Multi-technique assessment of spatial and temporal variability of methane fluxes in a peat meadow

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Abstract

Methane fluxes measured in a eutrophic peat meadow in the Netherlands dominated by vascular plants showed high spatial and temporal variability. To elucidate this variability as well as the underlying processes, various measurement techniques were used: soil gradients of methane concentrations, the chamber method, and the eddy covariance technique. Additionally, soil temperature at multiple depths, soil water level, root depth, carbon dioxide fluxes, incoming radiation, atmospheric pressure, wind speed, friction velocity latent, heat fluxes, and living biomass were monitored.

A comparison of the measurement techniques showed that: a) the soil gradient method and the chamber method showed comparable methane fluxes only at a site with low water table and shallow roots, while at other sites methane fluxes were underestimated with the soil gradient method by an order of magnitude; b) a footprint analysis showed that the chamber method and eddy covariance showed similar methane fluxes for the different land elements (dry land, wet land, and ditches plus borders). However, when up-scaling the chamber measurements over time using a regression model based on soil temperature, methane emissions were overestimated by 37% compared to the eddy covariance data. The chamber method was the best technique to assess spatial variability, while eddy covariance was best for assessing temporal variability as well as up-scaling. The soil gradient method for methane fluxes should be used with great care, and probably generates reliable results only in areas with low water table and shallow roots.

Both the chamber method and eddy covariance showed significant spatial variability, which was best explained by soil water level in combination with root depth patterns. Together, these variables probably determined the net methane production and the available mechanisms for methane transport to atmosphere. The methane fluxes showed strong temporal variability at different scales: diurnal cycles, significant day-to-day variability, and seasonal variations. Clear diurnal cycles of methane fluxes were observed synchronous to incoming radiation, latent heat and net ecosystem exchange, but not synchronous to temperature. This suggested that stomatal opening and/or pressurized convective throughflow were important mechanisms for gas transport through plants. The variability at the day-to-day scale was best explained by soil temperature below the water
level combined with soil water level. Highest methane fluxes were observed during summer and lowest fluxes during autumn and winter. The vegetation density together with temperature and length of day-light probably determined this seasonality. Also, temporal variability varied spatially, probably due to the water table and root depth.

5.1 Introduction

Methane is currently causing 20% of the part of the greenhouse gas effect that is added to the normal greenhouse gas affect by human action. Peatlands and wetlands are the most important natural source of methane (Foster et al., 2007). Changes in climate and management affect water dynamics and temperature in peatlands and wetlands, and as a result methane emission may become even larger (Thomas et al., 1996; Christensen et al., 2003; Denman et al., 2007). The variability of methane fluxes and the underlying processes have been the subject of investigation during the past decennia; however, it is not yet well understood.

When assessing the variation in methane fluxes from subsoil to atmosphere, two aspects should be considered. First, the microbial processes of methane production and oxidation in the soil, and second, the mechanisms that transport the methane gas to the atmosphere (e.g. Whalen, 2005; Van Huissteden et al., 2005). Methanogenesis by microbes is a form of anaerobic respiration, in which microbes use carbon as the terminal electron acceptor (Schlesinger, 1991; Whalen, 2005). Most favourable conditions for this process are water saturated (anaerobic) soils with high organic content and high temperatures (e.g. Christensen et al., 2003). The availability of easily decomposable organic matter (root exudates and fresh organic matter) enhances methanogenesis, while high soil acidity has a limiting impact (Dunfield et al., 1993; Whalen, 2005; Van Huissteden et al., 2005). Under aerobic conditions, methane is oxidized by methanotrophic microbes that use oxygen as terminal electron acceptor (Schlesinger, 1991; Whalen, 2005). Also, anaerobic oxidation of methane has been observed as a microbial process using sulphate or nitrate instead of oxygen as the terminal electron acceptor (Raghoebarsing et al., 2006; Thauer and Shima, 2006). However, at our research site no anaerobic methane oxidation was detected (personal communication, K.F. Ettwig).

The transport of methane from the subsoil to the atmosphere follows three main pathways: diffusion through the anaerobic and aerobic soil, transport through aerenchyma in roots and stems of vascular plants and ebullition (Schlesinger, 1991; Walter and Heimann, 2000; Christensen et al., 2003; Whalen, 2005; Van Huissteden et al., 2006). Passive molecular diffusion through soils is a relatively slow process during which a certain amount of methane is oxidized. Concerning gas transport through vascular plants, two major mechanisms have been identified: molecular diffusion and pressurized throughflow convection (Whiting and Chanton, 1996). Plant transport is found to be a very important transport mechanism, which can contribute up to 62% to total methane emissions in peat areas (Grünfeld and Brix, 1999). On the other hand, vascular plants transport oxygen down into the rhizosphere, creating aerobic conditions favourable for methane oxidation, thereby reducing methane emissions (Roura-Carol and Freeman, 1999; Wang and Han, 2005; Whalen, 2005). Ebullition occurs in anaerobic soils over-saturated with gases. Gas
bubbles that contain methane form resulting from the high gas concentrations and move upwards to the atmosphere through water logged soils. Ebullition shows high temporal and spatial variability (Christensen et al., 2003; Strack et al., 2005), but generally increases with temperature increase (Beckmann and Lloyd, 2001; Marik et al., 2002; Christensen et al., 2003).

Differences in methane emissions from varying peatland and wetland areas have been observed by many researchers (e.g. Moore and Knowles, 1990; Grünfeld and Brix, 1999; Van den Pol-van Dasselaar et al., 1999; Christensen et al., 2003; Best and Jacobs, 1997; Pelletier et al., 2007; Drösler et al, 2008). However, observations of high spatial variability within a research area are less frequent (Waddington and Roulet, 1996; Hirota et al., 2004; Van Huissteden et al., 2005; Waddington and Day, 2007). In previous research, spatial variability of methane fluxes was attributed most often to differences in water level (e.g. Moore and Knowles, 1990; Van den Pol-van Dasselaar et al., 1999; Van Huissteden et al., 2005; Waddington and Roulet, 1996; Jungkunst and Fiedler, 2007; Waddington and Day, 2007), nutrient availability and net carbon uptake (Whiting and Chanton, 1993; Christensen et al., 2003; Van Huissteden et al., 2005) and/or differences in soil temperature (Beckmann and Lloyd, 2001; Christensen et al., 2003). Also, vegetation differences were found to be an important driver of spatial variability (Sheppard and Lloyd, 2002; Van Huissteden et al., 2005; Ding et al., 2005; Jungkunst and Fiedler, 2007).

At the temporal scale, the temperature dependency of methane fluxes has been established by a wealth of studies (e.g. Zimov et al., 1997; Bubier et al., 2005; Treat et al., 2007; Rinne et al., 2007). However, longer term changes of methane fluxes have also been shown to be related to persistent water level changes with inherent vegetation transitions (Juuinen et al., 2003; Hirota et al., 2004; Treat et al., 2007; Pelletier et al., 2007). Additionally, at lakes and areas with high coverage of open water surfaces a relation between CH$_4$ fluxes and atmospheric pressure differences and/or (near) surface turbulence was observed. This was found to be the result of enhanced gas ebullition from water surfaces during periods with low atmospheric pressure or high turbulence (Mattson and Likens, 1990; Fan et al., 1992; Casper et al., 2000; Wille et al., 2008; Sachs et al., 2008). Considering shorter time scales, diurnal cycles were found in various studies and were reported to result from molecular diffusion and related to temperature (Schütz et al., 1989a; Schütz et al., 1989b; Whiting and Chanton, 1993; Friborg et al., 1997), while others reported diurnal cycles that resulted from stomatal opening (Knapp and Yavitt, 1992) and/or pressurized convective throughflow due to strong pressure- or temperature gradients between plant aerenchyma and atmosphere, and were closely related to light intensity (Brix et al., 1992; Chanton et al., 1993; Thomas et al., 1996; Hirota et al., 2004; Wang and Han, 2005). At other measurement locations, no diurnal variations were observed at all (Rinne et al., 2007; Sachs et al., 2008).

Measurement techniques of methane fluxes comprise mainly the chamber method and the micrometeorological eddy covariance technique. Although the number of landscape scale measurements using eddy covariance is increasing (e.g. Fan et al., 1992; Fowler et al., 1995; Friborg et al., 1997; Rinne et al, 2007; Lohila et al. 2007; Kroon et al., 2007; Wille et al., 2008; Sachs et al., 2008), the chamber method is dominant (Christensen et al, 2003).
Chapter 5

The two techniques cover different scales both in time and space: the chamber method covers up to 1 m$^2$ spatially and measurements are discontinuous, while eddy covariance covers 100 – 1000 m$^2$ and measurements are continuous over time. The comparability of the approaches has however not yet been very well established.

Methane fluxes have been described in previous research at varying temporal and spatial scales, and a multitude of environmental factors and processes have been shown to influence methane fluxes. However, integrated studies at the field scale were scarce, and sometimes the different studies appeared to contradict. In this paper, we aim to determine the comparability of the flux measurement techniques and the advantages and disadvantages of both techniques. Furthermore, we aim to elucidate the spatial and temporal variability of methane fluxes at the field scale (1 – 1000 m$^2$) as well as perform an integrated assessment of the underlying processes and their relative importance. For this purpose, various measurement techniques were used to determine methane fluxes: the chamber method, soil gradients of methane concentrations and the eddy covariance technique. Additionally, a number of environmental variables were monitored in order to assess the processes underlying methane fluxes. Based on previous findings, the data were subjected to a variety of statistical analyses. In this manner, the driving variables and processes underlying the spatial and temporal variability of methane fluxes could be investigated.

5.2 Methods and instrumentation

5.2.1 Site description

The Horstermeer measurement site is located on former agricultural land in a drained natural lake in the central part of the Netherlands (52.144° N, 5.043° E) and was described extensively by Hendriks et al. (2007). The research site has been taken out of agricultural production in 1998 and developed into semi-natural grassland. The two meter thick soil consisted of eutrophic peat, overlain with organic-rich lake deposits and was overlying eolian sands of Pleistocene age. After the site was taken out of agricultural production, the ditch water table was raised to approximately 10 cm below the land surface. The area had a surface elevation of -2.2 m (below sea level), a flat topography and consisted of grassland dissected by approximately equidistant, parallel ditches (width: 1 to 3 m) bordered with reeds (Fig. 5.1 and Fig. 5.2). The main wind direction in the area was southwest, while also a northern wind direction was often observed. The vegetation at the research site was transitional and consisted of vascular plants (main species: Holcus lanatus, Urtica dioica, Glyceria maxima, Juncus effusus, Phalaris arundinacea, Phragmites australis and Typha latifolia). The Horstermeer polder was situated in a moderate maritime climate with an average air temperature of 9.8 °C and an average precipitation of 793 mm yr$^{-1}$. The growing season started at the end of March or the beginning of April and ended at the end of August or the beginning of September (Hendriks et al., 2007). However, some plant species were evergreen (e.g. Holcus lanatus, Juncus effusus).
Multi-technique assessment of methane fluxes in a peat meadow

Figure 5.1: Overview of the Horstermeer site including all measurement locations.
5.2.2 Chamber measurements and additional soil observations

Chamber measurements of CH$_4$ fluxes and ecosystem respiration ($R_{eco}$) were made with a Photo Acoustic Field Gas-Monitor (type 1312, Innova AirTech Instruments, Ballerup, Denmark) connected with tubes to closed, dark chambers (Hendriks et al., 2007; Van Huissteden et al., 2005). Between January 2004 and November 2007, CH$_4$ fluxes have been measured at the sites A, B, C, D and E (Fig. 5.1 and Fig. 5.3). At each of these sites, 40 manual chamber measurements were made with a measurement frequency of once per two weeks during the growing season and once per month during the winter period. In January 2006 sites F, G and two sites at the ditch water surface (d1 and d2) were added. Between January 2006 and October 2007, 28 manual chamber measurements were made at both sites F and G, and 22 measurements were made at each ditch site. The measurement frequency here was the same as at the other chamber sites. Temperature at 0.01 m below the soil surface ($T_{s1}$) and temperature at 0.01 m below the ditch water surface ($T_{water}$) were measured at all sites simultaneous with the chamber measurements. Starting in September 2005, temperature at 0.15 m below the soil surface was measured ($T_{s15}$) at all land sites simultaneous with all chamber measurements too. Also, soil water level ($WL_{soil}$) was measured at all land sites simultaneous with the chamber measurements.

Figure 5.2: Satellite images of the south west part of the Horstermeerpolder. Image (a) shows the area around the eddy covariance tower (which was located at the cross) and the average 99% footprint area over the three measurement periods (with a diameter of 480 m). Also, vegetation patterns and ditches are visible in this image. Image (b) shows a close up of the footprint area around the eddy covariance tower and the land elements defined for the footprint analysis, consisting of dry land (green), relatively wet land (grey) and ditches plus ditch borders (blue). The measurement area of figure 1 is indicated by the rectangle around the tower.
measurements. In June 2007 soil cores of $1.17 \times 10^{-4}$ m$^3$ were cut at two locations and at three depths (0.05–0.10 m, 0.15-0.20 m and 0.25-0.30 m) near each chamber site. The water filled porosity was calculated as the water filled pore space percentage after saturation of the soil cores. After sampling the cores were saturated with water and weighed, next the samples were dried for two days in the oven and weighed again. The water filled pore space percentage was determined from the difference in weight between the cores in saturated state and dry state, knowing the density of water (1000 kg m$^{-3}$). The average water filled porosity per chamber site was calculated from the three depths at the two locations that were sampled for each site. The soil and root mass were measured at all flux chamber sites by sampling soil cores from three depths (-0.05 to -0.10 m, -0.15 to -0.20 m and -0.25 to -0.30m) every six weeks during the growing seasons of 2006 and 2007. Root mass was determined by peptization and sieving out the sediment and weighing the dry root material. With this method, root mass smaller than 10 g m$^{-3}$ could not be detected. Also, roots with a diameter smaller than 0.5 mm could be washed out during the sieving process.

5.2.3 Pore water sampling and analysis

At sites A, C and E, water samples were taken from soil filters in the pore water in the clayey peat at 0.10 m, 0.30 m, 0.50 m, 0.70 m, 0.90 m, 1.10 m and 1.30 m depth (Fig. 5.1) to determine the gradient of dissolved CH$_4$ concentrations in the soil. Samples were collected using glass filters installed in a borehole with a diameter of approximately 25 mm and connected to the surface using small diameter Teflon tubes with an internal diameter of 2 mm. Samples were taken using a 10 ml syringe for drawing up the water, after which the sample was sucked into a 10 ml vacuum extainer vial connected to the syringe by a three-way stopcock. All sampling was performed anaerobic. Dissolved CH$_4$ was analyzed with a gas chromatograph (Hewlett Packard 5890A, Avondale PA, USA) from triplicates of the gas-filled headspace of the extainers that occupied 10 to 30% of the extainer vial space. The total CH$_4$ concentration in the sampled water volume was calculated according to Henry’s law using the gas-water solubility coefficient of CH$_4$ (0.035) and the ratio between water and gas-filled part of the extainer. At each of the three sites (A, C and E), samples were taken at all seven depths at eleven days between January 2004 and June 2006. The measurement frequency was irregular, but sampling occasions were simultaneous with chamber measurements at these sites.

5.2.4 Flux calculation from soil CH$_4$ gradient

The CH$_4$ fluxes were calculated for the soil layer between -0.30 m and -0.10 m depth and for the top soil layer between -0.10 m depth and atmosphere according to the method of Pihlatie et al. (2007). Calculations were made as follows:

$$F_i = -D_i \frac{C_{i+1} - C_i}{(l_i + l_{i+1})/2}$$  (5.1)
Where $F_i$ is the CH$_4$ flux from the soil layer $i$ to the soil layer above it $i+1$ (mg m$^{-2}$ hr$^{-1}$), $D_i$ is the diffusion coefficient of CH$_4$ in the soil layer $i$ (m$^2$ hr$^{-1}$), $C_{i+1}$ is the gas concentration (mg m$^{-3}$) in the layer $i+1$, $C_i$ is the gas concentration (mg m$^{-3}$) in the layer $i$, and $l_i$ and $l_{i+1}$ are the thickness of the soil layers (m), respectively. In the case of the top soil layer to atmosphere it was assumed that most of the resistance to transport occurred in the soil: in this case $D_i$ is that of the top soil layer and the distance $(l_i + l_{i+1})/2$ is the depth of that layer divided by 2. Also, ambient atmospheric gas concentrations were assumed at the top of the top soil layer. The diffusion coefficients ($D$) of CH$_4$ in the soil were calculated from the diffusion coefficients in the water and air ($D_0$) according to Troeh et al. (1982):

$$
\frac{D}{D_0} = \left( \frac{\phi - u}{1 - u} \right)^h
$$

(5.2)

Where $\phi$ is the water or air filled porosity of the soil (m$^3$ m$^{-3}$) and $u$ and $h$ are empirical parameters obtained from the literature (Troeh et al., 1982). $D_0$ values for CH$_4$ in water solution and air based on literature were used (Tai et al., 1978; Winkelmann, 2007). $D_0$ for both CH$_4$ in water solution and in air were dependent on temperature and were calculated as follows:

CH$_4$ in water solution:

$$
D_0 = 2 \times 10^{-6} \times e^{(0.0583\times T_{15})}
$$

(5.3)

CH$_4$ in air:

$$
D_0 = 7 \times 10^{-6} \times e^{(0.0055\times T_{31})}
$$

(5.4)

The soil layer between -0.30 m and -0.10 m depth was water saturated permanently and $D_0$ was fully determined by eq. 5.3. The water saturation of the top soil layer varied over time and was estimated from WL$_{soil}$ at the sampling day. A weighed average of $D_0$ values for CH$_4$ in water solution and in air was used for this layer for each sampling occasion.

5.2.5 Eddy covariance measurements and continuous meteorological, soil temperature and water level measurements

CO$_2$ concentration and water vapour concentration were measured at 10Hz rate with a LI-7500 open path infrared gas analyzer (LI-COR Lincoln, NE, USA) directed into the prevailing wind direction (southwest), and wind speed in three dimensions and air temperature were measured with a Windmaster Pro 3-axis ultrasonic anemometer (GILL Instruments Limited, Hampshire, UK). Both instruments were installed at 4.3 m above the surface at the Horstermeer measurement site. CH$_4$ concentrations were measured at 10 Hz rate with a Fast Methane Analyser (type DLT-100, Los Gatos Research Ltd.), which was attached to a dry vacuum scroll pump (XDS35i, BOC Edwards, Crawly, UK). The gas inlet filter was positioned at 4.3 m above the surface and 0.2 m away from the LI-7500. The footprint of the eddy covariance tower consisted of abandoned peat meadow, dissected with parallel ditches (Fig. 5.2).
The EUROFLUX methodology (Aubinet et al., 2000) was applied to the eddy covariance data to calculate the fluxes of momentum, sensible and latent heat (H and LE), CO$_2$ and CH$_4$ on a thirty minute basis. The Webb correction for density fluctuations arising from variations in water vapour (measured with the LI-7500) was applied according to Leuning and Moncrieff (1990) for both the open path and the closed path eddy covariance set-up. The Webb correction for density fluctuations arising from variations in temperature that was applied to the open path CO$_2$ measurements, was not required for the closed path measurements of CH$_4$ (Leuning and Moncrieff, 1990). The tube length of the set-up was over 1000 times the inner diameter of the tube and therefore the air temperature in the measurement cell could be considered stable. Frequency loss corrections for closed-path systems were applied according to the theory of Leuning and King (1992). Since 3-axis ultrasonic anemometers were found to under measure wind speed at large angles, the method of Nakai et al. (2006) was used to apply the angle of attack dependent calibration (Gash and Dolman, 2003; Van der Molen et al., 2004). Recently, Burba et al. (2008) developed a correction for the effect of instrument surface heat exchange on open-path eddy covariance measurements of CO$_2$ fluxes during winter periods. In this research the so called Burba-correction was not applied to the data, because no measurements were made during winter and temperatures were above 10$^\circ$C for most of the measurement days. Moreover, in case of the CH$_4$ eddy covariance measurements, the correction was not applied since this method consisted of a closed-path set-up. A quality check was performed during which all outliers and unrealistic values were removed from all eddy covariance data and meteorological data (Hendriks et al., 2007; Hendriks et al., 2008). The quality of the eddy covariance data was good and was described extensively by Hendriks et al. (2007) and Hendriks et al. (2008).

Micrometeorological measurements were executed at a tower close by the eddy covariance set up (Fig. 5.1). Additional micrometeorological measurements were executed at a tower close by the eddy covariance set up. Incoming and reflected shortwave radiation (Kipp and zonen, Delft, the Netherlands) and longwave radiation (Eppley Pyrgeometers, Eppley laboratory Inc., Model Precision Infrared Radiometers) were measured. All radiometers were installed at a height of 2.5 m (except incoming long wave radiation at 1.6 m height). Wind direction was monitored with a wind vane (Campbell Scientific Ltd, W200P) on top of the tower (4.6 m height) and wind speed with a cup anemometer (Vector Instruments model A100M/A100ML) at 3.0 m height. Additionally, air pressure was measured (SensorTechnics pressure transducers, model 144SC1216BARO).

Continuous soil water level ($W_{\text{soil}}$) recordings were made with pressure sensors installed in access tubes in the clayey peat top layer at two locations. Additionally, soil temperature was measured at 0.01 m ($T_{s1}$), 0.15 m ($T_{s15}$) and 0.40 m ($T_{s40}$). A detailed description of all continuous meteorological and soil measurements at the Horstermeer site was given by Hendriks et al. (2007). Additionally, aboveground living biomass of the dominant vegetation types was sampled, dried and weighed between autumn 2005 and autumn 2007 for each season. Typical amounts of weighed averages of living biomass for spring, summer and autumn were determined.
All continuous measurements except CH₄ fluxes were measured continuously from January 2005 until August 2008. Measurements of CH₄ fluxes were only of high quality during three periods: a two week period in summer 2006, a six week period in autumn 2007 and a one week period in spring 2008. For the continuous data in this paper, we have focused on these periods. A quality check was performed during which all outliers, unrealistic values and data collected during periods with low turbulence (friction velocity \( u_* < 0.09 \) m s\(^{-1}\)) were removed from eddy covariance data and meteorological data (Hendriks et al., 2007 and 2008). Resulting gaps were never larger than several half hours during the selected periods. Gap filling was performed through methods that are similar to Falge et al. (2001), with consideration of both the co-variation of fluxes with meteorological variables and the temporal auto-correlation of the fluxes (Reichstein et al., 2005). Since relations between CH₄ fluxes and other environmental variables were not yet known, linear interpolation was used to gap fill these data series. The NEE fluxes were partitioned in gross primary production (GPP) and ecosystem respiration (R\(_{eco}\)) using the model of Reichstein et al. (2005), which uses air temperature to partition the data.

### 5.2.6 Footprint Analysis

Since the analyses of the chamber measurements suggested significant spatial variation of CH₄ fluxes between land elements (dry land, relatively wet land and ditches plus ditch borders), a footprint analysis was made to confirm this with the eddy covariance measurements. Also, the comparability of the two flux measurement techniques could be tested. The method of Neftel et al. (2008) was used to perform footprint analyses and determine the source areas of CH₄ fluxes measured with the eddy covariance technique. This footprint tool is based on the footprint model developed for boundary layer conditions with non-neutral stratification by Kormann and Meixner (2001). Using an aerial photograph and ArcGIS for Windows, land surface quadrangles of three land elements (dry land, relatively wet land and ditches plus ditch borders) were defined in a two-dimensional Cartesian coordinate system with respect to the eddy covariance tower.

Starting from meteorological data readily available from the eddy covariance and micrometeorological measurements, the footprint tool (software: Visual basic) first calculated the two-dimensional footprint density functions according to Kormann and Meixner (2001). This footprint was then overlaid with coordinates of the specified quadrangular surface areas, and each area’s fraction within the footprint was computed. Next, the footprint contributions of the user-defined fields were computed. For this purpose, a grid with the size of the considered domain was overlaid with the quadrangles. The grid was rotated into the wind direction and had a fixed resolution of 200 points along wind and 100 points in crosswind direction. For each grid point lying inside any of the quadrangles, its contribution to the footprint was calculated and summed up, finally yielding the footprint contribution of each field. The footprint contributions of the quadrangles were given as percentages of the integral over the considered domain. For each thirty-minute data period, the footprint model determined the land surface ellipse that contributed 99% to the signal at the eddy covariance tower as well as the contributions per quadrangle in percentages. A check for the consistency of meteorological inputs was performed. If the inputs of a record result in \( z_m/L \) (\( z_m \) = measurement height; \( L \) = Obukhov
length) values smaller than -3 or greater than +3, the record was ignored, ensuring that footprint calculations were limited to the in \( zm/L \) range used by Kormann and Meixner (2001). Similarly, records with a ratio \( u^*/U > 1 \) were ignored (\( u_* \) = friction velocity; \( U \) = wind speed).

The footprint model was executed for the non-gap filled eddy covariance data sets of all three periods. Only half hours for which at least 75% of the mapped area fell within the 99% surface ellipse determined by the footprint model were used for further analyses. In figure 5.2, the average 99% distances of the footprints around the tower is shown as well as the surface areas of the land elements that were defined for the analysis. It was expected that when the contribution to the footprint of a land element with significantly deviating \( \text{CH}_4 \) fluxes increased, this would result in a significant change in \( \text{CH}_4 \) fluxes measured at the eddy covariance tower. This hypothesis was tested with regression analyses.

5.2.7 Statistical techniques

Multiple comparison analysis (MCA) with balanced one-way ANOVA analyses was performed to compare the means of two or more data sets and to determine which pairs of means were significantly different (Hochberg and Tamhane, 1987; Milliken and Johnson, 1992). Pearsonian product-moment correlation coefficient analysis was applied to determine correlation coefficients between variables (R). The significance of the correlation (p) was determined with significance test based on the Student's t distribution (t-test) with a significance level of 95%. Also, single and multiple regressions analyses of the data were made to compile statistical models. To test the significance of the regression, the root mean square error (RSME) was compared to the variance of the regression line using F-tests (variance ratio test) with 95% significance levels. The statistical analyses described above were performed with the computer programmes Matlab 7.6.0 for Windows and SPSS 14.0 for Windows.

5.3 Results I: chamber measurements and additional soil observations

5.3.1 Data series

Large variability of \( \text{CH}_4 \) fluxes was observed both spatially and temporally (Table 5.1 and Fig. 5.3). Highest \( \text{CH}_4 \) emission was detected at sites F and G near the ditches, while relatively low \( \text{CH}_4 \) emission was observed at the relatively dry sites in the middle part of the research area (sites A to E) and intermediate \( \text{CH}_4 \) emission at the ditch sites (d1 and d2). \( R_{eco} \) was lowest at the ditch sites, also relatively low at sites A and G and highest at sites B, D and E. All sites showed highest \( \text{CH}_4 \) emissions and \( R_{eco} \) during summer, intermediate emissions during spring and low emissions or, incidentally, uptake of \( \text{CH}_4 \) during autumn and winter. During 2007 both \( \text{CH}_4 \) emission and \( R_{eco} \) were low compared to the other measurement years.

Both Ts1 and Twater were highest during summer and spring periods, intermediate during autumn periods and low during winter, but never dropped below 0 °C. Twater was on average 2.0 °C higher than Ts1, while the differences in Ts1 between the land sites were small (Table 5.1 and Fig. 5.3). Ts15 was on average 2.3 °C lower than Ts1 measured over
the same period. WLsoil was relatively low in the middle part of the research area and lowest at sites A and B. During summer periods, minimum WLsoil values of -0.51 m were observed at sites A and B, while during autumn and winter WLsoil reached the soil surface at all sites. WLsoil at sites F and G reached the soil surface almost permanently. During 2007 $T_{s1}$ and $T_{water}$ were on average lower, and WLsoil was higher, than in the previous years.

At sites A and B, significant amounts of root mass occurred only between -0.05 m and -0.10 m depth. At the other sites, root mass was high in all sampled layers, implying root depths of at least -0.30 m (Table 5.1). The species *Juncus effusus* and *Typha latifolia* that occurred at sites E and G, respectively, are known for their deep rooting system up to -0.70 m to -1.00 m depth. With our sampling technique, however, these deep roots could not be observed.

### 5.3.2 Spatial variability

The large differences in flux magnitude between the chamber sites suggested considerable spatial variability within the field site. To test this and to determine the driving factors of the spatial variability, statistical analyses were performed. MCA based on CH$_4$ fluxes pointed out that the chamber sites could be divided in four significantly different groups: a group FG with sites F and G, a group DD with the ditch sites, a group CDE with sites C, D and E and a group AB with sites A and B (Table 5.2). MCA based on WL$_{soil}$ resulted in the same groups of sites, indicating a strong spatial relation between WL$_{soil}$ and CH$_4$.

<table>
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<th>CH$_4$ flux ($\text{mg m}^2 \text{hr}^{-1}$)</th>
<th>$R_{\text{eco}}$ ($\text{g m}^2 \text{hr}^{-1}$)</th>
<th>$T_{s1}/T_{\text{water}}$ ($^\circ\text{C}$)</th>
<th>WL$_{soil}$ (m below surf.)</th>
<th>rooting depth (m below surf.)</th>
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</tr>
<tr>
<td>site C</td>
<td>average</td>
<td>3.7</td>
<td>1.09</td>
<td>13.4</td>
<td>-0.09</td>
</tr>
<tr>
<td></td>
<td>stdev</td>
<td>3.5</td>
<td>0.85</td>
<td>6.7</td>
<td>0.10</td>
</tr>
<tr>
<td>site D</td>
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<td>1.68</td>
<td>13.8</td>
<td>-0.11</td>
</tr>
<tr>
<td></td>
<td>stdev</td>
<td>6.6</td>
<td>1.25</td>
<td>6.3</td>
<td>0.14</td>
</tr>
<tr>
<td>site E</td>
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<td>13.2</td>
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</tr>
<tr>
<td></td>
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<td>5.4</td>
<td>1.03</td>
<td>6.2</td>
<td>0.11</td>
</tr>
<tr>
<td>site F</td>
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<td>20.5</td>
<td>1.11</td>
<td>14.1</td>
<td>-0.01</td>
</tr>
<tr>
<td></td>
<td>stdev</td>
<td>20.3</td>
<td>0.86</td>
<td>5.4</td>
<td>0.04</td>
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<td>site G</td>
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<tr>
<td>d1</td>
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<td>0.26</td>
<td>16.3</td>
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</tr>
<tr>
<td></td>
<td>stdev</td>
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<tr>
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<td>0.20</td>
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<td></td>
<td>stdev</td>
<td>8.7</td>
<td>8.94</td>
<td>6.9</td>
<td>-</td>
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</table>

* At site E the species *Juncus effusus* occurred with rooting depth up to -1.00 m.

Table 5.1: Summary of the results from the chamber measurements (CH$_4$ fluxes and $R_{\text{eco}}$), $T_{s1}/T_{\text{water}}$, WL$_{soil}$ and root depth. Average values, maxima, minima and standard deviations for the whole measurement period are listed per flux chamber site.
Figure 5.3A: Flux chamber data (\(\text{CH}_4\) fluxes and \(R_{\text{eco}}\)), \(T_{s1}\) and \(WL_{\text{soil}}\) data at sites A, B, C, D and E between January 2004 and October 2007.
Figure 5.3B: Flux chamber data (\(CH_4\) fluxes and \(R_{eco}\), \(T_{s1}\) and \(WL_{soil}\) data at sites F, G, d1 and d2 between January 2006 and October 2007.
fluxes. However, when MCA was based on $R_{\text{eco}}$ other significantly different groups were found and when based on $T_{s1}/T_{\text{water}}$ no significant distinction between chamber sites could be made at all.

A regression analysis using site averages showed a significant relation between $\text{CH}_4$ flux and $\text{WL}_{\text{soil}}$, with exponentially increasing $\text{CH}_4$ emissions with rising $\text{WL}_{\text{soil}}$ (Table 5.3). The relation between average $T_{s1}/T_{\text{water}}$ and the average $\text{CH}_4$ flux at the sites was less obvious. Clearly, sites F and G were outliers, with extremely high $\text{CH}_4$ emissions for relatively low $T_{s1}$. Apparently, the effect of the high $\text{WL}_{\text{soil}}$ on $\text{CH}_4$ production dominated over the influence of $T_{s1}$. For the other sites, a significant relation was found, with exponentially increasing $\text{CH}_4$ emissions at increasing $T_{s1}/T_{\text{water}}$. Average $R_{\text{eco}}$ showed no significant relation with average $\text{CH}_4$ flux. Mainly sites F, G, d1 and d2 showed outlying values with relatively high $\text{CH}_4$ emissions for low $R_{\text{eco}}$.

<table>
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<th>Variable</th>
<th>sites in group</th>
<th>F</th>
<th>df1</th>
<th>df2</th>
<th>p</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\text{CH}_4$ flux</td>
<td>group FG</td>
<td>F and G</td>
<td>11.92</td>
<td>8</td>
<td>144</td>
</tr>
<tr>
<td></td>
<td>group DD</td>
<td>d1 and d2</td>
<td>7.03</td>
<td>6</td>
<td>112</td>
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<tr>
<td></td>
<td>group CDE</td>
<td>C, D and E</td>
<td>2.14</td>
<td>4</td>
<td>90</td>
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<tr>
<td></td>
<td>group AB</td>
<td>C and D</td>
<td>2.14</td>
<td>4</td>
<td>90</td>
</tr>
<tr>
<td>$R_{\text{eco}}$</td>
<td>group BDE</td>
<td>B, D and E</td>
<td>10.42</td>
<td>8</td>
<td>144</td>
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<tr>
<td></td>
<td>group AG</td>
<td>A and G</td>
<td>10.42</td>
<td>8</td>
<td>144</td>
</tr>
<tr>
<td></td>
<td>group DD</td>
<td>d1 and d2</td>
<td>5.49</td>
<td>6</td>
<td>147</td>
</tr>
<tr>
<td>Sites F and C did not show a significant difference from the other sites.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| $T_{s1}/\text{water}$ | group all | all sites | -- | -- | -- | -- |
| $\text{WL}_{\text{soil}}$ | group FG | F and G | 8.71 | 6   | 168 | <<0.01 |
|               | group DD   | d1 and d2 | (*) | -- | -- | -- |
|               | group CDE | C, D and E | 4.48 | 4   | 274 | <<0.01 |
|               | group AB   | C and D | 4.48 | 4   | 274 | <<0.01 |
| (*) Sites d1 and d2 are ditch sites and had a water layer of app. 1.5m on top of the soil year round and were considered a separate group. |

Table 5.2: Results from the balanced one-way ANOVA analyses and MCA (F-value, degrees of freedom (df1 and df2) and significance (p)), respectively based on $\text{CH}_4$ fluxes, $R_{\text{eco}}$, $T_{s1}/T_{\text{water}}$ and $\text{WL}_{\text{soil}}$.

5.3.3 Temporal variability

Due to the differences in soil conditions between the four groups of sites, differences in temporal variability and driving factors were expected. Based on the four groups of sites that were defined with the MCA, regression analyses of the exponential relations between $\text{CH}_4$ fluxes and the environmental variables ($T_{s1}$, $T_{s15}$, $\text{WL}_{\text{soil}}$, and $R_{\text{eco}}$) were done to test these hypotheses (Table 5.3). Group AB had relatively high significance for all regressions, suggesting that all variables were related to the $\text{CH}_4$ fluxes. Group CDE had a relatively high significance for the relation of $\text{CH}_4$ fluxes with $R_{\text{eco}}$ and low significance for the relations with $T_{s1}$ and $\text{WL}_{\text{soil}}$. Group FG had moderate significance for the relation of $\text{CH}_4$ flux with $T_{s1}$ and $R_{\text{eco}}$ and a low significance for the relations with $\text{WL}_{\text{soil}}$. Group DD showed a relatively significant relation of $\text{CH}_4$ fluxes and $T_{\text{water}}$, but a low significance for the relation with $R_{\text{eco}}$. The exponential relations of $\text{CH}_4$ flux with $T_{s15}$ were similar to those with $T_{s1}$. Multi-linear regression analyses did not result in significant improvement of the significance for any of the groups, suggesting collinearity of the environmental variables.
The relation of CH$_4$ flux with WL$_{soil}$ was negative, and significances were relatively low (Table 5.3). At the same time, WL$_{soil}$ and T$_{s1}$ showed significant negative linear relations for all groups with decreasing WL$_{soil}$ at increasing T$_{s1}$, which hampered detection of the independent influence of the two variables on CH$_4$ fluxes. To properly analyse the relation between WL$_{soil}$ and CH$_4$ fluxes, CH$_4$ fluxes were corrected for the influence of T$_{s1}$ according to eq. 5.5:

$$CH_4\text{ flux, corr} = CH_4\text{ flux} - \exp(a + b \times T_{s1})$$  \hspace{1cm} (5.5)

where $a$ and $b$ are the coefficients of the regression equations (Table 5.3). The resulting temperature-corrected CH$_4$ fluxes (CH$_4$ flux,corr) were plotted over WL$_{soil}$ values of group AB and FG and trend lines were drawn through the data (Fig. 5.4). For group AB, CH$_4$ flux,corr generally decreased with increasing WL$_{soil}$, and CH$_4$ flux,corr showed large variations for WL$_{soil}$ lower than -0.20 m. Group FG showed a generally increasing CH$_4$ flux,corr with higher WL$_{soil}$, and variability of CH$_4$ flux,corr was also high when WL$_{soil}$ was high. CH$_4$ fluxes from group CDE showed low significance for both the relation with T$_{s1}$ and WL$_{soil}$, therefore no sensible CH$_4$ flux,corr values could be determined.
To assess seasonal trends, all data were averaged for each season per group of sites over the period 2005 to 2007 (Fig. 5.5). At all groups seasonal trends were observed for CH$_4$ fluxes, $R_{\text{eco}}$, $T_{\text{soil}}$ and $W_{\text{soil}}$. However, due to the high within-season variability, the inter-seasonal variability was not significant for any of the variables. Nonetheless interesting was the variation in the trends of CH$_4$ fluxes between the four groups. Group AB showed no significant CH$_4$ emission except during summer, while CH$_4$ emissions at group CDE were relatively stable throughout the year. At group FG CH$_4$ fluxes were high, except in autumn, and showed extremely high within-seasonal variability. Group DD showed high CH$_4$ emission in summer, low CH$_4$ emission in spring and no clear fluxes in winter and autumn.
5.4 Results II: Soil concentrations and surface fluxes from soil gradients

At all three sites, the lowest CH$_4$ concentrations were observed near the soil surface, and the highest CH$_4$ concentrations at -0.70 m depth (Fig. 5.6). Below -0.70 m depth CH$_4$ concentrations decreased gradually. Site C showed relatively low CH$_4$ concentrations and high temporal variability; while relatively little variation over depth was observed. At site A, CH$_4$ concentrations were approximately twice as high as at the others site, except near the soil surface. Above -0.30 m depth, a clear transition to low concentrations was observed. Site E showed intermediate CH$_4$ concentrations at all depths and a clear transition to lower CH$_4$ concentrations from -0.70 m upward was observed. An increase in CH$_4$ concentrations over the course of the year was observed at site A over the whole profile, while at sites C and E such a trend was less clear.

Figure 5.5: Seasonal averages of chamber measurements (CH$_4$ flux and R$_{eco}$), $T_{soil}$ and WL$_{soil}$ per group of sites. Error bars indicate standard deviations.
Subsequently, the fluxes that were calculated from the soil CH\textsubscript{4} gradient were plotted over time with the simultaneously measured CH\textsubscript{4} surface fluxes obtained with the chamber method (Fig. 5.7). At site A, the soil CH\textsubscript{4} gradient method from the top soil layer showed surface fluxes comparable with that of the chamber technique for most of the measurement days. The results of the soil CH\textsubscript{4} gradient method for the soil layer between -0.10 and -0.30 m below the surface showed significantly higher fluxes at this site, mainly due to the large concentration differences between these soil layers. At the other sites (C and E), results from the soil CH\textsubscript{4} gradient method for both layers were comparable, while the chamber method showed CH\textsubscript{4} fluxes that were approximately ten times higher. During the winter periods the techniques were comparable at some measurement days at sites C and E.

Figure 5.6: Results from the measurements of dissolved CH\textsubscript{4} concentrations in the pore water over a soil profile at flux chamber sites A, C and E. Each line represents one measurement day and the thick line represents the average of all measurements. Note that the horizontal axis has a logarithmic scale.
Figure 5.7: Comparison of surfaces fluxes observed with the soil CH$_4$ gradient method in the upper part of the soil profile and the chamber method. Site A is represented in two plots to clarify the small scale variability of the surface fluxes at this site.
Multi-technique assessment of methane fluxes in a peat meadow

Figure 5.8: Data series of CH$_4$ fluxes, NEE and LE measured with eddy covariance as well as continuous data series of SW, U, T, and WL for the three measurement periods. For CH$_4$ fluxes and NEE, half hourly data as well as daily averages and daytime and night time averages have been plotted. Also, the daily average of the regression model based on chamber data has been plotted in the upper graph.

2006 period
2007 period
2008 period
5.5 Results III: eddy covariance, meteorological and continuous soil data

5.5.1 Data series

CH$_4$ emission was on average highest during the 2006 summer period, and lowest during the 2007 autumn period, while day-to-day variability was relatively low for the 2008 spring period (Fig. 5.8). Over the 2006 summer period and the 2008 spring period, on average net uptake of CO$_2$ was observed, indicating growth of the vegetation, while during the 2007 autumn period on average net emission of CO$_2$ was observed. This was in accordance with the amount of aboveground living biomass that was highest in summer (5.88 kg m$^{-2}$), lowest in autumn (0.51 kg m$^{-2}$) and intermediate in spring (2.85 kg m$^{-2}$). Day-to-day variability of NEE was low during the 2007 autumn period compared to the other periods. Short wave incoming radiation (SW$_{in}$) and LE were highest during the 2006 summer period and lowest during the 2007 autumn period. The main wind direction in the area was southwest, while also a northern wind direction was observed often. Wind speed (U) was similar for all periods, while friction velocity ($u_*$) was relatively high during the 2008 spring period (Fig. 5.8). During the 2006 summer period atmospheric pressure (P) varied between 1012 hPa and 1023 hPa, during the 2007 autumn period P varied between 1006 hPa and 1035 hPa, and during the 2008 spring period P gradually decreased from 1016 hPa to 1006 hPa.

Also, $T_{s1}$, $T_{s15}$ and $T_{s40}$ were highest during the 2006 summer period and lowest during the 2007 autumn period (Fig. 5.8), while the variability of all temperatures was relatively high for the 2007 autumn period. On average temperatures decreased with depth, but differences between $T_{s1}$, $T_{s15}$ and $T_{s40}$ were very small in the 2007 autumn period and largest in the 2006 summer period. WL$_{soil}$ was lowest during the 2006 summer period and highest during the 2007 autumn period. The variability of WL$_{soil}$ was large during the 2006 summer period and very low during the 2007 autumn period. These WL$_{soil}$ characteristics corresponded to the magnitude and day-to-day variability of LE during the respective periods. Due to the lack of outflow through the highly compacted soil, the lowering of WL$_{soil}$ within one day (WL$_{diff}$) resulted mainly from the magnitude of evapotranspiration at that day. WL$_{diff}$ was on average largest during the 2006 summer period and smallest during the 2007 autumn period.

5.5.2 Spatial variability: footprint analysis

Regression analyses were performed with a 95% confidence interval to determine the effect of the three land elements in the footprint of the eddy covariance tower (dry land, wet land and ditches plus borders) on the measured CH$_4$ fluxes. Analyses were performed for daytime data and nighttime data separately because consistent differences between nighttime and daytime fluxes, due to the diurnal cycle, had a disturbing effect on the results. For the 2006 summer period, significant linear relations were found for the contribution of each land element to the CH$_4$ flux measured at the eddy covariance tower, both for nighttime and daytime (Table 5.4 and Fig. 5.9). For daytime periods a 10% increase of ditches plus ditch borders in the footprint area resulted in an increase of CH$_4$
flux of 3.7 ± 1.4 mg m\(^{-2}\) hr\(^{-1}\), while for relatively wet land an increase of 1.1 ± 0.4 mg m\(^{-2}\) hr\(^{-1}\) and for relatively dry land a decrease of -0.9 ± 0.3 mg m\(^{-2}\) hr\(^{-1}\) was observed. The CH\(_4\) flux at each land element during this summer period could be estimated by using the linear regression equations and assuming the contribution of the respective land elements 100%. Daytime CH\(_4\) fluxes of 35.5 ± 15.6 mg m\(^{-2}\) hr\(^{-1}\), 7.7 ± 5.5 mg m\(^{-2}\) hr\(^{-1}\) and -3.0 ± 4.1 mg m\(^{-2}\) hr\(^{-1}\) were estimated for ditches plus ditch borders, relatively wet land and relatively dry land respectively. This corresponded well with CH\(_4\) emissions during summer observed with the chamber method (Fig. 5.9), except that with the chamber method no negative CH\(_4\) fluxes were observed at the relatively dry land (group AB), but low emissions.

<table>
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<tr>
<th>dependent variable</th>
<th>independent variable (field contributing to footprint)</th>
<th>n</th>
<th>regression type</th>
<th>a</th>
<th>b</th>
<th>p</th>
<th>RMSE</th>
</tr>
</thead>
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<td>41</td>
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<td>a + bx</td>
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<td>7.60</td>
<td>&lt;0.001</td>
<td>0.39</td>
<td></td>
</tr>
<tr>
<td>CH(_4) flux ditch+border 2006 night</td>
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<td>a + bx</td>
<td>-1.93</td>
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<td>&lt;0.001</td>
<td>0.40</td>
<td></td>
</tr>
<tr>
<td>CH(_4) flux dry land 2006 day</td>
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<td>a + bx</td>
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<td>0.90</td>
<td>0.175</td>
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<td>1.01</td>
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<tr>
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<td>1.17</td>
<td>4.23</td>
<td>0.090</td>
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</table>

Table 5.4: Results from the regression analyses of night time and daytime data of the footprint analysis for the 2006 summer period and the 2007 autumn period: type of regression equation, constants (a and b), significance (p) and root mean square error (RMSE).

For the 2007 autumn period, similar trends were observed, but no significant regression was found between CH\(_4\) flux and footprint area. This pointed at smaller differences between the CH\(_4\) fluxes from the three land elements, which corresponded with the lack of difference between the groups of chamber sites in autumn measured with the chamber method (Fig. 5.5). The 2008 spring period was too short for proper analyses of the footprint results.

5.5.3 Temporal variability

Besides the seasonal variability of CH\(_4\) fluxes described above (highest emissions in summer, lowest emissions in autumn and intermediate emissions in spring), temporal variability at shorter time scales was also observed from the continuous measurements. CH\(_4\) fluxes showed clear diurnal cycles in all measurement periods as well as day-to-day variability.
All continuous data series, when averaged per 30-minute period of the day for each measurement period, showed more or less pronounced diurnal cycles (Fig. 5.10). The diurnal cycles of CH$_4$ fluxes, NEE, SW$_{in}$ and LE showed clear similarity: the start of daytime (SW$_{in}$ > 20 W m$^{-2}$) coincided with the clear increase of CH$_4$ emissions, the uptake of CO$_2$ and an increase of LE, while at the start of night time (SW$_{in}$ < 20 W m$^{-2}$) the opposite occurred. Also, peaks of CH$_4$ emission, CO$_2$ uptake, SW$_{in}$ and LE occurred simultaneous at approximately 13:00hr. Daily cycles of $T_{s1}$, $T_{s15}$ and $T_{s40}$ and WL$_{soil}$ however, were less pronounced and lagged approximately three hours behind the diurnal cycle of the other variables. Diurnal cycles of CH$_4$ fluxes showed the largest amplitude during the 2006 summer period, while the amplitudes of the other periods were similar but smaller. Net CO$_2$ fluxes showed the largest amplitude during the 2008 spring period, while the amplitudes of the other two periods were similar. SW$_{in}$, LE, $T_{s1}$ and WL$_{soil}$ showed largest amplitudes during 2006 summer period and smallest amplitudes during the 2007 autumn period. All periods showed similar night time CH$_4$ emissions of approximately 0.9 mg m$^{-2}$ hr$^{-1}$.

Regression analyses were performed with daily averaged values from all three measurement periods (Table 5.5). Single regression models showed high significance for all variables, except for P and $u$. The single linear regression models with the highest significance used $T_{s15}$ as independent variable, while also the regression models using $T_{s1}$, $T_{s40}$, WL$_{soil}$, and WL$_{diff}$ had relatively high significance. In order to compile a regression model to predict and, perhaps, gap-fill CH$_4$ fluxes at the Horstermeer site, multiple linear regression analyses were done. Only when $T_{s40}$ and WL$_{soil}$ were combined did a multi-linear regression model result in a significant improvement of the significance (p<<0.001):
Figure 5.10: Diurnal cycles of eddy covariance data (CH\textsubscript{4} fluxes, NEE, LE, ET), SW\textsubscript{in}, T\textsubscript{s1} and WL\textsubscript{soil} per measurement period. Also, the diurnal cycle of CH\textsubscript{4} fluxes from the regression model based on chamber data has been plotted in the second graph.
\[
CH_4\text{flux} = -1.274 + 0.181 \times T_{40} - 1.816 \times WL_{soil}
\] (5.6)

and explained 65% of the variance of CH₄ fluxes, while both independent variables added significantly to the significance of the regression equation (p = 0.002 and p = 0.001, respectively).

Similar to the results from the chamber data, the relation between WL_{soil} and CH₄ flux was negative (decreasing CH₄ flux with higher WL_{soil}) and the linear relation between T_{s1} and WL_{soil} was strong. After correcting the CH₄ fluxes for T_{s1} according to eq. 5.5, the CH₄flux,corr data were plotted over WL_{soil} and a trend line was drawn through the data (Fig. 5.4). CH₄flux,corr was relatively low at WL_{soil}, around -0.20 m while with decreasing and increasing WL_{soil}, CH₄flux,corr increased.

<table>
<thead>
<tr>
<th>Dependent variable</th>
<th>Independent variable</th>
<th>n</th>
<th>Regression type</th>
<th>a</th>
<th>b</th>
<th>p</th>
<th>RMSE</th>
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<td>exp(a + bx)</td>
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<td>0.34</td>
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<td>0.26</td>
</tr>
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<td>R_{eco}</td>
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<td>a + bx</td>
<td>-0.76</td>
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<td>0.001</td>
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</tr>
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<td>a + bx</td>
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<td>0.134</td>
<td>0.41</td>
</tr>
<tr>
<td>CH₄ flux</td>
<td>u*</td>
<td>48</td>
<td>a + bx</td>
<td>-0.43</td>
<td>0.45</td>
<td>0.373</td>
<td>0.42</td>
</tr>
<tr>
<td>CH₄ flux</td>
<td>T_{s1}</td>
<td>48</td>
<td>exp(a + bx)</td>
<td>-0.84</td>
<td>0.09</td>
<td>&lt;0.001</td>
<td>0.22</td>
</tr>
<tr>
<td>CH₄ flux</td>
<td>T_{s15}</td>
<td>48</td>
<td>exp(a + bx)</td>
<td>-1.43</td>
<td>0.13</td>
<td>&lt;0.001</td>
<td>0.21</td>
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<tr>
<td>CH₄ flux</td>
<td>T_{s40}</td>
<td>48</td>
<td>exp(a + bx)</td>
<td>-2.60</td>
<td>0.22</td>
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<td>0.22</td>
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<tr>
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<td>WL_{soil}</td>
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<td>a + bx</td>
<td>0.94</td>
<td>-3.02</td>
<td>&lt;0.001</td>
<td>0.28</td>
</tr>
<tr>
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<td>WL_{diff}</td>
<td>48</td>
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<td>1.16</td>
<td>12.91</td>
<td>&lt;0.001</td>
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</tr>
<tr>
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<td>-exp(a + bx)</td>
<td>-4.46</td>
<td>0.18</td>
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<td>0.00</td>
</tr>
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<td>WL_{diff}</td>
<td>T_{s1}</td>
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<td>-exp(a + bx)</td>
<td>-10.82</td>
<td>0.46</td>
<td>&lt;0.001</td>
<td>0.00</td>
</tr>
</tbody>
</table>

Table 5.5: Results from the regression analyses of the relation between CH₄ fluxes measured with eddy covariance and all other environmental variables: type of regression equation, constants (a and b), significance (p) and root mean square error (RMSE). Also, the relation between WL_{soil} and T_{s1} and the relation between CH₄flux,corr and WL_{soil} have described. Analyses were based on daily averages from all three measurement periods.

5.5.4 Comparison of eddy covariance data and a regression model based on chamber data

The eddy covariance data series were compared with the CH₄ chamber measurements using the regression models based on the exponential relation between CH₄ fluxes and T_{s1} for the different land elements. To scale up the chamber data to the ecosystem level, the contributions of the land elements (dry land, relatively wet land and ditches plus ditch borders) determined by the footprint model for each 30-minute period were used. The
results of the regression model were in the same range as the eddy covariance data, but
daily averages were overestimated with 0.58 mg m\(^{-2}\) hr\(^{-1}\) on average (Fig. 5.8). The model
did however follow the general trend with increasing CH\(_4\) emission during the 2006
summer period and the decreasing trend during the 2007 autumn period.

5.6 Discussion and conclusions

In this paragraph, the results of the research were discussed, and, where possible,
conclusions were drawn. First, the comparisons of the techniques for measuring CH\(_4\)
fluxes were discussed; both the comparison between the soil gradient method and the
chamber method, and the comparison between the chamber method and the eddy
covariance method. Next, the observed spatial variability of the CH\(_4\) fluxes as well as
underlying mechanisms was described. Finally, the temporal variability of the CH\(_4\) fluxes
at various scales and underlying mechanisms was discussed.

5.6.1 Technique comparisons

Comparisons were made between the soil gradient method and the chamber method as
well as between the chamber method and the eddy covariance method. While the soil
gradient method and the chamber method were applied at the same spatial scale (< 1 m\(^2\)),
the eddy covariance method was applied at a significantly larger scale (100 \(- 1000\) m\(^2\)).
As a result, small scale spatial differences were largely averaged out by the eddy
covariance measurements. However, spatial variability within the footprint is reflected by
the eddy covariance signal depending on the footprint area. Local hotspots of CH\(_4\)
emissions can strongly influence the eddy covariance measurements. Temporally, eddy
covariance data are continuous over time thereby giving a good representation of the
patterns in temporal variability of CH\(_4\) fluxes in the fetch. Flux data determined with the
soil gradient or chamber method, however, are point measurements and patterns in
temporal variability cannot be distinguished easily.

The comparison between the soil gradient method and the chamber method showed that
these flux measurement techniques were well comparable at a site with low WL\(_{soil}\), and
that probably had diffusion as main gas transport mechanism. The comparability, which
was in agreement with findings of Pihlatie at al. (2007), increased the confidence in the
chamber method. However, for sites with deep rooting plants and/or high WL\(_{soil}\), and
dominant transport mechanisms were probably plant transport or ebullition, the soil
gradient method largely underestimated the surface fluxes. Plant transport and ebullition
both provide a shortcut through the upper aerobe soil layers, enabling gas with high CH\(_4\)
concentrations to be emitted from the soil to the atmosphere rapidly, and prohibiting
oxidation of CH\(_4\) in the upper soil layers. According to the findings in this research, the
gradient method should only be used in case of low WL\(_{soil}\) and areas with diffusion as
main gas transport mechanism through the soil. Estimating CH\(_4\) fluxes with the soil
gradient method from relatively unknown areas will probably generate large uncertainties.

The chamber method and the eddy covariance technique were compared in two ways.
First, the results of a footprint analysis of eddy covariance data were compared with the
results of chamber measurements from three land elements with significantly different
Chapter 5

CH₄ fluxes (ditches plus ditch borders, relatively wet land and relatively dry land). Although the results of the footprint model had relatively high uncertainties, the comparison showed that during summer, the eddy covariance technique measured comparable CH₄ fluxes for all land elements. For this period, the footprint analysis showed CH₄ fluxes of 35.5 ± 15.6 mg m⁻² hr⁻¹, 7.7 ± 5.5 mg m⁻² hr⁻¹ and -3.0 ± 4.1 mg m⁻² hr⁻¹ for ditches plus ditch borders, relatively wet land and relatively dry land respectively. The chamber method showed CH₄ fluxes of 22.8 ± 17.3 mg m⁻² hr⁻¹, 5.5 ± 5.6 mg m⁻² hr⁻¹ and 2.8 ± 3.2 mg m⁻² hr⁻¹ for the three land elements respectively for summer periods. This finding was in agreement with previous comparisons between the chamber method and the eddy covariance technique (Fan et al., 1992; Lohila et al., 2007; Friborg et al., 1997; Neftel et al., 2008). During autumn no significant distinction could be made between the land elements with the footprint model. This however corresponded with the lack of difference of CH₄ fluxes between the land elements in autumn obtained with the chamber method.

Secondly, a comparison was made between the continuous eddy covariance observations and a regression model based on flux chamber data and Tₛ₁, taking into account the relative coverage of the three land elements in the footprint area. Although the results were very well comparable, the chamber method overestimated daily CH₄ emissions with approximately 37%.

The overestimation was probably mostly due to the fact that the regression model based on Tₛ₁ was not sensitive enough to simulate changes in CH₄ fluxes. Probably because chamber measurements were only made during daytime, the relatively low nighttime CH₄ fluxes could not be simulated resulting an overestimation of the daily CH₄ emission. Additionally, the regression models for CH₄ fluxes based on chamber data and Tₛ₁ did not have very high significance; the relation between CH₄ fluxes and Tₛ₁ was not strong enough to use Tₛ₁ as a reliable variable for up-scaling the chamber data over time, resulting in high uncertainties. Instead, a statistical model based on Tₛ₄₀ and WLₙₐ₅₉ₚ₅ might give better results, since the regression analysis of the eddy covariance data pointed out that these variables predicted the CH₄ fluxes best, when combined. However, in this research no measurements of Tₛ₄₀ were made at the chamber sites.

Both flux measurement techniques showed to have advantages as well as disadvantages. Since the eddy covariance technique averages over a larger area, small scale heterogeneities were largely averaged out and the total flux could be determined over longer periods for a whole ecosystem. Spatial variations were not easily detected and knowledge of hotspot locations acquired with the chamber method was indispensable for proper footprint analysis. Spatial variability was best observed with chamber measurements; however, up-scaling chamber measurements posed problems. To scale up in space, a very detailed knowledge of the spatial patterns of environmental variables was required. To scale up over time, a regression model using Tₛ₁ was not sufficient. Additionally, ebullition probably caused large variability both spatially and temporally (Christensen et al., 2003; Strack et al., 2005). Using the eddy covariance method, a relatively reliable average of the fluxes of the whole ecosystem could be obtained. However, when using the chamber method or when up-scaling results from either
technique, ebullition will cause large uncertainties and further research should be done to decrease these uncertainties.

Regression models that were determined from the eddy covariance data showed higher significances than regression models based on chamber measurements. They better represented the whole ecosystem since the measurements were made on a larger scale, and consisted of daily averages instead of point measurements (as was the case with the chamber method). In further research, regression models deduced from the eddy covariance data could be compared with the results from process-based models. Also, an attempt could be made to improve regression models by introducing diurnal cycles into the model equations. Results from these modelling comparisons could improve prediction and up-scaling of CH$_4$ fluxes from peat meadow areas. Additionally, strategies for gap filling could be explored.

5.6.2 Spatial variability of methane fluxes

CH$_4$ fluxes measured in a peat meadow in the Netherlands, dominated by vascular plants, showed high spatial variability. Within a small area (one hectare) four land elements with significantly different CH$_4$ fluxes were found. Highest fluxes were observed at the saturated land near the ditches, intermediate fluxes from the ditch water surface, lowest fluxes from dry land in the middle part of the area and somewhat higher fluxes from sites with intermediate WL$_{soil}$ and deep rooting plants. Average CH$_4$ emissions of a 3.5 year measurement period based on the chamber measurements were respectively 23.1 mg m$^{-2}$ hr$^{-1}$, 7.6 mg m$^{-2}$ hr$^{-1}$, 1.2 mg m$^{-2}$ hr$^{-1}$ and 4.1 mg m$^{-2}$ hr$^{-1}$. Similar differences between land elements were found in previous studies (Moore and Knowles; 1990; Waddington and Roulet, 1996; Hirota et al., 2004; Van Huissteden et al., 2005; Waddington and Day, 2007). A footprint analysis of the eddy covariance data from our site showed differences between land elements comparable to the chamber data.

The spatial variability of CH$_4$ fluxes in the area was probably determined by WL$_{soil}$ in combination with root depth. Similar results were found by Hirota et al. (2004), while other research pointed solely at WL$_{soil}$ differences (Waddington and Roulet, 1996 Van den Pol-van Dasselaar et al., 1999; Best and Jacobs, 1997). Besides affecting production and oxidation of CH$_4$, WL$_{soil}$ in combination with root depth was important for the available transport mechanisms which either enhanced or suppressed CH$_4$ fluxes. The spatial analysis combined with data on rooting depth patterns, suggested that four types of soil profiles occurred within the research area as a result of varying combinations of WL$_{soil}$ conditions and rooting depth (Fig. 5.11). The differences between the soil profiles affected both CH$_4$ production and oxidation as well as CH$_4$ transport mechanisms.

The distinct spatial variability also appeared to affect the temporal variability of CH$_4$ fluxes from the different areas. At sites where the whole soil profile was permanently anaerobic, diffusion, plant transport and ebullition were all possible transport mechanisms. Emissions were on average high but also showed high variability, resulting at least partly from the ebullition component (Christensen et al., 2003; Strack et al., 2005). At sites where WL$_{soil}$ was lower, but roots grew below the water table, diffusion and plant transport were probably the main transport mechanisms. However, plants also transported
oxygen into the rhizosphere increasing the amount of CH$_4$ that was oxidized thereby reducing the CH$_4$ emissions to a moderate magnitude (Waddington et al., 1996; Hirota et al., 2004; Whalen, 2005; Wang and Han, 2005). Emissions were stable throughout the year as a result of the constantly high CH$_4$ concentrations in the anaerobic subsoil surrounding the rhizosphere. At sites where WL$_{soil}$ was low and plant roots shallow, diffusion was probably the only transport mechanism for CH$_4$ gas. Since a large amount of the CH$_4$ was oxidized during diffusion, fluxes were low. This was in agreement with previous research (Strack et al., 2006) where vascular plants in combination with low WL$_{soil}$ generated relatively low CH$_4$ emissions, while vascular plants in combination with high WL$_{soil}$ generated relatively high CH$_4$ emissions. At the ditch sites, probably diffusion and ebullition were available as transport mechanisms, while plant transport was prohibited by the lack of plants. Additionally, fresh organic material was not available and methanogenic microbes were dependent on the inflow of organic material from the surrounding soil profiles. Probably as a result of these processes, CH$_4$ emissions were low compared to saturated soil with vegetation. This finding was in agreement with previous research (Waddington et al., 1996; Ding et al., 2005) where strong reductions of CH$_4$ emissions were observed after cutting vascular plants in both emerged and submerged environments. CH$_4$ emissions from ditches were highest in summer, while during autumn and winter no significant fluxes were observed. This was similar to the findings of Waddington and Day (2007) who described CH$_4$ emissions from ponds and ditches as seasonal hotspots. Sites that were saturated almost permanently, and where diffusion and/or ebullition were important transport mechanisms; $T_{s1}/T_{water}$ was the best explanatory variable. This could be explained by the fact that $T_{s1}$ affects methanogenesis and methane
oxidation (Schlesinger, 1991; Whalen, 2005) as well as ebullition from anaerobic soils (Beckmann and Lloyd, 2001; Marik et al., 2002; Christensen et al., 2003). At sites where plant transport was the dominant transport mechanism, $T_{s1}$ did not have a relation with $CH_4$ fluxes, while $R_{eco}$ did. This suggested that $CH_4$ fluxes were relatively independent of shallow microbial activity and ebullition, while plant activity was important for $CH_4$ fluxes.

The $CH_4$ concentration profiles and the comparison between the soil $CH_4$ gradient method and the chamber method showed several interesting results. The $CH_4$ concentrations from the soil $CH_4$ profiles showed that below -0.70 m depth $CH_4$ concentrations at all sites decreased. This was probably due to small productivity of labile organic material below the maximum rooting depth, and mixing with seepage ground water with relatively low $CH_4$ concentrations (Hendriks et al., 2007). Soil $CH_4$ concentrations at site E showed a clear transition to lower $CH_4$ concentrations from -0.70 m upward, which suggested transport of $CH_4$ to the atmosphere by deep rooting plants (Juncus effusus) bypassing the aerobic upper soil layer and $CH_4$ oxidation due to aerobic conditions in the root zone. The relatively high soil $CH_4$ concentrations over the whole soil profile and the sharp transition of $CH_4$ concentrations in the top soil layer at site A was probably the result of $CH_4$ diffusion to the atmosphere as main transport mechanism and $CH_4$ oxidation in the top soil. The comparison between the soil $CH_4$ gradient method and the chamber method showed that only at site A, with shallow rooting plants and low WL$_{soil}$ and at some occasions during winter at the other sites (C and E), the soil $CH_4$ gradient method showed results comparable to the chamber method. For sites C and E, with deep rooting plants and relatively high WL$_{soil}$, the methods were not comparable. At these sites $CH_4$ was probably effectively transported by plant roots from deeper soil layers to the atmosphere, while at site A and during winter periods, the main transport mechanism for $CH_4$ was diffusion. These results emphasised the importance of root characteristics and WL$_{soil}$, and suggested that at sites with plants that root below the water table, $CH_4$ surface fluxes were relatively high and that plant transport was a relatively effective mechanism for $CH_4$ transport.

5.6.3 Temporal variability of methane fluxes

Methane fluxes in the peat meadow in the Netherlands also showed high temporal variability at different scales: $CH_4$ fluxes showed a clear diurnal cycle during all seasons as well as significant day-to-day variability, and seasonal variations. Continuous eddy covariance measurements showed a clear diurnal cycle of $CH_4$ fluxes in spring, summer and autumn. During night time, emissions were similar for all seasons (approximately 0.90 mg m$^{-2}$ hr$^{-1}$), while the amplitude observed during daytime was largest in summer and lower, but comparable in spring and autumn. In previous research, at various research locations diurnal cycles of $CH_4$ fluxes were observed (Schütz et al., 1989a; Schütz et al., 1989b; Knapp and Yavitt, 1992; Brix et al., 1992; Whiting and Chanton, 1993; Chanton et al., 1993; Thomas et al., 1996; Friborg et al., 1997; Hirota et al., 2004; Wang and Han, 2005). At other measurement locations, no diurnal variations were observed at all (Rinne et al., 2007; Sachs et al., 2008). The observations of the diurnal cycle at the Horstermeer site were relatively clear compared to previous observations. This was probably partly due to the high precision and accuracy of the instrument used for the eddy covariance
measurements (Fast Methane Analyser, type DLT-100, Los Gatos Research Ltd.; Hendriks et al., 2008).

Previous studies reporting diurnal cycles, showed that the CH$_4$ fluxes were either synchronous to temperature or to light intensity. The diurnal cycles that were related to temperature were found to be caused by changes in molecular diffusion (Schütz et al., 1989a; Schütz et al., 1989b; Whiting and Chanton, 1993; Friborg et al., 1997). On the other hand, the diurnal cycles related to light intensity were due to either stomatal opening (Knapp and Yavitt, 1992) or pressurized convective throughflow (Brix et al., 1992; Chanton et al., 1993; Thomas et al., 1996; Hirota et al., 2004; Wang and Han, 2005). The similarity with the diurnal cycle of SW$_m$ and the dissimilarity with that of T$_{15}$, suggested that the diurnal cycle of CH$_4$ fluxes at the Horstermeer site was due to plant stomatal opening (Knapp and Yavitt, 1992) or convective throughflow (Chanton et al., 1993; Thomas et al., 1996; Hirota et al., 2004; Wang and Han, 2005). The high similarity of the CH$_4$ fluxes with diurnal cycles of NEE and LE supported this conclusion. Two plant species that occurred in the research area (Typha latifolia and Phragmites australis) have been shown to possess an outstanding capacity to vent underground tissue by pressurized throughflow (Whiting and Chanton, 1996; Brix et al., 1996). However, for the other species that occurred in the research area plant transport mechanisms have not yet been studied in detail. The fact that diurnal cycles have not been observed at all research locations (Rinne et al., 2007; Sachs et al., 2008), might be due to differences in vegetation. Vascular plants have the capacity to transport the CH$_4$ gas from the subsoil, while non-vascular plants (Bryophytes) do not have this capacity. As a result, mechanisms that were found to induce diurnal cycles of CH$_4$ fluxes (stomatal opening and pressurized convective throughflow) do not occur at areas dominated by Bryophytes.

The difference in magnitude of the amplitude of the diurnal cycle between the seasons was probably the result of the amount of living vascular plants available for transport and production of fresh organic material. Additionally, SW$_m$ and humidity (both determining the strength of the convective throughflow) as well as the amount of CH$_4$ produced and oxidized in the rhizosphere. All these factors were most favourable during summer, resulting in the highest daytime CH$_4$ emissions in this season. Similar results were found by Wang and Han (2005) who observed diurnal cycles of CH$_4$ fluxes from an organic site throughout the whole year, except in winter. Night time emissions were rather stable over all measurement periods, which might be due to relative constant diffusion rates through the soil and water from deeper soil layers that were permanently saturated with CH$_4$.

At the daily time scale, CH$_4$ fluxes measured with eddy covariance were highly related to T$_{15}$ and WL$_{soil}$, with higher CH$_4$ fluxes for higher T$_{15}$ and lower WL$_{soil}$. Significant linear regressions models were found for all environmental variables, except for P and u$. Previous investigations have shown clear relations between these two variables (P and u$)$ and CH$_4$ fluxes; however, in those researches measurements were made over lakes or areas with a higher coverage of open water surfaces (Mattson and Likens, 1990; Fan et al., 1992; Casper et al., 2000; Wille et al., 2008; Sachs et al., 2008). The open water surfaces in the footprint of the eddy covariance tower consisted of narrow ditches bordered with high reed vegetation, which took up only 2 - 10% of the area in the footprint. If an effect
of P or u on enhanced ebullition from these ditches existed, it was probably too small to be clearly registered by the eddy covariance set-up.

A multi-linear regression equation with $T_{s40}$ and $WL_{soil}$ explained over 65% of the day-to-day variations in CH$_4$ fluxes throughout the year. $WL_{soil}$ was highly correlated to $WL_{diff}$ and LE which were related to gas transport through plants, and to $T_{s1}$ which drove the shallow microbial activity. $T_{s40}$ on the other hand probably had a large influence on the methane production and formation of gas bubbles in deeper, anaerobic layers of the soil. Together $T_{s40}$ and $WL_{soil}$ best explained the largest part of the temporal variability of CH$_4$ fluxes at the Horstermeer site. The regression model based on $T_{s40}$ and $WL_{soil}$ will be useful to scale up CH$_4$ emissions over time at the Horstermeer site and to gap fill data series.

Both chamber measurements and eddy covariance measurements showed highest CH$_4$ emissions in summer, lowest emissions in autumn and winter and intermediate emissions during spring. These seasonal trends were comparable to a wealth of studies (e.g. Zimov et al., 1997; Rinne et al., 2007; Van der Molen et al., 2007; Drössler et al., 2008) and probably resulted from seasonal changes in changes in production and oxidation of CH$_4$ in the rhizosphere and plant gas transport. In previous research was found that living biomass is an important driver of CH$_4$ emissions due to two mechanisms (e.g. Whiting and Chanton, 1993; Van Huissteden et al., 2006; Hirot et al., 2004). First, living biomass provides labile organic carbon to the soil, especially in the root zone, which becomes available for methanogenesis (e.g. Whalen, 2005; Van Huissteden et al., 2006). Additionally, more plant gas transport is possible in case of high amounts of living biomass. Plant transport is found to be a very important transport mechanism, which can contribute up to 62% to total methane emissions in peat areas (e.g. Whiting and Chanton, 1996; Grünfeld and Brix, 1999). On the other hand, vascular plants transport oxygen down into the rhizosphere, creating aerobic conditions favourable for methane oxidation, thereby reducing methane emissions (e.g. Roura-Carol and Freeman, 1999; Wang and Han, 2005). Conditions at the Horstermeer site for high CH$_4$ production and transport were most favourable during summer and a lesser extent during spring when temperatures were high and amounts of living biomass high and the biomass was most productive. With respect to the transport, length of day was also important factor at the seasonal scale: longer day-light time periods resulted in higher total daily emissions; thereby increasing average CH$_4$ emissions at the end of spring and in summer.

The temporal effect of higher $WL_{soil}$ on increasing CH$_4$ emission was weak and of minor importance compared to the effects describes above. However, when CH$_4$ fluxes were corrected for the influence of $T_{s1}$, CH$_4$flux,corr was found to have a relation with $WL_{soil}$. CH$_4$flux,corr was lowest for $WL_{soil}$ around -0.20 m and increased with higher $WL_{soil}$, but also with lower $WL_{soil}$. Similar results were found by Rinne et al. (2007) and Fiedler and Summer (2007). The high CH$_4$flux,corr for low $WL_{soil}$ was caused by the fact that $WL_{soil}$ was also an indicator for CH$_4$ transport by plants from deeper anaerobic layers, which was highest when plant transpiration was highest (and $WL_{soil}$ low). Additionally, the high CH$_4$flux,corr for low $WL_{soil}$ might be caused by pressure release during rapid $WL_{soil}$ changes, as was found before by Moore and Knowless (1990) and Strack et al. (2005).
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References


Multi-technique assessment of methane fluxes in a peat meadow


131


