Factors influencing CO_2 and CH_4 emissions from coastal wetlands in the Liaohe Delta, Northeast China

L. Olsson[1,2], S. Ye[3], X. Yu[3], M. Wei[3], K. W. Krauss[4], H. Brix[1]

[1] {Department of Bioscience, Aarhus University, Aarhus, Denmark}

[2] {Sino-Danish Centre for Education and Research (SDC), Aarhus, Denmark}

[3] {Key Laboratory of Coastal Wetlands, China Geological Survey, Qingdao Institute of Marine Geology, Qingdao, China}

[4] {U.S. Geological Survey, National Wetlands Research Center, Lafayette, LA, USA}

Correspondence to: H. Brix (hans.brix@bios.au.dk)

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_4 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 CH_4 sinks whereas the *Phragmites* wetlands and the rice paddy were net CH_4 sources emitting 1.2-6.1 g CH_4 m^{-2} y^{-1}. The *Phragmites* wetlands emitted the most CH_4 per unit area and the most CH_4 relative to CO_2. The main controlling factors for the CH_4 emissions were water table, temperature, soil organic carbon and salinity. The CH_4 emission was accelerated at high and constant (or managed) water tables and decreased at water tables below the soil surface. High temperatures enhanced CH_4 emissions, and emission rates were consistently low (<1 mg CH_4 m^{-2} h) at soil temperatures <18ºC. At salinity levels >18 ppt, the CH_4 emission rates were always low (<1 mg CH_4 m^{-2} h^{-1}) probably because methanogens were outcompeted by sulphate reducing bacteria. Saline *Phragmites* wetlands can, however, emit significant amounts of CH_4 as CH_4 produced in deep soil layers are transported through the air-space tissue of the plants to the atmosphere. The CH_4 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.

Keywords: Coastal wetlands, common reed, greenhouse gas emissions, *Phragmites australis,* rice paddy, seablite, *Suaeda salsa*

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_2 and CH_4 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_2 as an end-product, whereas anaerobic decomposition is much slower and, along with CO_2, also produces CH_4. 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 CH_4 is 25 times greater than that of CO_2 on a 100 year time scale (IPCC, 2007) and high emissions of CH_4 can therefore have disproportionately adverse effects on the climate. According to Whalen (2005), wetlands contribute to about 24% of global CH_4 emissions from all sources, and are the largest natural source of CH_4. Due to the increasing concern of greenhouse gas emissions and global warming, it is important to gain more knowledge about the factors affecting CO_2 and CH_4 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, and therefore, both CO_2 and CH_4 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 ^{o}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_2, NO_3 and SO_4) (Mitsch and Gosselink, 2007). The position of the water table is therefore an important controlling factor on CH_4 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_4 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_4 ha^{-1} y^{-1} when the water table was above -20 cm. The more oxidized conditions associated with low water tables favor CH_4 oxidation by aerobic methanotrophic bacteria (Whalen, 2005), as well as aerobic decomposition of organic matter, both processes emitting CO_2. 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_2 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_4 emissions. Oxygen released from roots create aerobic microsites in the rhizosphere (Brix, 1994), which favors CH_4 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_4 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_4 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_4 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_2 emissions (Chambers et al., 2011; Weston et al., 2011). However, coastal wetlands with high salinity usually emit less CH_4 than less saline wetlands (Bartlett et al., 1987; Poffenbarger et al., 2011). This has been explained by the high concentration of sulphate ions

(SO_4^{2-}) 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 CH_4 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_4 than those with a lower salinity.

Although many factors are known to influence CO_2 and CH_4 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 CH_4 emission and ecosystem respiration in the dominant wetland communities in a coastal wetland ecosystem, (ii) to assess the seasonal variation in CH_4 emission and ecosystem respiration in different plant communities, and (iii) to determine the main controlling factors for CH_4 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 1,280 km^2 (Li et al. 2012). About 786 km^2 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

124 breeding areas for many endangered bird species, and are designated as a Shuangtaizihekou
125 (Liaohekou) National Nature Reserve since 1986 and also listed as a Ramsar site since 2004 (Li et
126 al., 2012). However, the wetlands are adversely affected by the polluted water from the Liaohe
127 River (Zhang et al., 2010) and oil extraction activities, as the Liaohe Delta contains the third largest
128 oil field in China (Zhu et al., 2010).

129 Five study sites were selected to embrace the main wetland types of the delta. The five study sites
130 included two *Suaeda* marshes, one created and one natural ('Suaeda1' at 40°52'11.09"N;
131 121°36'21.72"E and 'Suaeda2' at 40°57'38.62"N; 121°48'20.03"E, respectively), two *Phragmites*
132 wetlands for paper production, ('Phrag1' at 40°52'22.34"N; 121°36'08.89"E and 'Phrag2' at
133 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).

134 **2.2 Gas sampling and analysis**

135 Gas samples for estimation of CO_2 and CH_4 emission were collected monthly from April to
136 November 2012, using the static chamber method (Livingston and Hutchinson, 1995). Six
137 quadratic metal frames (0.6 x 0.6 m) were permanently installed in each study site, and wooden
138 boardwalks were built to facilitate access to the frames without disturbing the soil. Small holes
139 were drilled in the sides of the frames just at the ground surface to facilitate water exchange
140 between the inside of the frames and the surrounding wetland between sampling events. These
141 holes were plugged during sampling. At each sampling event, a white plastic chamber (0.55 x 0.55
142 x 0.30 m) was placed over the metal frame and an airtight seal was created by water (about 1 cm
143 deep) within a trough inside the frame. The chambers were modified from past designs deployed
144 in shaded forested wetlands (Krauss and Whitbeck, 2012; Yu et al., 2008) by using aluminum foil to
145 cover all inside walls to block out light and prevent photosynthesis and to minimize temperature
146 changes. Transparent and opaque chambers have been shown to provide similar CH_4 flux
147 estimates (Minke et al., 2014). If the vegetation was taller than the chamber, the plants were bent
148 to fit inside the chamber. At Phrag2, however, the plants grew so tall that they had to be cut in
149 June; we limited what we had to cut as much as possible. A small fan was used to mix the air inside
150 the chamber during sampling, and a PVC tube with the outer end placed in water was used to
151 equilibrate the air pressure inside the chamber with the outside air pressure. Gas samples were
152 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 minute 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 CO_2 flux in each chamber was also measured in situ during separate 1 minute incubations on the same day using a portable infrared gas analyzer (LI-COR 8100, Lincoln, NE, USA).

The concentrations of CO_2 and CH_4 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_4 and CO_2. Gas emissions in mg CH_4 m^{-2} h^{-1} and mg CO_2 m^{-2} h^{-1} (using the weight of the whole molecules of CH_4 and CO_2, 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_2 emissions are available from Phrag1 in April and from Suaeda1 and Suaeda2 in May, and no data on CO_2 and CH_4 emissions in August from Phrag1.

Cumulative CO_2 and CH_4 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, CO_2 emissions were assumed to be constant on a daily and nightly basis. And although some studies have found diurnal variations in CH_4 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_4 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 meter (Jenco Electronics, Ltd., Shanghai, China).

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 20x20 cm plots outside the frame with a similar plant density were harvested. The plants were dried at 60^{o}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^{o}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 hours 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 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_3COOH, 40.0 g NH_4NO_3, 1.12 g NH_4F, 1.68 mL concentrated HNO_3, 0.58 g EDTA and 1,600 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 minutes and then centrifuged for 20 minutes. 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).

2.5 Statistical analysis

The in situ measurements of CO_2 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 CH_4 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 (CH_4 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 CO_2 and CH_4 emissions, respectively. The response variables were CO_2 and CH_4 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 CO_2 and CH_4 emissions were log-transformed as described before to meet the assumptions of normality and equal variances. The original mixed effects model for CO_2 and CH_4 emissions, respectively, was in the form:

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

where 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 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^{o}C in May to 23-28^{o}C in August, and then declined to 0-7^{o}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.

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 Sueda2 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

Al 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.

3.1 CH_4 emissions

There were large variations in CH_4 emission rates both among sites and over the season (Fig. 1a) and these differences were statistically significant (Table 2). The highest CH_4 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_4 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_4 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_4 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_4 emission rates were close to zero throughout the sampling period. Means and ranges of CH_4 emission rates over the whole sampling period, and significant differences ($p<0.05$) among sites, are shown in Table 4.

The CH_4 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_4 emission rates were less than 1 mg m^{-2} h^{-1} at temperatures below 18^{o}C and at water tables below the soil surface. The highest CH_4 emission rates were measured at Phrag2 when both the temperature and the water table were high (Fig. 3). The CH_4 emissions decreased significantly (Table 3) with increasing salinity, as CH_4 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_4 emission rates were practically zero.

Cumulative CH_4 emissions over the entire growing season in 2012 were highest at Phrag2 with 6.1 g CH_4 m^{-2}, corresponding to 154 g CO_2-equivalents m^{-2} y^{-1} (Fig. 2, Table 4). These emissions were about 1.5 times higher than the cumulative CH_4 emissions from the rice paddy, and about five

295 times higher than the CH_4 emissions from Phrag1. CH_4 emissions from the *Suaeda* wetlands were
296 negligible.

297 **3.2 Ecosystem respiration (R_{eco})**
298 The measured flux of CO_2 in the darkened chamber is the sum of the flux of CO_2 from the soil and
299 the respiration of the plant tissue inside the chambers. We here refer to this as the ecosystem
300 respiration (R_{eco}). The ecosystem respiration rates varied significantly both among sites and over
301 time (Fig. 1b, Table 2). The highest ecosystem respiration rates at the rice paddy and at Phrag2
302 (2,400 and 2,300 mg CO_2 m^{-2} h^{-1}, respectively) were twice as high as the highest R_{eco} at Phrag1 and
303 three times higher than the R_{eco} at the two *Suaeda* wetlands. At Phrag2, R_{eco} was highest in June
304 and July, whereas at the rice paddy, the R_{eco} was low at this time of the year and highest in August
305 (Fig. 1b). It should, however, be mentioned that the *Phragmites* stems at Phrag2 were cut in June.
306 Hence, the biomass within the chambers from July and onwards was lower than the biomass in
307 the surrounding reed vegetation. Overall, the ecosystem respiration rates were significantly
308 related to plant biomass, soil temperature and salinity (Table 3) whereas water table had no
309 significant effect on R_{eco} ($p>0.05$).

310 The cumulative CO_2 emissions, without accounting for photosynthetic CO_2 uptake, varied between
311 1.7 kg m^{-2} y^{-1} in the *Suaeda* wetlands to 3.3-4.4 kg m^{-2} y^{-1} in the *Phragmites* (Table 4). The
312 cumulative CO_2 emission in the rice paddy was in-between this range (3.3 kg m^{-2} y^{-1}).

313
314

315 **4 Discussion**

316 **4.1 CH_4 emissions**
317 Over one growing season in 2012, the two *Phragmites* wetlands emitted on average 0.15 and 1.01
318 mg CH_4 m^{-2} h^{-1} (Phrag1 and Phrag2, respectively) and the rice paddy 0.75 mg m^{-2} h^{-1}, whereas the
319 emissions from the two *Suaeda* wetlands were negligible. The large differences in CH_4 emission
320 rates among the five sites can be explained by the differences in soil organic matter, salinity and
321 water tables, and, to some extent, vegetation type. For methanogenesis to take place there must
322 be a sufficient amount of labile organic substrate available (Mah et al., 1977), such as dead plant
323 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_4 emission rates with increasing content of soil organic 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_4 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_4 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_4 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_4 emission.

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 (Brix et al., 1996; Singh and Singh, 1995). When CH_4 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_4 otherwise could have been oxidized by methanotrophic bacteria (Whalen, 2005). Plant-mediated transport has been reported to be the main pathway of CH_4 transport from the soil to the atmosphere and constituting as much as 60-90 % of the CH_4 emissions (Butterbach-Bahl et al., 1997; Huang et al., 2005). In the present study, transport of CH_4 through the air-space tissue of the plants may explain the relatively high CH_4 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 CH_4 emissions, as CH_4 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 CH_4 transport through the plants, whereas the removal of fine roots and root

354 tips of coarse roots led to significant reductions in plant-mediated CH_4 transport. Thus, it is likely
355 that the extensive root system of the reeds at Phrag2 contributed to the high CH_4 emission rates
356 at this site.

357 At salinity levels above 18 ppt the CH_4 emission rates were always lower than 1 mg m^{-2} h^{-1} across
358 all sites (Fig. 4). This is consistent with Poffenbarger et al. (2011) who found a salinity threshold of
359 18 ppt, above which CH_4 emission rates were significantly lower than at lower salinity levels. The
360 effect of salinity has been explained by the high concentrations of SO_4^{2-} in seawater, which inhibits
361 CH_4 production due to competition from sulphate reducing bacteria (Bartlett et al., 1987; D'Angelo
362 and Reddy, 1999). Thus, the lack of CH_4 emissions at the *Suaeda* sites is most likely an effect of the
363 high salinity, particularly at the Suaeda1 site where salinities were up to 35 ppt. The salinity was,
364 however, significantly lower at the Suaeda2 site with salinities of 5-15 ppt, and yet there were no
365 CH_4 emissions as SO_4^{2-} concentrations were still high enough to inhibit methanogenesis. At Phrag2,
366 on the other hand, CH_4 emission rates were high although the water salinity was occasionally as
367 high as 15 ppt. These seemingly contradictory results can be explained by the fact that a high
368 salinity in the water mainly affects the upper soil layers, but not necessarily the deeper layers.
369 Therefore, methanogens may be outcompeted by sulphate reducing bacteria in the upper layers
370 of the soil, but CH_4 can still be produced in the deeper soil layers where all SO_4^{2-} have been
371 reduced. The roots of *P. australis* grow to a soil depth of at least 40-60 cm, and CH_4 can therefore
372 be transported from the deeper anoxic zone through the air-space tissue of the plants to the
373 atmosphere. Thus, the relatively high salinity at Phrag2 probably inhibited methanogenesis in the
374 upper soil layers, but the CH_4 produced in the deeper soil layers were still transported to the
375 atmosphere through the plants. At the *Suaeda* wetlands, the generally low and fluctuating water
376 tables indicate that the anaerobic zone where methanogenesis can take place was at a deeper soil
377 depth than at the *Phragmites* wetlands. The roots of *S. salsa* lack aerenchyma and are generally
378 restricted to the upper 20 cm of the soil, and are therefore ineffective conduits for CH_4 from the
379 deeper soil layers to the atmosphere. Thus, although salinity levels at Suaeda2 were not always
380 high, any CH_4 that may have been produced in the soil did not reach the atmosphere because of
381 CH_4 oxidation in the upper soil layer. At the rice paddy, the low salinity of around 2 ppt seemingly
382 had no inhibitory effect on the CH_4 production and emission.

The water table is an important parameter affecting the CH_4 emission rate. The highest CH_4 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_4 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_4 emission rates. The soils at the tidally influenced *Suaeda* wetlands were periodically drained and hence partly oxidized inhibiting CH_4 production. When water tables at the *Phragmites* wetlands and the rice paddy were below the soil surface, the CH_4 emission rates were always <1 mg CH_4 m^{-2} h^{-1} probably because CH_4 produced in deeper soil layers was oxidized in the upper oxic soil layers, reducing the amount of CH_4 reaching the atmosphere. When the water tables approached the soil surface, the CH_4 emission rates increased. This is in agreement with the findings of Zhu et al. (2014), who reported that the seasonal CH_4 emissions from an herbaceous peatland were highly linked to water table fluctuations, and that the water table was the main environmental driver for CH_4 emissions over a single growing season, whereas soil temperature was important on a longer time scale. The important effect of water table on CH_4 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_4 emission rates at Phrag1, Phrag2 and Rice were primarily related to the variations in soil temperatures. The highest CH_4 emission rates occurred during the summer months July-September, when temperatures were relatively high. We found an exponential relationship between soil temperature and CH_4 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^{o}C than at 10^{o}C. Also the plant-mediated CH_4 transport may be accelerated at higher temperatures as Hosono & Nouchi (1997) reported that the CH_4 transport through rice plants was twice as high at a rhizosphere temperature of 30^{o}C as compared to the transport at 15^{o}C. Thus, the high CH_4 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^{o}C, which occurred before June and after September, CH_4 emission rates were consistently low (<1 mg CH_4 m^{-2} h^{-1}). 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 all sites, which suggests that 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 CO_2 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_2 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 to increase after August, but the ecosystem

443 respiration decreased drastically, indicating that soil respiration constituted the main part of
444 ecosystem respiration at the rice paddy.

445 **4.3 Cumulative emissions**

446 The two *Suaeda* wetlands were net CH_4 sinks whereas the two *Phragmites* wetlands and the rice
447 paddy were net CH_4 sources during April to November 2012. Although the peak CH_4 emission rates
448 at the rice paddy were similar to those at Phrag2, the cumulative CH_4 emission rates from Phrag2
449 were 1.5 times higher than those from Rice. The cumulative CO_2 emitted from ecosystem
450 respiration followed a similar pattern, with Phrag2 emitting 1.3 times more CO_2 than the rice
451 paddy. Thus, on a yearly basis Phrag2 emitted the highest amounts of both CH_4 and CO_2 per unit
452 area, and also the most CH_4 relative to CO_2. Since CO_2 emissions from vegetated ecosystems are
453 counteracted by photosynthetic CO_2 uptake and possibly carbon sequestration, the CO_2 emissions
454 measured as ecosystem respiration does not contribute to the greenhouse effect. However, the
455 CH_4 emissions from wetland ecosystems contribute to the radiative forcing, and therefore CH_4
456 emission rates should be minimized. It is, however, the balance between carbon sequestrations on
457 the one hand and CH_4 emission on the other hand that determines if a particular wetland can be
458 considered to be a net source or a net sink for radiative greenhouse gases (Mitsch et al., 2013).
459 Based on the present study, it is unfortunately not possible to estimate the carbon sequestration
460 of the different wetland communities.

461 **4.4 CH_4 emission rates and R_{eco} compared to other studies**

462 The CH_4 emission rates and seasonal pattern at Phrag2 were similar to those measured by Huang
463 et al. (2005) from a reed wetland in the Liaohe delta, where CH_4 emission rates varied from -0.97
464 mg CH_4 m^{-2} h^{-1} in early May to 2.73 mg CH_4 m^{-2} h^{-1} in early September. The average CH_4 emission
465 rate at Phrag2 was within the range of CH_4 emission rates from reed wetlands in other parts of
466 China, varying from 0.75 mg m^{-2} h^{-1} (Xu et al., 2014) to 5.13 mg m^{-2} h^{-1} (Tong et al., 2010). The
467 *Suaeda* wetlands had CH_4 emission rates very similar to those from a *Suaeda salsa* marsh in the
468 Yellow River delta, China, with rates ranging from -0.74 to 0.42 mg m^{-2} h^{-1} (Sun et al., 2013). The
469 CH_4 emission rates from the rice paddy in the present study were lower than those reported from
470 continuously and intermittently flooded rice paddies in Nanjing, China, which emitted 1-3 mg m^{-2}
471 h^{-1} (Zou et al., 2005). This might be due to temperature differences or differences in soil
472 characteristics at the two sites.

The yearly cumulative CH_4 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 g m^{-2}). However, markedly higher cumulative CH_4 emissions have been measured from other reed wetlands, such as 39.5 g m^{-2} from a tidal reed marsh in southeast China (Tong et al., 2010) and 65.9 g m^{-2} from a restored reed fen in northeastern Germany (Koch et al., 2014). The yearly cumulative CH_4 emissions from the rice paddy in our study were about six times higher than the 0.54-0.58 g m^{-2} measured from rice paddies in eastern China (Zhang et al., 2014) but much lower than the 57 g m^{-2} measured over only two months from a rice paddy in the Philippines (Gaihre et al., 2014). The *Suaeda* wetlands in our study had no net CH_4 emissions over the sampling period, in contrast to a *Suaeda glauca* marsh in southeast China which emitted 0.399 g CH_4 m^{-2} y^{-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_2 emission rates at Phrag1 were somewhat lower than the 569.7 mg m^{-2} 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 mg CO_2 m^{-2} h^{-1}, Suaeda1 and 2 both had higher average CO_2 emissions.

4.4 Conclusions

We aimed at determining which factors are most important under field conditions for controlling CH_4 and CO_2 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_4 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_4 sinks whereas the *Phragmites* wetlands and the rice paddy were net CH_4 sources. The *Phragmites* wetlands had the highest climatic impact as they emitted the most cumulative CH_4 per unit area and the most CH_4 relative to CO_2 compared to the other wetland types. The main controlling factors for the CH_4 emissions were water table, soil organic carbon, temperature and

502 salinity. Methane emissions are accelerated at high and constant (or managed) water tables and
503 decrease at water tables below the soil surface, or fluctuating water tables. Methane emissions
504 are also accelerated at high temperatures and depressed at high salinity levels. Saline wetlands
505 can, however, emit significant amounts of CH_4 as aerenchymatous wetland plants with deep root
506 systems can transport CH_4 produced in the deeper soil layers to the atmosphere. The ecosystem
507 respiration of the wetland communities depends largely on temperature and the plant
508 aboveground biomass, but soil organic matter content and belowground biomass are also
509 important. It is, however, necessary to quantify not only the ecosystem respiration, but also the
510 balance between the net CO_2 exchange and the CH_4 emission to determine if a particular wetland
511 can be considered to be a net source or a net sink for radiative greenhouse gases. Our study
512 indicates that the CH_4 emissions from coastal wetlands can be reduced by managing the water in
513 the wetland in a way that creates fluctuating water tables, including water tables below the soil
514 surface, as well as by occasional flooding by high-salinity water. However, the effects of potential
515 water management schemes on the biological communities in the wetlands must be carefully
516 studied prior to the implementation of the management in order to avoid negative and
517 undesirable effects on the wetland communities.

518 **Author contribution**

519 S.Y., K.W.K and H.B. designed the study, L.O. and S.Y. performed the field and laboratory
520 measurements, and L.O. prepared the manuscript with contributions from all co-authors.

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695

696 **Table 1**. Physical/chemical topsoil characteristics (0-5 cm depth for bulk density, water content and redox
697 potential; else 0-4 cm depth) at the five wetland sites (two *Suaeda salsa* wetlands, two *Phragmites australis*
698 wetlands and one rice paddy) in the Liaohe Delta, northeast China. Data was collected in 2013 by Siyuan Ye
699 et al. (personal communication).

	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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703 **Table 2.** Results from repeated-measures ANOVAs with the response variables CH_4-flux and R_{eco},
704 respectively, the fixed factors Site and Time and their interaction, and the random factor Plot. Gas fluxes
705 were measured during April-November 2012 from six plots at two *Suaeda salsa* wetlands, two *Phragmites*
706 *australis* wetlands and one rice paddy in the Liaohe Delta, northeast China. All measurements from April,
707 May and August were excluded from the analysis due to missing data from some sites.

Response variable	**Factor**	**df**	**F-ratio**	**p**
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

708 df: degrees of freedom

Table 3. Results from linear mixed effects models, with CH_4 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 equation 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 CH_4 emissions >0 included (Phrag1, Phrag2 and Rice).

714 **Table 4.** Mean CH_4 emission and ecosystem respiration rates (R_{eco}) with ranges in parentheses, and
715 cumulative CO_2 equivalents from CH_4 and CO_2 emissions, respectively, from two *Phragmites australis*
716 wetlands and one rice paddy during April-November 2012 in the Liaohe Delta, northeast China. CH_4 fluxes
717 are converted to CO_2-eqvivalents using a factor of 25. Superscript letters represent significant differences
718 ($p<0.05$) among sites.

Site	**CH_4 emission rates** ($mg\ m^{-2}\ h^{-1}$)	**R_{eco}** ($mg\ CO_2\ m^{-2}\ h^{-1}$)	**Cumulative CO_2-equivalents**	
			CH_4 ($g\ CO_2\text{-eqv}\ m^{-2}\ y^{-1}$)	**CO_2** ($g\ CO_2\text{-eqv}\ m^{-2}\ y^{-1}$)
Suaeda1	0.01 (-0.31 - 0.44) [a]	278 (-3.6 - 814) [ab]	-0.4	1671
Suaeda2	-0.01 (-0.50 - 0.42) [a]	423 (4.6 - 954) [b]	-1.9	1730
Phrag1*	0.15 (-0.31 - 1.48) [ab]	484 (-14.8 - 1300) [c]	31.1	2963
Phrag2	1.01 (-0.28 - 6.38) [c]	811 (27.4 - 3357) [c]	153.7	4443
Rice	0.75 (-0.27 - 4.63) [b]	532 (-0.2 - 3181) [a]	91.6	3337

719 * No data from August

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Legends to figures:

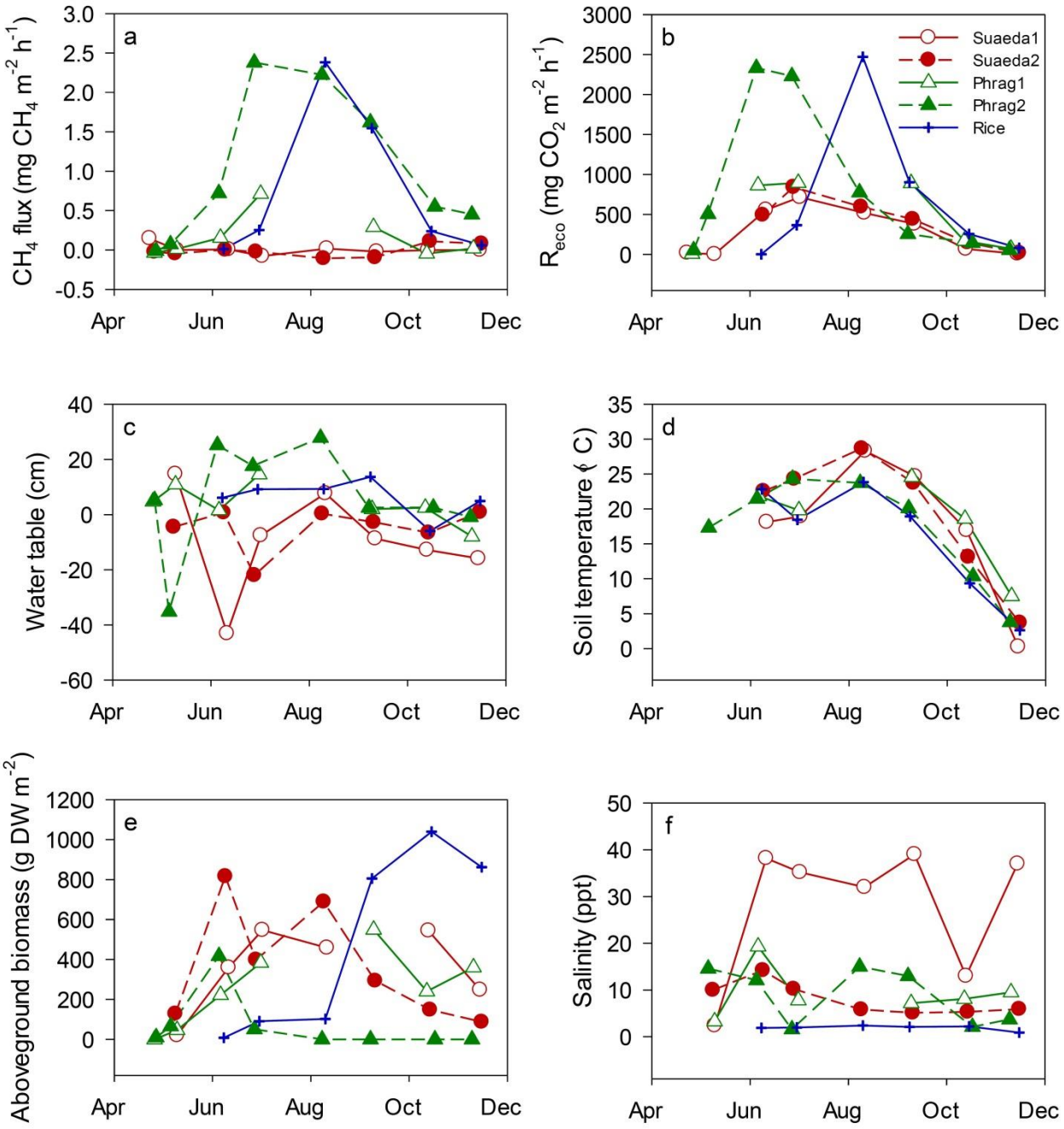
Figure 1. Seasonal variation in (a) CH_4 emission rates, (b) ecosystem respiration, (c) water table, (d) soil temperature, (e) aboveground dry biomass and (f) salinity in two *Suaeda salsa* wetlands, two *Phragmites australis* wetlands and one rice paddy during 2012 in the Liaohe Delta, northeast China. 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 CH_4 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.

Figure 3. Relationship between CH_4 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.

Figure 4. Relationship between salinity and CH_4 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.

741 Fig. 1



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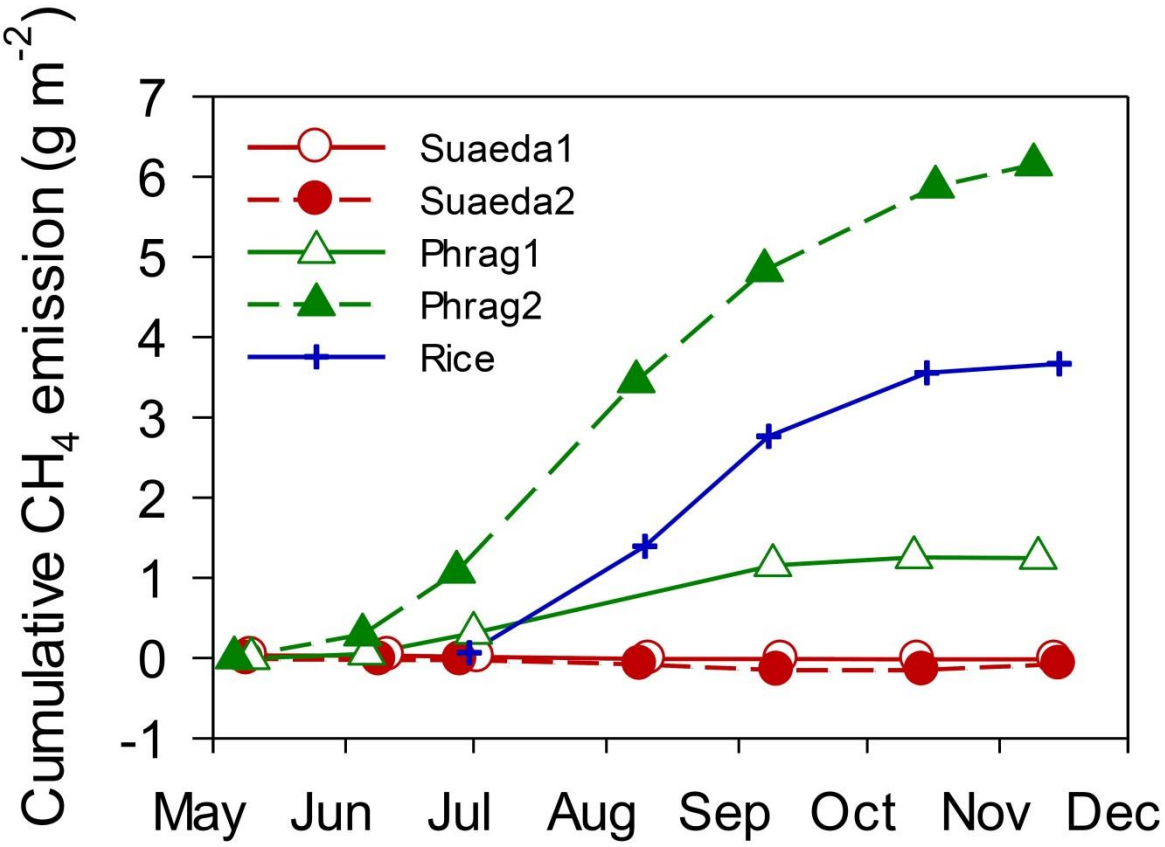
744 Fig. 2



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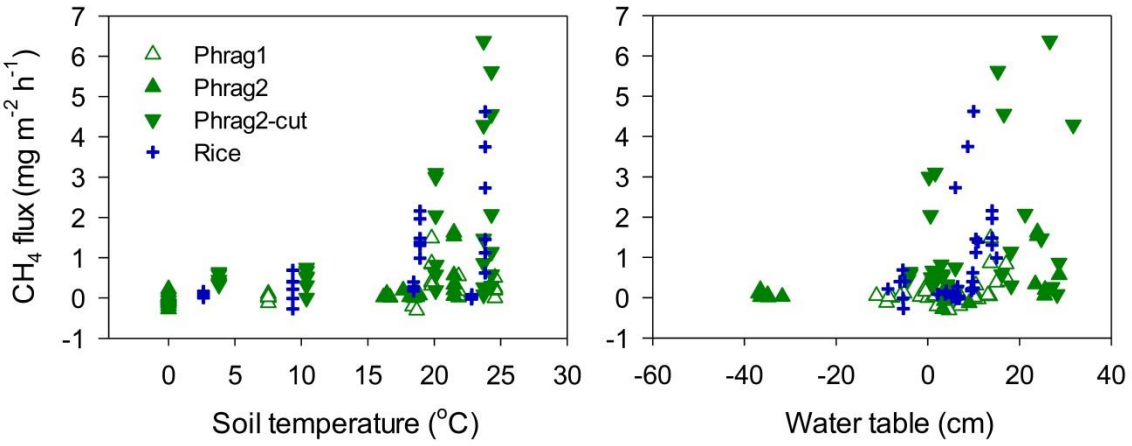
747 Fig. 3.



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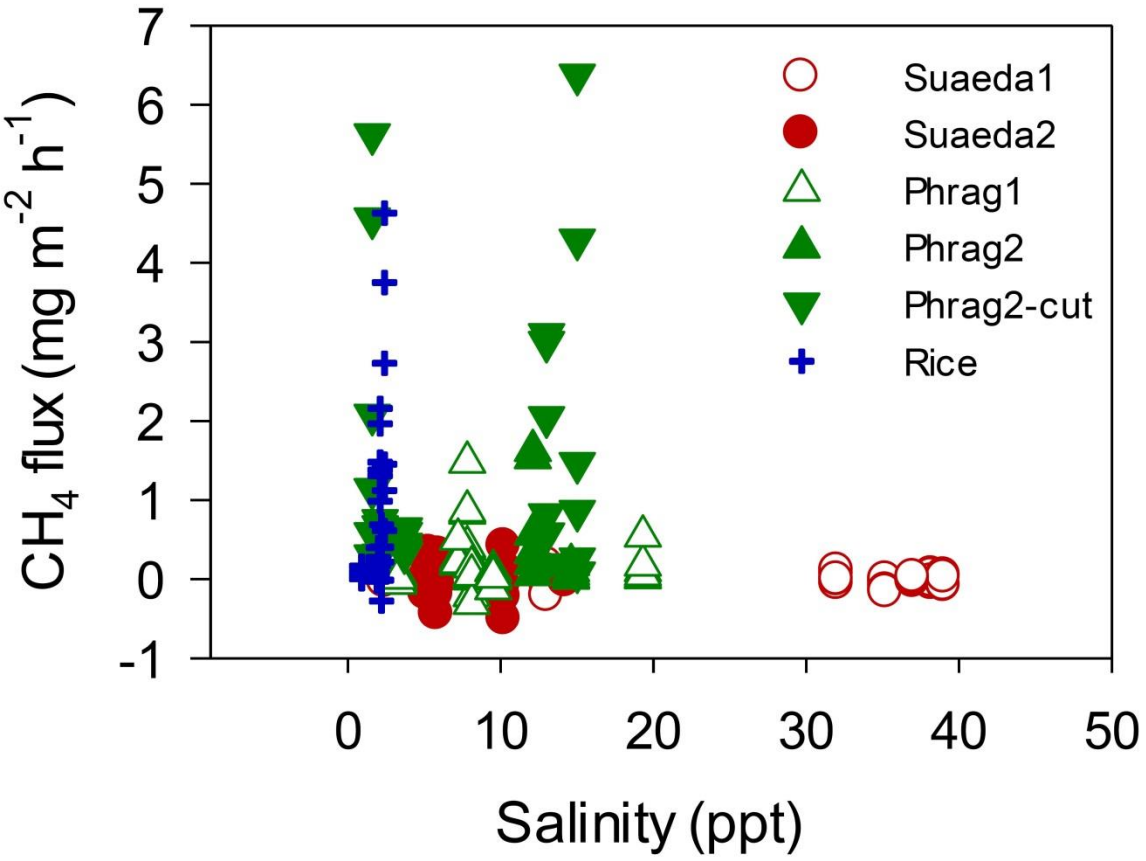
750 Fig. 4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6
5
4
3
2
1
0
-1
CH_4 flux (mg m^{-2} h^{-1})
0
10
20
30
40
50
Salinity (ppt)
Suaeda1
Suaeda2
Phrag1
Phrag2
Phrag2-cut
Rice

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