Effects of Water Regimes on Methane Emissions in Peatland and Gley Marsh

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The increasing frequency of extreme drought and intense precipitation events with global warming may affect CH\textsubscript{4} emissions from different types of wetlands by regulating drying–wetting cycles. To determine the effects of different water regimes on CH\textsubscript{4} emissions, a mesocosm experiment was conducted. Soil cores sampled from peatland and gley marsh were subjected to two drying–wetting cycles (i.e., fluctuating between −10 and 10 cm for 7 and 15 d, respectively) and three steady water table treatments (10, 0, and −10 cm). Alternation between drying and wetting stimulated CH\textsubscript{4} emissions ($F = 16.03$ for 7 d and $F = 31.85$ for 15 d, $P < 0.01$). The highest emission pulses were observed between 4 and 9 d after the water table increased according to the models. Peak pulse emissions significantly increased by 41% in peatland and 109% in gley marsh after rewetting compared with that in the steady 0-cm water table treatment. Peatland soils had higher CH\textsubscript{4} emissions than gley marsh soils under steady water table treatments ($P < 0.01$). This study shows that large pulses of CH\textsubscript{4} can be emitted during short-duration drying–wetting episodes. If these pulses are not accounted for in budgets, CH\textsubscript{4} emissions may be incorrectly assessed when comparing with field measurements during regularly spaced sampling intervals over only a few days or weeks by the static opaque chamber technique. Accurate estimates of CH\textsubscript{4} budgets not only depend on increased measurement frequency but by necessity should incorporate weather events that cause rapid changes in the soil moisture regime.

Wetlands at middle to high latitudes are among the most important biological sources for atmospheric CH\textsubscript{4} because of their vast areal coverage and potential for CH\textsubscript{4} production (Gorham, 1991; Bubier et al., 1995). In general, net CH\textsubscript{4} emissions from wetlands depend mainly on soil CH\textsubscript{4} production in the anaerobic environment and are offset to some extent by CH\textsubscript{4} consumption by methanotrophs in aerobic soil layers (Huttunen et al., 2003). The water table position is crucial to regulate O\textsubscript{2} availability and thus affects CH\textsubscript{4} emissions (Reddy and DeLaune, 2008). In general, global climate change is predicted to lead to longer drought periods and more intense and irregular precipitation events (Huntington, 2006; IPCC, 2015). This will affect element dynamics in wetlands, causing potential positive or negative feedbacks on greenhouse gas emissions (Austin et al., 2004; Gordon et al., 2008; Miller et al., 2005). Drying–wetting cycles influence oxidation and reduction of electron acceptors in sediments, e.g., NO\textsubscript{3}\textsuperscript{−}, Mn, Fe, and SO\textsubscript{4}\textsuperscript{2−}, which in turn indirectly affects CH\textsubscript{4} production and oxidation (e.g., Boon et al., 1997; Le Mer and Roger, 2001). Considering the diversity in wetland types and microtopography, responses to fluctuations in soil moisture conditions by variable precipitation regimes are complicated and hard to predict (Collins et al., 2008; Evans and Wallenstein, 2012). Therefore, high uncertainties remain for predictions of CH\textsubscript{4} emissions from different types of wetlands in a changing climate.

Previous studies have shown high interannual variability of precipitation and frequent fluctuations in precipitation events in the Sanjiang Plain in northeast China (Yang, 1989; Liu, 2005, p. 3–8). This variability has an important role in regulating CH\textsubscript{4} emissions associated with hydrological and biogeochemical processes. Based on our field observations, significant temporal variations in CH\textsubscript{4} emissions occur due to extended drought periods in the summer following natural rainfall events (Zhu et al., 2014). This raised
concerns about the accuracy of scenarios based on short-term disturbances in environmental and ecological variables (Blodau and Moore, 2003).

The Sanjiang Plain is known as the largest freshwater wetland complex, approximately 10,400 km², in China (Song et al., 2009). Natural freshwater wetlands in China are divided into gley marshes and peatlands (Liu, 2005, p. 3–8). Gley marsh usually occurs in floodplains along rivers, whereas peatlands are usually located in waterlogged depressions and ancient riverbeds (Wang et al., 2006). Although similar in plant communities, the two marsh types differ in soil structure, thermal properties, and the type and amount of substrate. Peatland marshes have significant peat layers. The differences in soil properties may lead to differences in their respective drivers of CH₄ emissions and patterns. In this study, we examined CH₄ emissions from peatlands and gley marshes in response to different water regimes, including steady and fluctuating water levels, using mesocosms. Based in part on our previous studies in the field (Zhu et al., 2014), our hypotheses were that: (i) wetting of dry soil leads to a considerable pulse of CH₄ emissions; (ii) there is a time lag for CH₄ pulses after water levels first increase; and (iii) under stable water table conditions, peatland soils emit more CH₄ than gley marsh soils.

Materials and Methods

Study Sites and Mesocosm Experiment

Our study was conducted in the Sanjiang Mire Wetland Experimental Station, Chinese Academy of Sciences, located in the Sanjiang Plain in northeast China (Fig. 1). The study site belongs to the seasonally frozen zone, where the water and soil are completely frozen from late October to early April in the following year (Ding et al., 2002). The mean annual temperature and precipitation are approximately 2.5°C and 560 mm, respectively, and characterized by substantial interannual variations (Song et al., 2009). Precipitation is the main water source to the marshes. The peat depth in our site is less than 1 m thick, but in some places as thick as 4 m (Wang et al., 2006). The peatland site (47°31' N, 133°22' E) is characterized by a peat layer >50 cm thick. A brown and fibrous root layer, a spongy peat layer, and a pale yellow and sticky gley soil layer were found in the soil profile. In the gley marsh (47°35' N, 133°31' E), the albic horizon was widely scattered at a depth of 30 to 50 cm below the surface. The texture of Gleyosols is loamy clay. The volume weight of the surface stratum is 0.1 to 0.8 g cm⁻³ (Zhang, 1981; Gong, 2007). Both the peatland and gley marsh were dominated by Carex lasiocarpa Ehrh. Non-dominant species include Glyceria spiculosa (F. Schmidt) Roshevitz, Carex meyeriana Kunth, Nymphaea tetragona Georgi, Caltha palustris L., Iris tectorum Maxim., and Bryophyte spp. Detailed descriptions of the peatland and gley marsh are given in Table 1.

Forty soil cores, 37 cm in diameter and 50 cm long, were collected from both the peatland and gley marsh in early May 2013 (80 total). The vegetation was left intact. All the cores were installed in polyacrylic cylinders of the same size. The mesocosm experiment was conducted in tanks (300 by 100 by 60 cm) with different water table levels in the field. We conducted three steady-level and two drying–wetting water table treatments, each with eight replications. The water table in the steady-level cores was adjusted to 10, 0, and −10 cm and held constant. In the fluctuating treatments, the water table was initially kept at a depth of 10 cm below the soil surface for 7 or 15 d, then flooded for 7 or 15 d, respectively (i.e., the water table at 10 cm above the soil surface), then drained again for 7 or 15 d, respectively (i.e., the water table again at 10 cm below the soil surface). The cylindrical containers were perforated at the depth of 10 cm below the surface. The water table was lowered as needed by raising the containers in the tanks above the water surface. The water table depth was kept constant by daily inspection and replenished as needed by the addition of collected rainwater to the tanks.
The core heights were elevated relative to the water table depth rather than lowered to the peat surface in the drying–wetting experiment. Drying of the undisturbed soil columns was achieved by covering the tanks to keep out precipitation. When the weather was clear, the plastic film was rolled up. Mesocosms were acclimatized at a water table level of ~0 cm for 2 mo before the experiment was started. Our study was maintained only during the growing season, from May to October, to avoid freeze damage to the cylindrical containers.

Methane Emission Measurements

Methane emission measurements were made using a cylindrical sampling container (Fig. 2). A vent on the top was used to keep pressure equilibration and was sealed with a septum plug when not sampling (Leppälä et al., 2011). Measurements were made about every 10 d from late June to early October in 2013 in the steady water table experiment. For the alternating drying–wetting experiment, the first measurement was made after 7 or 15 d of drying. Then we increased the measurement intensity to examine the instant response of CH₄ emissions to changes in the water level. On the first day, when the dry soil cores were submerged, we measured after 1, 2, 4, and 8 h and after 0, 1, 2, and 5 h in the 7- and 15-d drying–wetting experiments, respectively. Subsequent measurements were made on Days 1, 3, 5, and 7 and on Days 2, 4, 9, 13, and 15, respectively. When soil cores were dried again, measurements were made on Days 1, 3, and 7 and on Days 1, 3, 5, 9, 13, and 15, respectively. The gas samples were measured immediately by a gas chromatograph (Agilent 4890D, Agilent Co.) (Song et al., 2009). The CH₄ fluxes were calculated from the initial slope of a nonlinear regression of concentration against time (Kroon et al., 2008; Song et al., 2009; Zhu et al., 2014).

Statistical Methods

The mean CH₄ emission values were calculated from the average of the measured fluxes in the static water table experiment. One-way ANOVA (Tukey comparison) was used to test the differences in CH₄ emissions under steady water table conditions. An independent samples t-test was used to examine the differences between peatland and gley marsh under different water table regimes. Change in CH₄ emissions with time under drying–wetting conditions was tested by repeated measures ANOVA and modeled by linear and curve equations. In all analyses, the factors and the relationships tested were considered statistically significant at P < 0.05. All the statistical analyses were performed using SPSS 16.0.

Results

Methane Emissions under Steady Water Table Conditions

Methane fluxes varied with time, and obvious emission peaks occurred in August under all three stable water level regimes (Fig. 3). Mean CH₄ emissions ranged from 0.1 to 60.7 mg m⁻² h⁻¹ in the peatland and from 0.2 to 37.5 mg m⁻² h⁻¹ in the gley marsh. Temporal variations in CH₄ emissions were found, displaying parabolic or exponential shapes (Fig. 3). The ANOVA analysis showed a significant effect of water table level in the constant water table level treatments (Table 2). Emissions were lowest in both the gley and peatland soils under drawdown conditions (Fig. 3). Methane emissions in the peatland were more variable and higher than those in the gley marsh (P = 0.01, 0.029, and 0.005 for the 10, 0, and −10 cm water table treatments, respectively).

Methane Emissions under Drying–Wetting Water Table Conditions

Methane emissions in both peatland and gley marsh were modeled by a parabolic function during wetting and by an exponential function during redrying (Fig. 4). In the 7-d drying–wetting treatment, CH₄ emissions showed a drop (from 28.7 to 16.9 mg m⁻² h⁻¹) in the first hour after sudden wetting and then gradually increased for the first few hours before climbing rapidly after the first day (Fig. 4a). Peak emission (47.1 mg m⁻² h⁻¹), 64% higher than prior to wetting and 43% above that under the 0-cm steady water table condition during the same period (Fig. 4a), was observed on the third day of wetting in the peatland. According to the model, CH₄ emissions were highest on Day 5.74. A similar though slightly lower increase (14.5 mg m⁻² h⁻¹) was observed on the first day. Peak emission (22.49 mg m⁻² h⁻¹) was observed on the third day in the gley marsh soil (Fig. 4a), and the highest emission was on Day 4.6, a little earlier than in the peatland.
In the 15-d drying–wetting cycle experiment, emissions decreased initially on wetting and then increased rapidly by the second day. Peak emissions in peatland soils were reached on Day 8.9, whereas in the gley soils, peaks were reached after 4 d. The highest emission was on Day 7 (Fig. 4b). There was a decrease in CH$_4$ emissions on Day 8.9 and 7.3 in the peatland and gley marsh, respectively, according to the model (Fig. 4b). Methane emissions in both the peatland and gley marsh decreased exponentially after the water table was lowered to −10 cm (Fig. 4b). On the first day after drying, emissions in the peatland decreased by 28 and 38% in the 7- and 15-d cycles, respectively, compared with values just before the water table was lowered. Thereafter emissions declined steadily in both experiments. In the gley marsh, the decrease in CH$_4$ emissions was much smaller, decreasing by 10 and 1% in the 7- and 15-d cycles, respectively. A significant effect of time period on CH$_4$ fluxes was identified by ANOVA ($F = 16.03$ for 7 d and $F = 31.85$ for 15 d, $P < 0.01$).

### Discussion

**Effect of Steady Water Tables on Methane Emissions**

Methane emissions under steady water table conditions were significantly lower in the drawdown treatment in both peatland and gley marshes but did not differ between the 10- and 0-cm treatments (Table 2). Methane is produced in a strictly anaerobic process, and lower water tables and increasing the thickness of the aerated top layer can lower CH$_4$ emissions by simultaneously increasing methanotrophy in aerobic soil layers (e.g., Bubier et al., 1995; Deppe et al., 2010a).

Although the peatland and gley marsh have several common properties, such as vegetation, they differ in thermal properties, soil structure, and the type and amount of substrate. Based on the soil characteristics, soil total organic C and N in peatlands are much higher than in gley marsh (Table 1). These differences lead to differences in their respective drivers of CH$_4$ emissions (Carter et al., 2012). Therefore, more substrates such as organic acids, phenolic compounds, carbohydrates, and labile C were expected to be used by methanogenic bacteria to produce CH$_4$ in peatland. Peatland also had 65% greater biomass production than gley marsh (Table 1). More aboveground biomass in the peatland meant more aerenchyma were formed, which enhances CH$_4$ transport to the atmosphere. Therefore, more CH$_4$ production and more CH$_4$ transportation led to more CH$_4$ emissions in peatland.

**Effects of Drying–Wetting Cycle on Methane Emissions**

Considerable pulses of CH$_4$ emissions were observed with fluctuation in the water level in both drying-wetting cycle experiments (Fig. 4). These results were consistent with measurements in the constant −10 cm water table experiment, after a sudden rainfall event on 15 September; the mesocosm was inadvertently exposed to precipitation because the cover had been left off. The main reason may be that the physical disturbance, water table fluctuation, may directly cause a CH$_4$ burst from deep soil reservoirs where CH$_4$ concentrations are high (Dinsmore et al., 2009). In addition, after rewetting, soil substrate availability may be enhanced by diffusion and then a pulse of emissions might also be a biological. Couwenberg et al. (2011) found that a pulse in CH$_4$ fluxes after rewetting was caused by large inputs of labile organic matter. Moreover, methanogenic archaea survive and can be revived quickly in air-dried wetland soils during rewetting (Boon et al., 1997). The persistence of CH$_4$ emissions agrees with an interpretation that the activity of

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**Table 2. Methane emissions under steady water table level.** The mean CH$_4$ emission values were calculated from the average of the measured fluxes.

<table>
<thead>
<tr>
<th>Site</th>
<th>CH$_4$ emissions</th>
</tr>
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<tbody>
<tr>
<td></td>
<td>10 cm</td>
</tr>
<tr>
<td>Gley marsh</td>
<td>15.0 ± 2.3 c†</td>
</tr>
<tr>
<td>Peatland</td>
<td>23.0 ± 3.4a</td>
</tr>
</tbody>
</table>

† Mean ± standard error. Means followed by different letters in a row indicate significant differences among the different water table level treatments ($p < 0.05$).
methanogens was shielded by short periods of oxygen intrusion (Deppe et al., 2010a).

The significant pulses of CH$_4$ emissions occurred with 4 to 9 d of inundation according to the models, which agrees with a large body of research about the effect of water level fluctuations on anaerobic decay processes in wetlands (e.g., Hou et al., 2000; Deppe et al., 2010a; Li et al., 2011). Boon et al. (1997) also found that CH$_4$ emissions spiked within 3 to 6 d after flooding in summer. The following mechanisms may explain the observed lag. Rapid and short precipitation events may not influence CH$_4$ emissions immediately. Emissions mainly depended on the actual soil moisture by this reasoning. First, gas emission rates and diffusivity efficiency are regulated by soil porosity and moisture (Updegraff et al., 1995). Some air-filled porosity still occurred in the saturated zone when aerated soil was rapidly inundated, possibly because of trapped air during the initial stages of sudden rewetting (Baird and Waldron, 2003). The non-methanogenic electron acceptors, such as SO$_4^{2-}$, NO$_3^-$, Mn, and Fe, were replenished due to the variation of soil moisture and aeration. They can compete with methanogens for electron donors on the basis of the redox potential, and then CH$_4$ emissions under subsequent saturated conditions are delayed (Knorr and Blodau, 2009). Second, because of the hydrophobic characteristics of soils, dried soils in the top portion of the column were not easily rewetted even though the water level was increased. A depth of 10 cm above the water table may still have aerobic conditions (Silins and Rothwell, 1999). Third, methanogenic reducers can endure long periods of exposure and survive for extended periods of time under unsaturated conditions (Deppe et al., 2010b). A decrease in methanogens in the top peat layers under extended aerobic conditions appears to require extended periods of saturation to revive CH$_4$ production. Thus, until the O$_2$ is used up, CH$_4$ production starts and consumption is affected. Deppe et al. (2010a) have suggested that soil anaerobic conditions were not exactly correlated with water-saturated conditions. In our field study, when the water level was −10 cm, the water-filled pore space was about 30% and there were still CH$_4$ emissions (Zhu et al., 2014).

Peak pulse emissions significantly increased by 41% in the peatland and 109% in the gley marsh after rewetting compared with that in the steady 0-cm water table treatment. Generally,
rewetting increases CH$_4$ emissions, although in some situations low emissions have been found (Juottonen et al., 2012).

**Summary**

Our study clearly shows the impact of water table depth and fluctuation on CH$_4$ emissions on two different marsh types. Alternation between drying and wetting resulted in pulsed CH$_4$ emissions. Increases in CH$_4$ emissions lagged 4 to 9 d after increases in the water table. Higher CH$_4$ emissions occurred in peatland than in grey marsh in the constant water table treatments. Our study suggests that variability of CH$_4$ emissions in the mesocosms exposed to episodic wetting and drying increased compared with those with a constant water table. Our work highlights the importance of capturing the pulsed emissions after a sudden rise in water table levels in establishing accurate CH$_4$ emission budgets. Various climate change models predict an increased frequency of drought and intense rainfall events in northeast China (Zhao and Luo, 2007). Therefore, high-frequency chamber measurements are necessary to accurately estimate the magnitude of greenhouse gas emissions.

**Acknowledgments**

This research was financially supported by the National Key Research and Development Program of China (No. 2016YFA0602303); “Strategic Priority Research Program-Climate Change: Carbon Budget and Related Issues” of the Chinese Academy of Sciences (No. XDA05020502); the National Natural Science Foundation of China (No. 41125001); China Postdoctoral Science Foundation (No. 2015M580010) and Science and Technology Development Project of Jilin Province (No. 20170520081JH). We thank Christopher Martin Swarzenski, Chao Wang and Ying Ji for their assistance.

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