



# Factors influencing CO<sub>2</sub> and CH<sub>4</sub> 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 CH<sub>4</sub> emissions and ecosystem respiration ( $R_{\text{eco}}$ ) 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 CH<sub>4</sub> sinks whereas the *Phragmites* wetlands and the rice paddy were net CH<sub>4</sub> sources emitting 1.2–6.1 g CH<sub>4</sub> m<sup>-2</sup> yr<sup>-1</sup>. The *Phragmites* wetlands emitted the most CH<sub>4</sub> per unit area and the most CH<sub>4</sub> relative to CO<sub>2</sub>. The main controlling factors for the CH<sub>4</sub> emissions were water table, temperature, soil organic carbon and salinity. The CH<sub>4</sub> emission was accelerated at high and constant (or managed) water tables and decreased at water tables below the soil surface. High temperatures enhanced CH<sub>4</sub> emissions, and emission rates were consistently low (< 1 mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup>) at soil temperatures < 18 °C. At salinity levels > 18 ppt, the CH<sub>4</sub> emission rates were always low (< 1 mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup>) probably because methanogens were out-competed by sulphate-reducing bacteria. Saline *Phragmites* wetlands can, however, emit significant amounts of CH<sub>4</sub> as CH<sub>4</sub> produced in deep soil layers are transported through the air-space tissue of the plants to the atmosphere. The CH<sub>4</sub> 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.

## 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 CO<sub>2</sub> and CH<sub>4</sub> 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 CO<sub>2</sub> as an end-product, whereas anaerobic decomposition is much slower and, along with CO<sub>2</sub>, also produces CH<sub>4</sub>. Both gasses are known as greenhouse gasses, which cause global warming due to their ability to absorb solar radiation (IPCC, 2007). The global warming potential (GWP) of CH<sub>4</sub> is 25 times greater than that of CO<sub>2</sub> on a 100-year timescale (IPCC, 2007) and high emissions of CH<sub>4</sub> can therefore have disproportionately ad-

verse effects on the climate. According to Whalen (2005), wetlands contribute to about 24 % of global CH<sub>4</sub> emissions from all sources, and are the largest natural source of CH<sub>4</sub>. Due to the increasing concern of greenhouse gas emissions and global warming, it is important to gain more knowledge about the factors affecting CO<sub>2</sub> and CH<sub>4</sub> 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 (Bridgman 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, and therefore, both CO<sub>2</sub> and CH<sub>4</sub> 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 (O<sub>2</sub>, NO<sub>3</sub> and SO<sub>4</sub>; Mitsch and Gosselink, 2007). The position of the water table is therefore an important controlling factor on CH<sub>4</sub> 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 CH<sub>4</sub> emissions in peatlands were practically zero when the water table was below –20 cm, whereas the emissions varied between near zero and 500 kg CH<sub>4</sub> ha<sup>–1</sup> yr<sup>–1</sup> when the water table was above –20 cm. The more oxidized conditions associated with low water tables favour CH<sub>4</sub> oxidation by aerobic methanotrophic bacteria (Whalen, 2005), as well as aerobic decomposition of organic matter, both processes emitting CO<sub>2</sub>. 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 CO<sub>2</sub> 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 CH<sub>4</sub> emissions. Oxygen released from roots create aerobic microsites in the rhizosphere (Brix, 1994), which favors CH<sub>4</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> emissions depending on the specific site conditions and type of vegetation.

Acute saltwater intrusion to freshwater wetlands has been reported to increase soil respiration and lead to elevated CO<sub>2</sub> emissions (Chambers et al., 2011; Weston et al., 2011). However, coastal wetlands with high salinity usually emit less CH<sub>4</sub> than less saline wetlands (Bartlett et al., 1987; Poffenbarger et al., 2011). This has been explained by the high concentration of sulphate ions (SO<sub>4</sub><sup>2–</sup>) in sea water, and the consequent high activity of sulphate-reducing bacteria which out-compete methanogens for organic substrate (Bartlett et al., 1987). Poffenbarger et al. (2011) analyzed CH<sub>4</sub> 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 CH<sub>4</sub> than those with a lower salinity.

Although many factors are known to influence CO<sub>2</sub> and CH<sub>4</sub> 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 being 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 CH<sub>4</sub> emission and ecosystem respiration in the dominant wetland communities in a coastal wetland ecosystem, (ii) to assess the seasonal variation in CH<sub>4</sub> emission and ecosystem respiration in different plant communities, and (iii) to determine the main controlling factors for CH<sub>4</sub> emission and ecosystem respiration under field conditions.

## 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 km<sup>2</sup> (Li et al., 2012). About 786 km<sup>2</sup> 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 salt marshes 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 Shuangtaizihekou (Liaohekou) 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).

## 2.2 Gas sampling and analysis

Gas samples for estimation of CO<sub>2</sub> and CH<sub>4</sub> emissions were collected monthly from April to November 2012, using the static chamber method (Livingston and Hutchinson, 1995). Six quadratic metal frames (0.6 × 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 × 0.55 × 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 aluminium foil to cover all inside walls to block out light and prevent photosynthesis completely and to minimize temperature changes. Transparent and opaque chambers have been shown to provide similar CH<sub>4</sub> flux estimates (Minke et al., 2014). 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 immedi-

ately transferred into pre-evacuated 10 mL glass vials with a thick rubber cap and an aluminium 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 1 week before analysis. For comparison, the CO<sub>2</sub> 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).

The concentrations of CO<sub>2</sub> and CH<sub>4</sub> 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 ChemStation Software (Agilent Technologies, Inc., Santa Clara, CA, USA) and the peak areas used to calculate the concentrations of CH<sub>4</sub> and CO<sub>2</sub>. Gas emissions in mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup> and mg CO<sub>2</sub> m<sup>-2</sup> h<sup>-1</sup> (using the weight of the whole molecules of CH<sub>4</sub> and CO<sub>2</sub>, respectively) were determined from the increase in concentration in the chambers over time using linear regression analysis. Regression lines with a coefficient of determination ( $R^2$ ) < 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 CO<sub>2</sub> emissions are available from *Phrag1* in April and from *Suaeda1* and *Suaeda2* in May, and no data on CO<sub>2</sub> and CH<sub>4</sub> emissions in August from *Phrag1*.

Cumulative CO<sub>2</sub> and CH<sub>4</sub> emissions at each site were calculated as the integral of the mean gas emissions (in mg m<sup>-2</sup> d<sup>-1</sup>) from the monthly sampling campaigns. As the gas sampling chambers were darkened, CO<sub>2</sub> emissions were assumed to be constant on a daily and nightly basis. And although some studies have found diurnal variations in CH<sub>4</sub> emissions (Käki et al., 2001; Neubauer et al., 2000; Tong et al., 2013), no consistent pattern has been found. Hence, we assumed that the CH<sub>4</sub> 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 individual 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 metre (Jenco Electronics, Ltd., Shanghai, China).

The above-ground 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 × 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 (Eh) 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 at 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 and were 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 was treated with 4M 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 CH<sub>3</sub>COOH, 40.0 g NH<sub>4</sub>NO<sub>3</sub>, 1.12 g NH<sub>4</sub>F, 1.68 mL concentrated HNO<sub>3</sub>, 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 was 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 CO<sub>2</sub> emissions with the IRGA were used in the statistical analyses. Methane emissions and ecosystem respiration ( $R_{\text{eco}}$ ) 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 CH<sub>4</sub> emissions and  $R_{\text{eco}}$  were log-transformed to meet the assumption of equal variances, which was checked using Levene's test ( $p > 0.05$ ). Since the data set included a few negative gas flux values, a constant was added to the fluxes (CH<sub>4</sub> flux+0.6 and  $R_{\text{eco}}$ +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 CO<sub>2</sub> and CH<sub>4</sub> emissions, respectively. The response variables were CO<sub>2</sub> and CH<sub>4</sub> emissions. The fixed effects were plant species (categorical variable), soil temperature (SoilT), water table (WT), above-ground biomass (Biomass) and Salinity (continuous variables). The random effects were Site and Plot. An interaction effect between plant species and above-ground 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 CO<sub>2</sub> and CH<sub>4</sub> emissions were log-transformed as described before to meet the assumptions of normality and equal variances. The original mixed effects model for CO<sub>2</sub> and CH<sub>4</sub> emissions, respectively, was in the form:

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

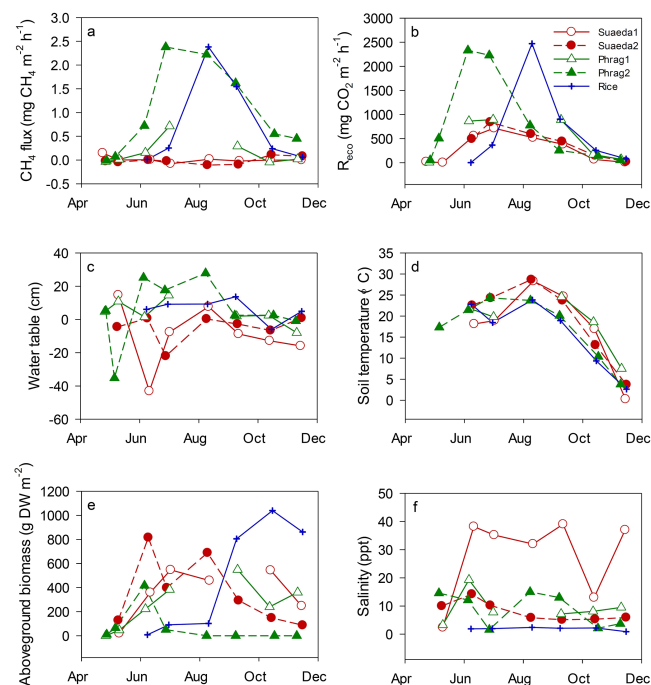
where  $\beta_1$  is a coefficient specific for plant species  $i$ ,  $\beta_2$ ,  $\beta_3$  and  $\beta_4$  are coefficients for fixed effects common for all plant species,  $b_1$  and  $b_2$  are coefficients for the random effects and  $\varepsilon_i$  is the residual error for plant species  $i$ .

## 3 Results

### 3.1 Environmental parameters

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 (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 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, whereafter the temperature increases over a few weeks.



**Figure 1.** Seasonal variation in (a) CH<sub>4</sub> emission rates, (b) ecosystem respiration, (c) water table, (d) soil temperature, (e) above-ground 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 are missing in August because it was not possible to sample due to extreme flooding. Above-ground biomass data from Suaeda1 are missing in September due to technical issues.

The amount of above-ground biomass was basically zero during the first sampling campaign in late April. Thereafter, both *Suaeda* and *Phragmites* grew rapidly reaching above-ground biomasses in June of  $\sim 800$  g dry mass  $m^{-2}$  for *Suaeda* and  $\sim 400$  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.

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 1). 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.

All topsoils consisted largely of fine silt and clay and had a low content of organic matter (Org-C < 2 % of the dry matter). However, the contents of organic carbon (12 %) and nitrogen (1 %) were markedly higher at Phrag2 than at the other sites (Table 1). At Phrag1, the contents of organic carbon (1.8 %) and nitrogen (0.17 %) were 2–3 times higher than at the *Suaeda* sites and the rice paddy. Differences in other analyzed mineral elements were less pronounced and probably reflected the predominantly mineral composition of the soils, except for the concentration of P which was higher at Phrag2 and the rice paddy than at the other sites.

### 3.2 CH<sub>4</sub> emissions

There were large variations in CH<sub>4</sub> emission rates both among sites and over the season (Fig. 1a) and these differences were statistically significant (Table 2). The highest CH<sub>4</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> emission rates were close to zero throughout the sampling period. Means and ranges of CH<sub>4</sub> emission rates over the whole sampling period, and significant differences ( $p < 0.05$ ) among sites, are shown in Table 4.

The CH<sub>4</sub> emission rates at sites with significant emissions (Phrag1, Phrag2 and Rice) were positively related to both soil temperature and water table (Table 3; Fig. 3). The CH<sub>4</sub> emission rates were less than  $1$  mg  $m^{-2}$   $h^{-1}$  at temperatures below  $18^{\circ}C$  and at water tables below the soil surface. The highest CH<sub>4</sub> emission rates were measured at Phrag2 when both the temperature and the water table were high (Fig. 3). The CH<sub>4</sub> emissions decreased significantly (Table 3) with increasing salinity, as CH<sub>4</sub> 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), CH<sub>4</sub> emission rates were practically zero.

Cumulative CH<sub>4</sub> emissions over the entire growing season in 2012 were highest at Phrag2 with  $6.1$  g CH<sub>4</sub>  $m^{-2}$   $yr^{-1}$ , corresponding to  $154$  g CO<sub>2</sub>-equivalents  $m^{-2}$   $yr^{-1}$  (Fig. 2, Table 4). These emissions were about 1.5 times higher than the cumulative CH<sub>4</sub> emissions from the rice paddy, and about five times higher than the CH<sub>4</sub> emissions from Phrag1. CH<sub>4</sub> emissions from the *Suaeda* wetlands were negligible.

**Table 1.** 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 were collected in 2013 by Siyuan Ye.

	Suaeda1	Suaeda2	Phrag1	Phrag2	Rice
Bulk density (g cm <sup>-3</sup> )	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 <sup>-1</sup> )	6735	4215	3817	2103	2239
Cu (μg g <sup>-1</sup> )	9.96	6.78	9.11	7.18	3.44
Fe (μg g <sup>-1</sup> )	282	434	396	343	343
K (μg g <sup>-1</sup> )	849	576	598	892	109
Mg (μg g <sup>-1</sup> )	2043	1120	1395	1687	216
Mn (μg g <sup>-1</sup> )	291	368	308	104	78
P (μg g <sup>-1</sup> )	19.7	27.8	9.9	46.7	37.0
Zn (μg g <sup>-1</sup> )	9.6	11.1	17.8	30.8	8.2

**Table 2.** Results from repeated-measures ANOVAs with the response variables CH<sub>4</sub>-flux and  $R_{\text{eco}}$ , 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	F-ratio	p
CH <sub>4</sub> -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
$R_{\text{eco}}$	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.

### 3.3 Ecosystem respiration ( $R_{\text{eco}}$ )

The measured flux of CO<sub>2</sub> in the darkened chamber is the sum of the flux of CO<sub>2</sub> from the soil and the respiration of the plant tissue inside the chambers. We refer to this here as the ecosystem respiration ( $R_{\text{eco}}$ ). The ecosystem respiration rates varied significantly both among sites and over time (Fig. 1b, Table 2). The highest ecosystem respiration rates at the rice paddy and at Phrag2 (2400 and 2300 mg CO<sub>2</sub> m<sup>-2</sup> h<sup>-1</sup>, respectively) were twice as high as the highest  $R_{\text{eco}}$  at Phrag1 and three times higher than the  $R_{\text{eco}}$  at the two *Suaeda* wetlands. At Phrag2,  $R_{\text{eco}}$  was highest in June and July, whereas at the rice paddy, the  $R_{\text{eco}}$  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 3) whereas water table had no significant effect on  $R_{\text{eco}}$  ( $p > 0.05$ ).

The cumulative CO<sub>2</sub> emissions, without accounting for photosynthetic CO<sub>2</sub> uptake, varied between 1.7 kg m<sup>-2</sup> yr<sup>-1</sup> in the *Suaeda* wetlands to 3.0–4.4 kg m<sup>-2</sup> yr<sup>-1</sup> in the *Phragmites* (Table 4). The cumulative CO<sub>2</sub> emission in the rice paddy was in-between this range (3.3 kg m<sup>-2</sup> yr<sup>-1</sup>).

## 4 Discussion

### 4.1 CH<sub>4</sub> emissions

Over one growing season in 2012, the two *Phragmites* wetlands emitted on average 0.15 and 1.01 mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup> (Phrag1 and Phrag2, respectively) and the rice paddy 0.75 mg m<sup>-2</sup> h<sup>-1</sup>, whereas the emissions from the two *Suaeda* wetlands were negligible. The large differences in CH<sub>4</sub> emission rates among the five sites can be explained by the differences in soil organic matter, salinity and water tables, and, to some extent, vegetation type. For methanogenesis to take place there must be a sufficient amount of labile organic substrate available (Mah et al., 1977), such as dead plant material from the previous growing season and root exudates from the standing vegetation (Mann and Wetzel, 1996; Zhai et al., 2013). Previous studies have reported increasing CH<sub>4</sub> emission rates with increasing content of soil organic



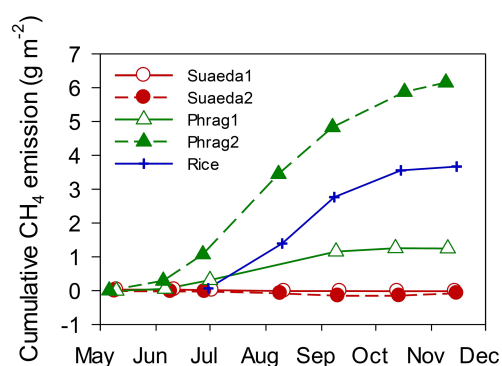
**Table 3.** Results from linear mixed-effects models, with CH<sub>4</sub> emission rate and ecosystem respiration rate ( $R_{\text{eco}}$ ) as response variables, and the fixed effects plant species, biomass, soil temperature, water table and salinity. Shown are the coefficients of the fixed effects to be included in Eq. (1), standard errors of the means and  $p$  values.

Response variable	Predictor	Coefficient	SE	$p$
CH <sub>4</sub> 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 <sub>4</sub> emission rate*	Water table	0.0071	0.0019	<0.001
	Soil temperature	0.0074	0.0034	<0.001
$R_{\text{eco}}$	Suaeda*Biomass	$-1.93 \cdot 10^{-5}$	$3.1 \cdot 10^{-4}$	0.003
	Phrag*Biomass	$7.1 \cdot 10^{-4}$	$2.5 \cdot 10^{-4}$	0.003
	Rice*Biomass	$9.2 \cdot 10^{-4}$	$3.0 \cdot 10^{-4}$	0.003
	Soil temperature	0.057	0.0042	<0.001
	Salinity	0.0095	0.0044	0.049

\* Only sites with CH<sub>4</sub> emissions >0 included (Phrag1, Phrag2 and Rice).

matter in different types of wetlands (Le Mer and Roger, 2001; Picek et al., 2007; Serrano-Silva et al., 2014; Sha et al., 2011; Tanner et al., 1997). At Phrag2, where CH<sub>4</sub> emission rates were significantly higher than at the other sites, there was a many-fold higher content of organic carbon and nitrogen in the soil compared to the soils at the other sites, and the reeds at Phrag2 had a very dense root system in the upper soil layers. Thus, the reason for the high CH<sub>4</sub> emission rates at Phrag2 was most likely the higher content of organic substrate for methanogenesis, originating from dead plant residues and from root exudates. At the rice paddy, where the second highest CH<sub>4</sub> 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 probably resulting from different plant inputs into the soil. A lower C:N ratio of the organic matter in the soil may increase organic matter lability by decreasing nitrogen limitation for decomposers (Hodgkins et al., 2014). However, the fact that the rice paddy was constantly flooded throughout the growing season probably also stimulated methanogenesis and CH<sub>4</sub> emission.

Both *P. australis* and rice have well-developed aerenchyma in roots, rhizomes and stems, which provides them with a high ability to transport gasses between the soil and the atmosphere through the plant tissue (Brix et al., 1996; Singh and Singh, 1995). When CH<sub>4</sub> 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 CH<sub>4</sub> otherwise could have been oxidized by methanotrophic bacteria (Whalen, 2005). Plant-mediated transport has been reported to be the main pathway of CH<sub>4</sub> transport from the soil to the atmosphere and constituting as much as 60–90% of the CH<sub>4</sub> emissions (Butterbach-Bahl et al., 1997; Huang et al., 2005). In the present study, transport of CH<sub>4</sub> through the air-space tissue of the plants may explain the relatively high CH<sub>4</sub> emission



**Figure 2.** Cumulative CH<sub>4</sub> 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.

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 above-ground biomass per se probably had no effect on the plant-mediated CH<sub>4</sub> emissions, as CH<sub>4</sub> 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 above-ground biomass was unimportant for the CH<sub>4</sub> transport through the plants, whereas the removal of fine roots and root tips of coarse roots led to significant reductions in plant-mediated CH<sub>4</sub> transport. Thus, it is likely that the extensive root system of

**Table 4.** Mean CH<sub>4</sub> emission and ecosystem respiration rates ( $R_{\text{eco}}$ ) with ranges in parentheses, and cumulative CO<sub>2</sub> equivalents from CH<sub>4</sub> and CO<sub>2</sub> emissions, respectively, from two *Phragmites australis* wetlands and one rice paddy during April–November 2012 in the Liaohe Delta, Northeast China. CH<sub>4</sub> fluxes are converted to CO<sub>2</sub>-equivalents using a factor of 25. Superscript letters represent significant differences ( $p < 0.05$ ) among sites.

Site	CH <sub>4</sub> emission rates (mg m <sup>-2</sup> h <sup>-1</sup> )	$R_{\text{eco}}$ (mg CO <sub>2</sub> m <sup>-2</sup> h <sup>-1</sup> )	Cumulative CO <sub>2</sub> -equivalents	
			CH <sub>4</sub> (g CO <sub>2</sub> -eqv m <sup>-2</sup> yr <sup>-1</sup> )	CO <sub>2</sub> (g CO <sub>2</sub> -eqv m <sup>-2</sup> yr <sup>-1</sup> )
Suaeda1	0.01 (−0.31–0.44) <sup>a</sup>	278 (−3.6–814) <sup>ab</sup>	−0.4	1671
Suaeda2	−0.01 (−0.50–0.42) <sup>a</sup>	423 (4.6–954) <sup>b</sup>	−1.9	1730
Phrag1*	0.15 (−0.31–1.48) <sup>ab</sup>	484 (−14.8–1300) <sup>c</sup>	31.1	2963
Phrag2	1.01 (−0.28–6.38) <sup>c</sup>	811 (27.4–3357) <sup>c</sup>	153.7	4443
Rice	0.75 (−0.27–4.63) <sup>b</sup>	532 (−0.2–3181) <sup>a</sup>	91.6	3337

\* No data from August.

the reeds at Phrag2 contributed to the high CH<sub>4</sub> emission rates at this site.

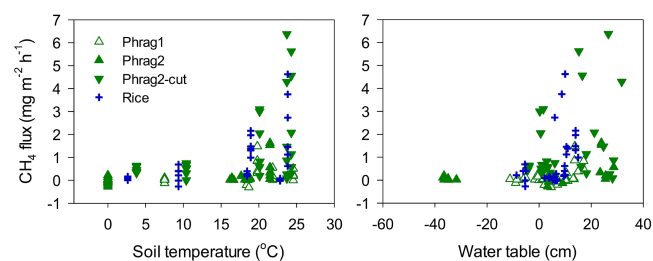
At salinity levels above 18 ppt the CH<sub>4</sub> emission rates were always lower than 1 mg m<sup>-2</sup> h<sup>-1</sup> across all sites (Fig. 4). This is consistent with Poffenbarger et al. (2011) who found a salinity threshold of 18 ppt, above which CH<sub>4</sub> emission rates were significantly lower than at lower salinity levels. The effect of salinity has been explained by the high concentrations of SO<sub>4</sub><sup>2-</sup> in seawater, which inhibit CH<sub>4</sub> production due to competition from sulphate reducing bacteria (Bartlett et al., 1987; D'Angelo and Reddy, 1999). Thus, the lack of CH<sub>4</sub> emissions at the *Suaeda* sites is most likely an effect of the high salinity, particularly at the Suaeda1 site where salinities were up to 35 ppt. The salinity was, however, significantly lower at the Suaeda2 site with salinities of 5–15 ppt, and yet there were no CH<sub>4</sub> emissions as SO<sub>4</sub><sup>2-</sup> concentrations were still high enough to inhibit methanogenesis. At Phrag2, on the other hand, CH<sub>4</sub> emission rates were high although the water salinity was occasionally as high as 15 ppt. These seemingly contradictory results can be explained by the fact that a high salinity in the water mainly affects the upper soil layers, but not necessarily the deeper layers. Therefore, methanogens may be out-competed by sulphate reducing bacteria in the upper layers of the soil, but CH<sub>4</sub> can still be produced in the deeper soil layers where all SO<sub>4</sub><sup>2-</sup> have been reduced. The roots of *P. australis* grow to a soil depth of at least 40–60 cm, and CH<sub>4</sub> can therefore be transported from the deeper anoxic zone through the air-space tissue of the plants to the atmosphere. Thus, the relatively high salinity at Phrag2 probably inhibited methanogenesis in the upper soil layers, but the CH<sub>4</sub> produced in the deeper soil layers were still transported to the atmosphere through the plants. At the *Suaeda* wetlands, the generally low and fluctuating water tables indicate that the anaerobic zone where methanogenesis can take place was at a deeper soil depth than at the *Phragmites* wetlands. The roots of *S. salsa* lack aerenchyma and are generally restricted to the upper 20 cm of the soil, and are therefore ineffective conduits for CH<sub>4</sub> from the deeper

soil layers to the atmosphere. Thus, although salinity levels at Suaeda2 were not always high, any CH<sub>4</sub> that may have been produced in the soil did not reach the atmosphere because of CH<sub>4</sub> oxidation in the upper soil layer. At the rice paddy, the low salinity of around 2 ppt seemingly had no inhibitory effect on the CH<sub>4</sub> production and emission.

The water table is an important parameter affecting the CH<sub>4</sub> emission rate. The highest CH<sub>4</sub> emissions occurred at the three sites where the water exchange and water table were managed to maximize the reed biomass (Phrag1, Phrag2) and crop yield (Rice) whereas very low CH<sub>4</sub> emission rates were found at the two *Suaeda* wetlands 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 CH<sub>4</sub> emission rates. The soils at the tidally influenced *Suaeda* wetlands were periodically drained and hence partly oxidized inhibiting CH<sub>4</sub> production. When water tables at the *Phragmites* wetlands and the rice paddy were below the soil surface, the CH<sub>4</sub> emission rates were always < 1 mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup> probably because CH<sub>4</sub> produced in deeper soil layers was oxidized in the upper oxic soil layers, reducing the amount of CH<sub>4</sub> reaching the atmosphere. When the water tables approached the soil surface, the CH<sub>4</sub> emission rates increased. This is in agreement with the findings of Zhu et al. (2014), who reported that the seasonal CH<sub>4</sub> emissions from an herbaceous peatland were highly linked to water table fluctuations, and that the water table was the main environmental driver for CH<sub>4</sub> emissions over a single growing season, whereas soil temperature was important on a longer timescale. The important effect of water table on CH<sub>4</sub> emission rates is in agreement with observations in other studies (e.g. Bridgham et al., 2006; Couwenberg et al., 2011; Le Mer and Roger, 2001; Serrano-Silva et al., 2014). However, in the present study both soil water table and temperature were important drivers.

The large seasonal variations in CH<sub>4</sub> emission rates at Phrag1, Phrag2 and Rice were primarily related to the vari-



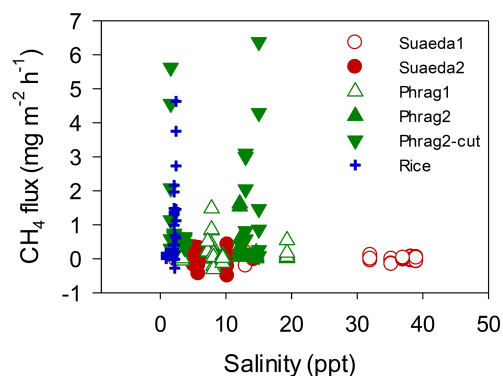


**Figure 3.** Relationship between CH<sub>4</sub> emission rates and (a) soil temperature, and (b) water table, in two *Phragmites australis* wetlands and a rice paddy in the Liaohe Delta, Northeast China. Data points after cutting the vegetation at Phrag2 are represented by downward triangles (Phrag2-cut). Measurements were done from April to November 2012.

ations in soil temperatures. The highest CH<sub>4</sub> emission rates occurred during the summer months July–September, when temperatures were relatively high. We found an exponential relationship between soil temperature and CH<sub>4</sub> 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 than at 10 °C. Also the plant-mediated CH<sub>4</sub> transport may be accelerated at higher temperatures as Hosono and Nouchi (1997) reported that the CH<sub>4</sub> 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 CH<sub>4</sub> 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, CH<sub>4</sub> emission rates were consistently low (< 1 mg CH<sub>4</sub> m<sup>-2</sup> h<sup>-1</sup>). In the spring, the low rates might be associated with a time-lag in the growth of methanogens as the temperature was increasing over a relatively short period. In the autumn the low rates might be influenced by low availability of organic carbon, as most carbon might have been burned off during the hot summer months.

#### 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



**Figure 4.** Relationship between salinity and CH<sub>4</sub> emission rates in two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy during 2012 in the Liaohe Delta, Northeast China. Data points after cutting the vegetation at Phrag2 are represented by downward triangles (Phrag2-cut). Measurements were done from April to November 2012.

all sites, which suggests that temperature was the main controlling factor for ecosystem respiration. This is in agreement with the findings of other studies (Bridgman 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 above-ground biomass was highest. At Phrag1, Suaeda1 and Suaeda2, the seasonal pattern of ecosystem respiration rates correlated to that of the above-ground 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 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 CO<sub>2</sub> 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 CO<sub>2</sub> 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 above-ground biomass was only about 100 g m<sup>-2</sup>. The above-ground rice biomass continued 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

The two *Suaeda* wetlands were net CH<sub>4</sub> sinks whereas the two *Phragmites* wetlands and the rice paddy were net CH<sub>4</sub> sources during April to November 2012. Although the peak CH<sub>4</sub> emission rates at the rice paddy were similar to those

at Phrag2, the cumulative CH<sub>4</sub> emission rates from Phrag2 were 1.5 times higher than those from Rice. The cumulative CO<sub>2</sub> emitted from ecosystem respiration followed a similar pattern, with Phrag2 emitting 1.3 times more CO<sub>2</sub> than the rice paddy. Thus, on a yearly basis Phrag2 emitted the highest amounts of both CH<sub>4</sub> and CO<sub>2</sub> per unit area, and also the most CH<sub>4</sub> relative to CO<sub>2</sub>. Since CO<sub>2</sub> emissions from vegetated ecosystems are counteracted by photosynthetic CO<sub>2</sub> uptake and possibly carbon sequestration, the CO<sub>2</sub> emissions measured as ecosystem respiration does not contribute to the greenhouse effect. However, the CH<sub>4</sub> emissions from wetland ecosystems contribute to the radiative forcing, and therefore CH<sub>4</sub> emission rates should be minimized. It is, however, the balance between carbon sequestrations on the one hand and CH<sub>4</sub> 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 gasses (Mitsch et al., 2013). Based on the present study, it is unfortunately not possible to estimate the carbon sequestration of the different wetland communities.

#### 4.4 CH<sub>4</sub> emission rates and $R_{\text{eco}}$ compared to other studies

The CH<sub>4</sub> 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 CH<sub>4</sub> emission rates varied from  $-0.97 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$  in early May to  $2.73 \text{ mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$  in early September. The average CH<sub>4</sub> emission rate at Phrag2 was within the range of CH<sub>4</sub> emission rates from reed wetlands in other parts of China, varying from  $0.75 \text{ mg m}^{-2} \text{ h}^{-1}$  (Xu et al., 2014) to  $5.13 \text{ mg m}^{-2} \text{ h}^{-1}$  (Tong et al., 2010). The *Suaeda* wetlands had CH<sub>4</sub> emission rates very similar to those from a *Suaeda salsa* marsh in the Yellow River delta, China, with rates ranging from  $-0.74$  to  $0.42 \text{ mg m}^{-2} \text{ h}^{-1}$  (Sun et al., 2013). The CH<sub>4</sub> emission rates from the rice paddy in the present study were lower than those reported from continuously and intermittently flooded rice paddies in Nanjing, China, which emitted  $1\text{--}3 \text{ mg m}^{-2} \text{ h}^{-1}$  (Zou et al., 2005). This might be due to temperature differences or differences in soil characteristics at the two sites.

The yearly cumulative CH<sub>4</sub> emissions from Phrag2 were similar to those reported by Xu et al. (2014) from a coastal saline grass flat dominated by *P. australis* in southeast China ( $6.28 \text{ g m}^{-2}$ ). However, markedly higher cumulative CH<sub>4</sub> emissions have been measured from other reed wetlands, such as  $39.5 \text{ g m}^{-2}$  from a tidal reed marsh in southeast China (Tong et al., 2010) and  $65.9 \text{ g m}^{-2}$  from a restored reed fen in northeastern Germany (Koch et al., 2014). The yearly cumulative CH<sub>4</sub> emissions from the rice paddy in our study were about six times higher than the  $0.54\text{--}0.58 \text{ g m}^{-2}$  measured from rice paddies in eastern China (Zhang et al., 2014) but much lower than the  $57 \text{ g m}^{-2}$  measured over only 2 months from a rice paddy in the Philippines (Gaijre et al., 2014). The

*Suaeda* wetlands in our study had no net CH<sub>4</sub> emissions over the sampling period, in contrast to a *Suaeda glauca* marsh in southeast China which emitted  $0.399 \text{ g CH}_4 \text{ m}^{-2} \text{ yr}^{-1}$  (Xu et al., 2014).

The average ecosystem respiration rates in this study were in a comparable range to those recorded from coastal saline wetlands in southeast China by Xu et al. (2014). The average CO<sub>2</sub> emission rates at Phrag1 were somewhat lower than the  $569.7 \text{ mg m}^{-2} \text{ h}^{-1}$  from the *Phragmites* wetland in their study, whereas the emissions from Phrag2 were higher. Compared to the *Suaeda glauca* marsh in Xu et al. (2014), which emitted on average  $248.6 \text{ mg CO}_2 \text{ m}^{-2} \text{ h}^{-1}$ , Suaeda1 and 2 both had higher average CO<sub>2</sub> emissions.

## 5 Conclusions

We aimed at determining which factors are most important under field conditions for controlling CH<sub>4</sub> and CO<sub>2</sub> 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 CH<sub>4</sub> 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 CH<sub>4</sub> sinks whereas the *Phragmites* wetlands and the rice paddy were net CH<sub>4</sub> sources. The *Phragmites* wetlands had the highest climatic impact as they emitted the most cumulative CH<sub>4</sub> per unit area and the most CH<sub>4</sub> relative to CO<sub>2</sub> compared to the other wetland types. The main controlling factors for the CH<sub>4</sub> emissions were water table, soil organic carbon, 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 CH<sub>4</sub> as aerenchymatous wetland plants with deep root systems can transport CH<sub>4</sub> produced in the deeper soil layers to the atmosphere. The ecosystem respiration of the wetland communities depends largely on temperature and the plant above-ground 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 CO<sub>2</sub> exchange and the CH<sub>4</sub> emission to determine if a particular wetland can be considered to be a net source or a net sink for radiative greenhouse gasses. Our study indicates that the CH<sub>4</sub> 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. How-

ever, the effects of potential water management schemes on the biological communities in the wetlands must be carefully studied prior to the implementation of the management in order to avoid negative and undesirable effects on the wetland communities.

*Author contributions.* S. Ye, K. W. Krauss and H. Brix designed the study, L. Olsson and S. Ye performed the field and laboratory measurements, and L. Olsson prepared the manuscript with contributions from all co-authors.

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