Growing season methane emission from a boreal peatland in the continuous permafrost zone of Northeast China: effects of active layer depth and vegetation

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Abstract. Boreal peatlands are significant natural sources of methane and especially vulnerable to abrupt climate change. However, the controlling factors of CH\textsubscript{4} emission in boreal peatlands are still unclear. In this study, we investigated CH\textsubscript{4} fluxes and abiotic factors (temperature, water table depth, active layer depth, and dissolved CH\textsubscript{4} concentrations in pore water) during the growing seasons in 2010 and 2011 in both shrub-sphagnum- and sedge-dominated plant communities in the continuous permafrost zone of Northeast China. The objective of our study was to examine the effects of vegetation types and abiotic factors on CH\textsubscript{4} fluxes from a boreal peatland. In an \textit{Eriophorum}-dominated community, mean CH\textsubscript{4} emissions were 1.02 and 0.80 mg m\textsuperscript{−2} h\textsuperscript{−1} in 2010 and 2011, respectively. CH\textsubscript{4} fluxes (0.38 mg m\textsuperscript{−2} h\textsuperscript{−1}) released from the shrub-mosses-dominated community were lower than that from \textit{Eriophorum}-dominated community. Moreover, in the \textit{Eriophorum}-dominated community, CH\textsubscript{4} fluxes showed a significant temporal pattern with a peak value in late August in both 2010 and 2011. However, no distinct seasonal variation was observed in the CH\textsubscript{4} flux in the shrub-mosses-dominated community. Interestingly, in both \textit{Eriophorum}- and shrub-sphagnum-dominated communities, CH\textsubscript{4} fluxes did not show close correlation with air or soil temperature and water table depth, whereas CH\textsubscript{4} emissions correlated well to active layer depth and CH\textsubscript{4} concentration in soil pore water, especially in the \textit{Eriophorum}-dominated community. Our results suggest that CH\textsubscript{4} released from the thawed CH\textsubscript{4}-rich permafrost layer may be a key factor controlling CH\textsubscript{4} emissions in boreal peatlands, and highlight that CH\textsubscript{4} fluxes vary with vegetation type in boreal peatlands. With increasing temperature in future climate patterns, increasing active layer depth and shifting plant functional groups in this region may have a significant effect on CH\textsubscript{4} emission.

1 Introduction

Methane (CH\textsubscript{4}), as one of the most important greenhouse gases, is 25 times more effective in absorbing heat in the atmosphere than carbon dioxide (CO\textsubscript{2}) on a 100-yr time horizon (IPCC, 2007). The atmospheric CH\textsubscript{4} abundance increased from 715 ppb in pre-industrial age to 1774 ppb in 2005. Increases in atmospheric CH\textsubscript{4} concentrations (148 \%) are greater than the other two greenhouse gases (CO\textsubscript{2} 35 \% and N\textsubscript{2}O 18 \%) over the same time period. In order to reduce uncertainties in future projections of Earth’s climate change, the current global CH\textsubscript{4} budget should be better known. Denman et al. (2007) estimated that more than 580 Tg yr\textsuperscript{−1} of CH\textsubscript{4} are emitted to the atmosphere, with 33 \% originating from natural ecosystem sources. However, the contribution of different CH\textsubscript{4} sources and sinks is still highly uncertain due to the sparseness of in situ observations.

Among all the natural ecosystem CH\textsubscript{4} sources, natural wetlands are regarded as the single largest methane source, accounting for 20 \% of the global CH\textsubscript{4} budget (Fung et al., 1987). While covering nearly 3 \% of Earth’s land surface, northern peatlands store a carbon pool of 455 Pg (Gorham, 1991), approximately accounting for one-third of the global soil carbon (Rydin and Jeglum, 2006), and could potentially release carbon in the form of CH\textsubscript{4} to the atmosphere. The
magnitude of CH$_4$ emission from peatland ecosystems is a comprehensive result of several processes including CH$_4$ production and oxidation in the peat profile and abiotic mechanisms such as gas bubbles, diffusion, and gas transport through vascular plant aerenchyma (Whalen, 2005).

Previous studies demonstrated that wetland methane emissions depend on a large amount of abiotic and biotic factors, among the most important of which are temperature, water table depth, vegetation type, substrate quality and supply (Bellisario et al., 1999; Whalen, 2005). Temperature controls methanogenesis and CH$_4$ oxidation by affecting methanogenic and methanotrophic bacteria. The wide range of $Q_{10}$ (reaction rate increase for a 10°C temperature increase) for methanogenesis and methane oxidation suggested a highly significant effect of temperature on CH$_4$ production and oxidation rates (Whalen, 2005). Substrate availability and supply originating from wetland plant litter and/or root exudates determine CH$_4$ production and oxidation. Otherwise, species composition of plants can affect CH$_4$ emissions and substrate availability for methanogens. Previous evidence showed that the vascular plants such as Eriophorum species (Ström et al., 2011) and Carex species (Ding et al., 2005) have a very strong effect on CH$_4$ emission in the northern wetlands, by supply of available substrate and/or gas transportation of aerenchyma. In addition, peatland soil aerobic (anaerobic) conditions resulting from a drop (increase) of the water table can influence CH$_4$ oxidation (production) and then affect CH$_4$ fluxes (Whalen, 2005).

Boreal regions are of close concern since they are expected to undergo large changes in temperature and precipitation (Turetsky et al., 2007). Large amounts of labile soil organic matter that is currently preserved by permafrost will be vulnerable to climate change and could result in changing CH$_4$ emissions through changing peatland hydrology and thermal conditions. For example, permafrost degradation caused by warming will lower the water table following increased drainage in the discontinuous permafrost zone (Riordan et al., 2006) and increase thermokarst lake areas in the continuous permafrost zone (Smith et al., 2005). In addition, boreal peatland soil moisture varied in different permafrost zones owing to increasing difference between potential summer evapotranspiration and precipitation that has been reported (Klein et al., 2005). Under ongoing climate changes, the uncertainties of CH$_4$ fluxes from boreal peatlands have increased, which might confuse the knowledge of the effects of climate change on the boreal peatland carbon cycle.

Many studies on peatland CH$_4$ emissions have been conducted in Siberia (Nilsson et al., 2001; Bohn et al., 2007) and subarctic or arctic regions (Zona et al., 2009; Jackowicz-Korczyńska et al., 2010). However, to our knowledge, there is no study reporting CH$_4$ emissions from boreal peatland in the continuous permafrost zone in China. Understanding CH$_4$ emission from peatland in the continuous permafrost zone can make us better understand CH$_4$ emission patterns and increase the accuracy of estimating a peatland CH$_4$ budget. The goal of this study was to provide a first dataset of CH$_4$ fluxes from a permafrost peatland in Northeast China, and to investigate the factors controlling the seasonal CH$_4$ fluxes from a permafrost peatland.

2 Materials and methods

2.1 Study site and experiment installation

The measurement was conducted in a minerotrophic peatland located in the north of Great Hing’an Mountains, Northeast China (52.94° N, 122.86° E). The study site is situated in the continuous permafrost zone. The climate of this area is cool continental, with a 30-yr (1980–2009) mean annual temperature of −3.9°C and mean annual precipitation of 452 mm, 203 mm of which falls in rainy season (July and August). The coldest monthly mean temperature is −28.7°C in January, and the warmest is 18.4°C in July. The surface of the peatland site is a mosaic of microforms, which are divided into hummock, tussock and hollow. Plants usually grow from early May to late September and the dominant evergreen shrubs are Chamaedaphne calyculata and Ledum palustre. Deciduous shrubs contain Vaccinium vitis-idaea and Betula fruticos. Hummocks were covered by Sphagnum mosses (S. capillifolium, S. magellanicum), Polytrichum commune and previously mentioned shrubs. Tussocks support sedges (Eriophorum vaginatum) as the dominant vascular plant species, as well as sparse shrubs (Vaccinium vitis-idaea, Ledum palustre). A scatter of bryophytes (Polytrichum juniperinum) were present in hollows. The soil type in our study site is classified as peat soil.

A set of twelve plots for gas sampling were selected, and eight of them were chosen so as to be representative of the dominant vegetation in the three microforms and to capture the variability for each of these situations. The intervals among these plots ranged from 5 to 20 m. Four plots were established on the tussock and hollow places where the dominant plant species was Eriophorum vaginatum (Eriophorum-dominated plots: EPs), and four plots on hummocks where dwarf shrubs and mosses were the dominant species (shrub–mosses-dominated plots: SPs). In order to determine the influence of peatland vascular vegetation (Eriophorum vaginatum) on methane emission, four other plots were established on the bare peat where the above-ground parts of dominant vascular plants (Eriophorum vaginatum) were carefully cut and removed before each measurement (bare-peat plots: BPs). In order to make a comparison among these types, flux observations were conducted on the same date.

2.2 Biomass determination and chemical analysis of soils

Above-ground biomass (ABG) was measured by clipping three $1 \times 1$ m quadrats for a shrub-mosses-dominated community and three $0.5 \times 0.5$ m quadrats for a Eriophorum-
dominated community in mid-August and sorting materials by species. We also collected mosses by clipping at the base of the capitulum following Moore et al. (2002). Plant tissues were oven-dried to a constant mass at 65 °C and then were weighted.

To determine pH, total carbon and nitrogen contents of soil from two varied-vegetation-dominated communities, three soil cores were collected from a depth of 0–20 cm on each community. Soil pH was determined by a glass electrode in a 1 : 5 soil: 10 mM CaCl2 solutions of fresh samples according to ISO 10390 standard. The soil samples used for carbon and nitrogen analysis were dried at room temperature and then milled and sieved using a 2 mm screen. Soil organic carbon and total nitrogen concentrations were analyzed by the Multi N/C 2100 Analyzer (containing an HT 1300 Solid Module, Analytik Jena AG, Germany) and the Kjeldahl digestion method using a Behr analyzer (Germany), respectively.

2.3 Gas flux determination

Gas fluxes were measured by the closed chamber and gas chromatograph techniques (Wang and Wang, 2003; Song et al., 2009). The closed chamber was made by stainless steel and consisted of two parts: a square base collar (length: 50 cm, width: 50 cm and height: 20 cm) and a top chamber (length: 50 cm, width: 50 cm and height: 50 or 70 cm) opened at the bottom. The collar was inserted directly into the peat layer to a depth of 15 cm, and kept in the soil during the entire observation period. The top chamber was put on the collar during gas sampling, and immediately removed after gas samples were collected. Two fans were fixed on the inside symmetrical corners of each chamber to keep the air mixed in the chamber closure during sampling. The chambers were wrapped with Styrofoam to prevent an increase in headspace air temperature due to heating when sampling. We built boardwalks to minimize disturbance on the plant and soil microenvironments around collars after the collars were installed.

Gas sampling started in June 2010 and continued until September 2011 at weekly interval during the two growing seasons. Gas samples were only collected in the morning (09:00–11:00 a.m.) because the flux during this period is almost equal to the daily mean flux (Tang et al., 2006). During the flux measurements, headspace samples (50 ml each) were drawn from the chamber every 10 min (including zero time) over half an hour period after enclosure using 60 ml syringes and stored in TedlarR air sample bags (100 ml, Delin Ltd, Liaoning, China), which had been pre-evacuated to close to 0 Pa. A total of four samples were taken during a flux measurement.

The collected gas samples were delivered to Sanjiang Experimental Station of Wetland Ecology, Chinese Academy of Sciences, and analyzed within a week. Gas concentrations were measured by a modified gas chromatograph (Agilent 4890D, Agilent Co., Santa Clara, CA, USA). The gas chromatograph was equipped with a flame ionization detector (FID) for CH4 analysis. The air bags with known standard concentration of CH4 were delivered with the collected samples to the laboratory to evaluate the leakage of trace gases during transport and analysis. No significant changes in the concentration of the standards were found during one week of transfer. The fluxes were calculated as the change in chamber concentration over time. The fluxes were rejected unless they yielded a linear regression with coefficient $R^2 > 0.8$ for CH4. More details of the flux calculation can be found in Song et al. (2009).

2.4 Dissolved methane concentration

Soil pore water was sampled at several depths to determine dissolved CH4 concentration if there was enough pore water for extracting. A set of stainless-steel tubes varied in length were installed before measuring at 10 cm intervals from peatland surface to 40 cm below the surface. Immediately after gas flux measurements, pore water samples (20 ml) were drawn from tubes using a syringe and then injected into evacuated vials (60 ml). Prior to determining CH4 concentration in pore water, vials were shaken for a few minutes to extract dissolved CH4. Subsequently, 40 ml of the headspace was sampled by a syringe and stored in a TedlarR air sample bag. CH4 concentration was analyzed as described above. The methods for calculating dissolved pore water CH4 ($\mu$mol l$^{-1}$) have been described by Ding et al. (2003).

2.5 Abiotic variables

Air temperature, soil temperature, depth of active layer and groundwater level were measured at the same time as gas sampling. Air temperature inside the chambers was measured with a thermometer inserted into the chambers, and soil temperature was measured 0, 5, 10, 15 and 20 cm below the peat surface next to the chambers using a portable digital thermometer (JM 624, Jinming Instrument CO., Ltd, Tianjin, China). Active layer depth was simultaneously measured by a steel rod. Groundwater level was monitored by digging a small well adjacent to the collar over the frost-free season. Daily precipitation data were manually recorded near the sampling site.

2.6 Data analysis

Correlation analysis (Spearman’s rank correlation test) was used for identifying the relationships between CH4 fluxes and environmental factors (i.e. temperature, water table depth, active layer depth and soil pore water CH4 concentration). In all analyses where $p < 0.05$, the factor tested and the relationships were considered statistically significant. The one-way analysis of variance (ANOVA) was conducted to test the differences in soil chemical characters for both communities. All the statistical analyses were conducted
by Software packages SPSS 13.0 (SPSS Inc., Chicago, IL, USA) and figures were prepared by Origin 8.0 (Origin Lab Corporation, USA) for Windows XP.

3 Results and discussion

3.1 Environment variables, biomass, soil chemical characteristics, CH$_4$ concentration in pore water and CH$_4$ fluxes

During the sampling period, monthly mean air temperature (MMAT) varied from 5.3°C (September 2011) to 20.3°C (July 2011; Fig. 1a). There was no marked discrepancy between the MMAT and the 30-yr mean value in the two measurement years. However, we observed extreme daily temperatures in the last few days of June, and the maximum daily temperature reached 39.4°C on 27 June 2010. Accumulative precipitations from May to September were 325.9 and 493.7 mm in 2010 and 2011, which were 11% lower and 34.8% greater than the 30-yr mean value during the same period, respectively (Fig. 1b). A heavy rain occurred on 23 August 2011 and the accumulative rainfall was 129.1 mm (data not shown). The seasonality of ground temperature and soil temperature were consistent with the seasonal patterns of air temperature during the sampling period in 2011. The in-chamber soil temperatures observed in different vegetation plots showed that soil temperatures at the SP site were a little higher than that at the EP site (Fig. 2). The presence of Sphagnum at the SP site preserved soil heat diffusion. The water table depth throughout the measurement period ranged from −10.7 to −24 cm (minus value means below the surface) at the SP site and from −10.5 to −36 cm at the EP site (Fig. 3). The water table depth was consistently higher at the SP site than at the EP site during the two growing seasons (Fig. 3), and the average difference in water table between the two sites was 4 cm. A similar seasonal variation of water table depth at the SP and EP sites was observed, and the lowest value occurred in late June or early July due to higher temperature and less precipitation.

At the beginning of the measurement, peatland surface soil was frozen. The active layer depth continuously increased with air and soil temperatures at the initial stage. In the late sampling period, the active layer depth still increased with decreasing air and soil temperatures. This might be on account of heat in deep soil transferring slower than that in upper soil layers and the atmosphere. The maximum active layer depth reached 72.4 cm and 80.7 cm by the end of the observation period in 2010 and 2011, respectively.

The above-ground biomass of shrubs, sedges and mosses from both communities in the peatland is given in Table 1. The total ABG from the SP site was two times higher than that from the EP site, whereas the ABG of sedges was much lower at the SP site. There was no significant difference in soil chemical characteristics between the SP and EP sites ($p=0.260$ for SOC and 0.236 for TN; Table 1). Soil organic carbon content was a little higher at the SP site (424.7 ± 40.5 g kg$^{-1}$) than at the EP site (403.7 ± 20.6 g kg$^{-1}$). The inverse pattern was observed in the total nitrogen content. pH was slightly lower at the EP site compared to the SP site.

The details and seasonal fluctuations in pore water concentration of CH$_4$ measured in the peatland soil profile can be seen in Fig. 4. Pore water CH$_4$ concentration at 20 cm below
Fig. 2. Temperatures recorded by digital thermometer at SP (shrub–mosses plot) and EP (Eriophorum plot) sites during sampling in 2011. (A) Air temperature inside the chamber; (B) peat surface temperature; (C) soil temperature at 5 cm depth; (D) soil temperature at 10 cm depth.

Fig. 3. The seasonal variation of net CH$_4$ fluxes and environmental variables (water table and active layer depth) observed at the study site during the growing seasons of 2010 and 2011.
Figure 3 shows that the seasonal variations of CH$_4$ flux exist for both sites. A similar seasonal trend of CH$_4$ fluxes in disparate observation years was found at the SP and EP sites. However, the variation in CH$_4$ emissions at the SP site is lower than that at the EP site. Except for the vascular plants regulating methane emissions, methane oxidation in situ conditions may play a more important role in hummocks than in tussocks. CH$_4$ emissions gradually increased with the development of growing season and peaked in late August in both years. Unlike other previous studies that reported no seasonal variation of CH$_4$ fluxes from peatlands, we found a distinct temporal variation in methane emissions where CH$_4$ fluxes peaked in late summer when the active layer reached the gas-contained layer, and which was consistent with peak pore water CH$_4$ concentration. Our results were consistent with Moore and Knowles (1990), who found CH$_4$ fluxes peaked in the later growing season from a subarctic fen in Quebec.

### 3.2 Controls on CH$_4$ flux

Previous studies have shown that temperature (Bellisario et al., 1999; Pelletier et al., 2007; Sun et al., 2011) and water table depth (Moore et al., 2011) were primary factors that controlled peatland CH$_4$ emissions. The relationships between CH$_4$ fluxes and environmental factors such as temperature, water table depth and active layer depth in an independent observation year were examined. The site-specific CH$_4$ fluxes did not show any relationship with soil or air temperature and water table depth, indicating a complicated conjunct effect of variables on CH$_4$ flux. It was consistent with Christensen et al. (1995), who found no correlations between environmental factors and CH$_4$ emission in Siberian mesic tundra. Ström et al. (2011) also found no correlations between seasonal mean CH$_4$ fluxes and water table depth and soil temperature in an arctic wetland. In the present study, the controls on seasonal variation of CH$_4$ flux were distinct at different stages of plant growth. In the early growing season (Period I), when moisture was adequate to support methanogenesis, temperature played a critical role in peatland CH$_4$ emission (Table 2). However, there was a lag time between rising temperatures and CH$_4$ flux in the early season because microbial communities and vegetation required time to become established. The following mechanisms might interpret temperature-dependence CH$_4$ fluxes during the early growing season. Firstly, temperature was an important control on methanogenesis. The widely reported $Q_{10}$ values for methanogenesis ranging from 1 to 35 in boreal peatland soils (Whalen, 2005) suggested that temperature sensitivity of the underlying microbial processes involved in the production of CH$_4$ was high under appropriate substrate and moisture conditions. The lack of CH$_4$ production capacity under low temperature magnified the effect of temperature on CH$_4$ emission. Secondly, temperature controlled plant growth, which could provide not only substrate for methanogenesis but also the peatland surface showed no seasonal variation and the mean CH$_4$ concentration in pore water was 14.37 µmol l$^{-1}$. However, a significant seasonal variation of CH$_4$ concentration in 30 and 40 cm below peat surface was observed. CH$_4$ concentrations at 30 and 40 cm depths increased following the development of the growing season. Correlation analysis showed that average CH$_4$ concentration between 20 cm and 40 cm was related to soil temperature at 40 cm depth ($r = 0.573, p = 0.05$). Figure 4 also shows that pore water CH$_4$ concentrations increased with depth. At the depth from 20 cm to 40 cm, the concentration of CH$_4$ increased sharply by 2 to 10 times magnitude.

Generally, the peatland emitted CH$_4$ to the atmosphere during the two growing seasons, although CH$_4$ absorption might occur occasionally. At the SP site, CH$_4$ fluxes were in the range of −0.02 to 0.51 mg m$^{-2}$ h$^{-1}$, with a mean value of 0.21 mg m$^{-2}$ h$^{-1}$ in the measuring period from June to September in 2010. In 2011, CH$_4$ fluxes ranged from 0.02 to 1.35 mg m$^{-2}$ h$^{-1}$ during the entire growing season at the SP site, and the mean seasonal flux was 0.56 mg m$^{-2}$ h$^{-1}$. CH$_4$ fluxes measured from the EP site were significantly higher than that from the SP site, which ranged from −0.01 to 2.28 mg m$^{-2}$ h$^{-1}$ with a mean flux of 1.02 mg m$^{-2}$ h$^{-1}$ in 2010 and −0.08 to 3.51 mg m$^{-2}$ h$^{-1}$ with a mean flux of 0.80 mg m$^{-2}$ h$^{-1}$ in 2011. In the present study, CH$_4$ fluxes obtained through static chambers during the growing seasons (−0.08−3.51 mg m$^{-2}$ h$^{-1}$) are greatly higher than that from Alaskan upland tundra (Bartlett et al., 1992), and they are similar in range to those from boreal raised bog (Pelletier et al., 2007) and subarctic/arctic fen (Christensen, 1993). The CH$_4$ emissions are much lower than those from the BOREAS peatlands (Bubier et al., 1995) and Xiaoxing'an Mountain peatlands (Sun et al., 2011).

Figure 4. Seasonal variation of dissolved pore water CH$_4$ concentrations at different soil depths was determined for the study site during sampling in 2011.
an efficient pathway for methane to liberate from peat to the atmosphere (Joabsson et al., 1999). In addition, as temperature increased, thaw depth of permafrost gradually increased, which can create appropriate soil circumstances such as saturation status and re-release of substrate previously preserved in the frozen layer for methanogenesis and methanogenesis (Yavitt et al., 2006). Therefore, the magnitude of CH$_4$ dependent on soil temperature was the important limiting factor for the CH$_4$ emission rate in the early growing season. The weak statistical relationship between methane emission and temperature at the peatland site during the growing season probably reflected the high spatial variability in emission rates at the plots, fluctuations in water table position, and seasonal changes in vegetation cover.

In general, water table position acted as a creation of aerobic and anaerobic conditions in the peat soil profile, which determined peatland CH$_4$ emissions. Studies have revealed that CH$_4$ fluxes increased from soils under elevated water tables, or high soil moisture contents (Moore and Knowles, 1989). In this study, soil moisture was large due to low evapotranspiration in the early growing season, but CH$_4$ fluxes were very low. A possible reason was that CH$_4$ production in anaerobic conditions was constrained by low soil temperature and limited substrate supply, and part of CH$_4$ might be consumed in the aerobic layer during the process of transmission to the atmosphere. As the growing season developed (Period II), the positive correlation between CH$_4$ emission and water table depth was shown (Table 2). This suggests that the effects of water table depth on methane emission will be enhanced under appropriate temperature conditions. It was consistent with other studies that found similar relationships, conducted in boreal peatlands (Roulet et al., 1993). A higher water table depth caused by summer precipitation and permafrost thaw might result in a larger anoxic CH$_4$ production zone and stimulate emissions.

This study was performed in the mountain peatland located in the southern margin of the Eurasian permafrost zone where the active layer depth has been increasing in recent decades (Jin et al., 2000). Some previous studies have shown that CH$_4$ flux correlated well with active layer depth in peatlands underlain by permafrost (van Huissteden et al., 2005). In our study, we found a positive correlation between thaw depth and the gas fluxes of CH$_4$ (Table 2), which was consistent with the above mentioned studies. However, Wille et al. (2008) reported that CH$_4$ flux did not correlate with the thaw depth in arctic tundra. The reasons they drew were that the majority of CH$_4$ originated from the upper soil layers, and the contribution of deep soil layers to methane emissions was small due to the temperature gradient in the thawed active layers and temperature dependence of microbial activity. However, recent studies reported that layers nearest the top of the permafrost (50–100 cm) in Alaska and Siberia contained higher CH$_4$ concentrations, which suggest that the majority of CH$_4$ will release from the eroding permafrost (Michaelson et al., 2011). Song et al. (2012) observed high CH$_4$ concentration in the refrozen active layer and upper permafrost layer in our study region, which could partly explain high CH$_4$ flux in the late growing season when the active layer reached tens of centimeters. The high CH$_4$ content in the permafrost might be originated from modern methanogenesis by cold-adapted methanogenic archaea in permafrost soil (Wagner et al., 2007) and release of trapped CH$_4$ formed in the unfrozen active layer during previous winter. It is also possible that CH$_4$ production took place in the freshly thawed permafrost due to the recovery of the bacteria from the upper permafrost (Coolen et al., 2011). In our study, we observed decreasing CH$_4$ flux with increasing thaw depth during the late growing season. This can be explained by decreasing air and surface soil temperatures constraining CH$_4$ production and little root survival in deeper soil layers, which limits CH$_4$ transport and emission.

The magnitude of CH$_4$ concentration in soil pore water increasing with depth indicated that CH$_4$ production was high in the deep saturated soil layer. The seasonal variation in CH$_4$ emission was significantly correlated with mean soil pore water CH$_4$ (Table 2). It implied that the magnitude of soil pore water CH$_4$ controlled CH$_4$ emission rates in the peatland. Our results are in agreement with Nouchi and Mariko (1993), who reported that CH$_4$ emission rate was proportional to pore water CH$_4$ concentration. Soil pore water containing high CH$_4$ concentrations was in correspondence with the EP site CH$_4$ flux rates recorded in late growing season.

### Table 2. Correlation coefficients between mean CH$_4$ fluxes and abiotic factors during the sampling period of 2011.

<table>
<thead>
<tr>
<th>Temperature* (°C)</th>
<th>Water table depth (cm)</th>
<th>Active layer depth (cm)</th>
<th>Pore water CH$_4$ concentration (µmol l$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Period I</td>
<td>Period II</td>
<td>Entire</td>
</tr>
<tr>
<td>Mean CH$_4$ flux</td>
<td>0.721*</td>
<td>−40.491</td>
<td>0.033</td>
</tr>
<tr>
<td>(mg m$^{-2}$ h$^{-1}$)</td>
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<tr>
<td>Pore water CH$_4$</td>
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<td>concentration (µmol l$^{-1}$)</td>
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* Correlation is significant at 0.05 levels; ** correlation is significant at 0.01 levels. Period I and II were arbitrarily defined at before and after 8 July 2011.

* Average temperature between 5 and 10 cm below peatland surface.
This suggests that plants at the EP site are more effective at transporting CH4.

We found that CH4 emission from the EP site was significantly higher than that from the SP site (Fig. 3). This can be partly explained by the presence of sedges (*Eriophorum vaginatum*) between the two sites. At the EP site, the dominant plant was *Eriophorum vaginatum*, classified as a vascular plant, while the SP site was dominantly covered by *Sphagnum* species, dwarf shrubs, and sparse *Eriophorum vaginatum*. We observed that the above-ground biomass of *Eriophorum vaginatum* from the SP site was much lower than that from the EP site (Table 1). The vascular plants of peatland could play an important role in gas exchange between the land and the atmosphere (Joabsson et al., 1999). In addition, CH4 transport through *Eriophorum* was the major pathway for CH4 fluxes (Frenzel and Rudolph, 1998). We also found that methane fluxes would decrease 77% and 73% from the EP site in 2010 and 2011 after cutting the above-ground part of *Eriophorum vaginatum* (Fig. 5). However, vascular plants might act as conduit for transferring oxygen to the rhizosphere, which both inhibits archael CH4 production and enhances methanotrophy. Yet, Frenzel and Rudolph (1998) found that oxidation of CH4 was negligible during its passage through *E. angustifolium*. In addition, root exudates and fine root litter of *Eriophorum* could stimulate CH4 production. Ström et al. (2011) reported that *Eriophorum* secreted more organic acids than other highly bio-available organic matters that could be easily utilized by methanogens in arctic wetland. Mosses contributed less significantly to active gas transport since they did not develop real root systems in peat (Sheppard et al., 2007). Otherwise, CH4 oxidation was reported from mosses originating from high-latitude wetlands, which decreased CH4 emissions from anoxic conditions (Larmola et al., 2010). So, different compositions of vegetation in peatland can explain the spatial variation of CH4 fluxes.

**4 Conclusions**

Seasonal methane fluxes were measured from a boreal peatland ecosystem in a continuous permafrost zone for two consecutive years. Seasonal average CH4 fluxes ranged from 0.21 to 1.02 mg m$^{-2}$ h$^{-1}$, with an apparent seasonal variation. Our results showed that environmental factors such as temperature and water table level were not responsible for regulating temporal variations of methane emission. CH4 emission rates during the growing season were strongly controlled by plant, active layer depth and CH4 concentrations in soil pore water. It implies that permafrost peatland under warming conditions can create a positive feedback to climate change due to increased CH4 emission through altering plant composition and increasing active layer depth. As CH4 emission from ecosystems depended on the balance of CH4 production and oxidation, the determination of the seasonal potential CH4 productions and oxidations in soil layers might provide some evidence for explanation of the seasonal and spatial variations of CH4 fluxes from boreal peatland ecosystems. In addition, future studies should focus on exploring the origination of plenty of CH4 in lower permafrost layers and soil pore water at tens of centimeters depth in peatland, which might promote our understanding of methane emission from peatlands in permafrost zones.

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