6). However, APs are expected to accumulate a great amount of organic matter in the surface soil due to the excessive feeding and excretion of intensively cultivated aquaculture species (Yang et al., 2019). To avoid eutrophication of APs, the removal of animal excrement through dredging is a typical management practice in APs (Herbeck et al., 2013; Molnar et al., 2013). This would significantly decrease the accumulated organic matter. Therefore, due to the typical drainage of aquaculture ponds between harvesting and start-up of the next cultivation cycle, low C inputs (due to dredging) and high decomposition rates would cause a significant decrease in S_{STC} and EOC (Fig. 2). Previous studies also identified those processes as an important cause of C loss in converted wetlands (Kauffman et al., 2018; Sasmito et al., 2020; Yang et al., 2018a). Furthermore, the respiration of high stocks of aquatic animals in APs leads to high CO_2 and dissolved inorganic C concentrations in surface water (Yang et al., 2018b) and thus increases the concentrations of soluble SIC in soil porewater.

4.2. Different patterns of C responses to LULCC in salt marshes and mangroves

In this study, we found that the patterns of responses in plant–soil C variables to LULCC were different between salt marshes and mangroves (Fig. 3). The responses of W_{STC} and W_{SOC} to LULCC were comparable between salt marshes and mangroves, with a mean change rate of -27.0 ± 2.4 % (Fig. 3A and C). However, the change rates of S_{STC} and S_{SOC} due to LULCC in mangroves were relatively smaller than those in salt marshes, which might be because conversion of mangroves was associated with a stronger increase in bulk density (BD) (Fig. S2C). On the other hand, the increase in BD after LULCC could offset the decrease in soil C content (Alongi, 2020; Ellert and Bettany, 1995; Wallenius et al., 2011).

Previous studies reported that mangroves could export greater amounts of dissolved inorganic C to adjacent coastal waters than salt marshes (Alongi, 2020). In this study, the W_{SIC} and S_{SIC} values were nearly 5 times lower in mangroves than in salt marshes. Hence, under similar human-induced impacts of LULCC, such as irrigation in DFs and PFs and C emissions (respiration of aquatic animals) in APs, mangroves might have a higher percentage change in W_{SIC} and S_{SIC} than salt marshes (Kindler et al., 2011; Yang et al., 2018b; Zamanian et al., 2016; Zhang et al., 2019).

Within the focus of this study, as it only concerned the C dynamics in surface soil (0–30 cm depth), the S_{PBC} contributed a high proportion of EOC, especially for mangrove ecosystems (Figs. 2 and 5). Therefore, vegetation shifts and elimination of original trees caused by LULCC would result in a higher loss and change rate of S_{PBC} and EOC in mangroves (Kauffman et al., 2018; Lovelock et al., 2017; Sasmito et al., 2020).

Our results, including the change in plant–soil C content and stock due to the conversion of natural salt marshes and mangroves into LULCC types, were comparable to most of the previous studies on a global scale (Table 2). However, the effect of conversion to DFs and PFs on plant and soil C dynamics in salt marshes remains controversial. Xu et al. (2019) and Zhang et al. (2019) found that the conversion to agricultural land could increase soil C stocks, which may be related to the high intensity of agricultural practices such as organic fertilization and irrigation in the research area. Thus, the intensity of management practices in agricultural land as an important factor influencing soil C dynamics after LULCC should be considered in future research.

4.3. Latitudinal patterns of C responses to LULCC

As shown in Fig. 4, the response of W_{STC} to latitude increases in PFs might be related to temperature controls on C dynamics. Temperature has a well-documented positive effect on the decomposition rate and CO_2 emissions (Davidson and Janssens, 2006; Kirwan and Mudd, 2012; Lovelock et al., 2017). Higher temperatures in tropical and subtropical regions normally result in a higher decomposition rate of soil organic C, while decomposition rates are smaller in temperate regions. As the latitude increases, the response of S_{PBC} and EOC to all LULCC types shifts from a significant decrease in S_{PBC} and EOC at lower latitudes to a neutral response or increased values at higher latitudes. This is mainly related to the geographical distribution of the types of coastal wetlands. The vegetation types of coastal wetlands at lower latitudes are typically mangroves, which have a higher biomass productivity (Mcleod et al., 2011; Lu et al., 2017; Fig. S2A). The vegetation shifts caused by LULCC would lead to a great loss of S_{PBC} and EOC at low latitudes (Fig. 3G and H). At higher latitudes, coastal wetlands are dominated by salt marsh species with relatively lower biomass productivity, such as *Suaeda glauca*. Therefore, the conversion to RWs, DFs and PFs increased the PB and S_{PBC} in the Yellow River Estuary and Liao River Estuary (Figs. 1 and 4).

This study showed that the responses of W_{SIC} and S_{SIC} to LULCC shifted from an increased value at lower latitudes to a decreased value at higher latitudes. As mentioned above, W_{SIC} and S_{SIC} in mangroves at lower latitudes under similar LULCC practices would have a higher percentage change and increase rate compared to salt marshes at higher latitudes. Due to the complex climatic and geophysical effects, more data are needed to interpret the reaction of the C budget in salt marshes and mangroves to LULCC practices at the spatiotemporal scale.

4.4. Effect of LULCC on potential C emissions

Significant loss of EOC (including S_{SOC} and S_{PBC}) due to LULCC might lead to large potential C emissions (IPCC, 2014; Sasmito et al., 2019). As estimated in this study, the mean annual potential CO_2 emissions of all LULCC types were 7.92 ± 2.94 Mg CO_2-eq ha^{-1} yr^{-1}. demonstrating a notable greenhouse effect as a result of LUCC in China's coastal wetlands. Based on the area of lost coastal wetlands (see Table S3), we estimate that LULCC in Chinese coastal wetlands caused an annual CO_2 emission of 4.57 ± 1.65 Tg CO_2-eq yr^{-1}, with a total potential CO_2 emission of 123.42 ± 32.91 Tg CO_2-eq during 1979–2020 (for the studied nine provinces, Fig. S1b).

The conversion to APs resulted in the largest net EOC loss and CO_2 emission, followed by RWs (Fig. 2 and Graphical abstract). The great loss and decomposition of aboveground plant biomass due to periodic harvesting in DFs and PFs and elimination of vegetation in APs can cause notable CO_2 emissions in a short time. However, the decomposition of belowground and dead biomass will emit CO_2 continuously for 5–10 years after land-use conversion (Lovelock et al., 2011; Sasmito et al., 2020). Increased pore space and soil drainage in LULCC types turned soil anaerobic conditions into aerobic conditions, and thus, enhanced soil microbial activity and biochemical functioning will increase the decomposition potential and CO_2 emissions (Xiao et al., 2015; Ghani et al., 2003). The lower annual potential CO_2 emissions in DFs and PFs were mainly because the fertilization of organic manure in DFs and PFs could replenish soil organic C, resulting in a relatively lower S_{SOC} loss and CO_2 emissions from these LULCC types. Compared to RWs and DFs, the inundation in the rice planting period in PFs and the long-term flooding in APs could inhibit the oxidation and decomposition of SOC, preventing the emission of CO_2. However, an uncertainty regarding C emission calculations needs to be taken into account. The approach we used might lead to an overestimation of CO_2 from DFs and PFs because plants or crops harvested seasonally or yearly from these systems are not fully converted to CO_2. Moreover, soil erosion induced by irrigation, tillage and dredging in the converted wetlands would also lead to C losses, mainly including soil lateral redistribution and leaching of dissolved carbon into groundwater (Li et al., 2019; Van Oost et al., 2007). Whereas, not all parts of the C losses contributed to potential CO_2 emissions.

On the other hand, methane (CH_4) is often released in PFs and APs because inundation and low-salinity conditions are favorable for methanogen activity and CH_4 production (Olsson et al., 2015; Tan et al., 2020; Yang et al., 2018b). CH_4 has a 34 times greater global warming potential than CO_2 on a per unit mass basis over the 100-year time horizon (IPCC, 2014). However, in this study, the potential C loss due to CH_4 emissions was not estimated, which might lead to an underestimation of the greenhouse effect of LULCC when NCWs were converted to PFs and APs but an overestimation if NCWs were converted to RWs and DFs under drying conditions. A global meta-analysis by Tan et al. (2020) showed that CH_4 is the dominant greenhouse gas (e.g., CO_2, CH_4 and nitrous oxide) in the conversion of natural coastal wetlands into constructed wetlands and aquaculture ponds, but CH_4 emissions decreased in the conversion to cropland. Therefore, CH_4 emissions deserve more attention in future studies on the effect of LULCC on C loss in coastal wetlands.

5. Conclusions

This study focused on the effect of popular LULCC types in Chinese coastal provinces on plant–soil carbon contents and stocks against the original natural salt marshes and mangroves. Conversion to reclaimed wetlands, dry farmlands, paddy fields and aquaculture ponds from both salt marshes and mangroves resulted in a decrease in most C content and stock variables while slightly increasing W_{SIC} and S_{SIC}. The relative loss rate of ecosystem organic C due to LULCC was greater in mangroves than in salt marshes. The annual potential CO_2 emissions caused by the loss of EOC were dependent on the LULCC types, with a greater impact from aquaculture ponds and reclaimed wetlands than from dry farmlands and paddy fields. With decreasing latitude, the loss rate of EOC in all LULCC types showed a significantly increasing trend. The response of plant and soil C variables to LULCC was sensitive to changes in plant biomass, median grain size, soil water content and soil NH_4^+-N content. Our results can be used to support land-use policy-making, the development of strategies to mitigate greenhouse gas emissions, and global C budget accounting. For instance, C budget assessments such as in the IPCC Evaluation Guidelines on Wetlands and global carbon budget studies (Hiraishi et al., 2014; Friedlingstein et al., 2020) do not account for the separate effects of various LULCC types on potential CO_2 emissions. This study might be helpful for policy-makers to develop practices of C emission reduction at a national scale. The dataset could also support improving the accuracy of global C model forecasting of climate change in response to various LULCC types in coastal wetlands.

CRediT authorship contribution statement

Li-Shan Tan: Conceptualization, Methodology, Investigation, Formal analysis, Writing – review & editing. **Zhen-Ming Ge:** Conceptualization, Methodology, Investigation, Formal analysis, Writing – review & editing. **Shi-Hua Li:** Investigation, Formal analysis. **Ke Zhou:** Investigation, Formal analysis. **Derrick Y.F. Lai:** Writing – review & editing. **Stijn Temmerman:** Writing – review & editing. **Zhi-Jun Dai:** Writing – review & editing.

Data availability

Data will be made available on request.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.164206.

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