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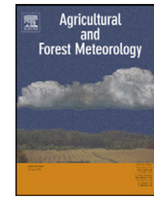


# Interpreting the variations in atmospheric methane fluxes observed above a restored wetland

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ABSTRACT

The eddy flux of methane ($CH_4$) was measured over 14 months above a restored wetland in western Denmark. The average annual daily $CH_4$ flux was 30.2 mg m$^{-2}$ d$^{-1}$, but the daily emission rates varied considerably over time. Several factors were identified that explained some of this variation. (1) Grazing cattle moving through the source area of the eddy flux mast increased the measured emission rates by one order of magnitude during short time periods. (2) Friction velocity exerted a strong control on the $CH_4$ flux whenever there were water pools on the surface. (3) An exponential response of the daily $CH_4$ flux to soil temperature at 20 cm depth was found for most of the study period, but not for parts of the summer season that coincided with a low water level in the river flowing through the wetland. (4) Additional variations in the $CH_4$ emission rates were related to the spatial heterogeneity of the source area. This area covered not only different plant communities but also a gravel road and a river surface, and it had a microtopography that visibly induced a large spatial variability in the wetness of the top soil. It is shown that the control mechanisms for the methane emission from restored wetlands are more complex than those reported for natural wetlands, since they include both management activities and slow adaptive processes related to changes in vegetation and hydrology. On the basis of eddy fluxes of carbon dioxide measured at the same site it is finally demonstrated that the variability in the $CH_4$ fluxes strongly affects the greenhouse gas sink strength of the restored wetland.



## 1. Introduction

Over the last century around half of the pristine wetland area in the countries of the European Union (EU) has been drained or made subject to other types of management practices, which has caused a loss in biodiversity and important bird habitats (Joosten and Clarke, 2002). As a consequence a number of national and international initiatives (with the Ramsar convention being the most prominent) have been implemented to protect wetlands against further loss and extinction. Whilst the effects of wetland loss on biodiversity have been well documented, there is as yet only little information available about the effects of drainage and restoration on the exchange of greenhouse gases between wetlands and the atmosphere.

The increase in the atmospheric concentration of methane ($CH_4$), in relation to the pre-industrial level, has resulted in a radiative forcing of 0.48 W m$^{-2}$ (IPCC, 2007). This makes $CH_4$ the second most important greenhouse gas after carbon dioxide ($CO_2$) whose increase has caused a radiative forcing of 1.66 W m$^{-2}$. Wetlands are the largest natural $CH_4$ source, whereas rumination, rice production and fossil fuel use are the largest anthropogenic sources. As far as vegetated land surfaces are concerned, $CH_4$ is produced by microbes in anaerobic sediment and is emitted through three pathways, namely diffusion through the top soil, ebullition from the soil pore water, and plant-mediated transport in terms of diffusion through aerenchyma of plants (Lai, 2009).

Since the substrate for methanogenesis is organic carbon compounds in the soil that come from photosynthesis via root exudation, root turnover and litter production, Whiting and Chanton (1993) dubbed net ecosystem production the 'master variable' for the control of methane emissions from wetlands. However, the emission rate of $CH_4$ is the difference between its production rate in anaerobic soil layers and its oxidation rate in the aerated top soil, and thus the water table, discerning anaerobic and aerobic layers, will in practice often be an overriding control on methane emissions (Bubier and Moore, 1994). Both the water table height and the soil temperature that influences production and oxidation rates have routinely been identified as main drivers of the $CH_4$ flux in field studies for more than 20 years (e.g. Crill et al., 1988; Roulet et al., 1992; Bubier et al., 1993; Dise et al., 1993). Until now, a large number of physical and biological controlling factors have been identified that are responsible for the huge spatial and temporal variation in methane fluxes from the soil into the atmosphere (Lai, 2009; Saarnio et al., 2009).

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Nevertheless, we still lack a full account of the environmental control of the $CH_4$ flux that would enable robust predictions of the $CH_4$ emissions under a changing climate and land use. It is already difficult to explain why previous studies have revealed such a high variability in the daily $CH_4$ emission rates per unit ground area of wetland averaged over the growing season, with a factor of about 20 between the lowest (Hargreaves and Fowler, 1998; Sachs et al., 2008) and highest (Treat et al., 2007; Song et al., 2009) observations. This uncertainty can only be overcome with new monitoring programs that make use of the latest developments in sensor technology, enabling spatially representative long-term measurements that fully cover the various driving forces behind the $CH_4$ exchange between wetlands and the atmosphere (Drösler et al., 2008). Such measurements need to cover all seasons, since continuously measured annual $CH_4$ budgets are still very rare (Rinne et al., 2007).

The collection of new data should include those wetland types that are underrepresented in the available literature, such as for example restored wetlands. Although they cover globally a much smaller area than pristine wetlands, a better understanding of the gas exchange of restored wetlands is important because they are amenable to management practices that may help optimize their role in the regional greenhouse gas budget. Previous studies have found that the atmospheric greenhouse gas exchange changes slowly after the restoration and remains different from that of pristine wetlands over many years (Tuittila et al., 2000; Waddington and Day, 2007). The speed of change depends on the water table height, the vegetation type before restoration and on management activities such as cutting the vegetation or introducing target species (Drösler et al., 2008) that are relevant for the ecological function of the upper peat layer (Pfadenhauer and Grootjans, 1999). The measurement and analysis of atmospheric $CH_4$ fluxes from restored wetlands faces the additional challenge that more control mechanisms than in pristine wetlands are involved. These additional control factors include, for example, a faster and spatially more variable vegetation succession and activities like hay making or introducing grazing animals that influence the soil structure and the vegetation. The complexity of the processes occurring in restored wetlands makes the choice of the most appropriate measurement techniques even more crucial than for pristine wetlands.

This study is entirely based on micrometeorological measurements and benefits from the availability of a new generation of laser gas analysers suitable for long-term continuous unattended data collection. A couple of methodological studies have demonstrated the ability of such methane analysers to be incorporated in continuously measuring eddy flux stations (e.g. Kroon et al., 2007, 2010; Hendriks et al., 2008). The present study does not focus on principal technical questions about eddy covariance and gas analysers, but on the variability of the eddy fluxes of $CH_4$ observed in the field. The two aims of this study were (1) to quantify the $CH_4$ emissions from a restored and managed wetland and (2) to identify the most relevant factors that influence the $CH_4$ flux from this wetland type.

## 2. Materials and methods

### 2.1. Site description

The study was carried out at 'Skjern Meadows' in western Denmark (Fig. 1A), which is one of the largest restored wetlands in Northern Europe. 4000 ha of peatlands, wet grasslands and marshes in the valley of the Skjern River, Denmark's largest river in terms of water volume, were drained in 1968 and claimed for agricultural cultivation. Recently, 2200 ha of this area have been restored. The movement of 2.7 million $m^3$ soil in order to excavate a new river course, filling the old channelised river stretches and removal of dikes and pumping stations took almost four years and was completed in 2002. Today the area consists of meadows, wetlands, lakes and meandering water courses (Pedersen et al., 2007; Nielsen and Schierup, 2007). The upper part of the restored wetland, mainly covered by fens and lakes, is characterised by Histosols, whereas the floodplain around the study site, as well as the marshes close to the mouth of the Skjern River, are dominated by Fluvisols, according to the FAO soil classification system. 'Skjern Meadows' is covered by various international conventions. The whole area is categorised as 'EU Habitat Area' and parts of it also as 'EU bird conservation area' and as 'Wetland of International Importance' according to the 'Ramsar Convention' (http://www.ramsar.org).

At the study site on the floodplain near the river delta (Fig. 1B) the meadows are grazed by cattle during the summer and are cut once a year for hay production. The vegetation on the former cultivated fields changed rapidly as a consequence of the restoration and the changed hydrological conditions (Pedersen et al., 2007). It is expected to develop further over the next decades, with management activities limiting the colonisation of tall grasses or bushes and trees. By 2003, the coverage of the wetland by the soft-rush (*Juncus effusus*) meadow community had increased from 2 to 26%, which turned this community into the most abundant one, followed by the reed-canary grass (*Phalaris arundinacea*) community covering 21% of the land surface and the perennial ryegrass (*Lolium perenne*)/white clover (*Trifolium repens*) community with a coverage of 13%, a percentage that remained nearly unaffected by the restoration (Andersen et al., 2005; Pedersen et al., 2007). The numbers of livestock on the three meadows closest to the study site, together with the harvest dates, are listed in Table 1. The aim of the grazing and hay making activities is to limit the vegetation height on the meadows and to prevent the development into a woodland.

The long-term annual precipitation in western Denmark amounts to 781 mm and is spread over 131 rain days on average (Danish Meteorological Institute, data from 1961 to 1990). The wettest months are usually October and November with long-term averages of 91 and 92 mm of rain, respectively, and the driest months are February and April with 41 and 42 mm, respectively. Within the investigation period, lasting from August 2008 until October 2009, the highest monthly precipitation was recorded in August and October 2008 (133 and 155 mm, respectively) whilst only 10 mm of rain fell during April 2009. The long-term annual mean temperature is 7.5 °C with a maximum in July (15.4 °C) and a minimum in January (−0.9 °C). No extended periods with frost occur in the area. During the flux measurement campaign the warmest month was August 2009 (16.8 °C) and the coldest was February 2009 (0.9 °C). The prevailing wind direction is west. As the water table is not regulated, the restoration area comprises both permanently and seasonally wet areas.

### 2.2. Eddy covariance measurements

An instrument mast was erected in the western part of the restored wetland (Fig. 1B), at 55°54′46″N and 8°24′17″E and an elevation of 2 m above sea level (a.s.l.). Due to legal access restrictions arising from the protected status of Skjern Meadows, the mast was positioned in a 3 m high hedgerow that grows on a shallow earth bank close to a small gravel road (Fig. 1C). A sonic anemometer (R3-50, Gill Instruments Ltd., Lymington, UK) was installed at the top of the mast, resulting in a measurement height of 7 m above the wetland surface. This relatively large measurement height was chosen in order to minimize the influence of the hedgerow on the turbulence.

The mole fraction of $CH_4$ in the air was determined by means of a gas analyser DLT-100 (Los Gatos Research Inc., Mountain View, CA, USA) based on off-axis integrated cavity ringdown spectroscopy. The suitability of the DLT-100 for eddy covariance measurements

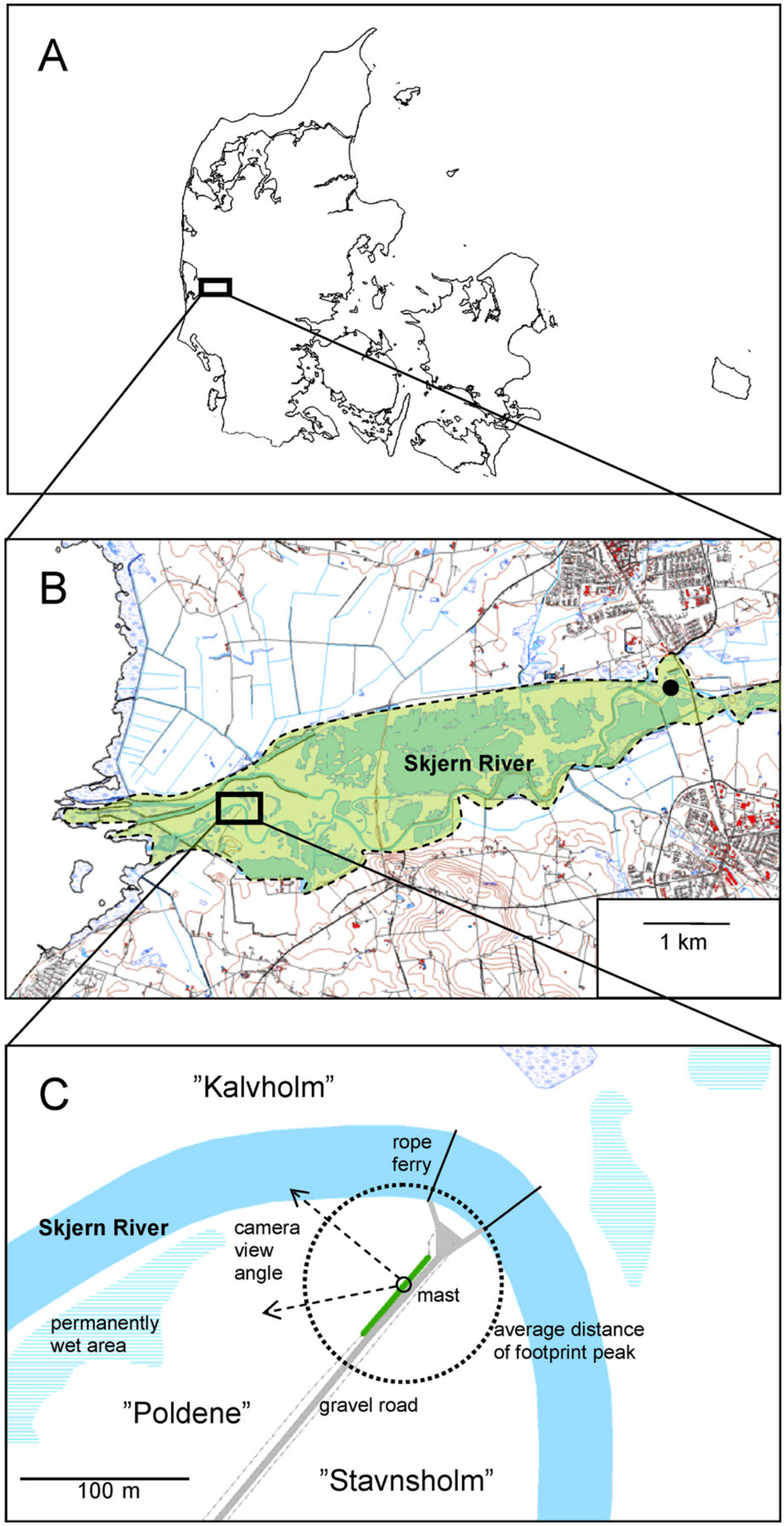


**Fig. 1.** (A) Location of the restored valley of the Skjern River. (B) Location of the research site (rectangle) and the site where the water level in the river was monitored (large black dot). The green area indicates the extension of the restored wetland. It continues further towards the east. (C) Position of the instrument mast and the average distance of the footprint peak (black circles) in relation to the spatial distribution of the various surface types. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

**Table 1**
Overview about grazing activities on the meadows closest to the instrument mast.

| Location | Area [ha] | Number of cattle | Time of grazing | Time of grass cut |
|---|---|---|---|---|
| West ("Poldene") | 23 | 2008: 30 | 28 May–29 September | Around 15 July |
| | | 2009: 45 | 15 September–5 October | On 29 June |
| East ("Stavnsholm") | 55 | 2008: 20 | 28 May–1 July | Around 15 July |
| | | 2009: ca. 50 | 18 May–5 October | Around 20 July |
| North ("Kalvholm") | 120 | 2008: 140 | 15 May–2 October | From 25 July |
| | | 2009: 105 | 19 May–1 October | From 15 July |

was demonstrated by Hendriks et al. (2008). A closed path infrared gas analyser (LI-7000, LI-COR Inc., Lincoln, NE, USA) was used to measure the mole fraction of $H_2O$, necessary for the dilution correction of the $CH_4$ flux. All measurements were taken at a nominal frequency of 10 Hz. High frequency $CO_2$ data were recorded as well, both with the LI-7000 and with an open path gas analyser (LI-7500, LI-COR Inc., Lincoln, NE, USA) mounted close to the sonic anemometer, but they are subject of a separate study (Herbst et al., in press). The data were stored on a data logger (CR3000, Campbell Scientific Ltd., Shepshed, UK) in a small hut next to the base of the hedgerow and transferred to Copenhagen University once a day.

The air was collected at 7 m height close to the sonic anemometer and sucked through separate pipes into the gas analysers in the hut. A membrane pump (N 89 KNDC, KNF Neuberger, Freiburg, Germany) with a nominal pumping speed of 9 l $min^{-1}$ was used for the LI-7000 and a vacuum pump (TriScroll 300, Varian Inc., Palo Alto, CA, USA) with a nominal pumping speed of 250 l $min^{-1}$ for the DLT-100. The air was filtered through a 1 μm Gelman filter in front of the inlet to the LI-7000 and through an external 5 μm filter plus an internal 2 μm Swagelok filter in case of the DLT-100. The filters were changed every 2 months on average. The flow charateristics of the two setups are summarised in Table 2. The average delay time given in Table 2 was determined experimentally from the best correlation between the vertical wind component and the gas concentrations. Flow rates, response times and cut-off frequencies were calculated on the basis of these measurements. Due to a sufficient measurement height of 7 m above the meadow surface, the relatively low cut-off frequencies were not considered as a major problem (Eugster and Zeeman, 2006), but a frequency response correction (Moore, 1986) was nevertheless essential and thus part of the flux calculations. On average, this correction increased the water vapour fluxes by 21.8% and the methane fluxes by 8.9%.

## 2.3. Flux calculations

The turbulent fluxes of $CH_4$ were calculated using the 'Alteddy' software, version 3.5 (Alterra, University of Wageningen, The Netherlands). This version includes a dilution correction (Webb et al., 1980) for the $CH_4$ flux. To perform this correction the water vapour fluctuations inside the DLT-100 cell must be known. However, water vapour is not measured in the DLT-100, and the only information available was the $H_2O$ mole fraction recorded in the LI-7000 cell. Given the very rapid flow through the tube towards the $CH_4$ analyser (Table 2) it was assumed that the $H_2O$ fluctuations in the DLT-100 were similar to those in the air at 7 m. Thus, the actual $H_2O$ fluctuations at 7 m, derived from the mole fractions measured in the LI-7000 and corrected for frequency losses owing to the LI-7000 setup, were used in the dilution correction of the $CH_4$ fluxes. The difference in the lag times for the two gas analysers was accounted for.

All fluxes were corrected for errors caused by tilt of the anemometer relative to the mean streamline coordinate system by use of the planar fit method (Wilczak et al., 2001). In this way the modification of the wind field by the presence of the hedgerow was accounted for. Table 3 shows the angles necessary to rotate the sonic into a plane parallel to the streamlines, with $\alpha$ being the 'pitch' angle about the original $y$-axis and $\beta$ being the 'roll' angle about the intermediate $x$-axis (Wilczak et al., 2001). The angles for the 18 wind sectors were calculated from a preliminary 'Alteddy' run using 17132 10-min data records collected between days 248 and 366 in 2008. Each wind sector was represented by several hundred measurements.

The relatively large variability of the tilt angle corrections (Table 3) had some implications for the calculation of the covariances between the vertical wind component and the $CH_4$ concentration, as it affected the choice of the most appropriate averaging time. On occasions with a rapid change in the wind field over the hedgerow, the commonly used 30-min averaging sometimes produced erroneous and physically impossible flux values (such as isolated, large negative spikes), for example when a change in the angle of attack within an averaging interval coincided with a change in the background $CH_4$ concentration, as it repeatedly happened around sunrise and sunset. Shortening the

**Table 2**
Flow characteristics for the measurements of the turbulent fluxes of $H_2O$ and $CH_4$ using closed-path gas analysers LI-7000 and DLT-100, respectively.

| | LI-7000 | DLT-100 |
|---|---|---|
| Cell size [l] | 0.011 | 0.550 |
| Inner tube diameter [mm] | 6 | 6 |
| Tube length [m] | 10.65 | 10.65 |
| Average delay time [s] | 3.5 | 0.5 |
| Actual flow rate [l $min^{-1}$] | 5.16 | 36.1 |
| Average cell pressure [kPa] | 94 | 16 |
| True flow rate in cell [l $min^{-1}$] | 5.56 | 229 |
| Response time [s] | 0.117 | 0.144 |
| Cut-off frequency [Hz] | 1.36 | 1.10 |
| Average Reynolds number | 1440 | 8200 |

**Table 3**
Angles of the first two axis rotations of the sonic anemometer data for different wind directions. See text for further explanations.

| Wind direction [°] | 'Pitch' angle $\alpha$ [°] | 'Roll' angle $\beta$ [°] |
|---|---|---|
| 10–30 | +3.12 | −5.95 |
| 30–50 | −0.35 | −0.24 |
| 50–70 | −2.56 | +1.18 |
| 70–90 | −5.93 | +3.40 |
| 90–110 | −1.75 | +3.99 |
| 110–130 | −3.24 | +2.65 |
| 130–150 | −1.32 | +5.73 |
| 150–170 | −2.45 | +3.50 |
| 170–190 | −2.96 | +3.40 |
| 190–210 | −2.51 | +3.21 |
| 210–230 | +0.60 | −1.86 |
| 230–250 | +2.94 | −4.55 |
| 250–270 | +2.34 | −4.53 |
| 270–290 | +4.04 | −4.42 |
| 290–310 | +3.23 | −4.73 |
| 310–330 | +3.23 | −5.15 |
| 330–350 | +2.05 | −5.88 |
| 350–10 | +2.50 | −3.24 |

averaging interval from 30 to 10 min improved the precision of the tilt angle correction and thus avoided the aforementioned flux errors and the necessity to isolate, reject and gapfill the respective data.

Whether 10 min were still long enough to capture fluctuations at all relevant frequencies was tested by comparing the average of three 10-min fluxes ($F_{10}$) with the corresponding 'true' 30-min flux ($F_{30}$) over the last 40 days of the measurement campaign, which covered the entire range of flux magnitudes observed. Only periods with valid $F_{30}$ data were used for this comparison, which meant that nine 30-min intervals had to be rejected. The two resulting flux data sets were practically indistinguishable ($F_{30} = 0.99 \cdot F_{10} + 0.54$, $R^2 = 0.95$), with the $F_{10}$ total being even 0.5% higher than the $F_{30}$ total. This demonstrates that practically no low frequency contributions to the flux were lost through the 10-min block averaging. It was concluded that, at this particular field site, the turbulent fluxes should be calculated at this shorter than usual time step in order to maximise the number of valid data records.

Strictly speaking, the turbulent flux accounts only for a part of the total gas exchange between the surface and the atmosphere, and in some circumstances storage and advection can contribute substantially to the total flux (Massman and Lee, 2002). Therefore the storage flux ($S$) was estimated from the changes in the average $CH_4$ concentration ($\Delta CH_4$) over the 10-min intervals ($t$) as $S = \Delta CH_4 \cdot h/(n \cdot t)$ with $n$ being the molar volume of an ideal gas and $h$ being the measurement height assuming that the $CH_4$ concentration was representative for the entire air column below the sensors. The total $CH_4$ flux ($F_{CH_4}$) was finally calculated as $F_{CH_4} = F_{10} + S$. Following de-spiking, $F_{CH_4}$ was averaged further to give half-hourly, hourly and daily fluxes, provided there were at least 12 h of valid data available per day.

The measurements ran continuously over most of the 14-months investigation period, except for a few occasions with power cuts at the field site, mostly caused by lightning. The DLT-100 analyser required more maintenance work than the LI-7000, because the cavity inside the instrument needed to be cleaned whenever the mirror ringdown time approximated the threshold of 3 μs below which no accurate measurements were possible. This was the case 21 times during the 14-months study period. Out of the entire 415 days of the study, the gaps in the LI-7000 data amounted to 21 days (5%) and those in the DLT-100 to 49 days (12%). In order to estimate annual totals, the gaps were filled with estimates based on soil temperature and season, using the response functions shown in Fig. 5.

The average distance of the point with the highest contribution to the flux measurements is indicated in Fig. 1C. It was calculated using the equations by Schuepp et al. (1990), assuming a displacement height of 0.2 m and a roughness length of 0.04 m. For neutral atmospheric stability, the distance would be 90 m, and 80% of the measured fluxes would come from a distance less than 800 m from the mast. However, using the stability correction method of Dyer (1974) and assuming a typical Monin–Obukhov length of $L = -40$ m, the distances are reduced to about 60 m (circle in Fig. 1C) and 580 m, respectively, which makes it highly unlikely that any areas outside 'Skjern Meadows' could have contributed to the fluxes (Fig. 1B). Since west was the most frequent wind direction, the typical footprint was dominated by the meadow named 'Poldene' (Fig. 1C) where soft rush and marsh foxtail (*Alopecurus geniculatus*) were the most abundant plant species.

Further data analysis in terms of curve fitting and statistics was carried out using the SigmaPlot 9.0 software (SSI, San Jose, California, USA). The exponential function used to describe the relation between $F_{CH_4}$ in mg m$^{-2}$ d$^{-1}$ and the soil temperature at 20 cm depth in °C ($t$) was $F_{CH_4} = a \cdot e^{bt}$, with $a$ and $b$ being empirical parameters.

### 2.4. Auxiliary data

Soil temperature sensors (thermocouples type 105-T, Campbell Scientific Ltd., Shepshed, UK) were installed about 50 m west of the mast at 5, 10 and 20 cm depth. The water level in the Skjern River was logged a few kilometers upstream between the towns of Skjern and Tarm (Fig. 1B). For the period of this study this was the only available information about the water level in the wetland, however from 1999 to 2003 a network of water level measurements was operated (Andersen et al., 2005) that covered a large part of the restoration area (albeit not the vicinity of the mast erected in 2008). These measurements showed that the rise and fall of the water table occurred largely synchronously across the entire wetland. This behaviour can be explained by the small size (1500 km$^2$) and flat topography (<100 m elevation) of the river catchment that leads to similar rainfall patterns between the river's upland and the meadow site. Nevertheless a large spatial variation in the absolute position of the water table in relation to the soil surface was found which is caused by the heterogeneous microtopography of the site. Due to the area's protected status there is no permission to install enough new bore holes to derive a representative local average of the water level for the source area of the instrument mast. This means that only the seasonal dynamics, and not the actual depth of the local water table were available for comparison with the methane flux data.

The leaf area index (LAI) on 'Poldene' and 'Stavnsholm' (Table 1) was measured during the growing season of 2009 with an optical sensor (LAI-2000, LI-COR, Lincoln, NE, USA). About once a month, three above-canopy readings and 30 below-canopy readings were taken on each of the two meadows and all data averaged. The measurements were taken in a transect from the south-eastern to the north-western edge of the circle shown in Fig. 1C that indicates the peak of the source area of the flux measurements. Along this transect three of the most abundant plant communities in the source area were covered by the LAI measurements, namely a meadow dominated by perennial ryegrass, a soft-rush meadow and a reed canary grass community located close to the river bank. As a supplement to the LAI measurements a web camera (TN-TV-IP400, Trendnet.dk, Denmark) was installed at 5 m height on the instrument mast in the early spring of 2009 in order to document the visual development of the vegetation. The camera was pointed towards the west (being the main wind direction), and it took one picture every day at 11:00 local time.

## 3. Results

### 3.1. Annual course of $CH_4$ fluxes

The daily average atmospheric methane fluxes over the entire measurement period are shown in Fig. 2 along with daily averages of soil temperature and the water level in the river. Upward fluxes were defined as positive. The $CH_4$ release decreased from values between 100 and 150 mg m$^{-2}$ d$^{-1}$ in August and September 2008 to less than 20 mg m$^{-2}$ d$^{-1}$ over most of the winter period (Fig. 2A). This decrease followed the respective trend in the soil temperature (Fig. 2B), and the relation to the temperature measured at 20 cm depth was closer than to the values obtained at 5 and 10 cm (data not shown). Except for one single day in the summer of 2009, no $CH_4$ uptake was observed on a daily basis. The connection between temperature and $CH_4$ emission appeared to weaken in mid April 2009. By 10 April two areas in the view of the web camera, where the water table was above the surface during winter, had fallen dry for the first time in 2009, at a time when the water table in the Skjern River had dropped to a local level of 75 cm (Fig. 2B). After it had dropped further to <64 cm on 29 April (D. 119), both the small

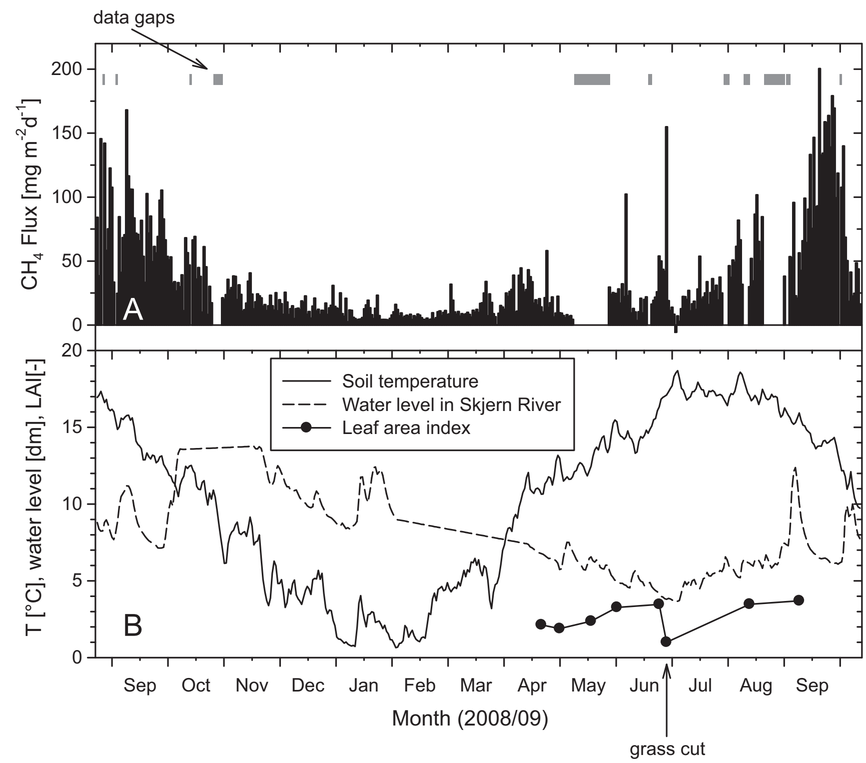


**Fig. 2.** Time course of the daily totals of the $CH_4$ fluxes (A) over the 415-day investigation period. Periods with missing data are indicated as grey bars. Daily averages of the soil temperature at 20 cm depth and the local water level in the Skjern River are given in panel (B), together with the leaf area index (LAI) measured over the growing season of 2009.

remaining $CH_4$ flux and the few observed peaks (Fig. 2A) remained largely unrelated to temperature, until the same water level threshold was reached again in mid August. Following heavy rainfall in the catchment in early September 2009, the water level in the river rose sharply, the water pools seen by the camera re-appeared and the highest $CH_4$ emissions of the entire measurement period (between 140 and 180 mg m$^{-2}$ d$^{-1}$ on average) were observed. The largest fluxes occurred during the second half of September 2009, at a time with stable soil temperatures around 14 °C and cattle being present on the meadow west of the instrument mast.

### 3.2. $CH_4$ fluxes and cattle

Looking at the days with the highest $CH_4$ fluxes in a higher temporal resolution, it turned out that large flux peaks occurred once or twice a day, which exceeded the relatively constant background flux by almost an order of magnitude. According to the quality check of the flux data using the criteria of Foken et al. (2004) these numbers were no artefacts, and they were only observed during periods when cattle were grazing in the source area of the instrument mast. A 10-day period with nearly constant westerly winds was selected to illustrate this phenomenon (Fig. 3). Under these conditions the view of the web camera captured the main part of the source area, and the daily picture taken at 11:00 local time could be used to compare the appearance of this area with the size of the fluxes at that time. Due to a camera fault pictures were only available for seven out of the ten selected days (Fig. 3A). According to the photographic evidence, cattle were present at 11:00 on days 264, 265, 266, 268 and 272. On all of these days a $CH_4$ flux peak was observed around 11:00. No cattle were visible on the photographs taken on days 267 and 273, and on these days the observed $CH_4$ flux peaks occurred at other times of the day (around 09:00 and 13:00 on day 267 and around 15:00 on day 273, see Fig. 3A). The peaks were unrelated to variations in atmospheric pressure (Shurpali et al., 1993). Thus, it seems likely that cattle moving across the source area caused a substantial part of the observed $CH_4$ fluxes. All comparable peaks found in the data set were consequently removed from the data record and new daily average fluxes calculated on the basis of the remaining hours. These were subsequently used in order to identify other factors that controlled the $CH_4$ flux over Skjern Meadows (Section 3.3). Over the entire 14 months investigation period, the removal of the peaks reduced the $CH_4$ flux by 11%.

### 3.3. $CH_4$ fluxes in relation to turbulence, temperature, season and fetch

No clear diurnal pattern in the hourly average $CH_4$ flux was observed in any season (Fig. 4A). In order to analyse a possible relation between the $CH_4$ flux and atmospheric turbulence, the method suggested by Long et al. (2010) was adopted. All eddy flux data from one season were rank ordered by friction velocity ($u_*$) and separated into equally sized groups, with each of them containing 10% of all observations. The averages and standard errors of $u_*$ and $F_{CH_4}$ were calculated for each of the groups (Fig. 4B). There was a tight linear relationship between $F_{CH_4}$ and $u_*$ from October to March and no significant relation from May to August. Data from April and September were omitted from this analysis because the large changes in soil temperature and water level (Fig. 2B) overshadowed any potential $u_*$ effects.

From the annual course of the $CH_4$ emissions shown in Fig. 2 it was expected that both soil temperature and season (primarily through changes in the water table position) would have a role in explaining the variability in the daily $CH_4$ fluxes. In Fig. 5A the $CH_4$ flux is plotted against soil temperature at 20 cm depth for two subsets of the data, representing periods distinguished by different water levels in the Skjern River and by the visibility of water above the surface west of the mast. Whilst the correlation between temperature and $CH_4$ flux was insignificant between 29 April and

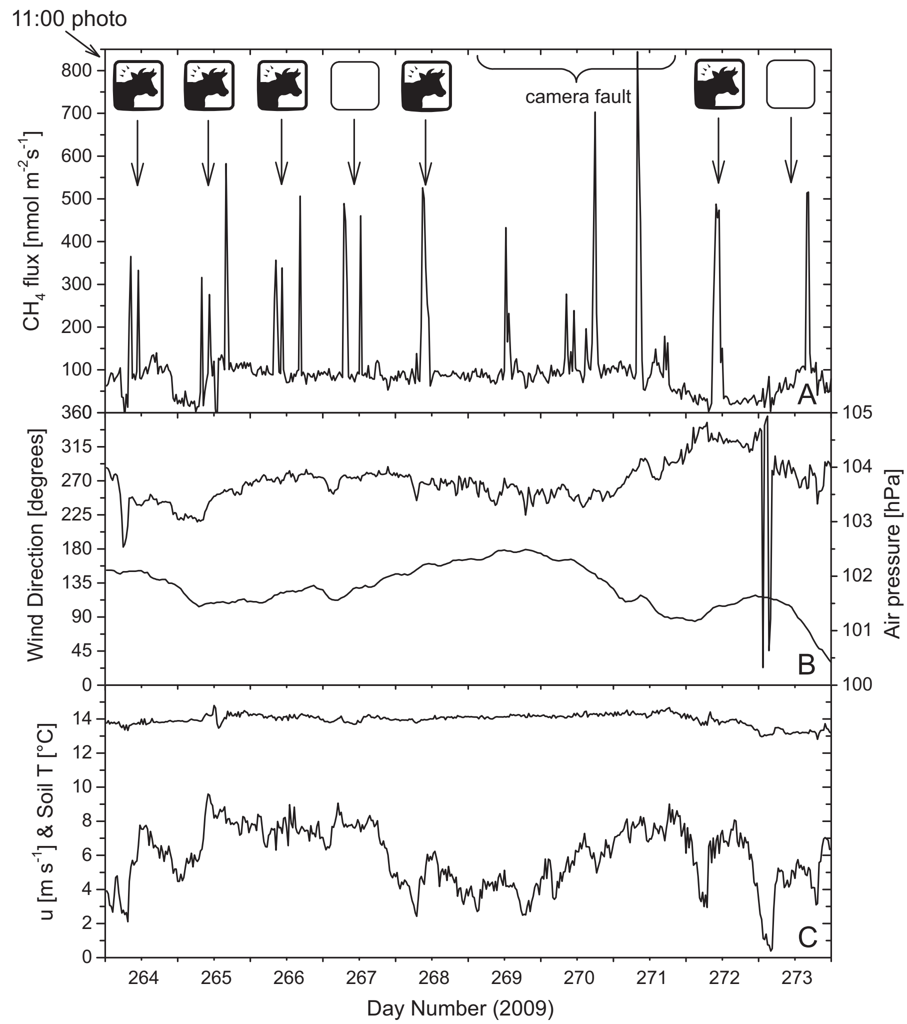


**Fig. 3.** Half-hourly averages of the $CH_4$ flux from 21st to 30th September 2009 (A). The presence of cattle in the footprint area at 11:00 local time each day is indicated through the symbols at the top. Wind direction (thin line) and air pressure (B), as well as wind speed (thin line) and soil temperature (C) for the same period are also shown.

15 August 2009 (DOY 119-227), there was a significant relationship for the rest of the investigation period (Fig. 5A). However, the exponential function shown in Fig. 5A left 35% of the variance in the daily $CH_4$ fluxes unexplained by the combination of season and soil temperature.

The largest positive deviations from the fitted curve in Fig. 5A were observed on days when the wind came from the west and the fluxes were influenced by the area where patches with water on the surface were seen through the camera. In contrast, no positive deviations ever occurred during periods when the wind came from a sector between NNW and NE (Fig. 5B). In this case, the average position of the peak of the source area was located above the river surface (Fig. 1C), and this is evidence that the river surface was a lower methane emitter than the land surface. On the basis of topographical data from an airborne survey of the Danish Ministry for the Environment the possible influence of the microtopography on the $CH_4$ fluxes is illustrated in Fig. 5B, too. The distribution of three surface classes (river and areas below and above 0.7 m a.s.l.) within the first 50% of the cumulative footprint is indicated for eight wind sectors. The sector with the largest positive deviations from the predicted temperature response (SW to W) had the highest proportion of low lying areas, whereas the three sectors with less than 50% low lying areas showed mostly negative deviations.

However, as both the width and the length of the source area generally vary with turbulence characteristics such as atmospheric stability and standard deviation of the cross wind component (Soegaard et al., 2003), a more sophisticated footprint model (and probably a longer data record) would be needed to assess the sensitivity of the $CH_4$ flux to the footprint comprehensively. This will be subject of a future study, and for now it was assumed that the upper envelope of the data points in Fig. 5A represented the most typical "wetland" response. The upper envelope, calculated as the average plus one standard deviation of all $CH_4$ fluxes obtained within temperature classes spanning 2 °C (Fig. 6), corresponds to situations when the main contribution to the flux came from areas with wet organic soils and typical wetland vegetation.

Using the temperature response function from Fig. 6 and the measured soil temperatures, the theoretical $CH_4$ emission for a situation with a water table permanently high enough to prevent aeration of the top soil (but without cattle) was estimated (Fig. 7A). Whilst the seasonal course of the actual $CH_4$ flux closely followed the theoretical course during the first half of the measurement period, there were large differences for the spring and summer 2009, when the $CH_4$ emission from large parts of the footprint area was apparently switched off. In contrast, the actual flux exceeded the expected one by a factor of two when cattle moved through the footprint area. From the large differences between the annual $CH_4$ emissions actually observed and calculated from the temperature response function it can be concluded that management effects such as water table regulation or grazing have the potential to affect the greenhouse gas budget of wetlands more than meteorological variations.

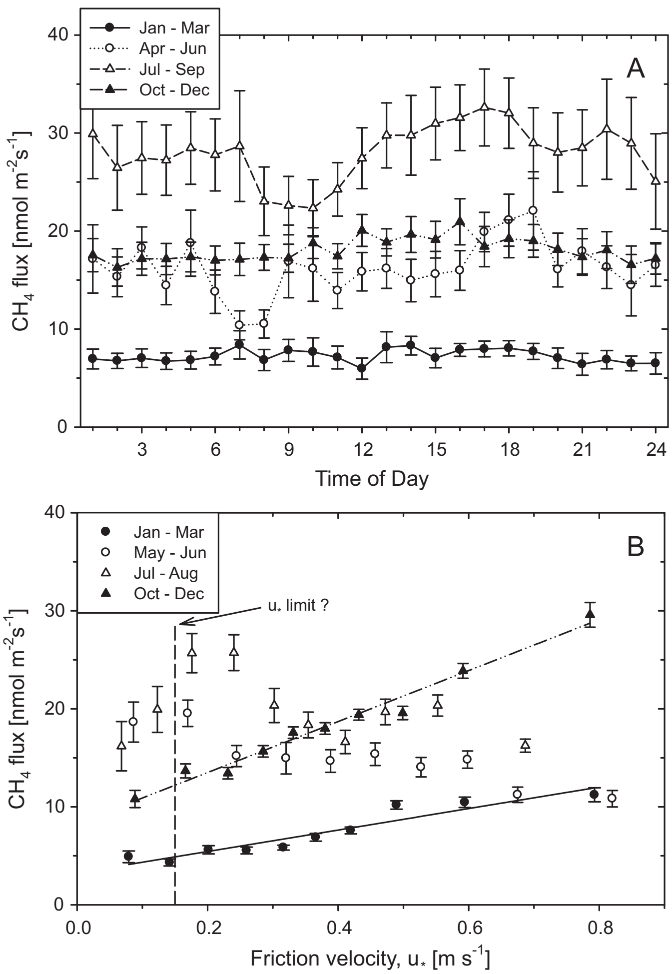


**Fig. 4.** (A) Average diurnal courses of the $CH_4$ flux for different seasons. (B) Average $CH_4$ flux rates plotted against average friction velocity. Half-hourly $CH_4$ flux means were rank ordered and grouped by friction velocity, with each data point representing 10% of the total data set. All days with cattle being present on the meadows were excluded from the calculations. The regression lines are $y = 10.95x + 3.25$ ($R^2 = 0.90$, $P < 0.001$) for the period January–March and $y = 26.00\,x + 8.30$ ($R^2 = 0.98$, $P < 0.0001$) for the period October–December. The error bars in both panels indicate the standard error of the mean values.

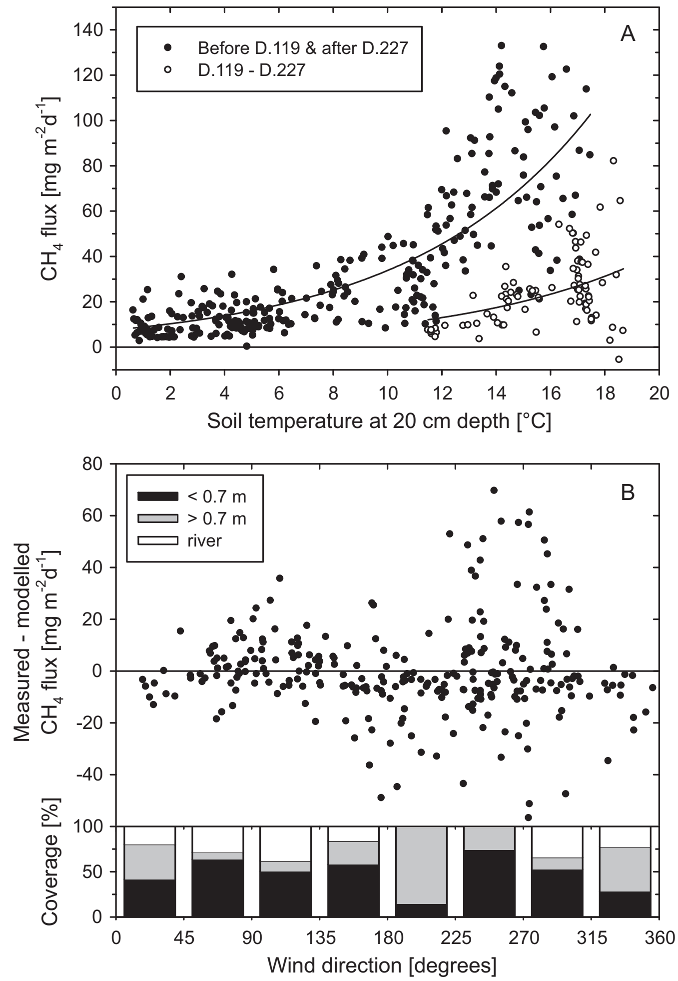


**Fig. 5.** (A) Daily methane flux versus soil temperature for the periods prior to 29 April and after 15 August 2009 (filled circles) and for the period in between (hollow circles). The curves with the best fit are based on the function $y = a{\cdot}e^{bx}$ and the parameters are $a = 7.63\,\mathrm{mg\,m^{-2}\,d^{-1}}$ ($P < 0.001$) and $b = 0.149\,^{\circ}\mathrm{C}^{-1}$ ($P < 0.001$) for the major part of the investigation period ($R^2 = 0.65$) and $a = 2.26\,\mathrm{mg\,m^{-2}\,d^{-1}}$ ($P = 0.16$) and $b = 0.146\,^{\circ}\mathrm{C}^{-1}$ ($P < 0.001$) for the period from 29 April to 15 August ($R^2 = 0.19$). (B) Deviations between measured and modelled fluxes versus daily mean wind direction (without days 119 to 227). The distribution of three surface types (river, land surface below 0.7 m and land surface above 0.7 m a.s.l.) within the first 185 m from the mast (corresponding to a cumulative footprint of 50%) is indicated for eight wind sectors. The peak of the footprint was located above the river in the sectors 0–45° and 315–360°.

## 4. Discussion

### 4.1. Universal factors controlling the $CH_4$ flux from wetlands

The position of the water table has been routinely shown to be the primary control of $CH_4$ emissions from wetlands (e.g. Roulet et al., 1992; Bubier et al., 1993; Hargreaves and Fowler, 1998). This is unsurprising since methanogenesis only occurs in anaerobic soils and oxidation through methanotrophic microorganisms in the aerobic top soil can prevent the $CH_4$ produced in deeper layers from being emitted into the atmosphere. Although it has been suggested that the water table height acts as a switch for the $CH_4$ emission (Drösler et al., 2008), the change in $CH_4$ production with changing water table in practice can occur more smoothly where the wetland surface has a microtopography that creates locally different water table heights (Bubier et al., 1993). As an example, Hendriks et al. (2007) reported $CH_4$ emission rates from an abandoned peat meadow that differed on the micro-scale by a factor ten between drier and wetter patches and by a factor of three between wet areas and ditches.

Unfortunately, the local water table in the source area of the mast could not be measured in this study, and therefore no quantification of the water level effect was possible. Nevertheless, two periods could be discerned that differed with respect to the correlation between soil temperature and $CH_4$ flux. During the period showing a strong correlation any hydrological effects must have been negligible, which could mean that the average water level remained above (or close to) the threshold of 0.1 m depth suggested by Drösler et al. (2008). The remaining period (spring and summer 2009) coincided with a decrease in the local water level in the river to less than 64 cm and began about two and a half weeks after the water table on the lowest parts of the meadows had dropped below the surface, according to the camera recordings.

Practically all studies of $CH_4$ emissions from wet soils have found that the $CH_4$ flux increased with enhanced temperatures (e.g. Svensson and Rosswall, 1984; Crill et al., 1988; Hargreaves and Fowler, 1998; Rinne et al., 2007; Song et al., 2009; Long et al., 2010). However, as the depth of the aerobic top soil increases, this dependence disappears (Svensson and Rosswall, 1984) because not only the $CH_4$ production, but also its oxidation is temperature dependent. As a consequence, a clear relation between $F_{CH_4}$ and

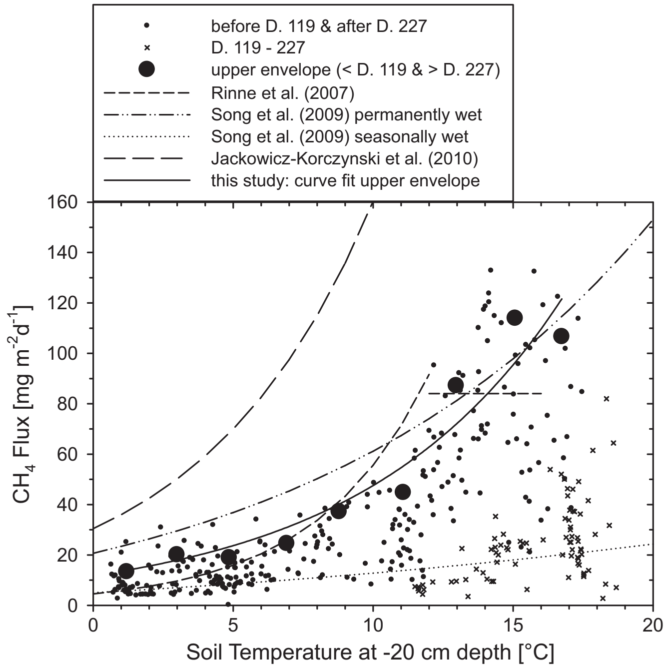


**Fig. 6.** Curve fit to the upper envelope (large black dots) of the data points, with the period from day 119 to day 227 being excluded. The fitted function has the type $y = a \cdot e^{bx}$ ($R^2 = 0.93$) and the parameters are $a = 11.81$ mg m$^{-2}$ d$^{-1}$ ($P = 0.010$) and $b = 0.139$ °C$^{-1}$ ($P < 0.001$). The corresponding functions derived by Rinne et al. (2007), Song et al. (2009) and Jackowicz-Korczyński et al. (2010) using the same function type are also shown. It should be noted that Song et al. (2009) used air temperature instead of soil temperature.

soil temperature can only be expected for situations where $CH_4$ oxidation in the top soil is negligible. For a range of natural wetlands it has been shown that an exponential temperature response function with only two parameters (see Section 2.3) is sufficient to explain most of the observed variance in the fluxes (Rinne et al., 2007; Song et al., 2009; Jackowicz-Korczyński et al., 2010). The same function type was used in the present study, and this enabled a direct comparison of both the curvature ('$b$') and the scaling factor ('$a$') between the restored Skjern Meadows and various natural wetlands (Fig. 6).

The parameters for a Finnish wetland site (Rinne et al., 2007), if converted into the same units as in this study, equal $a = 4.56$ mg m$^{-2}$ d$^{-1}$ and $b = 0.25$ °C$^{-1}$, compared to $a = 11.81$ mg m$^{-2}$ d$^{-1}$ and $b = 0.139$ °C$^{-1}$ for Skjern Meadows. This means that the function derived for the Finnish site is steeper than the response found for Skjern Meadows (Fig. 6). However, no further increase of $F_{CH_4}$ at temperatures higher than 12 °C was reported by Rinne et al. (2007), and looking at Fig. 6 it appears likely, that a response function fitted to the entire Finnish data set would have had a curvature similar to this study. In contrast, the study from China (Song et al., 2009), which distinguishes between different functional types of wetland, has much higher '$a$' and much lower '$b$' values if converted into the same units ($a = 31.88$ mg m$^{-2}$ d$^{-1}$ and $b = 0.082$ °C$^{-1}$ for the permanently wet sites and $a = 16.19$ mg m$^{-2}$ d$^{-1}$ and $b = 0.039$ °C$^{-1}$ for the temporarily wet sites), despite the very high average flux rate for the wet sites during the growing season of 350 mg m$^{-2}$ d$^{-1}$. However, Song et al. (2009) used air instead of soil temperature, measured over the large temperature range from −25 to +35 °C and added an offset of minus 11.2 mg m$^{-2}$ d$^{-1}$ to all fluxes. Despite these modifications, the curves fitted to the entire data set do not represent the data measured between 10 and 20 °C very well, as can be seen from Fig. 4 in Song et al. (2009). This demonstrates that the temperature range used for the curve fit is important, if $CH_4$ flux studies are being compared with each other. Finally, the response curve for a subarctic wetland (Jackowicz-Korczyński et al., 2010) has a similar curvature ($b = 0.167$ °C$^{-1}$) as the function derived in this study, but a much higher scaling factor ($a = 30.48$ m$^{-2}$ d$^{-1}$). From this comparison it seems possible that a universal value for the curvature '$b$' could be derived from a global $F_{CH_4}$ data set comprising both pristine and restored wetlands, provided a common temperature range is used for the curve fits. Such a result would leave modellers with one site specific scaling factor '$a$' that would depend on hydrology and possibly vegetation type and soil organic carbon.

Substrate supply to the methanogenic microorganisms in the soil can control the $CH_4$ production rates, and it has been shown, both from theoretical considerations and from field measurements, that the $CH_4$ release into the atmosphere is coupled to the net primary production (NPP) of the vegetation (Friborg et al., 2000; Joabsson and Christensen, 2001; Whiting and Chanton, 1993, 2001; Nykänen et al., 2003), with vascular plants providing the substrate for the methanogenesis and transporting $CH_4$ from the soil to the surface, thus preventing oxidation in the upper soil layers. Due to the time lag between $CO_2$ uptake through photosynthesis and increase in carbohydrate content in the soil, the ratio between $CH_4$ release and $CO_2$ fixation typically exhibits a seasonal variability and is highest in the late summer and autumn (Shurpali et al., 1993; Whiting and Chanton, 2001; Liikanen et al., 2006). However, in practice the NPP effect is often overridden by water level effects (Joabsson and Christensen, 2001; Lai, 2009). It cannot be excluded that the $CH_4$ flux rates observed in September 2009, which were unusually high in comparison to the average temperature response (Fig. 7A) were partly due to such a delayed response to the high NPP rates that were measured at Skjern Meadows in the spring and summer 2009 (Fig. 7B).

Not only has the amount of carbon fixation through the vegetation, but also the vegetation type been identified as a controlling factor for the $CH_4$ release from wetlands (Ström et al., 2003; Petrescu et al., 2008). Both the substrate quality and plant-mediated gas transport have a role in this relation (Joabsson and Christensen, 2001). The presence of sedges, for example, creates an effective additional diffusion pathway for $CH_4$ through the plants (Treat et al., 2007), and this is reflected in higher $CH_4$ emission totals. Also Tuittila et al. (2000) emphasised the importance of species-specific biomass dynamics and litter production for the $CH_4$ fluxes and demonstrated that this can explain the observed spatial variability in $CH_4$ emissions to a large extent. If this variability is large, then also the eddy fluxes obtained over a heterogeneous wetland will vary with the source area, as was observed in this study.

The presence of open river surfaces can also alter the average $CH_4$ release of a wetland area. Silvennoinen et al. (2008) measured a noticeable release of 66 mg $CH_4$–C m$^{-2}$ d$^{-1}$ from a river draining a peatland, but on average the $CH_4$ emission from river surfaces is much lower than from lakes, mires and marshes (Saarnio et al., 2009). This confirms the interpretation given in this study for the relatively low $CH_4$ fluxes for a defined sector where the Skjern River flows through the source area of the eddy flux mast. Other controlling factors that have been discussed include, for example, the content of organic acids in the soil solution (Christensen et al., 2003) or peat acidity in general (Lai, 2009), air pressure (Tokida et al., 2007) and the atmospheric turbulence near the surface in terms of $u_*$ (Sachs et al., 2008; Wille et al., 2008).

In several studies $CH_4$ fluxes measured at low turbulence were excluded from further analysis with cutoff points ranging from $u_* = 0.1$ m s$^{-1}$ (Friborg et al., 2000) to 0.17 m s$^{-1}$ (Long et al., 2010) and even 0.2 m s$^{-1}$ (Rinne et al., 2007) because it was expected that under these circumstances the turbulence was not sufficient for the $CH_4$ emissions to be detected by the instrumentation. How-

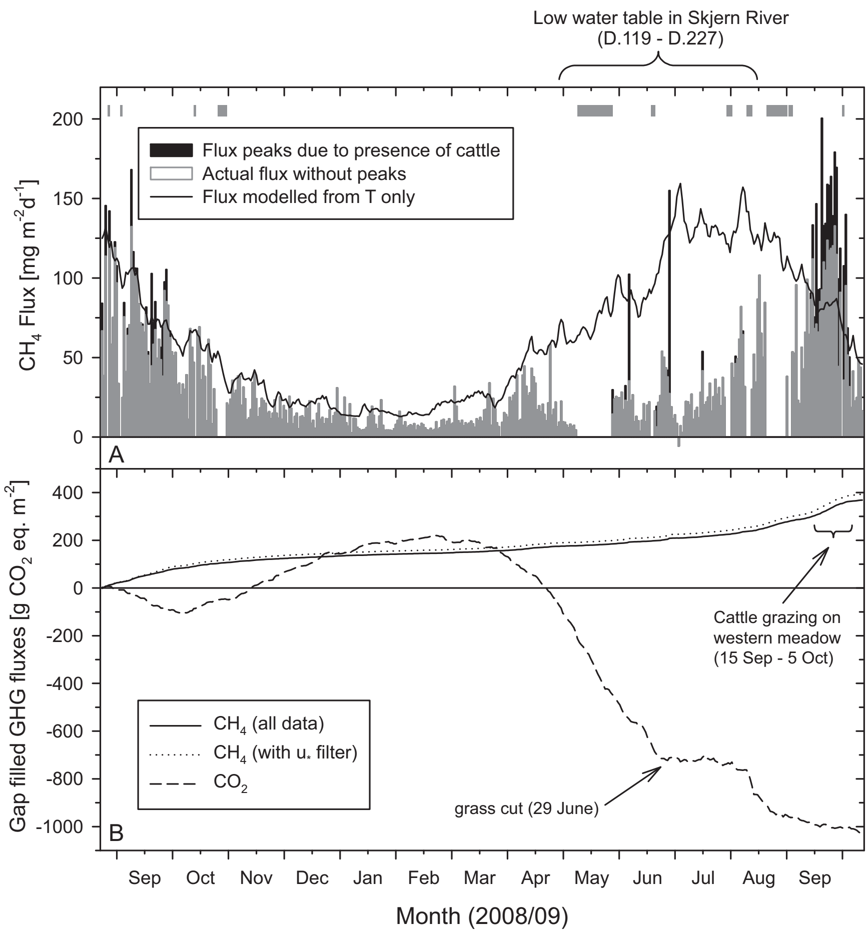


**Fig. 7.** (A) Time course of the measured daily $CH_4$ emission from Skjern Meadows (bars). The estimated fraction of the flux caused by the presence of cattle is indicated in black. Periods with missing data are indicated as grey bars. The line shows the theoretical $CH_4$ flux calculated from the function describing the upper envelope in Fig. 5. (B) Cumulative fluxes of $CH_4$ and $CO_2$, expressed in $CO_2$ equivalents. The dotted line is based on the rejection of all data with $u_* < 0.15\,\mathrm{m\,s^{-1}}$ (see text for further explanations).

ever, if $u_*$ (or wind speed, to which it is linearly related according to Wille et al., 2008) really controls the actual fluxes and not just their detectability, the application of a $u_*$ threshold induces a bias in the flux totals. Therefore only spikes were removed from the data but not periods with low $u_*$ in general, especially so since the storage flux was included in the calculation of the total flux.

Nevertheless the effect of a $u_*$ filtering was tested by removing all 10 min fluxes obtained at $u_* < 0.15\,\mathrm{m\,s^{-1}}$. According to Fig. 4B this appeared to be the most likely detection limit, if there was such a limit at all. Apart from this, the part of the total $CH_4$ emission coming from storage change, and not from turbulent flux, was largest at low $u_*$, and the storage flux had a relatively large uncertainty since the $CH_4$ concentration of the air was only measured at one height. Applying the $u_*$ filter removed 18% of the data, and the gaps were filled by the average rates from the same day. This procedure was justified because of the absence of diurnal variations in the $CH_4$ flux (Fig. 4A), and it ensured that gaps were filled with data representing similar environmental conditions. The result of this exercise was a modest increase of the total $CH_4$ emissions by 6.6% (Fig. 7B), and a small change in the parameters of the response functions in Fig. 5 to $a = 8.13$ and $b = 0.148$ for the 'wet' period and to $a = 1.94$ and $b = 0.155$ for the 'dry' period, with unchanged $R^2$ values.

Despite the removal of some uncertainty associated with storage and advection through $u_*$ filtering, it appears likely that an even larger uncertainty would be introduced, given that a strong control of $u_*$ on the actual $CH_4$ fluxes was indeed observed by Wille et al. (2008) and Sachs et al. (2008) who explained it by the fact that a relatively large number of ponds contributed to the source area. In one of the earliest $CH_4$ eddy flux studies, also Fan et al. (1992) observed a tight correlation between wind speed and $CH_4$ flux above Arctic tundra, but only for wind directions where open water contributed to the source area of the eddy fluxes. It is well known that the $CH_4$ flux across the water–air interface of wetland ponds increases approximately with the square of wind speed (Sebacher et al., 1983). Therefore it seems plausible that a similar effect of $u_*$ on the $CH_4$ fluxes existed at Skjern Meadows where it was apparent during the winter season but not in the summer, when the water pools on the wetland surface had disappeared. The $u_*$ effect would not have been detectable by common chamber techniques, and this not only questions previous results purely based on chamber measurements (Sachs et al., 2008) but demonstrates why it is important to complement them with measurement techniques that have minimum impact on the natural environment.

### 4.2. $CH_4$ emissions from restored wetlands

With the focus of previous research being on pristine wetlands, only a few studies have addressed effects of land use change and management on $CH_4$ emissions. For example, the time since the restoration of a wetland can strongly influence its greenhouse gas exchange with the atmosphere. Within a four year study starting with the rewetting of their research site, Waddington and Day (2007) observed a noteworthy $CH_4$ emission only in the fourth year and only in an area where no cutover of the vegetation took place. From a model analysis based on three years data, Tuittila et al. (2000) concluded that the $CH_4$ emissions following the rewetting of a peatland will remain lower than at pristine sites over many years. The review article of Drösler et al. (2008) confirms this and also points out that the time after rewetting can determine the sign of the $CO_2$ balance and thus the role of a wetland as a source or sink of greenhouse gases. Long-term increases in the $CH_4$ and $CO_2$ fluxes by factors of three and two, respectively, have also been reported for a constructed wetland over a period of ten years (Liikanen et al., 2006). Given that this study started about five years after the restoration of Skjern Meadows, it cannot be excluded that both the $CO_2$ and $CH_4$ exchange rates will increase over the coming years.

Prior to this study, areally averaged, seasonal $CH_4$ emission rates from restored wetlands have only been reported twice (Hendriks et al., 2007; Waddington and Day, 2007). The average $CH_4$ flux from a restored wetland in the Netherlands with an annual average temperature of 9.8 °C equalled 87 mg m$^{-2}$ d$^{-1}$ (Hendriks et al., 2007), which is about three times as high as the corresponding rate given by Waddington and Day (2007) for a restored wetland in Canada, where 4.2 g m$^{-2}$, observed over a growing season of 150 days with an average soil temperature around 15 °C, corresponded to a seasonal average of only 28 mg m$^{-2}$ d$^{-1}$. The emission rates for the Canadian wetland refer to the fourth year after the rewetting, whereas the Dutch study refers to a wetland 10 years after its restoration. There was neither cutting nor grazing at these two sites, and the only management activity was the regulation of the water table which was kept between 0 and 0.45 m under the surface at the Dutch site and between 0.1 and 0.7 m under the surface at the Canadian site.

When these $CH_4$ flux rates are compared to the present study, where an annual average rate (October 2008 to September 2009) of 30.2 mg m$^{-2}$ d$^{-1}$, with a corresponding soil temperature of 9.8 °C, was observed 5 years after the restoration of Skjern Meadows, it becomes clear that the annual $CH_4$ fluxes from restored wetlands cannot be related to a single control factor. For example, temperatures were similar for the Dutch and the Danish site, whereas flux rates and time after rewetting were similar for the Danish and the Canadian site. However, from the curve shown in Fig. 7 it was estimated what the $CH_4$ flux at Skjern Meadows would have been if it had shown a universal temperature response throughout all seasons. This calculation resulted in a theoretical emission rate of 61.8 mg m$^{-2}$ d$^{-1}$, which is still lower than, but much closer to, the value given by Hendriks et al. (2007) for similar climatic conditions but in the absence of grazing and cutting and with a regulated water table. This comparison points towards a possible influence of both the management regime and the time scale on the methane emissions from restored wetlands. A large scale relation between seasonal average temperatures and methane fluxes, as was found for natural wetlands (Christensen et al., 2003), might therefore be difficult to derive when restored wetlands are included.

Management effects on the $CH_4$ release from wetlands are currently least well understood. In particular, none of the papers dealing with gas exchange of restored wetlands mentions the potential influence of grazing animals on $CH_4$ flux measurements, despite the awareness of grazing effects on the $CO_2$ budget of grasslands (Klumpp et al., 2007). The amount of $CH_4$ emissions from cattle is relatively well known. For Denmark it has been found that dairy cattle on average emit 126 kg $CH_4$ per head per year through respiration and eructation and 19 kg through manure, whilst non-dairy cattle (bulls and calves) emit 37 kg (of which 2 kg occur through manure) on average (Nielsen et al., 2008). However, the exact emission rates vary between individuals and depend on the diet, which can cause variations in the fraction of energy intake converted into $CH_4$ from 2 to 12% (Johnson et al., 1994). If the average emission of 345 mg $CH_4$ per day by one dairy cow is compared to typical $CH_4$ emission rates from vegetation (with the majority of studies reporting seasonal averages between 20 and 150 mg m$^{-2}$ d$^{-1}$), it becomes clear that cattle can be $CH_4$ hotspots that must be included in spatially averaged $CH_4$ balances where grazing is a part of the management concept for peat meadow restoration.

In this context, the application of suitable micrometeorological techniques can be particularly valuable for two main reasons. Firstly, the transferability of $CH_4$ emission rates per animal based on indoors experiments to the field is generally questionable (Johnson et al., 1994) and secondly, micrometeorological measurements cover not only the methane produced by the ruminants themselves but also possible changes in $CH_4$ emissions from the soil. Such changes could occur when animals trample the vegetation and compact the soil, which might stimulate the ebullition of $CH_4$ from the soil pore water. Prior to this study, no eddy covariance measurements, but several other micrometeorological techniques such as the flux-gradient method as well as boundary-layer budgeting and mass budgeting approaches have been used to estimate $CH_4$ emissions from grazing animals in the field (Denmead et al., 2000; Laubach and Kelliher, 2004). They successfully verified inventory predictions of the $CH_4$ emissions from cattle and sheep herds, but required much more instrumentation and more complicated sampling and data analysis than conventional eddy covariance measurements do. The animals were also restricted to comparatively small paddocks where they grazed in a higher density than the cattle on Skjern Meadows, and it has yet to be investigated if those other micrometeorological methods would also be able to detect fluxes from non-intensive grazing against the background of simultaneous soil emissions.

Therefore it seems advantageous to use a routine eddy covariance setup to monitor both soil and animal emissions from restored wetlands. If both the cattle and the source area 'seen' by the eddy covariance sensors move freely across the meadows, then a sufficiently long observation period will result in a representative average flux rate, provided the two movements occur randomly and are independent of each other. A suitable footprint model could help shortening the time period required to obtain such an average rate. Another precondition for the applicability of the eddy covariance method is that the data quality must not be affected by the presence of the animals. During all cattle induced spikes in $F_{CH_4}$ the Foken et al. (2004) quality criteria were matched. Neither the ratio between $u_*$ and the fluctuation of the vertical wind component ($\sigma_w$), as an additional quality indicator, nor the ratio of $u_*$ over $u$ (horizontal wind speed), as a proxy for the surface roughness, did show any differences between spikes and background data. The density of the cattle was too low to affect the roughness length ($z_0$) of the source area noticeably. According to the method of Laubach & Kelliher (2004) $z_0$ was calculated from height ($h$) and facial area ($A_f$) of the bulls and ground area occupied by the herd ($A_g$) as $z_0 = 0.5h^*A_f/A_g$. It equalled 0.3 mm if the animals were distributed randomly or, more realistically, 7 mm if they were grouped on 1 ha during grazing. This $z_0$ value was still negligible compared to the estimated $z_0$ of 4 cm for the vegetation (calculated as 0.13 $h_{veg}$, with $h_{veg}$ being the average vegetation height of 30 cm).

Based on the low stocking density of 45 bulls grazing an area of 23 ha on 'Poldene' (Table 1) during a period in autumn 2009 with prevailing westerly winds, the numbers given by Nielsen et al. (2008) would predict that a modest additional methane flux of 20 mg m$^{-2}$ d$^{-1}$ originating from the animals' mouths and their manure could be expected for this part of Skjern Meadows. The average of the black bars in Fig. 7 for this period, corresponding to the measured contribution of cattle to the eddy flux, is 38 (±25) mg m$^{-2}$ d$^{-1}$. The large standard deviation indicates that the selected three weeks period probably was too short to determine a robust average flux rate. Therefore it remains unclear if the difference between the inventory based estimate and the flux calculated from the eddy covariance data corresponds to a real increase in soil

$CH_4$ emissions due to the cattle's presence. Nevertheless it seems possible that this question can be answered when a longer time series of data will have been collected. To our knowledge, this is the first study that documents the actual influence of grazing on in situ $CH_4$ eddy fluxes.

### 4.3. Implications for the full GHG balance

On a 100-year timescale, the GWP of methane is currently assumed to be 25 (IPCC, 2007). The direct comparison of global warming effects of $CH_4$ and $CO_2$ in relation to ecosystems is still a matter of debate (Frolking and Roulet, 2007), but for the present study we adapt the convention commonly used for anthropogenic emissions. In terms of pure carbon, this corresponds to a ratio of about 9 between the impact of $CH_4$–C and $CO_2$–C. On this background it has been pointed out that a wetland can be a carbon sink and greenhouse gas source at the same time (Whiting and Chanton, 2001; Friborg et al., 2003; Rinne et al., 2007). From chamber studies conducted over a large latitudinal gradient, Whiting and Chanton (2001) concluded that, on a 100-year time scale, wetlands in the temperate zone would function as greenhouse gas sinks whereas boreal wetlands would have a gas exchange close to the "greenhouse compensation point". However, the assumptions underlying these estimates hold for peatlands being in equilibrium with their environment and do not apply to areas that have been subject to land-use changes such as drainage, restoration or grazing. Since these are responsible for a substantial part of the observed variability in methane emissions between sites, further studies should not be restricted to natural wetlands, but account for land use changes too.

The comparison between the true $CH_4$ flux and the hypothetical 'wet' flux shown in Fig. 7A indicates that seasonality in terms of a drying of the top soil largely determined the sink strength of Skjern Meadows for greenhouse gases in terms of their GWP. Over the entire 415 day period the $CH_4$ emission was 369 g $CO_2$ eq. $m^{-2}$ without $u_*$ filtering or 393 g $CO_2$ eq. $m^{-2}$ with $u_*$ filtering (Fig. 7B). On an annual basis (1 October 2008 to 30 September 2009) 11 g $CH_4$ $m^{-2}$ $a^{-1}$ were emitted, corresponding to 276 g $CO_2$ eq. $m^{-2}$ $a^{-1}$. The $CO_2$ uptake over the same time interval was 908 g $m^{-2}$ $a^{-1}$ (Fig. 7B). Thus, in terms of GWP, the actual $CH_4$ emission already reduced the $CO_2$ uptake at Skjern Meadows by about 30%. A management scenario involving a regulated, high water table (which would be likely to double the $CH_4$ emissions according to Fig. 7A) and an increased grazing intensity could therefore easily counterbalance the complete $CO_2$ uptake in terms of the greenhouse effect. This projection is based on the observation that most of the seasonal variation in the $CO_2$ flux at Skjern Meadows could be explained by irradiance, temperature and leaf area index alone (Herbst et al., in press), which makes changes in the annual $CO_2$ budget due to an altered soil water regime unlikely. The absence of a clear response of the $CO_2$ flux to the water table depth was also reported by Moosavi and Crill (1997), and their conclusions were recently backed up by Parmentier et al. (2009) who showed that the $CO_2$ exchange of a Dutch peatland was unaffected by changes in the water table.

A better understanding of the various environmental control mechanisms for the atmospheric $CH_4$ flux requires more continuously operating monitoring stations. Both the spatial variability and the episodic nature of the $CH_4$ flux (Treat et al., 2007; Lai, 2009) call for more long-term eddy flux studies, as a necessary supplement to chamber measurements, that cover the variety of methane producing ecosystem types described by Drösler et al. (2008). The latest technical improvements in high-frequency gas analysis have made a wider application of this technique possible. In order to get a more balanced picture of the role of wetlands in the global greenhouse gas budget, it appears especially important that further studies include a number of areas in the temperate zone (Saarnio et al., 2009), where management practices could possibly help to mitigate the release of greenhouse gases from wetlands into the atmosphere.

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