

**Factors influencing
CO₂ and CH₄
emissions**

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Factors influencing CO₂ and CH₄ emissions from coastal wetlands in the Liaohe Delta, Northeast China

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Abstract

Many factors are known to influence greenhouse gas emissions from coastal wetlands, but it is still unclear which factors are most important under field conditions when they are all acting simultaneously. The objective of this study was to assess the effects of water table, salinity, soil temperature and vegetation on CH₄ emissions and ecosystem respiration (R_{eco}) from five coastal wetlands in the Liaohe Delta, northeast China: two *Phragmites australis* (common reed) wetlands, two *Suaeda salsa* (sea blite) marshes and a rice (*Oryza sativa*) paddy. Throughout the growing season, the *Suaeda* wetlands were net CH₄ sinks whereas the *Phragmites* wetlands and the rice paddy were net CH₄ sources emitting 1.2–6.1 gCH₄ m⁻² y⁻¹. The *Phragmites* wetlands emitted the most CH₄ per unit area and the most CH₄ relative to CO₂. The main controlling factors for the CH₄ emissions were water table, temperature and salinity. The CH₄ emission was accelerated at high and constant (or managed) water tables and decreased at water tables below the soil surface. High temperatures enhanced CH₄ emissions, and emission rates were consistently low (< 1 mg CH₄ m⁻² h) at soil temperatures < 18 °C. At salinity levels > 18 ppt, the CH₄ emission rates were always low (< 1 mg CH₄ m⁻² h⁻¹) probably because methanogens were outcompeted by sulphate reducing bacteria. Saline *Phragmites* wetlands can, however, emit significant amounts of CH₄ as CH₄ produced in deep soil layers are transported through the air-space tissue of the plants to the atmosphere. The CH₄ emission from coastal wetlands can be reduced by creating fluctuating water tables, including water tables below the soil surface, as well as by occasional flooding by high-salinity water. The effects of water management schemes on the biological communities in the wetlands must, however, be carefully studied prior to the management in order to avoid undesirable effects on the wetland communities.

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1 Introduction

Wetlands play an important role in the global carbon cycling as they function both as carbon sinks, by storing carbon in soils and vegetation, and as carbon sources, by releasing CO₂ and CH₄ into the atmosphere (Brix et al., 2001; Kayranli et al., 2010; Mitsch et al., 2013; Whiting and Chanton, 2001). Carbon dioxide is fixed by plants and autotrophic microorganisms through photosynthesis and thereby transformed to organic compounds locked away from the atmosphere, a process called carbon sequestration (Kayranli et al., 2010). Wetlands can store organic carbon vectored into the soil for a long time due to the generally slow decomposition rates in anaerobic wetland soils (Mitsch et al., 2013). Decomposition of organic matter does however still take place, both through aerobic and anaerobic processes. Aerobic processes are more efficient and mainly form CO₂ as an end-product, whereas anaerobic decomposition is much slower and, along with CO₂, also produces CH₄. Both gases are known as greenhouse gases, which cause global warming due to their ability to absorb solar radiation (IPCC, 2007). The global warming potential (GWP) of CH₄ is 25 times greater than that of CO₂ on a 100 year time scale (IPCC, 2007) and high emissions of CH₄ can therefore have disproportionately adverse effects on the climate. According to Whalen (2005), wetlands contribute to about 24 % of global CH₄ emissions from all sources, and are the largest natural source of CH₄. Due to the increasing concern of greenhouse gas emissions and global warming, it is important to gain more knowledge about the factors affecting CO₂ and CH₄ emissions in different wetland systems, and understand how the balance might be affected by management actions.

Previous work has shown that environmental factors like water table (Altor and Mitsch, 2008; Couwenberg et al., 2011; Hargreaves and Fowler, 1998), soil temperature (Bridgham and Richardson, 1992; Inglett et al., 2012), salinity (Bartlett et al., 1987; Weston et al., 2011) and vegetation biomass and type (Inglett et al., 2012; Kandel et al., 2013) may have strong controlling effects on greenhouse gas emissions from wetlands. Decomposition of organic matter in wetland soil is strongly dependent on temperature,

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and therefore, both CO₂ and CH₄ emissions from decomposition processes tend to increase with increasing soil temperature (Herbst et al., 2011; Inglett et al., 2012). The optimum temperature for methanogenesis is around 20–30 °C, depending on the community of methanogenic archaea (Svensson, 1984). However, methanogens are strictly anaerobic, and for methanogenesis to take place the redox potential must be as low as –200 mV, and other competing terminal electron acceptors must have been reduced (O₂, NO₃ and SO₄) (Mitsch and Gosselink, 2007). The position of the water table is therefore an important controlling factor on CH₄ emissions, as high water tables lead to oxygen depletion and thus low redox potentials, which favors methanogenesis in the wetland soil (Grunfeld and Brix, 1999). Couwenberg et al. (2011) found that CH₄ emissions in peatlands were practically zero when the water table was below –20 cm, whereas the emissions varied between near zero and 500 kg CH₄ ha^{–1} y^{–1} when the water table was above –20 cm. The more oxidized conditions associated with low water tables favor CH₄ oxidation by aerobic methanotrophic bacteria (Whalen, 2005), as well as aerobic decomposition of organic matter, both processes emitting CO₂. It can therefore be difficult to predict gas emissions under field conditions, as both soil temperatures and water tables may be subject to large seasonal variations.

The presence of vegetation affects CO₂ fluxes primarily by photosynthesizing and by increasing the total ecosystem respiration (Han et al., 2013; Kandel et al., 2013). However, the vegetation may also affect CH₄ emissions. Oxygen released from roots create aerobic microsites in the rhizosphere (Brix, 1994), which favors CH₄ oxidation by aerobic methanotrophs (Grunfeld and Brix, 1999). On the other hand, a high primary production also increases the available carbon substrate for methanogens via biomass decomposition and root exudation and can thus lead to higher CH₄ emissions (Van der Nat and Middelburg, 2000; Whiting and Chanton, 1993). In addition, wetland plants with internal air spaces (aerenchyma) provide an additional gas transport pathway, apart from diffusion and ebullition from the sediment, that can enhance CH₄ emissions (Brix et al., 1996; Henneberg et al., 2012; Sorrell and Boon, 1994). Methane produced in the soil can be transported through the aerenchyma of the plant tissue and bypass the

water column, where it otherwise could have been oxidized by methanotrophs before reaching the atmosphere (Whalen, 2005). Thus, wetland vegetation can both decrease and enhance CH₄ emissions depending on the specific site conditions and type of vegetation.

5 Acute saltwater intrusion to freshwater wetlands has been reported to increase soil respiration and lead to elevated CO₂ emissions (Chambers et al., 2011; Weston et al., 2011). However, coastal wetlands with high salinity usually emit less CH₄ than less saline wetlands (Bartlett et al., 1987; Poffenbarger et al., 2011). This has been explained by the high concentration of sulphate ions (SO₄²⁻) in sea water, and the consequent high activity of sulphate reducing bacteria which outcompete methanogens for organic substrate (Bartlett et al., 1987). Poffenbarger et al. (2011) analyzed CH₄ and salinity data from a number of coastal wetlands and found a threshold salinity level of 18 ppt, above which the wetlands emitted significantly less CH₄ than those with a lower salinity.

15 Although many factors are known to influence CO₂ and CH₄ emissions from coastal wetlands, it is still unclear which factors are most important under field conditions when they are all acting simultaneously. Knowledge of the interactive effects of the factors driving greenhouse gas emissions is a prerequisite to be able to manage wetlands in a way that minimizes greenhouse gas emissions, and to predict the effects of future climate change on greenhouse gas emissions from wetlands. The objectives of this study were (i) to quantify the CH₄ emission and ecosystem respiration in the dominant wetland communities in a coastal wetland ecosystem, (ii) to assess the seasonal variation in CH₄ emission and ecosystem respiration in different plant communities, and (iii) to determine the main controlling factors for CH₄ emission and ecosystem respiration under field conditions.

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2 Materials and methods

2.1 Study sites

The Liaohe Delta is situated in the Liaoning Province in northeast China and comprises a wetland area of around 1280 km² (Li et al., 2012). About 786 km² of that is marsh vegetated by common reed (*Phragmites australis* (Cav.) Trin. Ex Steud). The reed marshes in the Liaohe Delta represent probably the largest reed fields in the world (Brix et al., 2014). The growing conditions for common reed in the delta marshes have been improved since the 1960s by a freshwater irrigation management practice, that has washed away much of the soil salinity, and as a result, led to an expansion of the reed fields and an increase in productivity (Ji et al., 2009). The reed biomass is extensively used for paper production (Ma et al., 1993), and the hydrology is therefore regulated to maximize the biomass yield (Brix et al., 2014). Apart from reed marshes, the main wetland types in the Liaohe Delta are tidal saltmarshes vegetated by *Suaeda salsa* (L.) Pall., III (seablite), and rice paddies planted with *Oryza sativa* L. (Asian rice). The wetlands of the Liaohe Delta are important breeding areas for many endangered bird species, and are designated as a Shuangtaizihoukou (Liaohoukou) National Nature Reserve since 1986 and also listed as a Ramsar site since 2004 (Li et al., 2012). However, the wetlands are adversely affected by the polluted water from the Liaohe River (Zhang et al., 2010) and oil extraction activities, as the Liaohe Delta contains the third largest oil field in China (Zhu et al., 2010).

Five study sites were selected to embrace the main wetland types of the delta. The five study sites included two *Suaeda* marshes, one created and one natural (“Suaeda1” at 40°52′11.09″ N; 121°36′21.72″ E and “Suaeda2” at 40°57′38.62″ N; 121°48′20.03″ E, respectively), two *Phragmites* wetlands for paper production, (“Phrag1” at 40°52′22.34″ N; 121°36′08.89″ E and “Phrag2” at 41°09′33.75″ N; 121°47′42.71″ E) and a rice paddy (“Rice” at 41°10′38.69″ N; 121°41′17.28″ E).

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2.2 Gas sampling and analysis

Gas samples for estimation of CO₂ and CH₄ emission were collected monthly from April to November 2012, using the static chamber method (Livingston and Hutchinson, 1995). Six quadratic metal frames (0.6 m × 0.6 m) were permanently installed in each study site, and wooden boardwalks were built to facilitate access to the frames without disturbing the soil. Small holes were drilled in the sides of the frames just at the ground surface to facilitate water exchange between the inside of the frames and the surrounding wetland between sampling events. These holes were plugged during sampling. At each sampling event, a white plastic chamber (0.55 m × 0.55 m × 0.30 m) was placed over the metal frame and an airtight seal was created by water (about 1 cm deep) within a trough inside the frame. The chambers were modified from past designs deployed in shaded forested wetlands (Krauss and Whitbeck, 2012; Yu et al., 2008) by using aluminum foil to cover all inside walls to block out light and prevent photosynthesis completely. If the vegetation was taller than the chamber, the plants were bent to fit inside the chamber. At Phrag2, however, the plants grew so tall that they had to be cut in June; we limited what we had to cut as much as possible. A small fan was used to mix the air inside the chamber during sampling, and a PVC tube with the outer end placed in water was used to equilibrate the air pressure inside the chamber with the outside air pressure. Gas samples were taken from the chamber through a rubber septum using a 15 mL plastic syringe, and immediately transferred into pre-evacuated 10 mL glass vials with a thick rubber cap and an aluminum lid. The first sample was taken immediately after placing the chamber onto the frame, and four additional samples were taken with 20 min intervals. The temperatures at a soil depth of 10 cm and the air temperature in the chamber were recorded at each sampling time. The gas samples were stored at room temperature for a maximum of one week before analysis. For comparison, the CO₂ flux in each chamber was also measured in situ during separate 1 min incubations on the same day using a portable infrared gas analyzer (LI-COR 8100, Lincoln, NE, USA).

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The concentrations of CO₂ and CH₄ in the gas samples were analyzed in 0.6 mL injections on a TRACE Ultra GC-TCD (Thermo Fischer Scientific Inc., Waltham, MA, USA) at Qingdao Institute of Marine Geology and an Agilent 7890A at the Ocean University of China, respectively. Signals from the GCs were recorded in GC/MSD Chem-
 5 Station Software (Agilent Technologies, Inc., Santa Clara, CA, USA) and the peak areas used to calculate the concentrations of CH₄ and CO₂. Gas emissions in mg CH₄ m⁻² h⁻¹ and mg CO₂ m⁻² h⁻¹ (using the weight of the whole molecules of CH₄ and CO₂, respectively) were determined from the increase in concentration in the chambers over time using linear regression analysis. Regression lines with a coefficient of deter-
 10 mination (R^2) < 0.6 were not included, except in cases where it was obvious that the low R^2 value was due to extremely low gas fluxes (zero or near-zero fluxes). In a few cases, extremely deviant data were excluded. Because of technical problems, no data on CO₂ emissions are available from Phrag1 in April and from Suaeda1 and Suaeda2 in May, and no data on CO₂ and CH₄ emissions in August from Phrag1.

Cumulative CO₂ and CH₄ emissions at each site were calculated as the integral of the mean gas emissions (in mg m⁻² d⁻¹) from the monthly sampling campaigns. As the gas sampling chambers were darkened, CO₂ emissions were assumed to be constant on a daily and nightly basis. And although some studies have found diurnal variations in CH₄ emissions (Käki et al., 2001; Neubauer et al., 2000; Tong et al., 2013), no
 15 consistent pattern has been found. Hence, we assumed that the CH₄ emissions were also constant on a daily basis.

2.3 Environmental parameters

The water table was measured in a piezometer at each study site, and the soil surface level differences among the six plots at each site were used to calculate indi-
 20 vidual water tables for each plot. Water samples for salinity and pH analyses were taken from the piezometer, and measured using a Jenco 6010 microcomputer based pH/mV/temperature portable meter (Jenco Electronics, Ltd., Shanghai, China).

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The aboveground biomass inside the plots was estimated using a non-destructive method. In the *Phragmites* wetlands, the heights of all shoots inside the frames were measured, and 25 shoots encompassing the range of heights in the frames were harvested outside the frames. In the *Suaeda* wetlands, the plant density inside the frames was counted and 20 cm × 20 cm plots outside the frame with a similar plant density were harvested. The plants were dried at 60 °C and weighed, and the biomass inside the plots was calculated from a regression analysis between plant height and dry mass (*Phragmites*) and between plant density and dry mass (*Suaeda*). In the rice paddy, five rice plants outside the frames were harvested, dried and weighed, and the biomass within the frames was estimated based on the number of plants.

Soil core samples were taken to 5 cm depth from the topsoil near each frame using a 5 cm diameter steel cylinder. The samples were dried to constant weight at 60 °C for determination of bulk density and water content. Soil redox potentials were measured using platinum electrodes installed at a depth of 10 cm at least 24 h before measuring. Redox electrodes were referenced against a calomel electrode.

Two soil core samples were collected to 4 cm depth at each site the following year, mixed and analyzed for selected mineral elements and available nutrients. Total N and TC were analyzed on oven-dried subsamples ground to pass a 2 mm sieve, on a Perkin Elmer 2400 Series II CHNS/O elemental analyzer (Perkin Elmer, Inc., Waltham, MA, USA). For determination of Org-C, another set of subsamples were treated with 4 M HCl (Craft, 2007) to remove inorganic carbon before analysis on the same instrument. Available nutrients were extracted by the Mehlich-III method (Mehlich, 1984), using an extraction solution prepared from 22.98 mL concentrated CH₃COOH, 40.0 g NH₄NO₃, 1.12 g NH₄F, 1.68 mL concentrated HNO₃, 0.58 g EDTA and 1600 mL deionized water, diluted to 2 L. Air-dried soil subsamples were ground to pass a 1 mm mesh. 2.5 g of the ground soil were shaken with 25 mL extraction solution on a reciprocating oscillator for 5 min and then centrifuged for 20 min. The supernatant was diluted ten times and analyzed for Ca, Cu, Fe, K, Mg, Mn, P and Zn by ICP-OES (Optima 2000 DV, Perkin Elmer, USA).

2.4 Statistical analysis

The in situ measurements of CO₂ emissions with the IRGA were used in the statistical analyses. Methane emissions and ecosystem respiration (R_{eco}) were analyzed by Site and Time with Plot as a random factor nested within Site, in a repeated-measures setup using the General Linear Model (GLM) procedure of Statgraphics Centurion XVI (Statpoint Technologies, Inc., Warrenton, Virginia, USA). The Bonferroni post-hoc test was used to identify significant differences between different sites at the 5% significance level. Data of CH₄ emissions and R_{eco} were log-transformed to meet the assumption of equal variances, which was checked using Levene's test ($p > 0.05$). Since the dataset included a few negative gas flux values, a constant was added to the fluxes (CH₄ flux +0.6 and R_{eco} +25, respectively) before applying the log-transformations. Data from April, May and August were excluded from the analyses due to missing data at some sites.

Linear mixed effects models (multiple regressions) using R version 3.0.1 (Team, 2013) were used to assess the relations between the measured environmental factors and CO₂ and CH₄ emissions, respectively. The response variables were CO₂ and CH₄ emissions. The fixed effects were plant species (categorical variable), soil temperature (SoilT), water table (WT), aboveground biomass (Biomass) and Salinity (continuous variables). The random effects were Site and Plot. An interaction effect between plant species and aboveground biomass was also included. The effect of each variable or interaction was evaluated by removing the variable/interaction from the original model and using a likelihood ratio chi-square test to test for significant differences at the 5% significance level between the original model and the model excluding the variable/interaction. Data of CO₂ and CH₄ emissions were log-transformed as described before to meet the assumptions of normality and equal variances. The original mixed effects model for CO₂ and CH₄ emissions, respectively, was in the form:

$$\log_{10}(\text{gas flux})_i = \beta_{1_i} \cdot \text{Biomass}_i + \beta_2 \cdot \text{SoilT} + \beta_3 \cdot \text{Salinity} + \beta_4 \cdot \text{WT} + b_1(\text{Site}) + b_2(\text{Plot}) + \varepsilon_i \quad (1)$$

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where β_1 is a coefficient specific for plant species i , β_2 , β_3 and β_4 are coefficients for fixed effects common for all plant species, b_1 and b_2 are coefficients for the random effects and ε_i is the residual error for plant species i .

3 Results

3.1 CH₄ emissions

There were large variations in CH₄ emission rates both among sites and over the season (Fig. 1a) and these differences were statistically significant (Table 1). The highest CH₄ emission rates were found at Phrag2 and at the rice paddy. Peak emissions were 2.5 mg m⁻² h⁻¹ at both sites although the peak values were measured in July at Phrag2 and in August at the rice paddy (Fig. 1a). The highest CH₄ emission rates at Phrag1 (around 0.7 mg m⁻² h⁻¹) were only a fourth of those at Phrag2. At the two *Phragmites* wetlands, the CH₄ emission rates were close to zero in April–May, increased rapidly from June to July, and declined again after August. At the rice paddy, the CH₄ emission rates were near zero in June, low in July (0.25 mg m⁻² h⁻¹), increased very sharply from July to August and thereafter declined. At the *Suaeda* wetlands, the CH₄ emission rates were close to zero throughout the sampling period. Means and ranges of CH₄ emission rates over the whole sampling period, and significant differences ($p < 0.05$) among sites, are shown in Table 3.

The CH₄ emission rates at sites with significant emissions (Phrag1, Phrag2 and Rice) were positively related to both soil temperature and water table (Table 2; Fig. 3). The CH₄ emission rates were less than 1 mg m⁻² h⁻¹ at temperatures below 18 °C and at water tables below the soil surface. The highest CH₄ emission rates were measured at Phrag2 when both the temperature and the water table were high (Fig. 3). The CH₄ emissions decreased significantly (Table 2) with increasing salinity, as CH₄ emission rates were less than 1 mg m⁻² h⁻¹ at salinity levels above 18 ppt (Fig. 4). At the highest salinity levels at Suaeda1 (32–38 ppt), CH₄ emission rates were practically zero.

Cumulative CH₄ emissions over the entire growing season in 2012 were highest at Phrag2 with 6.1 gCH₄m⁻², corresponding to 154 gCO₂-equivalents m⁻²y⁻¹ (Fig. 2, Table 3). These emissions were about 1.5 times higher than the cumulative CH₄ emissions from the rice paddy, and about five times higher than the CH₄ emissions from Phrag1. CH₄ emissions from the *Suaeda* wetlands were negligible.

3.2 Ecosystem respiration (R_{eco})

The measured flux of CO₂ in the darkened chamber is the sum of the flux of CO₂ from the soil and the respiration of the plant tissue inside the chambers. We here refer to this as the ecosystem respiration (R_{eco}). The ecosystem respiration rates varied significantly both among sites and over time (Fig. 1b, Table 1). The highest ecosystem respiration rates at the rice paddy and at Phrag2 (2400 and 2300 mgCO₂m⁻²h⁻¹, respectively) were twice as high as the highest R_{eco} at Phrag1 and three times higher than the R_{eco} at the two *Suaeda* wetlands. At Phrag2, R_{eco} was highest in June and July, whereas at the rice paddy, the R_{eco} was low at this time of the year and highest in August (Fig. 1b). It should, however, be mentioned that the *Phragmites* stems at Phrag2 were cut in June. Hence, the biomass within the chambers from July and onwards was lower than the biomass in the surrounding reed vegetation. Overall, the ecosystem respiration rates were significantly related to plant biomass, soil temperature and salinity (Table 2) whereas water table had no significant effect on R_{eco} ($p > 0.05$).

The cumulative CO₂ emissions, without accounting for photosynthetic CO₂ uptake, varied between 1.7 kgm⁻²y⁻¹ in the *Suaeda* wetlands to 3.3–4.4 kgm⁻²y⁻¹ in the *Phragmites* (Table 3). The cumulative CO₂ emission in rice paddy was in-between this range (3.3 kgm⁻²y⁻¹).

3.3 Water table, soil temperature, biomass and salinity

The water tables varied greatly over the season, particularly at Phrag2 where the water table ranged from –35 to +27 cm, and at Suaeda1 where it ranged from –43 to +15 cm

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(Fig. 1c). At the two *Phragmites* wetlands, the water tables were managed to maximize the yield of *Phragmites* biomass. Hence, the water tables at these sites were above the soil surface during most of the growing season. The water tables at the two *Suaeda* wetlands were fluctuated greatly due to tidal variations, but the water tables were at the time of sampling usually below the soil surface. At the rice paddy, the water table was fairly stable around +10 cm from June to September due to regulation according to agricultural practice.

Soil temperatures at all sites increased from 18–22 °C in May to 23–28 °C in August, and then declined to 0–7 °C in November (Fig. 1d). We do not have temperature data from the months prior to our sampling, but usually the soils in the delta are frozen until April, where after the temperature increases over a few weeks.

The amount of aboveground biomass was basically zero during the first sampling campaign in late April. Thereafter, both *Suaeda* and *Phragmites* grew rapidly reaching aboveground biomasses in June of ~ 800 g dry mass m⁻² for *Suaeda* and ~ 400 g dry mass m⁻² for *Phragmites* before the cutting in June (Fig. 1e). In the rice paddy, the rice plants were planted in late June. Hence the development of biomass in the rice paddies occurred much later than in the natural *Suaeda* and *Phragmites* wetlands.

The salinity at Suaeda1 was 32–39 ppt during most of the sampling period (Fig. 1f). At Suaeda2 the salinity was lower: 10–15 ppt from May to July and then decreasing to 5–6 ppt in August to December. In the *Phragmites* wetlands, the salinities varied between 2 and 19 ppt depending on the water management scheme. The highest salinities were found at Phrag1. At the rice paddy the salinity was constantly low at around 2 ppt.

3.4 Soil characteristics

Soil bulk density varied between 0.93 g cm⁻³ at Phrag2 to 1.50 g cm⁻³ at Suaeda1, and soil water content between 27 % at Suaeda1 and 48 % at Phrag2 (Table 4). The mean redox potential was highest at Suaeda1 (+101 mV) and lowest at Phrag1 (-127 mV). The mean soil water pH was in the interval 7.12–7.70 at all sites

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residues and from root exudates. At the rice paddy, where the second highest CH₄ emissions were measured, the organic content of the soil was low, but the soil C:N ratio was lower (8.4) than the ratios at the other sites, indicating more labile organic matter at this site and therefore the presence of suitable substrates for methanogens.

Both *P. australis* and rice have well developed aerenchyma in roots, rhizomes and stems, which provides them with a high ability to transport gases between the soil and the atmosphere through the plant tissue (Singh and Singh, 1995). When CH₄ is transported from the soil through the air-space tissues of the plants, it bypasses the aerobic zone in the upper part of the soil and the water column, where CH₄ otherwise could have been oxidized by methanotrophic bacteria (Whalen, 2005). Plant-mediated transport has been reported to be the main pathway of CH₄ transport from the soil to the atmosphere and constituting as much as 60–90 % of the CH₄ emissions (Butterbach-Bahl et al., 1997; Huang et al., 2005). In the present study, transport of CH₄ through the air-space tissue of the plants may explain the relatively high CH₄ emission rates from the *Phragmites* wetlands and the rice paddy, while the lack of well-developed aerenchyma in *S. salsa* is consistent with the negligible emission rates from the *Suaeda* wetlands. The aboveground biomass *per se* probably had no effect on the plant-mediated CH₄ emissions, as CH₄ has been shown to be mainly emitted through micropores in the basal parts of rice plants (Nouchi et al., 1990) and through the basal internodes of *P. australis* (Brix, 1989). Also, Henneberg et al. (2012) showed in a manipulation experiment with *Juncus effusus* that aboveground biomass was unimportant for the CH₄ transport through the plants, whereas the removal of fine roots and root tips of coarse roots led to significant reductions in plant-mediated CH₄ transport. Thus, it is likely that the extensive root system of the reeds at Phrag2 contributed to the high CH₄ emission rates at this site.

At salinity levels above 18 ppt the CH₄ emission rates were always lower than 1 mg m⁻² h⁻¹ across all sites (Fig. 4). This is consistent with Poffenbarger et al. (2011) who found a salinity threshold of 18 ppt, above which CH₄ emission rates were significantly lower than at lower salinity levels. The effect of salinity has been explained by the

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with a natural tidal hydrology. At the rice paddy, the soil was continuously flooded from June until September, and the two *Phragmites* wetlands were more or less flooded from June until October, resulting in low redox potentials and relatively high CH₄ emission rates. The soils at the tidally influenced *Suaeda* wetlands were periodically drained and hence partly oxidized inhibiting CH₄ production. When water tables at the *Phragmites* wetlands and the rice paddy were below the soil surface, the CH₄ emission rates were always < 1 mg CH₄ m⁻² h⁻¹ probably because CH₄ produced in deeper soil layers was oxidized in the upper oxic soil layers, reducing the amount of CH₄ reaching the atmosphere. When the water tables approached the soil surface, the CH₄ emission rates increased. This is in agreement with the findings of Zhu et al. (2014), who reported that the seasonal CH₄ emissions from an herbaceous peatland were highly linked to water table fluctuations, and that the water table was the main environmental driver for CH₄ emissions over a single growing season, whereas soil temperature was important on a longer time scale. The important effect of water table on CH₄ emission rates is in agreement with observations in other studies (e.g. Bridgham et al., 2006). However, in the present study both soil water table and temperature were important drivers.

The large seasonal variations in CH₄ emission rates at Phrag1, Phrag2 and Rice were primarily related to the variations in soil temperatures. The highest CH₄ emission rates occurred during the summer months July–September, when temperatures were relatively high. We found an exponential relationship between soil temperature and CH₄ emission rates (Fig. 3) similar to those reported elsewhere (Herbst et al., 2011; Inglett et al., 2012) in accordance with the temperature dependency of the methanogenic bacteria. Furthermore, the amount of labile organic carbon substrates from root exudates can be stimulated by high temperatures as Zhai et al. (2013) found significantly higher root exudation rates from *P. australis* roots at 20 °C than at 10 °C. Also the plant-mediated CH₄ transport may be accelerated at higher temperatures as Hosono and Nouchi (1997) reported that the CH₄ transport through rice plants was twice as high at a rhizosphere temperature of 30 °C as compared to the transport at 15 °C. Thus, the high CH₄ emission rates at both Phrag2 and Rice during the warmest months of the

year were probably due to the high temperature and its stimulating effect on the activity of the methanogenic bacteria, the root exudation rates and the effectivity of the plant-mediated transport. At soil temperatures below 18 °C, which occurred before June and after September, CH₄ emission rates were consistently low (< 1 mg CH₄ m⁻² h⁻¹).

4.2 Ecosystem respiration (R_{eco})

Ecosystem respiration rates were highest in June–July at the *Phragmites* wetlands, June–August at the *Suaeda* wetlands and August at the rice paddy. The differences among the sites can be explained by the differences in soil organic matter and biomass, whereas the variations over time can be explained mainly by soil temperature and to some extent by differences in biomass. The seasonal pattern of ecosystem respiration was closely related to that of soil temperature at all sites, which suggests that soil temperature was the main controlling factor for ecosystem respiration. This in agreement with the findings of other studies (Bridgham and Richardson, 1992; Han et al., 2013; Happell and Chanton, 1993; Kandel et al., 2013; Krauss et al., 2012; Pulliam, 1993). However, biomass respiration also contributed to the ecosystem respiration rates, particularly late in the season when the aboveground biomass was highest. At Phrag1, Suaeda1 and Suaeda2, the seasonal pattern of ecosystem respiration rates correlated to that of the aboveground biomass, indicating that plant respiration may have constituted a large part of the total ecosystem respiration at these sites. This is in agreement with Kandel et al. (2013), who found that plant respiration contributed with about 50 % of the total ecosystem respiration in a cultivated peatland during the summer months, and Xu et al. (2014), who found ten times higher CO₂ emissions from marshes with plant communities than from those without. Also, the difference in ecosystem respiration rates between the two *Suaeda* wetlands corresponded to the differences in *Suaeda* biomass. However, at Phrag2 nearly all CO₂ emissions came from the soil and the belowground biomass, since only short stems were left behind after cutting the reeds in June. At the rice paddy, the ecosystem respiration peaked in August when the aboveground biomass was only about 100 g m⁻². The aboveground rice biomass continued

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to increase after August, but the ecosystem respiration decreased drastically, indicating that soil respiration constituted the main part of ecosystem respiration at the rice paddy.

4.3 Cumulative emissions and GWP

The two *Suaeda* wetlands were net CH₄ sinks whereas the two *Phragmites* wetlands and the rice paddy were net CH₄ sources during April to November 2012. Although the peak CH₄ emission rates at the rice paddy were similar to those at Phrag2, the cumulative CH₄ emission rates from Phrag2 were 1.5 times higher than those from Rice. The cumulative CO₂ emitted from ecosystem respiration followed a similar pattern, with Phrag2 emitting 1.3 times more CO₂ than the rice paddy. However, the CH₄/CO₂ emission ratio based on the cumulative CO₂-equivalents was 1.3 times higher at Phrag2 than at Rice. Thus, on a yearly basis Phrag2 emitted the highest amounts of both CH₄ and CO₂ per unit area, and also the most CH₄ relative to CO₂. Since CO₂ emissions from vegetated ecosystems are counteracted by photosynthetic CO₂ uptake and possibly carbon sequestration, the CO₂ emissions measured as ecosystem respiration does not contribute to the greenhouse effect. However, the CH₄ emissions from wetland ecosystems contribute to the radiative forcing, and therefore CH₄ emission rates should be minimized. It is, however, the balance between carbon sequestrations on the one hand and CH₄ emission on the other hand that determines if a particular wetland can be considered to be a net source or a net sink for radiative greenhouse gases (Mitsch et al., 2013).

4.4 CH₄ emission rates and R_{eco} compared to other studies

The CH₄ emission rates and seasonal pattern at Phrag2 were similar to those measured by Huang et al. (2005) from a reed wetland in the Liaohe delta, where CH₄ emission rates varied from $-0.97 \text{ mgCH}_4 \text{ m}^{-2} \text{ h}^{-1}$ in early May to $2.73 \text{ mgCH}_4 \text{ m}^{-2} \text{ h}^{-1}$ in early September. The average CH₄ emission rate at Phrag2 was within the range

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We aimed at determining which factors are most important under field conditions for controlling CH₄ and CO₂ emissions from coastal wetlands in order to be able to predict the effects of future climate change on greenhouse gas emissions from wetlands and potentially to be able to manage coastal wetlands in a way that minimizes greenhouse gas emissions. Hence, we quantified the CH₄ emissions and ecosystem respiration from April to November 2012 in five coastal wetlands in the Liaohe Delta, northeast China, and determined the main controlling factors for the seasonal variations and the differences among the sites. Over the study period, the two *Suaeda* wetlands were net CH₄ sinks whereas the *Phragmites* wetlands and the rice paddy were net CH₄ sources. The *Phragmites* wetlands had the highest climatic impact as they emitted the most cumulative CH₄ per unit area and the most CH₄ relative to CO₂ compared to the other wetland types. The main controlling factors for the CH₄ emissions were water table, temperature and salinity. Methane emissions are accelerated at high and constant (or managed) water tables and decrease at water tables below the soil surface, or fluctuating water tables. Methane emissions are also accelerated at high temperatures and depressed at high salinity levels. Saline wetlands can, however, emit significant amounts of CH₄ as aerenchymatous wetland plants with deep root systems can transport CH₄ produced in the deeper soil layers to the atmosphere. The ecosystem respiration of the wetland communities depends largely on temperature and the plant aboveground biomass, but soil organic matter content and belowground biomass are also important. It is, however, necessary to quantify not only the ecosystem respiration, but also the balance between the net CO₂ exchange and the CH₄ emission to determine if a particular wetland can be considered to be a net source or a net sink for radiative greenhouse gases. Our study indicates that the CH₄ emissions from coastal wetlands can be reduced by managing the water in the wetland in a way that creates fluctuating water tables, including water tables below the soil surface, as well as by occasional flooding by high-salinity water. However, the effects of potential water management schemes

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Table 1. Results from repeated-measures ANOVAs with the response variables CH₄-flux and R_{eco} , respectively, the fixed factors Site and Time and their interaction, and the random factor Plot. Gas fluxes were measured during April–November 2012 from six plots at two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy in the Liaohe Delta, northeast China. All measurements from April, May and August were excluded from the analysis due to missing data from some sites.

Response variable	Factor	df	<i>F</i> ratio	<i>p</i>
CH ₄ -flux	Site	4	19.9	< 0.001
	Time	4	7.5	< 0.001
	Site × Time	16	5.9	< 0.001
	Plot (random factor)	25	2.0	0.007
R_{eco}	Site	4	23.7	< 0.001
	Time	4	379.4	< 0.001
	Site × Time	16	55.7	< 0.001
	Plot (random factor)	25	1.9	0.010

df: degrees of freedom.

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Table 2. Results from linear mixed effects models, with CH₄ emission rate and ecosystem respiration rate (R_{eco}) as response variables, and the fixed effects Plant species, Biomass, Soil temperature, Water table and Salinity. Shown are the coefficients of the fixed effects to be included in Eq. (1), standard errors of the means and p values.

Response variable	Predictor	Coefficient	SE	p
CH ₄ emission rate	Water table	0.0054	0.0014	< 0.001
	Soil temperature	0.0017	0.0023	< 0.001
	Salinity	-0.0023	0.0030	< 0.001
CH ₄ emission rate ^a	Water table	0.0071	0.0019	< 0.001
	Soil temperature	0.0074	0.0034	< 0.001
R_{eco}	Suaeda Biomass	-1.93 10 ⁻⁵	3.1 10 ⁻⁴	0.003
	Phrag Biomass	7.1 10 ⁻⁴	2.5 10 ⁻⁴	0.003
	Rice Biomass	9.2 10 ⁻⁴	3.0 10 ⁻⁴	0.003
	Soil temperature	0.057	0.0042	< 0.001
	Salinity	0.0095	0.0044	0.049

^a Only sites with CH₄ emissions > 0 included (Phrag1, Phrag2 and Rice).

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Table 4. Physical/chemical topsoil characteristics (0–5 cm depth for bulk density, water content and redox potential; else 0–4 cm depth) at the five wetland sites (two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy) in the Liaohe Delta, northeast China. Data was collected in 2013 by Siyuan Ye et al. (personal communication, 2014).

	Suaeda1	Suaeda2	Phrag1	Phrag2	Rice
Bulk density (g cm ⁻³)	1.50	1.20	1.07	0.93	1.36
Water content (% of FW)	27	37	41	48	30
Redox potential (mV)	101	24	-127	-91	-82
TN (% of DW)	0.08	0.07	0.17	1.02	0.10
TC (% of DW)	0.95	0.83	1.81	12.59	0.88
Org-C (% of DW)	0.53	0.69	1.67	11.81	0.69
C : N ratio	12.4	12.0	9.8	12.3	8.4
Ca (μg g ⁻¹)	6735	4215	3817	2103	2239
Cu (μg g ⁻¹)	9.96	6.78	9.11	7.18	3.44
Fe (μg g ⁻¹)	282	434	396	343	343
K (μg g ⁻¹)	849	576	598	892	109
Mg (μg g ⁻¹)	2043	1120	1395	1687	216
Mn (μg g ⁻¹)	291	368	308	104	78
P (μg g ⁻¹)	19.7	27.8	9.9	46.7	37.0
Zn (μg g ⁻¹)	9.6	11.1	17.8	30.8	8.2

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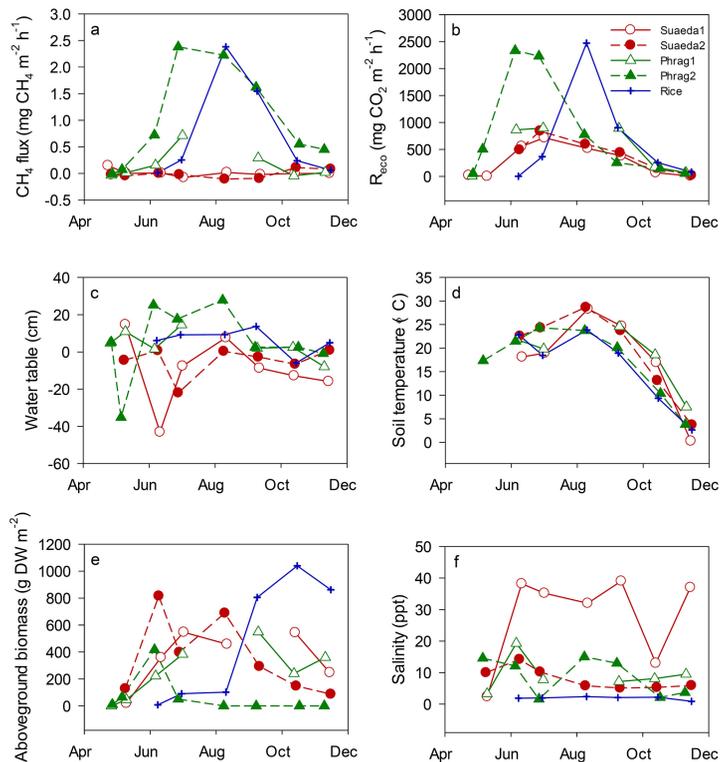



Figure 1. Seasonal variation in (a) CH₄ emission rates, (b) ecosystem respiration, (c) water table, (d) soil temperature, (e) aboveground dry biomass and (f) salinity in two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy during 2012 in the Liaohe Delta, northeast China. Plotted values are the averages for six plots at each site. Data from Phrag2 is missing in August because it was not possible to sample due to extreme flooding. Aboveground biomass data from Suaeda1 is missing in September due to technical issues.

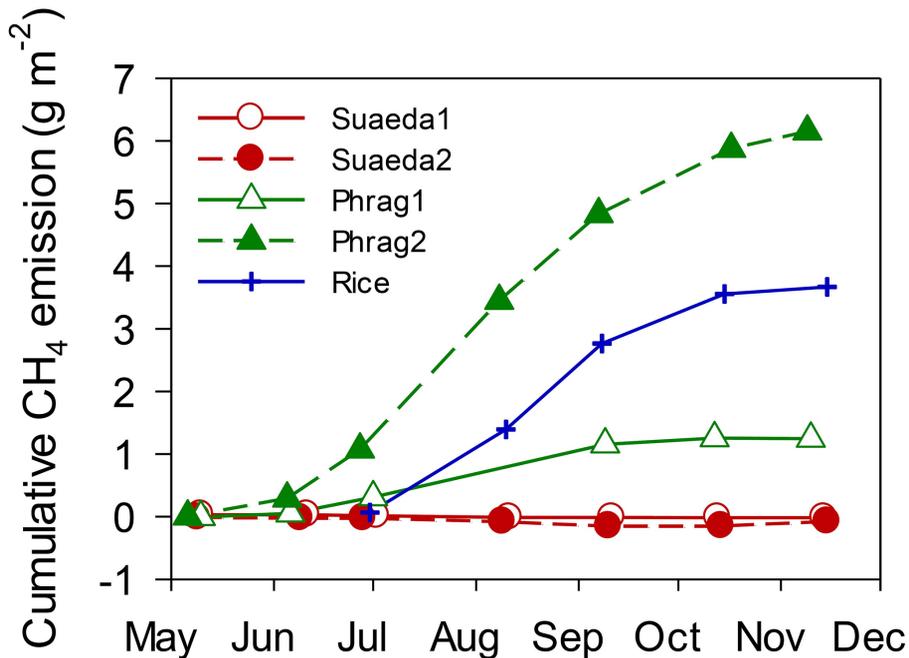


Figure 2. Cumulative CH₄ emissions during the growing season 2012 from two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy during 2012 in the Liaohe Delta, northeast China. The points represent integrals of the monthly mean values from six plots at each site. Measurements are missing from Phrag1 in August due to flooding.

Factors influencing CO₂ and CH₄ emissions

L. Olsson et al.

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