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Responses of soil CO_2 and CH_4 emissions to changing water table level in a coastal wetland

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ABSTRACT

Global climate change and in particular sea level rise have resulted in water table level rise in the coastal wetland, which may alter the magnitude and direction of carbon flux. However, the degree to which different water table level affects soil CO_2 and CH_4 emissions remains uncertain in coastal wetland. Here, a soil microcosm experiment with five water table levels (−40, −30, −20, −10, 0 cm) was conducted in the Yellow River Delta, China. The water table level was controlled by manual. The soil CO_2 and CH_4 emissions of each water table levels were measured during 150-days incubation in 2018. Our results showed that water table level rise decreased soil CO_2 emissions, while increased soil CH_4 emissions. However, there was no significant difference in soil CO_2 and CH_4 emissions from −20 to −40 water table levels, respectively. In addition, water table level rise significant alter soil physical and chemical properties in the uppermost soil layer (0−10 cm) in coastal wetland, in particular soil moisture and salinity, which probably jointly affected soil CO_2 and CH_4 emissions. Furthermore, cumulative soil CH_4 emission was positively significantly correlated to soil organic carbon and total carbon, suggesting that carbon component can supply energy and nutrients and benefit for soil CH_4 production. Additionally, there was a significant relationship between cumulative soil CO_2 emission and dissolved organic carbon, which indicated that CO_2 was mainly contributed from dissolved organic carbon. Cumulative soil CO_2 emission was significantly correlated with soil microbial biomass carbon, suggesting that microbial activity played an important role in CO_2 emissions in coastal wetlands. Our results also indicate that water table level rise caused by sea level rise may contribute to the storage of soil organic carbon and produces a lower global warming potentials of CH_4 and CO_2 in the further climate change. Therefore, it is necessary to estimate the effect of hydrological, especially water table level on carbon cycles in coastal wetland when evaluating the climate−carbon feedback scenarios.

1. Introduction

It is currently recognized that carbon dioxide (CO_2) and methane (CH_4) emissions are the main causes of climate change primarily characterized by global warming (IPCC, 2013). The CO_2 concentration has reached 405.0 ± 0.1 ppm in 2017, which is more than 46% higher than the pre-industrial level in 1750 (Dlugokencky and Tans, 2018). Despite CH_4 being present at lower concentrations in the atmosphere, its global warming potentials (100 years) is 34 times more than CO_2, contributing to over 20% of recent global warming (IPCC, 2013). Increasing CO_2 and CH_4 concentrations will lead to increasing concerns about the potential environmental impact of ongoing climate change. Therefore, quantifying the magnitude of the CO_2 and CH_4 source or sink is critical for accurately evaluating global carbon budgets, formulating scientifically sound management strategies of ecosystems and emission reduction measures (Yang et al., 2018a).

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Although coastal wetlands only account for 0.22–0.34% of the Earth's land surface (Fennessy, 2014), the organic-rich soils of many coastal wetlands contain exceptionally large C stocks, which can be two to three times higher than those in most terrestrial ecosystems (Han et al., 2018). Coastal wetlands including mangrove, seagrass, and salt marsh play an important role in the global carbon sequestration, the carbon buried in coastal wetlands is therefore called "blue carbon" (Lovelock et al., 2017). Previous study has demonstrated the global carbon burial rates of salt marshes are estimated to be 5 to 87 Tg C yr $^{-1}$, which are comparable to the carbon burial rate of forests ecosystems (53.0, 78.5 and 49.3 Tg C yr $^{-1}$ for temperate, tropical and boreal forests, respectively) (Mcleod et al., 2011). However, the carbon burial rates in salt marsh exist uncertainty. CH_4 emissions are typically minimal in salt marsh because an abundance of sulfate (an anion present in water) suppresses microbial CH_4 production and emission (Poffenbarger et al., 2011). Thus, with high rates of net C storage and minor CH_4 emission, coastal wetlands, including salt marshes, generally play an important role in climate mitigation. People have a great interest in the soil carbon stocks in coastal wetlands, which due to the C pool alterations will have a significant influence on the global C balance (Chambers et al., 2013). However, coastal wetland is very sensitive to global climate changes, especially sea level rise (Kirwan and Megonigal, 2013), which has exceeded the global mean sea level rise over the past 10 years (IPCC, 2014). The sea level rise could lead to the changes in hydrological cycle (e.g. water table level) in coastal wetlands. For example, sea level rise will stimulate the water table level rise in coastal wetland (Rotzoll and Fletcher, 2012; Cowling, 2016). Previous study showed that a global map of water table level indicates a strong correlation between coastal wetlands and water table level at 0 m (Fan et al., 2013). Thus, water table level is an important controlling factor in hydrological cycle in coastal wetland (Cowling, 2016; Han et al., 2018).

Water table level play an important role in altering the structure and functions of the wetland (Webb and Leake, 2006). In general, water table level has been recognized as one of the most important controls on CO_2 and CH_4 emissions from wetlands, since water table level determine the aerobic/anaerobic zones and redox state in the soil profile which in turn affect the decomposition rates of soil organic matter (Dinsmore et al., 2009). A continuous increase in water table level will decrease the diffusion of oxygen and limit aerobic microbial activity, and further inhibit CO_2 emissions in the water-saturated soil. Conversely, oxygen diffusion into soils could increase more efficient aerobic decomposition and lead to an increase in CO_2 emissions from soils when the water table drops (Juszczak et al., 2013; Yang et al., 2013). For example, previous studies have demonstrated mean CO_2 emissions were lowest under high water tables in a freshwater wetland, while a lowered water table (−11 to 0 cm) resulted in up to 120% increased CO_2 emission rates (Yang et al., 2013). In a Zoige peatland, low water table significantly increased soil CO_2 emissions compared to high water table (Cao et al., 2017). CH_4 production are high by increasing methanogens, including Methanobacteriaceae, Methanosaetaceae, Methanoregulaceae, Methanosarcinaceae and Methanomicrobiales when the water table rise (Horn et al., 2003; Zhang et al., 2018). On the contrary, CH_4 emissions decreased with the water table drops, which mainly due to the decreased CH_4 production potential (Wang et al., 2017) or increased CH_4 oxidation potential (Koh et al., 2009; Yang et al., 2013). Therefore, changes in water table level may have a pronounced effect on wetland ecosystem, as well as potential carbon-climate feedbacks (Rotzoll and Fletcher, 2012; Cowling, 2016; Taylor et al., 2013; Carretero and Kruse, 2012). However, existing studies have primarily focused on the relationship between greenhouse gases emissions and water table levels in peatlands, while studies in salt marshes are lacking (Chimner and Cooper, 2003; Turetsky et al., 2008; Berglund and Berglund, 2011; Ishikura et al., 2017; Yang et al., 2014; Cao et al., 2017; Wang et al., 2017; Olsson et al., 2015; Yamochi et al., 2017). Therefore, more investigations are required to elucidate the influence of water table level on soil CO_2 and CH_4 in coastal wetlands.

The water table level in many salt marshes can both be oligohaline and polyhaline, depending on the geological conditions, elevation, salt-water intrusion and connectivity to the open sea (Cowling, 2016; Han et al., 2018). Water table level rise can lead to the high salinity in coastal wetland, which is an important characteristic that different from other wetlands. In general, high salinity not only affect biogeochemical conditions but also alter microbial community composition and microbial activity (Neubauer, 2013; Zhang et al., 2018). Consequentially, elevated salinity probably potentially increases microbial respiration, stimulating organic C loss from wetland soils or decrease soil respiration, promoting C storage (Stagg et al., 2017). Additionally, high salinity can reduce soil CH_4 emissions, which probably because sulfate reducing bacteria (*desulfovibrio desulfuricans*) usually compete with methanogens for use of substrates to inhibit methanogens (Olsson et al., 2015). Therefore, soil salinity is an important environmental factor in affecting the rate of C cycling in coastal wetlands (Wilson et al., 2015; Servais et al., 2019; Wen et al., 2019).

The Yellow River Delta is one of the most active regions of land-ocean interaction among the many river deltas in the world. Due to near the sea and low elevation, the water table level is shallow and salt water, which cause soil salinization and alkalization. However, sea level rise is becoming a threat for this Delta and it is predicted to rise to 35–40 cm in 2050 (Sun et al., 2015), which might also result in water table level rise and high salinity. Changing in water table level caused by sea level rise may modify soil CO_2 and CH_4 emissions in the future climate change. Therefore, we conducted a soil microcosm experiment to improve the knowledge of the water table level (salt water) on soil CO_2 and CH_4 dynamics in the Yellow River Delta in 2018, our objectives are (1) to assess the effect of water table level on soil CO_2 and CH_4 fluxes in a salt marsh, (2) to evaluate how soil physical properties will influence the interaction between water table level and carbon fluxes in a salt marsh.

2. Materials and methods

2.1. Site description

The research was conducted at the Research Station of Coastal Wetland in the Yellow River Delta (37°45′50″ N, 118°59′24″ E), Chinese Academy of Sciences, in Kenli County, Shandong Province, China. This area is characterized by a continental monsoon climate. The annual average temperature is 12.9 °C. The average annual precipitation is 550–640 mm, with of which above 74% is rainfall between June and September which the salt marsh was always inundated throughout the wet season. The experimental soil texture is mainly sandy clay loam and is classified as a Salic Fluvisols (WRB, 2006; Jiao et al., 2019). Soil physico-chemical at different soil layers are showed in Table 1. The vegetation is dominated by *Suaeda salsa*, *Phragmites australis*, *Tamarix chinensis* and *Imperata cylindrica*. The water table level is shallow (mean 1.14 m) around the Yellow River Delta (Fan et al., 2012), and soil salinization at the soil surface (<20 cm) is generally severe (5.32–9.50 g kg^{-1}) and widespread.

2.2. Experimental design

The soil microcosms, 21 cm diameter and 54 cm long Polyvinyl chloride (PVC) pipes, filled with native soil and incubated in a

Table 1
Soil properties (mean ± standard deviation, n = 4) of the incubated soils in the Yellow River Delta. Ec, electrical conductivity; TC, total carbon; TN, total nitrogen; SOC, soil organic carbon; DOC, dissolved organic carbon and SMBC, soil microbial biomass carbon.

Depth (cm)	Soil moisture (%)	pH	Ec (ms cm^{-1})	TC (g kg^{-1})	TN (g kg^{-1})	SOC (g kg^{-1})	DOC (mg kg^{-1})	SMBC (mg kg^{-1})
0–10	26.27 ± 0.65	7.78 ± 0.06	9.86 ± 0.24	18.56 ± 1.42	0.84 ± 0.12	5.09 ± 0.72	42.45 ± 13.97	32.42 ± 3.09
10–20	27.89 ± 0.51	7.97 ± 0.02	8.55 ± 0.63	13.80 ± 0.84	0.41 ± 0.06	2.43 ± 0.21	13.72 ± 1.63	19.20 ± 1.16
20–30	26.73 ± 0.65	8.05 ± 0.01	7.17 ± 0.31	11.06 ± 0.15	0.23 ± 0.07	1.12 ± 0.02	10.47 ± 0.90	16.01 ± 1.54
30–40	27.39 ± 1.14	8.13 ± 0.07	6.81 ± 0.60	10.84 ± 0.20	0.20 ± 0.08	0.84 ± 0.05	8.38 ± 0.95	13.42 ± 1.24
40–50	28.47 ± 0.33	8.18 ± 0.01	7.78 ± 0.84	10.72 ± 0.10	0.18 ± 0.03	0.74 ± 0.02	8.42 ± 1.52	12.88 ± 1.12

greenhouse where the temperature was kept at 25 °C. The pipes were installed in early April 2018 near the station (37°46′13″ N, 118°58′52″ E). In the undisturbed area, ground vegetation was sparse and before taking the soil microcosms, the plants were removed by hand carefully. The pipes were vertically driven 50 cm into the soil, leaving 4 cm as headspace volume. The soil around the pipe was dug away and pulled the pipe from the ground carefully. The 20 pipes were transported to the greenhouse and put into the five water tanks (4 pipes were put in each tank). Five water table levels were conducted, denoted as 0 (at the soil surface), −10, −20, −30, −40 cm below soil surface. Therefore, each tank represents a water table level. The water salinity is 7 ppt via adding sea salt particles, which corresponded to the mean groundwater salinity of the sampling site. The water in tank was added by manual when the water table level dropped below the set points.

2.3. Gas flux measurements and soil analyses

Soil CO_2 and CH_4 fluxes were measured once every 10 days from 23 April to 23 September 2018, by attaching a LGR Ultraportable Greenhouse Gas Analyzer (UGGA) chamber (Los Gatos Research, Inc., San Jose, USA) to the top of the pipe. Additionally, soil cores (5 cm diameter, 10 cm high) were taken using an auger from four soil depths (0–10, 10–20, 136 20–30 and 30–40 cm) in each treatment and replicate. A part of soil samples was passed through a 2-mm sieve and then the soil samples was analysed for soil microbial biomass carbon (SMBC). Other were air-dried and ground to analyse physico-chemical.

Gravimetric soil moisture was measured by the drying method. Soil pH of centrifuged solutions were then tested by portable pH meter. Soil electrical conductivity (Ec) was measured as a proxy for soil salinity in a 1:5 soil: deionised water suspension with an Ec meter (2265FS, Spectrum Technologies, Inc.) after 1 h shaking at 25 °C. The soil total carbon (TC) and total nitrogen (TN) concentrations were quantified using vario MACRO element analyzer. The dissolved organic carbon (DOC) was obtained with highly purified water and was measured using high-temperature catalytic combustion in a total organic carbon analyzer (TOC-L CPN, Shimadzu). SMBC was measured by the fumigation-extraction method and was measured with a Total Organic Carbon Analyzer (Elementar vario TOC, Elementar 147 Co., Germany) (Joergensen, 1996).

As CO_2 emission was significantly correlated with DOC concentration, depending on microbial decomposition, we estimated the relative CO_2 emission rate contributed to the total emissions by each soil depth, as DOC varied over the soil column (Equation 1).

2.4. Global warming potentials (GWPs)

The CO_2 is usually used as the reference gas for estimating GWPs. The constants to calculate GWP for CH_4 is 34 (based on a 100-year time horizon, IPCC, 2013). The GWPs was thus calculated as (Equation 2):

2.5. Statistical analysis

The soil physico-chemical (soil moisture, pH, Ec, TN, TC, DOC, and SMBC concentrations) and soil CO_2 and CH_4 fluxes were statistically analysed with the analytical software SPSS 16.0 (IBMSPSS, USA). The difference between soil physico-chemical under different water table levels and different soil depths were tested by two-way ANOVA. Effects of main factors and their interaction were analysed, with the main factors being "Soil depth" and "Water table level". The relationships between CO_2 and CH_4 fluxes and soil properties were tested pearson correlation coefficients. Significance was accepted at the $P < 0.05$ level of probability.

3. Results

3.1. Effect of water table level on soil CH_4 and CO_2 emissions

Soil CH_4 and CO_2 emissions showed considerable fluctuations under different water table levels during the incubation (Fig. 1a and b). Soil CH_4 emission at −20, −30 and −40 cm water table levels increased during the initial 15 days, peaked after 60 days (0.22 ± 0.06, 0.27 ± 0.05 and 0.18 ± 0.03 nmol CH_4 m^{-2} s^{-1}, respectively), then decreased slowly from 60 to 150 days (Fig. 1a). Soil CH_4 emissions under 0 and -10 cm water table levels showed an increasing trend over the first 60 days, peaked at 110 and 90 days (0.37 ± 0.03 and 0.31 ± 0.08 nmol CH_4 m^{-2} s^{-1}, respectively), then fluctuated around the same concentration until 150 days (Fig. 1a). The dynamics of soil CO_2 emissions were overall similar for the different water table levels during the incubation period. Soil CO_2 emissions increased rapidly at the beginning of the incubation, peaked at 10 days (0.29 ± 0.03, 0.41 ± 0.03, 0.53 ± 0.04, 0.63 ± 0.08 and 0.60 ± 0.03 μmol CO_2 m^{-2} s^{-1} at 0, 10, −20, −30 and −40 cm treatments, respectively), and then gradually decreased with fluctuating emissions from 10 to 150 days (Fig. 1b).

The cumulative CH_4 emissions in almost all water table levels steadily increased at a constant rate (Fig. 1c). Average CH_4 emissions were generally lowest and highest at −40 and 0 cm water table level, respectively (0.07 ± 0.01 and 0.22 ± 0.01 nmol CH_4 m^{-2} s^{-1}, respectively; Fig. 2a). CH_4 emissions under 0 and -10 cm water table levels were significantly greater than at −20, −30 and −40 cm over the 150-day period ($P < 0.01$). However, there was no significant difference in CH_4 emissions between 0 and -10cm water table levels (Fig. 2a). In addition, soil CH_4 emissions increased exponentially with increase of water table level (Fig. 2b). The cumulative CO_2 emissions also increased at a constant rate (Fig. 1d). However, in contrast to CH_4 emissions, average CO_2 emissions were generally lowest and highest at 0 and -30 cm water table level, respectively (0.21 ± 0.01 and 0.32 ± 0.03 μmol CO_2 m^{-2} s^{-1}, respectively; Fig. 2c). CO_2 emission at 0 cm water table level was significantly smaller than that for the −20, −30 and −40 cm treatments ($P < 0.01$), but no remarkable difference was observed among −20, −30 and −40 cm water table levels, and neither among the 0 and -10 cm water table levels during the incubation (Fig. 2c). Additionally, soil CO_2 emissions decreased linearly with increase of water table level

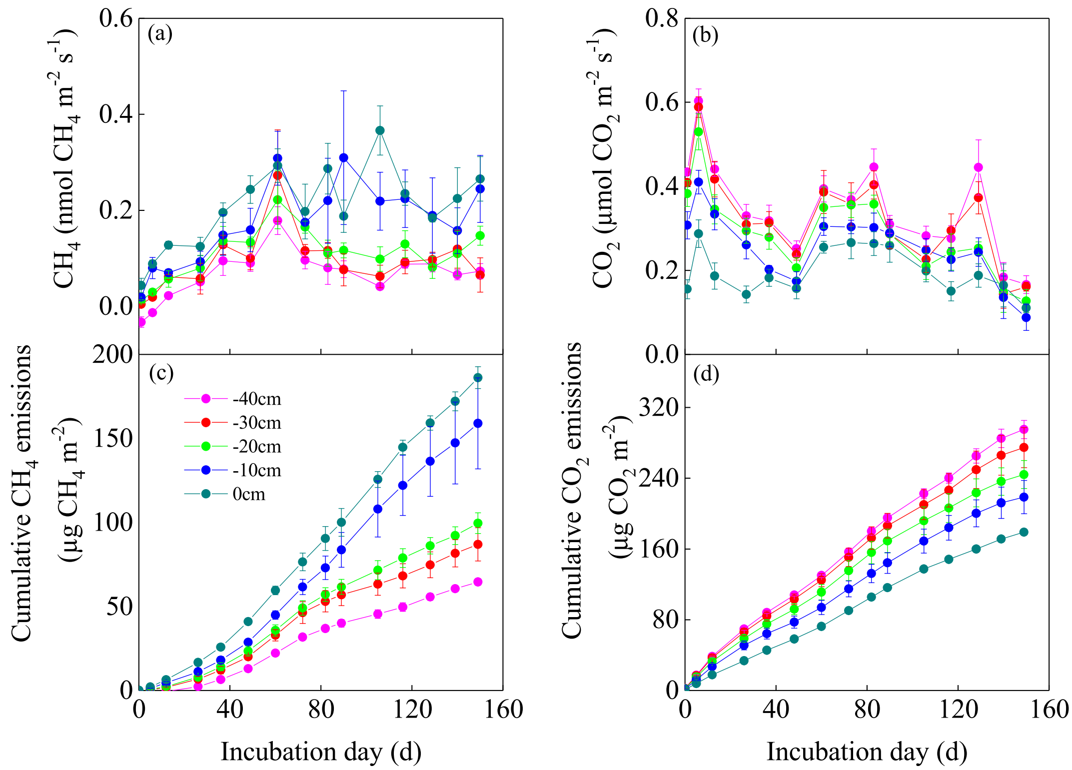

Fig. 1. Dynamics of soil CH_4 emissions (a) and CO_2 emissions (b), cumulative CH_4 emissions (c) and cumulative CO_2emissions (d) during the incubation period under −40, −30, −20, −10 and 0 cm water table levels in a salt marsh of coastal wetland. The error bars indicate the standard deviation of the mean (n = 4).

(Fig. 2d). Moreover, the GWPs of CH_4 and CO_2 decreased as the water table level rise. Average GWP were generally lowest and highest at 0 and -40 cm water table levels, respectively (186.19 ± 3.03 and 298.36 ± 10.17 mg CO_2−eq m^{-2}, respectively; Fig. 3).

3.2. Effect of water table level on soil properties in the microcosms

Soil properties were significantly affected by changing water table level in our microcosms (except TN), and water table level interacted with soil depth for all measured parameters except SOC and DOC (Table 2). Soil moisture was significantly higher at high water table levels, in particular at 0–10 and 10–20 cm soil layer ($P < 0.05$). However, no remarkable differences in soil moisture were observed among any treatments below the 30 cm soil layers (Fig. 4a). In the uppermost soil layer (0–10 cm), pH at the 0 cm water table level was significantly higher than pH at all other water table levels ($P < 0.01$). In general, pH at a water table level of 0 cm was fairly constant throughout all soil layers, while pH from −10 to −40 cm water table levels steadily increased with soil layers (Fig. 4b). The electrical conductivity (Ec) at 0 cm water table level was lower than that other treatments in the uppermost soil layer ($P < 0.001$) and it stayed fairly stable throughout all soil layers. Ec from −10 to −40 cm water table levels decreased with increasing soil depth (Fig. 4c).

There was a significant difference in SOC concentrations among all treatments at the uppermost soil layer ($P < 0.05$), whereas SOC concentrations were similar among different water table levels below the 10 cm soil layers (Fig. 4d). The SOC concentration ranged from 0.90 to 5.64 g kg^{-1} (Fig. 4d). There was significant difference in TC contents among all treatments at 0–10 and 10–20 cm soil depth ($P < 0.05$; Fig. 4e). TC at the uppermost soil layer under 0 and -10 cm water table levels were significantly higher than other treatments (Fig. 4e). Overall, TC concentrations ranging from 10.70 to 18.56 g kg^{-1} (Fig. 4e). There was no significant difference in TN concentration between any water table level, but TN decreased with soil layer (Fig. 4f). TN concentrations ranged from 0.14 to 0.79 g kg^{-1} (Fig. 4f).

DOC concentrations at the uppermost soil layer were higher than other water table levels under −30 and −40 cm water table levels ($P < 0.01$). However, no differences in DOC concentrations were observed among any water level treatment below the 0–10 cm soil layer (Fig. 4g). DOC concentrations significantly decreased with increasing soil depth (Table 2), ranging from 9.16 to 27.46 mg kg^{-1} (Fig. 4g). Additionally, the relative CO_2 emission rate contributed to the total emissions was the highest at the 0–10 cm soil layer compared to other soil layer. DOC concentrations under the −40 cm and −30 cm water table levels were higher than other treatments (Fig. 6). SMBC concentrations in the uppermost soil layer were higher than other treatments under the −30 and −40 cm water table levels ($P < 0.05$). SMBC concentrations decreased with increasing soil depth and SMBC showed similar among all treatments below uppermost soil layer (Fig. 4h).

4. Discussion

Soil CH_4 emissions increased exponentially with rising water table level in our microcosm experiment (Fig. 2b). Averaged CH_4 emissions more than doubled when the water table level rose from −40 to 0 cm. The correlation between elevated CH_4 emissions at higher water table levels was evidenced by previous studies in freshwater ecosystems (Moore and Dalva, 1993; Jungkunst et al., 2008; Hou et al., 2013; Karki et al., 2014; Yang et al., 2013, 2014; Wang et al., 2017; Hoyos-Santillan et al., 2019) and saline soils (Furukawa et al., 2005; Yamamoto et al., 2011). The soil physico-chemical properties are significantly affected by different water

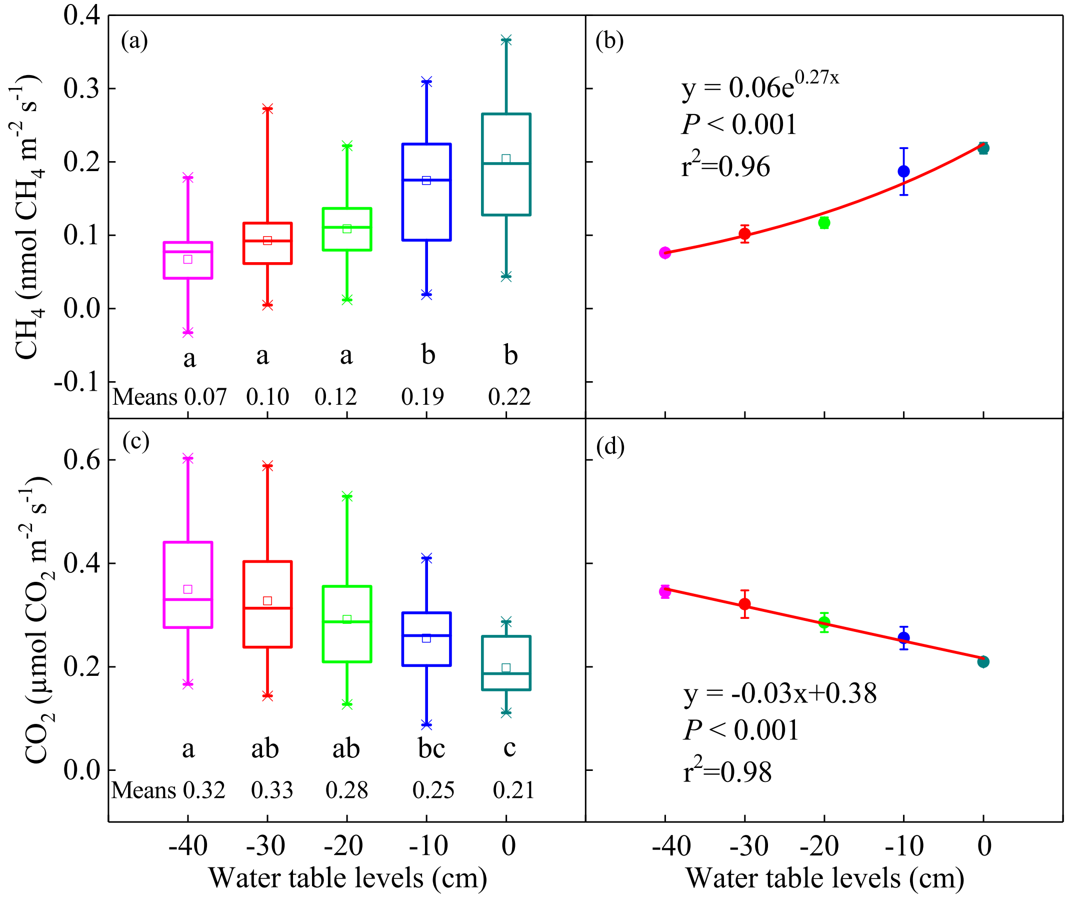

Fig. 2. Soil CH_4 (a, b) and CO_2 (c, d) emissions under different water table levels (mean ± SE) in a salt marsh of coastal wetland. Different letters on the error bars indicate significant differences at $P < 0.05$.

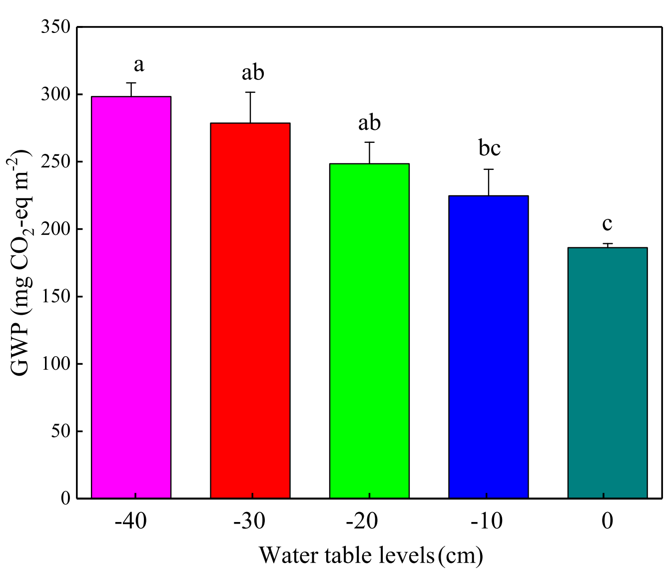

Fig. 3. Global warming potential (GWP) under different water table levels (mean ± SE) in a salt marsh of coastal wetland. Different letters on the error bars indicate significant differences at $P < 0.05$.

table levels, in particular in the uppermost soil depth (0–10 cm), which in turn regulate the soil CH_4 and CO_2 emissions (Berglund and Berglund, 2011). Soil cumulative CH_4 emissions were positively related to soil moisture at 0–10 cm soil depth ($P < 0.01$, Fig. 5), which indicated that soil moisture is the main factor affecting CH_4 production and consumption (Matysek et al., 2019; Strack et al., 2004; Schaufler et al., 2010). On the one hand, water-saturated and -logged soils caused by water table level rise, which could decrease the diffusion of oxygen. The decreased oxygen concentrations are beneficial to anaerobic decomposition by methanogenic bacteria, which can promote the CH_4 production (Yang et al., 2013). On the other hand, at low water table level, oxygen diffusion is facilitated into the soil, which may lead to increasing CH_4 oxidation (Lombardi et al., 1997; Strack et al., 2004).

Additionally, soil salinity is an especially important factor in salt marsh, which could alter the microbial processes, and change the future carbon sequestration (Wilson et al., 2015; Wen et al., 2019). Salinization causes higher Na^+, Cl^- and SO_4^{2-} concentrations in soils. Soil microbial communities are directly or indirectly affected by the osmotic and ionic stress these ions pose, which may decrease C cycle rate (Setia et al., 2010). Coastal wetland in particular salt marsh has high concentrations of SO_4^{2-}, which inhibit CH_4 production due to competition of sulfate reducing bacteria with methanogens (Olsson et al., 2015). Previous study has demonstrated that soil sulphur had significant relationship with salinity in the Yellow River Delta (Lu et al., 2015). And the average concentrations sulphur in 0–30 cm soil was about 822.43 mg kg^{-1}, which was higher than the average value of total sulphur in world (Yu et al., 2014). This indicated that the more sulphur, the more salinity, and finally inhibited CH_4 production (Poffenbarger et al., 2011; Wen et al., 2019). For example, in a coastal saline rice fields,

Table 2
Results (F-values) of two-way analysis of variance (ANOVA) on the effects of different water table level (W), soil depth (D), and their interactions on soil properties. Ec, electronic conductivity; SOC, soil organic carbon; TC, total carbon; TN, total nitrogen; DOC, dissolved organic carbon and SMBC, soil microbial biomass carbon.

	Soil moisture	pH	Ec	SOC	TC	TN	DOC	SMBC
W	15.84***	44.22***	139.86***	213.55***	374.86***	1.49	140.19***	335.61***
D	9.73***	1.12	1.29	0.92	1.67	163.97***	6.42***	3.64*
W*D	3.26**	4.14**	9.69***	1.09	3.64**	2.14*	1.57	2.47*

*, ** and ***: statistically significant at $P < 0.05$, $P < 0.01$ and $P < 0.001$.

lower CH_4 emissions are observed, which probably due to higher soil salinity limited methanogens (3.96 dS m^{-1}; Datta et al., 2013). Tidal marshes with salinities over 18 ppt have significantly lower CH_4 emissions than freshwater, oligohaline and mesohaline marshes (Poffenbarger et al., 2011). In the present study, cumulative soil CH_4 emissions were negatively significantly correlated to soil salinity (Fig. 5). Water table level rise significantly reduced Ec and SMBC at the 0–10 cm soil layer from 9.52 to 4.89 ms cm^{-1} (Fig. 4c) and from 27.59 to 21.04 ms cm^{-1}, respectively (Fig. 4h). Meanwhile, SMBC was positively correlated to Ec (Fig. 5), which indicated that the soil salinity may alter the activities of soil microbial structures and communities and in turn affect CH_4 emissions finally (Poffenbarger et al., 2011). On the contrary, soil CH_4 emission was not affected by salinity in a brackish marsh due to elevating salinity did not alter microbial processes (Wilson et al., 2018). Overall, the CH_4 emissions probably jointly affected by soil salinity and moisture when the water table level rise in coastal wetland. Furthermore, there were some evidences in the literature on the effects of SOC on CH_4 emission in the wetlands (Koh et al., 2009; Xiang et al., 2015). In our experiment, soil CH_4 emission was positively significant correlated to SOC and TC concentration (Fig. 5), which suggested that carbon component supply substrate or nutrients for methanogens, stimulating soil CH_4 emissions. Our research site is located in the Yellow River Delta, where the elevation is low, near the sea and the groundwater table is shallow. Therefore, the soil of coastal wetland is easy to become saturated during the wet season (from July to September), which indicated that the delta during the wet season will become a major source of methane, and this will have an adverse impact on climate change.

In contrast to CH_4 emissions, soil CO_2 emissions decreased with increased water table level (Fig. 2d). Soil CO_2 emissions under −40 cm water table level were 1.5 times higher than under 0 cm (Fig. 2c). Previous studies have demonstrated that soil CO_2 emissions decreased with increasing of water table level in the laboratory (Jungkunst et al., 2008; Kane et al., 2013; Yang et al., 2013, 2017; Matysek et al., 2019) and field experiments (Furukawa et al., 2005; Miao et al., 2013; Yamochi et al., 2017; Cao et al., 2017; Hoyos-Santillan et al., 2019). On the one hand, the mineralization of organic matter is increased by O_2 diffusion into deeper soil layers. Hydrolytic enzymes in the soil contribute to organic matter decomposition, but these enzymes are inhibited by phenolic compounds. Phenol oxidase is high under low water table levels and reduces the concentration of phenolic substances, thereby enhancing CO_2 emissions out of the soil. (Freeman et al., 2001, 2004). On the other hand, low water table levels also enhance soil microbial activities, which use up labile organic C substrates, directly leading to higher CO_2 emissions (Chimner and Cooper, 2003). Furthermore, anoxic conditions in high water table levels result in toxic byproducts (e.g., HS^{-}) that limit microbial growth and activities (Marton et al., 2012), and then reduce CO_2 emissions. In our study, cumulative CO_2 emissions were negatively related to soil moisture at 0–10 cm soil depth ($P < 0.01$, Fig. 5). Water table level rise would lead to the increase in soil moisture, which limit CO_2 emissions by limiting microbial activities and decomposition rates (Jimenez et al., 2012; Yang et al., 2014).

Salinity strongly affects soil carbon process, however, soil CO_2 flux responses to salinity are contradictory (Stagg et al., 2017). For example, soil CO_2 emissions in seawater have been shown to be significantly higher than in freshwater can be attributed to the increased availability of SO_4^{2-} to serve as a terminal electron acceptor in anaerobic microbial respiration (Chambers et al., 2011; Weston et al., 2011). On the contrary, previous studies have observed declines in CO_2 emission associated with increasing salinity because the reduced activities of enzymes associated with the hydrolysis of cellulose and the oxidation of lignin (Neubauer, 2013) and a lower sulfate reduction (Yang et al., 2018b). In addition, elevating salinity had little effect no soil CO_2 emissions in a brackish coastal wetland, which probably due to elevating salinity probably did not alter microbial processes (Wilson et al., 2018). In our experiment, soil CO_2 emissions were positively significantly correlated to salinity (Fig. 5). The CO_2 emissions decreased with the decrease of soil salinity (from 8.96 ms cm^{-1} under −40 cm water table level to 4.89 ms cm^{-1} under 0 cm water table level), which probably attributed to the contribution of higher sulfate reduction to the microbial respiration and carbon mineralization (Weston et al., 2006). Therefore, the interaction of soil moisture and salinity may affect the soil CO_2 emissions when changing water table in coastal wetland. Additionally, there was a significant positive correlation between soil cumulative CO_2 emissions and SMBC (Fig. 5), indicating that soil microbial activity play an important role in CO_2 emissions (Yang et al., 2014).

Dissolved organic carbon (DOC) concentrations are strongly affected by hydrological conditions in wetlands, especially the water table level (Strack et al., 2008, 2019). In the present study, a lower water table level significantly increased DOC concentrations (Fig. 4g), which was consistent with previous studies (Frank et al., 2014; Strack et al., 2019). The aerated soil resulting from water table level drawdown, the oxygen-rich soil layers can promote organic matter decomposition, contributing to higher DOC concentrations (Liu et al., 2016; Strack et al., 2019). On the contrary, a laboratory study suggested that drought (low water table level) decreased DOC concentration due to the increasing SO_4 and soil acidity (Tang et al., 2013). In a peatland, the DOC concentration in the −30 cm water table treatment was 1.2 times higher than the one in the −50 cm treatment (Matysek et al., 2019). There was no significant difference in DOC concentrations from −50 to 0 cm water tables in a Zoige Peatlands, which due to changing water table did not alter CO_2 emissions (Yang et al., 2017). However, high water level (anaerobic conditions) hinder gas exchange between soil and atmosphere and limit the diffusion of O_2 availability in water-saturated soils, which results in limited organic matter mineralization and limited DOC production (Jimenez et al., 2012; Yang et al., 2014). Additionally, the significant correlation between soil CO_2 emission and DOC concentrations (Fig. 5) was consistent with previous studies (Liu et al., 2017; Li et al., 2020) and indicated that CO_2 was mainly contributed from DOC (Chow et al., 2006; Liu et al., 2017). DOC is the most active organic substrate for microorganisms, which can supply energy and nutrients for microbial metabolism and further promote CO_2 production (Kane et al., 2013; Yang et al., 2018b). Therefore, the water table level lowering promoted the production of DOC and stimulated microbial activity,

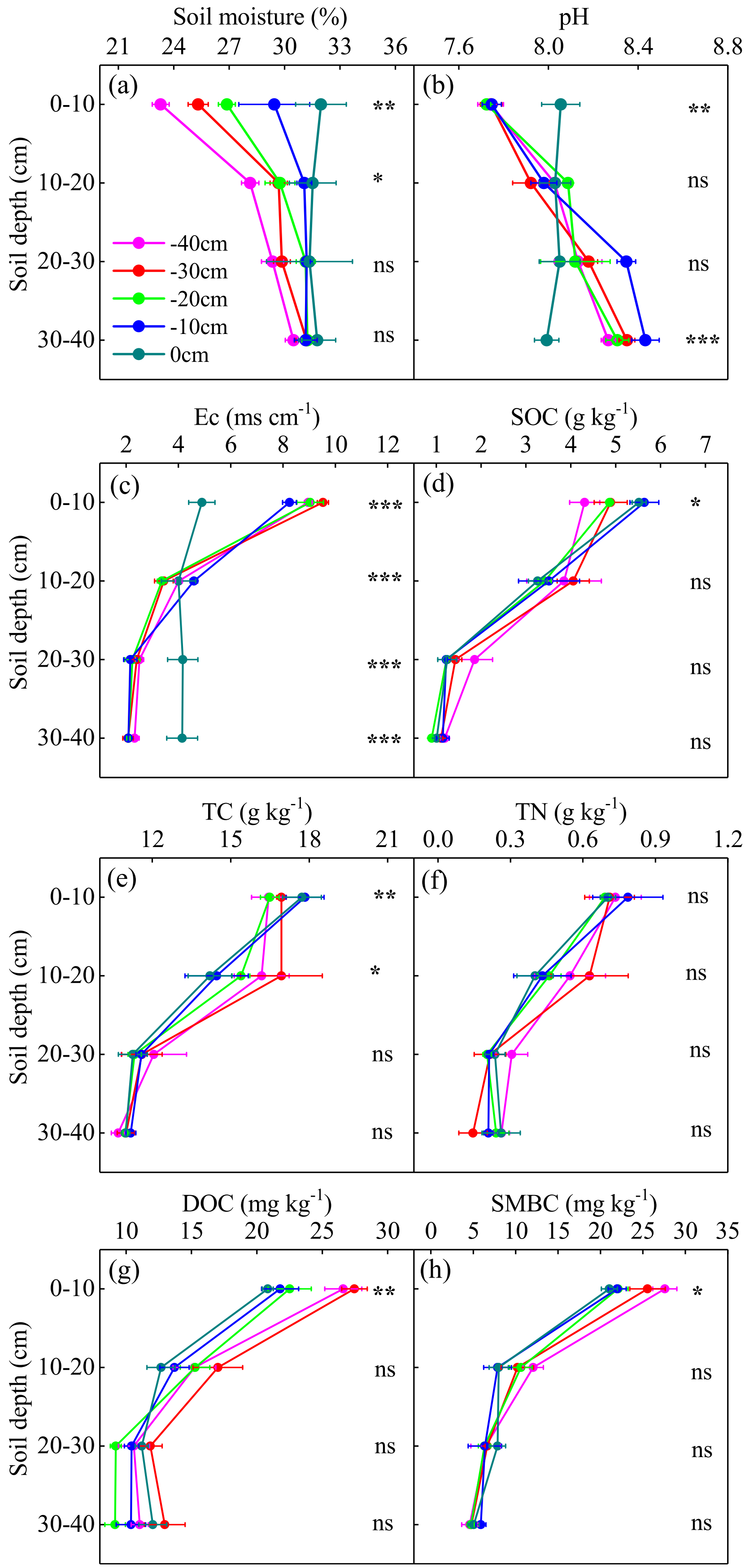

Fig. 4. Soil moisture (a), pH (b), Ec (c), SOC (d), TC (e), TN (f), DOC (g) and SMBC (h) at different soil depths under five water table levels. Means followed by different lettering have significant differences among water table levels at $*P < 0.05$, $**P < 0.01$ and $***P < 0.001$. Bars indicate standard errors (n = 4).

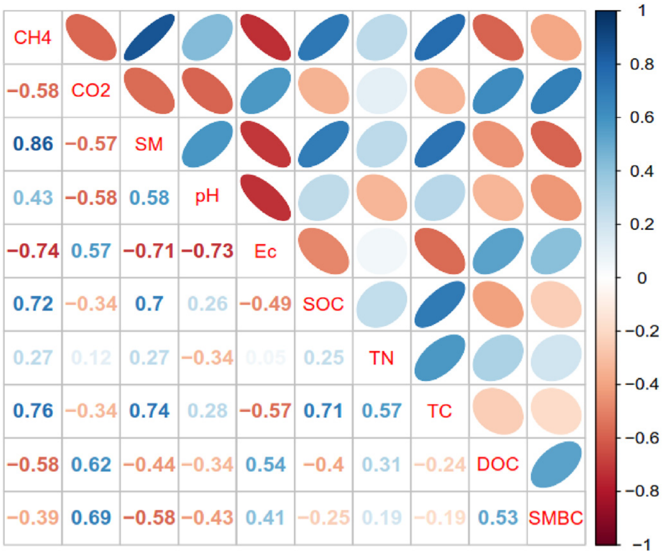

Fig. 5. Pearson correlation coefficients between SM (soil moisture), pH, Ec (electrical conductivity), SOC (soil organic carbon), TN (total nitrogen), TC (total carbon), DOC (dissolved organic carbon) and SMBC (soil microbial biomass carbon) and cumulative CH_4 emissions (CH_4) and cumulative CO_2 emissions (CO_2) at 0–10 cm soil depth.

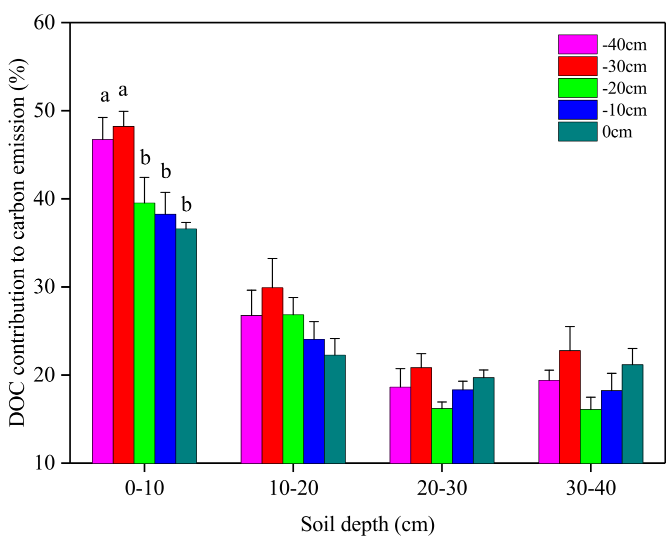

Fig. 6. Relative dissolved organic carbon (DOC) contribution to total carbon emissions under different soil depths and different water table levels. Different letters on the error bars indicate significant differences at $P < 0.05$.

further increased soil CO_2 emissions. This indicated that the DOC variations caused by different water table levels have pronounced effect on soil CO_2 emissions. Although DOC concentrations occupy only a small part of SOC, it has an important effect on blue carbon pool of coastal wetlands (Barrón and Duarte, 2016). Therefore, more attention must be paid to the effect of DOC production and on the mechanisms of CO_2 emissions in a coastal wetland under future climate change. Furthermore, we also found the DOC contributed most to CO_2 emissions in the uppermost soil layer compared to other soil depths, regardless of water table level (Fig. 6), which probably due to the deeper soil layers may contain more recalcitrant carbon than topsoil (Knorr et al., 2005). This result suggested that water table level rise will decelerate the carbon export from surface layer due to the decreased organic matter mineralization and DOC production.

5. Conclusions

Our study demonstrates that water table level rise decreased soil CO_2 emissions, while increased soil CH_4 emissions in salt marsh. In addition, water table level rise caused by sea level rise may contribute to the storage of organic carbon and produces a lower GWPs of CH_4 and CO_2. Moreover, water table level rise significant alter soil moisture, Ec, pH, TC, SOC, DOC and SMBC at the top soil. Among them, soil moisture and salinity are important factors affecting soil CH_4 and CO_2 fluxes in salt marsh. These results emphasize that water table level rise caused by sea level rise will modify the magnitude of soil CH_4 and CO_2 fluxes as well as soil properties. However, plant was not considered in our study. Additionally, methanogenesis and soil hydrolytic enzymes are important for CH_4 and CO_2 fluxes, respectively, but we did not consider. These limited data will increase the uncertainty about the effect of changing water table level on soil carbon fluxes in coastal wetland. Therefore, further research and more continuous data sets of carbon fluxes, coastal wetland hydrology, and other environmental factors are needed to understand how carbon fluxes respond to changing water table level and to assess the direction and magnitude of future carbon changes in salt marsh.

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.

CRediT authorship contribution statement

Mingliang Zhao: Software, Investigation, Writing - original draft, Supervision. **Guangxuan Han:** Conceptualization, Writing - review & editing, Funding acquisition. **Juanyong Li:** Investigation. **Weimin Song:** Project administration. **Wendi Qu:** Writing - review & editing. **Franziska Eller:** Writing - review & editing. **Jianping Wang:** Investigation. **Changsheng Jiang:** Writing - review & editing, Funding acquisition.

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