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Climate and site management as driving factors for the atmospheric greenhouse gas exchange of a restored wetland

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Abstract

The full atmospheric greenhouse gas (GHG) budget of a restored wetland in Western Denmark could be established for the years 2009–2011 from eddy covariance measurements of carbon dioxide (CO_2) and methane (CH_4) fluxes. The water table in the wetland, being restored in 2002, was unregulated, and the vegetation height was limited through occasional grazing by cattle and grass cutting. The annual net CO_2 uptake varied between 195 and 983 $g\,m^{-2}$ and the annual net CH_4 release varied between 11 and 17 $g\,m^{-2}$. In all three years the wetland was a carbon sink and removed between 42 and 259 $g\,C\,m^{-2}$ from the atmosphere. However, in terms of the annual GHG budget (assuming that 1 g CH_4 is equivalent to 25 g CO_2 with respect to the greenhouse effect) the wetland was a sink in 2009, a source in 2010 and neutral in 2011. Complementary observations of meteorological factors and management activities were used to explain the large inter-annual variations in the full atmospheric GHG budget of the wetland. It is shown that the largest impacts on the annual GHG fluxes, eventually defining their sign, came from site management through changes in grazing duration and animal stocking density and from extreme weather patterns through an unusually long period of snow cover in the second year of observations. Since integrated CO_2 and CH_4 flux data from restored wetlands are still very rare, it is concluded that more long-term flux measurements are needed to predict the role of this land use type in the atmospheric GHG budget more accurately.

1 Introduction

The exchange of greenhouse gases (GHG) such as carbon dioxide (CO_2) and methane (CH_4) between terrestrial ecosystems and the atmosphere is essential for the atmospheric concentration of these gases. Potential impacts of land use transformations on the GHG budget are therefore accounted for in international agreements such as the Kyoto protocol. To enable a comparison of the climate effect of individual GHGs,

BGD
9, 9029–9064, 2012

their global warming potential (GWP) is often expressed in units of CO_2 equivalents that would have the same radiative forcing effect over a time horizon of 100 yr (IPCC, 2007). In this way the full GHG budget of a land surface is quantified, and the climate effect of the individual gases can be compared. Considering the role of wetlands in the GHG budget of the land surface, there has thus been an increasing awareness that a carbon sink can be a greenhouse gas source, with CH_4 emissions counterbalancing the CO_2 uptake, if calculated according to the aforementioned method (Whiting and Chanton, 2001; Friborg et al., 2003; Schulze et al., 2009).

The environmental control of CO_2 fluxes above land ecosystems is meanwhile fairly well understood, not least because the eddy covariance (EC) observation network "Fluxnet" (Baldocchi et al., 2001) has enabled a global analysis of long-term data that has made it possible to parameterise and validate physiologically based land-atmosphere carbon exchange models for many vegetation types (Houborg et al., 2009). Nevertheless there are still uncertainties in these parameterisations, for example regarding the influence of the water table height in wet soils on ecosystem respiration (Lloyd, 2006; Parmentier et al., 2009) or regarding climate-management interactions, since managed sites are still underrepresented in "Fluxnet" (Klumpp et al., 2011).

In contrast, CH_4 has only recently received attention in the "Fluxnet" community, and as yet the few available long-term data are highly variable and difficult to interpret (Baldocchi et al., 2012). As a consequence, models of ecosystem CH_4 fluxes are not yet as robust, uniform and transferable as those for CO_2, although the underlying processes and the control of CH_4 emissions by water table height and soil temperature were already well described 20 yr ago by numerous studies based on chamber measurements mostly in natural wetlands in North America (e.g. Crill et al., 1988; Roulet et al., 1992; Bubier et al., 1993). Recent advances in sensor technology have meanwhile made it convenient to observe also CH_4 fluxes directly at the canopy scale using the EC technique. Several sensor configurations have been proven to be reliable and robust enough for continuous, unattended long-term studies (Rinne et al., 2007; Hendriks et al., 2008; Detto et al., 2011; Dengel et al., 2011), and the reported fluxes are in good agreement with up-scaled chamber measurements (Schrier-Uijl et al., 2010). The EC technique enables in situ observations of potential influence factors on the CH_4 flux that are out of reach for chamber studies, such as e.g. wind, flooding or grazing (Sachs et al., 2008; Meijide et al., 2011; Herbst et al., 2011a).

As yet there are not many multi-year EC studies of both CH_4 and CO_2 fluxes (Kroon et al., 2010), and we are not aware of any study that analyses specifically the interannual variability of the full GHG budget rather than its long-term average. This kind of analysis is particularly important for restored wetland areas that are not yet in equilibrium (Waddington and Day, 2007; Höper et al., 2008) and for wetlands that are amenable to management. Despite the advantages of the EC technique it should be noted that for spatially heterogeneous sites only a long-term observation will produce a representative integrated flux estimate, whereas on a short term basis changes in the source area of the fluxes can override their environmental controls (Forbrich et al., 2011; Baldocchi et al., 2012). Thus, a detailed process analysis dealing with the various pathways of CO_2 and CH_4 emissions from restored wetlands would require an accurate footprint modelling and the mapping of surface properties at high spatial resolution, which will be subject of a separate study. The aims of this paper are (1) to quantify the inter-annual variability of the atmospheric GHG exchange of a restored wetland in Western Denmark and (2) to assess the relative importance of the various climatic and site management factors that control the reported variability.

2 Materials and methods

2.1 Research site

"Skjern Meadows" in Western Denmark is one of the largest restored wetlands in Northern Europe. In 1968, roughly 4000 hectares of peatlands, wet grasslands and marshes in the valley of the Skjern River were drained and converted into agricultural land. Between 1999 and 2002, 2200 hectares of this area were restored by re-filling the channelised river stretches and excavating a new, meandering river course (Nielsen and Schierup, 2007). The study site is located on a floodplain not far from the mouth of the Skjern River where the soil is dominated by Fluvisols, according to the FAO soil classification system. "Skjern Meadows" is covered by various international conventions. For example, it is categorised as "Wetland of International Importance" according to the "Ramsar Convention" (http://www.ramsar.org).

The vegetation on the former cultivated fields changed rapidly as a consequence of the restoration and the changed hydrological conditions (Pedersen et al., 2007). 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 % (Andersen et al., 2005; Pedersen et al., 2007).

The vegetation height is limited by both grazing and hay making. These activities also prevent the development of the meadows into woodland. The vegetation at our site was cut for hay production on 29 June 2009 and on 8 August 2011. The total annual duration of grazing by cattle ranged from three weeks in 2009 to about four months in 2010 (see Figs. 3 and 4). The numbers of livestock varied between 2 bulls (15–24 months old) per hectare in 2009 and 4.5 bulls per hectare in the other two years, with the size of the herd being 45 and about 110, respectively.

2.2 Instrumentation

The atmospheric fluxes of CO_2 and CH_4 were determined with the eddy covariance technique. The instrument mast is located at 55°54′46″ N and 8°24′17″ E, at an elevation of 2 m 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 on a shallow earth bank close to a small gravel road (see Herbst et al., 2011a, for further details). A sonic anemometer (R3-50, Gill Instruments Ltd., Lymington, UK) was installed at the top of the mast at a measurement height of 7 m above the wetland surface. The relatively large measurement height was chosen in order to minimize the influence of the hedgerow on the turbulence. The CO_2 concentration in the air was measured with an open-path infrared gas analyser (LI-7500, LI-COR Inc., Lincoln, NE, USA) mounted close to the sonic anemometer, and 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 output spectroscopy. The analyser was placed in a hut near the mast and was connected to the measurement point near the sonic anemometer through a 10.65 m long tube having an inner diameter of 6 mm. A vacuum pump (XDS35i, Edwards, Sanborn, NY, USA) dragged the air into the DLT-100 measurement cell at a true pumping speed of 229 $l\,min^{-1}$, ensuring a fully turbulent flow with a Reynolds number of 8200 and resulting in an average cell pressure of 16 kPa. The response time was 0.14 s and the cut-off frequency was 1.1 Hz (Herbst et al., 2011a). The H_2O concentration necessary for the Webb correction was not measured inside the DLT-100, but in the air close to the tube inlet of the DLT-100 by means of a closed path infrared gas analyser (LI-7000, LI-COR Inc., Lincoln, NE, USA). Since 2010 this instrument was no longer available for the study, and the H_2O concentration from the LI-7500 was used instead. All measurements were taken at a nominal frequency of 10 Hz.

Auxiliary data collected near the instrument mast comprised soil temperatures at three depths, air temperature and humidity as well as up- and downward short- and longwave radiation. All data (fast eddy covariance data plus half hourly auxiliary data) were logged on a CR3000 data logger (Campbell Scientific Ltd., Shepshed, Loughborough, UK). Since all data were acquired and synchronised digitally, potential noise due to digital to analogue transformations (Eugster and Plüss, 2010) was avoided. The synchronisation of the sonic anemometer and the Licor gas analysers was achieved through polling the instruments by the data logger, however the DLT-100 sent its data independently at an average frequency of about 9.4 Hz. As a consequence, about every 20th record of the 10 Hz sonic data had no matching CH_4 record and was thus

BGD
9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

discarded for the CH_4 flux calculations. Every night the raw data were transferred automatically from the logger to an on-site server computer and further to Copenhagen University.

A web camera (TN-TV-IP400, http://www.trendnet.com) being installed at 5 m height on the instrument mast and pointing into the main wind direction (west) has been in operation since the early spring of 2009. It takes one picture every day at 11:00 local time which documents the visual development of the vegetation and the approximate height of the water table as indicated by water pools in certain places. For more details about the instrumentation including instrument maintenance and performance we refer to Herbst et al. (2011a) and Ringgaard et al. (2011).

2.3 Flux calculations and gap filling

The turbulent fluxes of CO_2 and CH_4 were calculated from the covariances of the vertical wind component and the respective gas concentrations by means of the "Alteddy" software, version 3.5 (Alterra, University of Wageningen, The Netherlands). A dilution correction (Webb et al., 1980) was performed for both the CO_2 and the CH_4 fluxes, and the CO_2 fluxes were additionally corrected for errors due to surface heating of the LI-7500 sensor head using the standard parameterisation of Burba et al. (2008). All fluxes were corrected for errors caused by the tilt of the anemometer relative to the mean streamline coordinate system by means of the "planar fit" method (Wilczak et al., 2001).

Following a quality control of the flux data according to Foken et al. (2004), only data having one of the three best quality flags were accepted for further analysis. All other data, along with a few spikes undetected by Foken's method, were rejected and, in case of CO_2, replaced by values estimated by the standard gap-filling method used by the FLUXNET community which is available as online tool at http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/ (Moffat et al., 2007). This affected a bit more than one third of all half-hourly CO_2 flux data. For methane, daily averages were calculated, and gaps in the daily data, arising from situations when less than 12 h of good data were available for a particular day, were filled with estimates based on soil temperature and season, using response functions of the form

$$F_{CH_4} = A \cdot e^{B \cdot t} \tag{1}$$

where A and B are empirical parameters and t is soil temperature at 20 cm depth in °C (Herbst et al., 2011a). About 12 % of all daily CH_4 fluxes had to be estimated in this way.

Since the total atmospheric fluxes of CO_2 and CH_4 consist of contributions from the turbulent fluxes measured with eddy covariance and the flux into storage in the air column below the sonic anemometer, the changes in storage were estimated from the changes in CO_2 and CH_4 mole fraction over the respective time interval and added to the turbulent fluxes.

According to the most commonly used GWP, the radiative forcing of 1 g CH_4 is equivalent to that of 25 g CO_2 (IPCC, 2007). The CH_4 fluxes measured at Skjern Meadows (in g m^{-2}) were thus multiplied by 25 and added to the measured CO_2 flux to give the total GHG budget. Another important greenhouse gas, nitrous oxide (N_2O), was not measured because only small fluxes were expected (see Sect. 4.2). Thus N_2O was neglected in the total GHG balance.

The observed net CO_2 fluxes (NEE) were separated into day and night time and related to incoming solar radiation (R_G) and air temperature (t) in °C, respectively. The non-linear regressions used were

$$\mathrm{NEE}_{\mathrm{night}} = R_{10} \cdot e^{308.6 \cdot \left(\frac{1}{56} - \frac{1}{t+46}\right)} \tag{2}$$

for the night-time data where R_{10} is the respiration rate at 10 °C (Lloyd and Taylor, 1994) and

$$\mathrm{NEE}_{\mathrm{day}} = \frac{a \cdot b \cdot R_G}{a \cdot R_G + b} + c \tag{3}$$

BGD

9, 9029–9064, 2012

for the day-time data (Ruimy et al., 1995; Frolking et al., 1998), where *a* corresponds to the initial slope of the curve, *b* is the maximum net CO_2 uptake rate, and *c* is the net CO_2 emission when the radiation is zero.

The curve fitting was performed with SigmaPlot 9.0 (SSI, San Jose, CA, USA). Its fitting routine is based on the Marquardt-Levenberg algorithm (Marquardt, 1963) which uses least squares analysis.

3 Results

3.1 Climatic conditions and annual atmospheric GHG exchange

The seasonal and annual average temperatures were fairly similar in 2009 and 2011, except for the warmer autumn in 2011 (Table 1). In contrast, 2010 was more than two °C colder than the other years. This was mainly due to an unusually cold winter and spring, whereas the mean temperature from July to September did not differ from the other years. 2010 was also drier than 2009 and 2011, with a similar seasonal distribution of the precipitation for all three years (Table 1). There was always more precipitation in the second half of the year than in the first half, and April was the driest month in all years (data not shown).

The daily totals of the CO_2 and CH_4 exchange between the wetland and the atmosphere are shown in Fig. 1 for the first 3.5 yr of the observations. For the CO_2 fluxes the expected general tendency towards negative (downward) fluxes in the summer and positive (upward) fluxes in the winter is evident, however positive daily totals were observed for several days in the summer, too. In contrast to CO_2, the CH_4 exchange was almost always directed upwards, and in all years the largest CH_4 emissions occurred in late summer and early autumn (Fig. 1). Apparently, the daily peak emission rates have increased from year to year since the start of the measurements in the late summer of 2008.

BGD
9, 9029–9064, 2012

If the daily CH_4 fluxes are converted into CO_2 equivalents and balanced against the daily CO_2 fluxes, the total greenhouse gas exchange (with respect to the global warming potential of the two gases) can be calculated and cumulated over an entire calendar year. The inter-annual comparison of the cumulative GHG budget (Fig. 2) revealed marked differences between the three years. Whilst Skjern Meadows was a substantial GHG sink in 2009, it turned into a moderate source in 2010 and became about neutral in 2011, considering the typical uncertainty of the flux totals (see Table 2 and Sect. 4.1). The net GHG uptake started late in the cold and dry year (2010) compared to the other two years. In 2010 the period of a net GHG uptake ended already in late May, whereas it otherwise lasted until early August (2011) or even late August (2009).

Regardless of these differences, the research site always remained a carbon sink (Table 2). Table 2 also demonstrates that the inter-annual variability of the CO_2 flux so far has been larger than the variability in CH_4 emissions. The highest and lowest annual totals differed by a factor of five for CO_2 but only by a factor of about one and a half for CH_4. In the following paragraphs the annual budgets and the temporal variability in relation to various environmental factors will be treated separately for the two greenhouse gases.

3.2 Temporal variability of CO_2 fluxes

The large differences between the individual annual courses of the CO_2 flux were related to specific management and weather events (Fig. 3). Due to the similar seasonal rainfall distribution the periods with low and high water tables hardly differed between the years. However, the duration of the snow cover varied considerably, with 2010 having an exceptionally long period of snow cover, which, according to Denmark's Meteorological Institute, was indeed the longest since the start of their recordings more than a hundred years ago. This situation is mirrored in the CO_2 flux, especially if early 2009 is compared with early 2010. The turning point of the cumulative net flux was reached more than a month later in 2010 than in 2009 due to the delayed start of the growing season (Fig. 3).

The further seasonal course and its interannual differences correlated with management activities rather than weather events. The long period of grazing in 2010 brought the net CO_2 uptake to a halt already in late spring, whereas the net uptake continued into August in 2011 and well into autumn in 2009. The re-growth following the late grass cutting in 2011 could not compensate any more for the respirative CO_2 emissions, whilst the early cut in 2009 enabled a full recovery of the vegetation in terms of its leaf area index (Herbst et al., 2011b) and thus a continued net CO_2 uptake over the rest of the growing season (Fig. 3).

3.3 Temporal variability of CH_4 fluxes

Most of the steepest parts of the cumulative annual CH_4 flux curves coincided with periods of grazing (Fig. 4). The continuous grazing in the summer of 2010, for example, caused considerable CH_4 emissions even during the summer when the low water table must have inhibited direct CH_4 emissions from the soil due to oxidation of CH_4 in the aerated top soil. This indicates that CH_4 emissions from rumination must have contributed to the measured flux. Despite a reduced grazing intensity in 2011 the total annual CH_4 efflux was even higher than in 2010, not least because the emissions rates in the autumn, after the end of the grazing period, were higher than in both other years. This means that, during the observation period, the annual CH_4 emissions have so far increased from year to year (Table 2 and Fig. 4). The varying timing and duration of the snow cover did not have any influence on the CH_4 emissions rates, according to Fig. 4.

The interaction of water table height, soil temperature and presence of grazing cattle in driving the CH_4 emissions from Skjern Meadows is illustrated in Fig. 5. The respective symbols indicate a clear functional switch between situations with low and high water table and a strong exponential temperature influence for periods with high water table. Daily CH_4 fluxes for days with grazing are often found at the higher end of the range of values for the respective temperatures (under both dry and wet soil conditions), however a clear separation was not possible. When the water table was high and the soil temperature exceeded a threshold of about 9 °C, the CH_4 emissions rates appeared to increase from year to year, even when no cattle were present in the source area of the flux measurements (Fig. 5). Possible reasons for this observation will be discussed in Sect. 4.4. The curve fits (Table 3) indicate a higher basal rate and lower temperature sensitivity for the year with the most intense grazing (2010).

4 Discussion

4.1 Accuracy and source area variability

The typical uncertainty for long-term eddy covariance measurements of CO_2 and CH_4 has often been estimated to be about 15 % of the total flux when expressed as a relative error (e.g. Schrier-Uijl et al., 2010; Herbst et al., 2011b). This estimate was used in this study, too. Potential sources of uncertainty are sensor resolution and calibration errors, flux corrections due to, for example, frequency losses and density fluctuations and uncertainties in the gap filling. According to Kroon et al. (2010) the uncertainty in annual eddy covariance CH_4 fluxes is even smaller than 10 % if the data coverage exceeds 80 % and a multivariate regression is used for gap filling. However, as the named errors can occur independently for CO_2 and CH_4 fluxes (being based on data from different gas analysers and subjected to different control factors), the estimated absolute errors for the two gas fluxes were added to estimate the uncertainty of the total annual GHG budget (Table 2). As a result, it can be reasonably assumed that the statements about Skjern Meadows being a GHG sink in 2009 and a GHG source in 2010 are certain. In contrast, the small net GHG emission estimated for 2011 is well within the uncertainty limit of the method used, which means that the 2011 budget was too close to zero to define it with certainty as a sink or a source.

The main focus in this paper was on annual balances, because these are less sensitive to additional uncertainties caused by changes in the source area of the observed fluxes (Forbrich et al., 2011; Baldocchi et al., 2012). Methane fluxes in particular are known to be spatially highly variable, and thus changes in the source area due to

BGD
9, 9029–9064, 2012

changes in wind direction or atmospheric stability can dominate the observed variations in the fluxes and mask their environmental control over time scales of hours to days (Forbrich et al., 2011). Herbst et al. (2011a) presented evidence that this plays a role at Skjern Meadows, too, not only with respect to flux peaks coming from the cattle but also regarding the background fluxes that depended on the prevailing wind direction.

4.2 Atmospheric fluxes and the total GHG budget

In order to avoid any misinterpretations of this study it should be pointed out that only the atmospheric budget was considered here. A total field budget of greenhouse gases would have to include import and export of CO_2 (for example through manure application and hay making, respectively) as well as a potential leaching of CO_2 and CH_4 to the river and the weight gain of the animals (Allard et al., 2007; Soussana et al., 2007). A true farm scale GHG budget would have to cover even more processes with respect to meat production, transport etc. (Byrne et al., 2007). None of the named processes was subject of this study, however for the field budget this did not play a major role. Leaching and animal weight gain can be neglected in the bigger picture (Allard et al., 2007), and there was no import of CO_2. The only modification in order to arrive at a field budget would be to calculate the amount of biomass having been removed after the two cuttings in 2009 and 2011.

The last uncertainty when assessing total GHG budgets is the potential role of GHGs other than CH_4 and CO_2. The main suspect for a managed grassland would be nitrous oxide (N_2O), which is mostly emitted at medium soil water content but reduced in wet soil (van Beek et al., 2011). At a nearby site in the Skjern river valley, however outside the restoration area, Petersen et al. (2012) observed considerable N_2O emissions of $1.2\,g\,m^{-2}\,yr^{-1}$ (or $358\,g\,m^{-2}\,yr^{-1}$ in terms of CO_2 equivalents) from grassland on decomposing peat. In a seasonally wet grassland, however, N_2O accounted for only 1 % of the total atmospheric GHG flux when expressed as CO_2 equivalents (Gleason et al., 2009), and this percentage becomes higher only if the site is fertilized (Kroon et al., 2010), which is not the case at Skjern Meadows. Kroon et al. (2010) concluded that the N_2O flux from rewetted and unfertilized former agricultural land is negligible, and the same conclusion was reached in the synthesis paper by Maljanen et al. (2010). Therefore we consider it as unlikely that Skjern Meadows could have emitted significant amounts of N_2O during this study. In summary, the atmospheric GHG budgets presented here do almost represent a complete field budget, apart from the biomass removed through hay making.

4.3 Impacts of climate and management on CO_2 fluxes

Theoretically, wet grasslands can be expected to function as CO_2 sinks as they accumulate a major part of the carbon fixed through photosynthesis as peat, however the exact budget will depend on the vegetation and its canopy photosynthetic capacity and can be difficult to predict if a wetland is not yet in equilibrium, for example following restoration (Drösler et al., 2008). It is known that site management plays a role, too, and Jacobs et al. (2007) reported a rather unexpected average annual CO_2 release of $220 \pm 90\,\mathrm{g\,C\,m^{-2}}$ for four Dutch grasslands on organic soils having a relatively high water table and including both managed and unmanaged sites. In contrast, Hendriks et al. (2007) observed a consistent net CO_2 uptake between 232 and $446\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ for a restored wet grassland with neither grazing nor cutting. We are not aware of any other long-term study of CO_2 fluxes above grazed restored wetlands. However, looking at an example for a drained former wetland site that has not (yet) been restored, Hatala et al. (2012) determined the GHG budget of a grazed degraded peatland in California and showed that the site was a strong CO_2 source emitting up to $300\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ as CO_2 (and additionally 3 g C as methane). These numbers agree well with earlier studies from Europe where grasslands on drained peat soils are always net CO_2 sources because of peat decomposition, independent of the specific site management (Maljanen et al., 2010). The rewetting of such sites can therefore always be considered as an improvement with respect to the CO_2 (and GHG) balance, regardless of the exact annual net CO_2 flux and its variation.

BGD
9, 9029–9064, 2012

Regarding the NEE of an unfertilized, grazed grassland on mineral soil Klumpp et al. (2011) reported an interannual variability by one order of magnitude (net CO_2 uptake between 49 and 486 $g\,C\,m^{-2}\,yr^{-1}$), with the higher net uptake rates occurring in relatively cool years. If a similar response to the annual temperature was also present in this study, it was overshadowed by other control factors, because the coolest year (2010) actually showed the lowest net CO_2 uptake. The impact of the position of the water table on the CO_2 fluxes remains unclear, too, not only in this study but also in the literature. Lloyd (2006) observed a linear reduction in R_{10} (Eq. 1) with raising water table, which is in agreement with theory and expectations (Drösler et al., 2008). Parmentier et al. (2009), in contrast, reported exactly the opposite and concluded there was no influence of the water table height on ecosystem respiration. In the present study changes in the position of the water table can only have affected the seasonal course of NEE but not its interannual variability as the hydrological conditions were very similar in all 3 yr. This means also that the apparent, small effect on R_{10} (Table 4) cannot certainly be ascribed to water table differences because these were confounded with seasonality during the first 3 yr of measurements.

The only significant climatic influence on the annual net CO_2 flux was the delayed start of the growing season in 2010 due to the long lasting snow cover which produced an offset in the cumulative CO_2 flux during springtime of at least 200 $g\,m^{-2}$ (Fig. 3), corresponding to >20 % of the annual net CO_2 exchange. Even without the longer grazing period and the larger number of cattle this extreme weather pattern would thus have reduced the annual net carbon uptake by Skjern Meadows. A similar effect was observed at a subarctic site where, compared over many years, the snow melt date had the largest influence on the annual CO_2 budget (Aurela et al., 2004).

The grass cutting was the other major influence factor, as evident from Fig. 3. Looking at the critical periods in July and August in more detail, the net CO_2 flux data were separated into day and night and analysed in terms of Eqs. (3) and (2), respectively (Table 4). The variations in parameter "*a*" indicate that the daytime net CO_2 flux was reduced to about one half during the weeks following the cut (July 2009 and August 2011)

BGD
9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

due to the reduction in leaf area index (Herbst et al., 2011b). In contrast, differences in night-time fluxes, representing ecosystem respiration rates, and their relation to soil temperature were small between years and treatments (parameter R_{10} in Table 4). This seems plausible since the soil carbon pool, forming the substrate for respiration, does not immediately change with vegetation cover.

Grazing had a weaker effect than cutting on the daytime net CO_2 flux, however it remains unclear whether, and to what extent, cattle respiration contributed to the reduction in net CO_2 uptake compared to an undisturbed situation. The unexpected net CO_2 emissions on some days in the summer of 2010 (Fig. 1) may well have depended on the location of the herd compared to the flux footprint (Herbst et al., 2011a; Baldocchi et al., 2012). Grazing did not create an offset (change in parameter "*c*", Table 4) in the response of NEE to R_G but rather a reduction in daytime maximum NEE (parameter "*b*"). The most likely explanation for this result is that at night (when $R_G = 0$) the cattle usually congregated in an area several hundred meters away from the mast where they at most can have contributed to the "tail" of the flux footprint, and even this only for certain wind directions. Night-time ecosystem CO_2 fluxes would thus have been unaffected by cattle respiration. Due to the absence of a clear separation between background CO_2 flux and cattle respiration, we did not attempt to partition the daytime CO_2 flux further and to quantify the percentage of cattle respiration in the total respiration flux.

Half-hourly net CO_2 exchange rates observed during both day and night did generally not differ between periods with low water table (July) and high water table (August; Table 4). In summary, growing season length, cutting frequency and grazing intensity controlled the interannual CO_2 flux variability more than changes in temperature or water table position did, and for the management effects there is evidence that changes in CO_2 uptake rather than ecosystem respiration were responsible for most of the observed variability.

BGD
9, 9029–9064, 2012

4.4 Impacts of climate and management on CH_4 fluxes

The lack of any noticeable interannual differences in the seasonal courses of temperature and water table height, being the most important control factors for CH_4 emissions in general, means that the observed interannual variability in CH_4 emissions was not caused by climatic variations. Nevertheless does the switch-on-off function of the water table (Christensen et al., 2003; Meijide et al., 2011) have the potential to induce substantial changes in the CH_4 emissions from Skjern Meadows. For example, a permanently high water table during the warm season would probably double the annual CH_4 flux (Herbst et al., 2011a), whereas climatically realistic variations in temperature would have a much smaller effect, according to Fig. 5. The role of wind speed and friction velocity on CH_4 emissions from wetlands is still a matter of debate (Sachs et al., 2008; Schrier-Uijl et al., 2010; Kroon et al., 2010). Despite the well-known fact that CH_4 emissions, particularly from open water surfaces, can be triggered by changes in wind speed, it remains unclear whether this would only affect the timing or also the amount of CH_4 being released into the atmosphere.

The interannual variations in the CH_4 emissions of up to 60 % must therefore be attributed to effects of site management rather than meteorological conditions. Several studies have shown that ruminating animals can contribute considerably to atmospheric CH_4 fluxes at the ecosystem scale (Dengel et al., 2011; Detto et al., 2011; Herbst et al., 2011a; Baldocchi et al., 2012). The size of this contribution depends not only on the stocking density but also on the selection of the animals. For example, dairy cows emit more CH_4 than bulls and heifers, which again emit more than sheep. An exact quantification of CH_4 coming from rumination is difficult on the basis of eddy covariance data alone because the movements of the animals and the source area are not necessarily random and independent (Baldocchi et al., 2012), but a rough estimation for Skjern Meadows came to the conclusion that the CH_4 emitted through rumination amounted to approximately 11 % of the total annual flux in 2009 (Herbst et al., 2011a). Taking the extended grazing periods and enhanced stocking densities in the following years into account, most of the observed increase in the annual CH_4 emissions could be explained by rumination. The CH_4 emissions from the cattle alone would not have been enough, though, to equalize the CO_2 fixation in terms of the total GHG effect. This observation agrees with Soussana et al. (2007) who found that, regarding the annual GHG budget, the CH_4 emission from cattle was never enough to counterbalance the net CO_2 uptake of grasslands on mineral soils where no additional CH_4 emissions from the soil occur.

However, it cannot be excluded that the presence of the cattle had indirect effects on the CH_4 fluxes, too. For example, it seems possible that trampling may have compacted the soil and reduced its aeration, and it became clearly visible during the investigation period that an increase in the cover of soft rush communities, as documented for the first four years following the rewetting (see Sect. 2.1), is still ongoing and accelerated when the area is grazed instead of cut (see Fig. 6). Soft rush, which is avoided by the cattle, contributes to plant mediated transport of CH_4 from the soil into the atmosphere because it has aerenchymous tissues through which CH_4 can bypass the aerated top soil without being oxidized. Chamber measurements of CH_4 emissions from a soft rush covered, but unsaturated meadow in the same region (Schäfer et al., 2012) revealed emission rates of up to $3.3\,mg\,m^{-2}\,h^{-1}$ (or $0.8\,g\,m^{-2}\,d^{-1}$), amounting to almost half of the maximum eddy fluxes from wet soils observed in this study (Fig. 5). The emission rates depended linearly on the dry weight of *Juncus effusus* in the respective plots (Petersen et al., 2012).

The impact of the percentage cover of aerenchymous plants, and of the species composition in general, on the CH_4 budget was also emphasized by Brix et al. (2001) and Levy et al. (2012). However, even without grazing an increase in annual CH_4 emissions over many years is likely in restored wetlands due to slow changes in the vegetation following the rewetting of a formerly drained site and due to a gradual colonisation of the soil by methanogenic microorganisms (Tuittila et al., 2000). In other case studies, Waddington and Day (2007) observed a continuous increase in CH_4 emissions over four years following the restoration of a peatland, and Liikanen et al. (2006) reported

BGD
9, 9029–9064, 2012

an increase from 140 to 400 $g\,m^{-2}$ between the 9th and the 15th year after the construction of a new wetland.

CH_4 fluxes are generally much more difficult to interpret than CO_2 fluxes due to a larger number of control factors and a larger spatial variability (Baldocchi et al., 2012) and also due to much fewer sites with long-term data. Future work at managed wetland sites like the one presented in this study should aim at a reliable footprint modelling and the characterisation of the spatial variability in surface properties. Ideally, this should include mobile methane sources like ruminating animals. Such studies would further have to be complemented by an annual monitoring of the plant species composition.

4.5 General relevance of the results

In an overview of peat soils in Scandinavia and their management, Maljanen et al. (2010) pointed out that, amongst the Northern European countries, Denmark has the highest fraction (ca. 90 %) of original wetlands being drained and converted into agricultural land, but also the highest percentage of those drained areas (>10 %) being rewetted again in recent years. Also in other countries of the temperate and boreal regions, there is an increasing trend towards restoration of formerly drained wetlands (Höper et al., 2008). Skjern Meadows is Denmark's largest restoration area and considered as a role model for Northern Europe. Thus it can be considered as a suitable site for a case study assessing the GHG budget of a wetland following restoration, although the development of a specific site will depend on the start conditions (Höper et al., 2008). In general, the results of this study are in line with earlier assessments of restored wetlands that concluded that such sites can become CO_2 sinks after some time, depending on the development of the vegetation, and that they always are CH_4 sources (Drösler et al., 2008). However, this study demonstrated also that an area can switch from sink to source and back from year to year, mostly depending on site management and extreme weather events, but less depending on gradual climatic variations. The results indicate that more long-term studies of full GHG budgets from wetlands

BGD
9, 9029–9064, 2012

are needed because the interannual variability in radiative forcing of a specific land use type can be particularly large where more than one GHG is affected and where different GHGs are controlled by different environmental factors. Whilst the response of GHG fluxes to climatic factors is increasingly well understood, many global models and prediction schemes fail to account for site management factors.

Some of the first flux tower sites that were equipped with sensors accounting for the full GHG balance (e.g. Rinne et al., 2007; Sachs et al., 2008; Kroon et al., 2010) have meanwhile been in operation for several years, and this will soon make a more comprehensive assessment of interannual variations in the total GHG flux from wetland sites possible. At present, comparisons between such sites are still hampered by the fact that there is as yet no standardised CH_4 flux measurement and gap filling protocol in place, as it is the case for CO_2 fluxes as a result of the "Euroflux" and "Fluxnet" activities (Aubinet et al., 2000; Papale et al., 2006). Recent instrument intercomparisons are encouraging (Detto et al., 2011) though, and both a standardisation of eddy flux measurements and a complementary application of chamber measurements at flux tower sites could help improving both the process understanding and the transferability of results to other climate and vegetation types where CO_2 and CH_4 contribute to the atmospheric GHG budget.

5 Conclusions

This case study has demonstrated that the annual CO_2 fluxes above a restored wetland were mainly controlled by site management and growing season length and that the annual CH_4 emissions depended strongly on grazing effects. The interannual variability was higher for the CO_2 than for the CH_4 fluxes, but the site remained always a net CO_2 and a net carbon sink. However, adding the CH_4 emissions as CO_2 equivalents to the balance made the sign of the total GHG budget switch between years. Somewhat unexpectedly, the site turned into a significant GHG source in the coolest and driest year. It was shown that site management played a large role in this observation, and with an increasing amount of former wetlands being restored there is a need to improve the prediction of the radiative forcing of wetland ecosystems by incorporating such anthropogenic effects. The eddy covariance technique has proven to be a suitable tool to monitor full GHG budgets from a wide range of wetland sites, and more such flux studies will be needed to understand the role of these ecosystems and their management in the atmospheric greenhouse gas budget.

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BGD
9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

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BGD
9, 9029–9064, 2012
Greenhouse gas exchange of a restored wetland
M. Herbst et al.

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BGD

9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

Table 1. Meteorological conditions during the 3 full years of investigation.

Time	Mean temperature [°C]			Total precipitation [mm]		
	2009	2010	2011	2009	2010	2011
Jan–Mar	2.0	−1.1	1.3	157	99	142
Apr–Jun	11.4	9.7	11.8	101	104	135
Jul–Sep	15.6	15.3	15.1	236	279	338
Oct–Dec	5.2	2.3	6.9	305	229	225
Total	8.6	6.6	8.8	799	711	840

BGD
9, 9029–9064, 2012

Table 2. Annual atmospheric greenhouse gas budget for the 3 full years of measurements. Upward fluxes are defined as positive.

Year	CO_2 flux $G\,m^{-2}$	$g\,C\,m^{-2}$	CH_4 flux $g\,m^{-2}$	$g\,C\,m^{-2}$	$g\,CO_2\,eq\,m^{-2}$	GHG budget $g\,CO_2\,eq\,m^{-2}$	Carbon budget $g\,C\,m^{-2}$
2009	−983 ± 147	−268 ± 40	11 ± 2	9 ± 1	284 ± 43	−700 ± 190	−259 ± 41
2010	−195 ± 29	−53 ± 8	14 ± 2	11 ± 2	359 ± 54	163 ± 83	−42 ± 10
2011	−387 ± 58	−105 ± 16	17 ± 3	13 ± 2	432 ± 65	45 ± 123	−92 ± 18

BGD
9, 9029–9064, 2012

Table 3. Results from the curve fitting for the CH_4 flux versus soil temperature plots shown in Fig. 7, using Eq. (1).

	High water table	Low water table
2009		(D 119–D 227)
A [$\mathrm{g\,m^{-2}\,s^{-1}}$] ± SE	0.0086 ± 0.0013	0.0034 ± 0.0031
B [$^{\circ}\mathrm{C}^{-1}$] ± SE	0.151 ± 0.011	0.129 ± 0.056
2010		(D 127–D 227)
A [$\mathrm{g\,m^{-2}\,s^{-1}}$] ± SE	0.0129 ± 0.0014	0.0109 ± 0.0060
B [$^{\circ}\mathrm{C}^{-1}$] ± SE	0.128 ± 0.008	0.104 ± 0.034
2011		(D 124–D 222)
A [$\mathrm{g\,m^{-2}\,s^{-1}}$] ± SE	0.0124 ± 0.0017	(0 ± 0)
B [$^{\circ}\mathrm{C}^{-1}$] ± SE	0.162 ± 0.010	(0.576 ± 0.097)

BGD
9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

Table 4. Results from the curve fitting of night-time net CO_2 fluxes against soil temperature and daytime net CO_2 fluxes against incoming solar radiation. R_{10} refers to Eq. (2) and a, b and c refer to Eq. (3).

	Jul (low water table)	Aug (high water table)
2009	Cut	Recovered
R_{10} [μmol m^{-2} s^{-1}] ± SE	2.77 ± 0.08	2.87 ± 0.07
a [μmol J^{-1}] ± SE	−0.040 ± 0.005	−0.083 ± 0.008
b [μmol m^{-2} s^{-1}] ± SE	−22.83 ± 2.19	−26.59 ± 1.83
c [μmol m^{-2} s^{-1}] ± SE	4.86 ± 0.25	4.01 ± 0.25
2010	Grazed	Grazed
R_{10} [μmol m^{-2} s^{-1}] ± SE	2.87 ± 0.09	2.32 ± 0.02
a [μmol J^{-1}] ± SE	−0.094 ± 0.009	−0.072 ± 0.010
b [μmol m^{-2} s^{-1}] ± SE	−18.80 ± 0.72	−17.27 ± 1.35
c [μmol m^{-2} s^{-1}] ± SE	6.62 ± 0.21	3.88 ± 0.26
2011	Undisturbed	Cut
R_{10} [μmol m^{-2} s^{-1}] ± SE	2.52 ± 0.08	2.10 ± 0.15
a [μmol J^{-1}] ± SE	−0.095 ± 0.006	−0.034 ± 0.005
b [μmol m^{-2} s^{-1}] ± SE	−23.52 ± 0.77	−19.95 ± 3.44
c [μmol m^{-2} s^{-1}] ± SE	5.16 ± 0.21	3.48 ± 0.25

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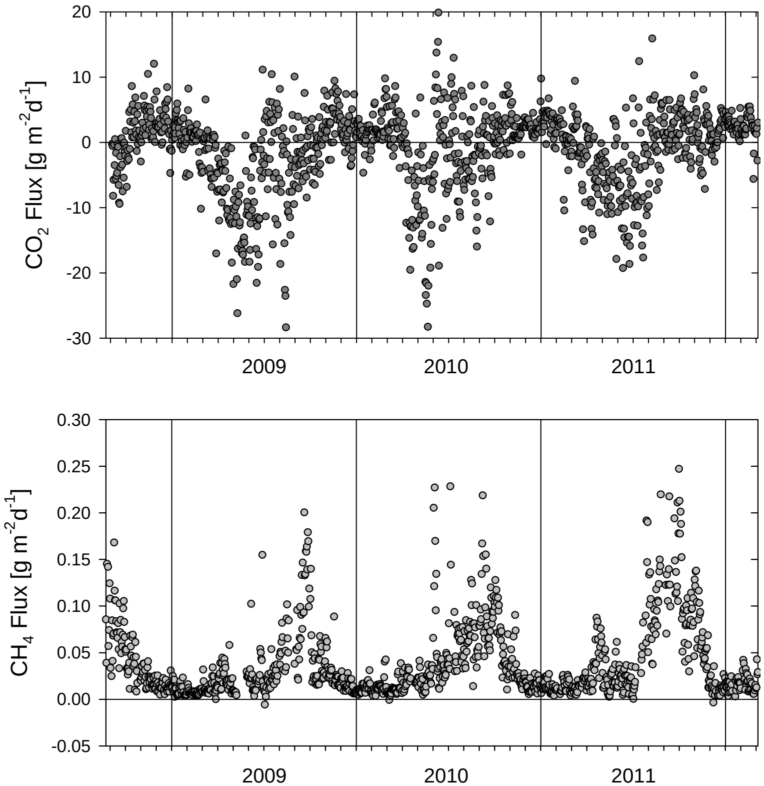

Fig. 1. Daily atmospheric fluxes of CO_2 (upper panel) and CH_4 (lower panel) over the first 3.5 yr of measurements at Skjern Meadows. Upward fluxes are defined as positive.

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Greenhouse gas exchange of a restored wetland

M. Herbst et al.

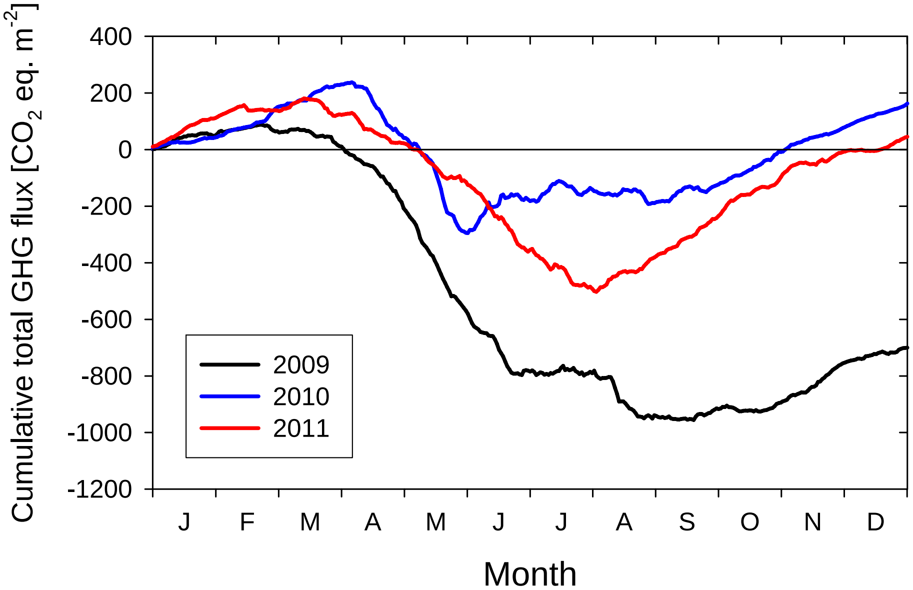

Fig. 2. The total cumulative greenhouse gas fluxes for the three full calendar years of measurements, expressed as CO_2 equivalents on the basis that 1 g CH_4 is equivalent to 25 g CO_2. Upward fluxes are defined as positive.

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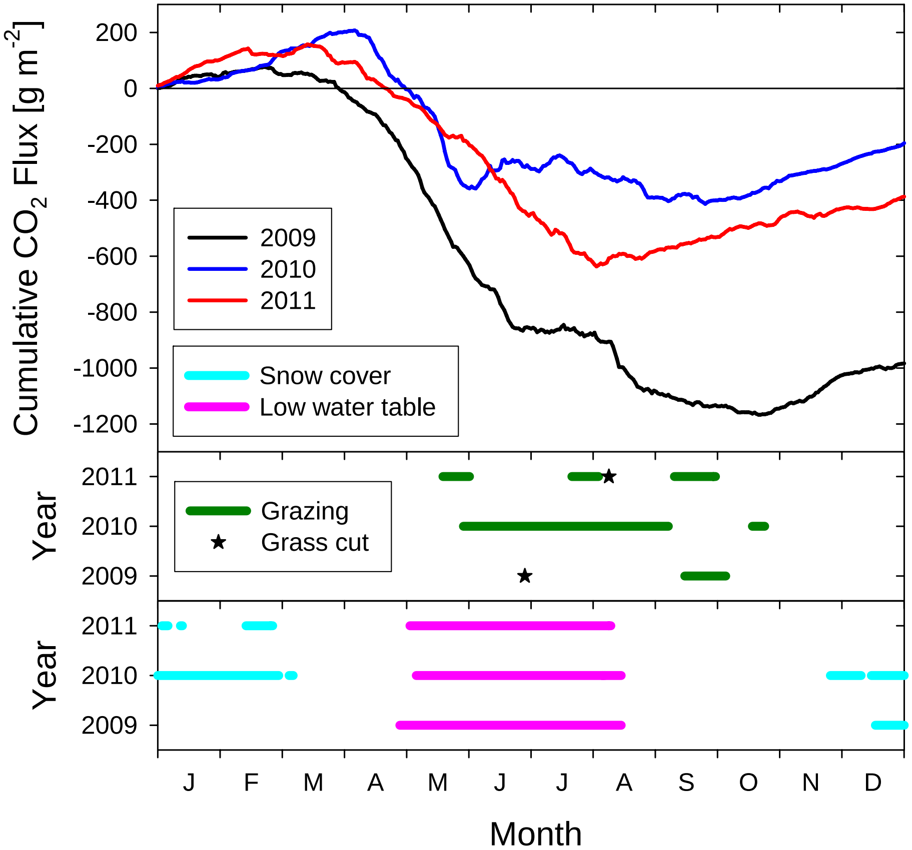

Fig. 3. Cumulative CO_2 fluxes (upper panel) shown together with management activities (middle panel), snow cover duration and water table height (lower panel). See text for the definition of "low" and "high" water table.

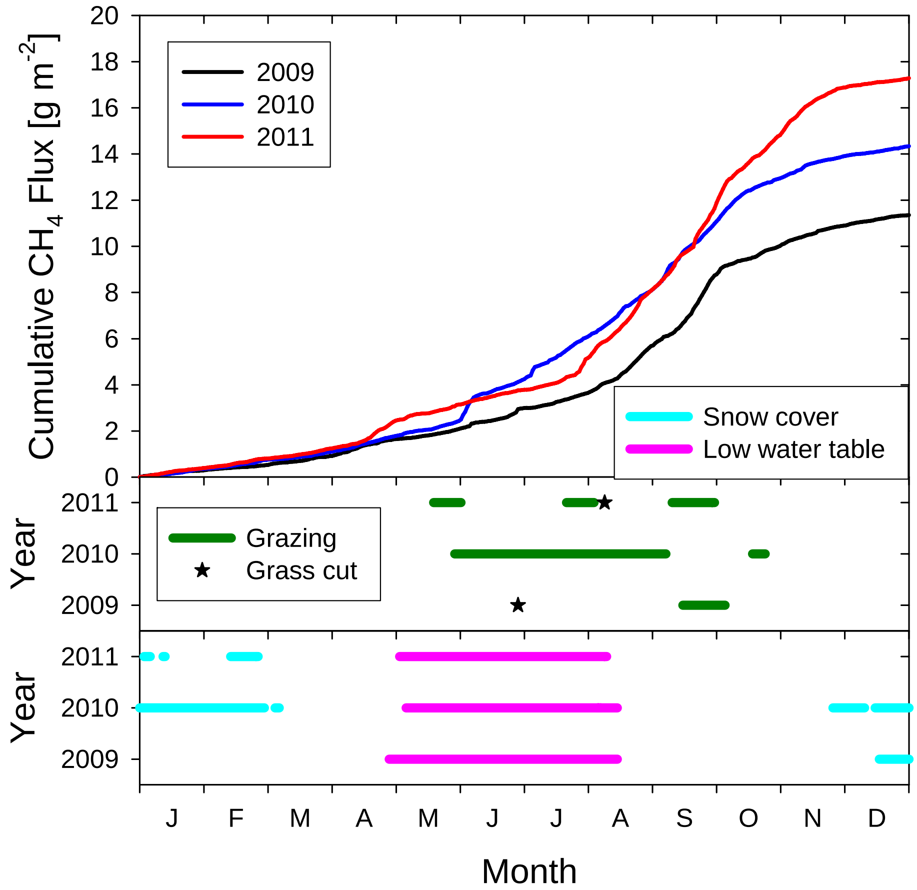

Fig. 4. Same as Fig. 3 but for CH_4 instead of CO_2.

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Greenhouse gas exchange of a restored wetland

M. Herbst et al.

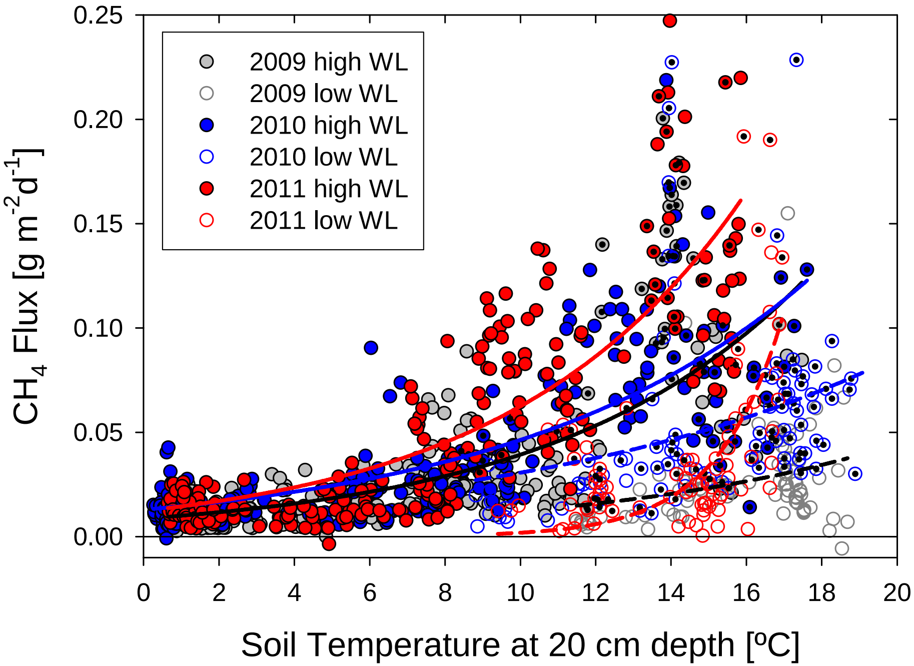

Fig. 5. Daily CH_4 fluxes observed over three years in relation to soil temperature at 20 cm depth. Different symbols indicate periods with low and high water table and periods with (black dots) and without grazing, respectively.

BGD
9, 9029–9064, 2012

Greenhouse gas exchange of a restored wetland

M. Herbst et al.

Title Page
Abstract | Introduction
Conclusions | References
Tables | Figures
◂◂ | ▸▸
◂ | ▸
Back | Close
Full Screen / Esc
Printer-friendly Version
Interactive Discussion

Fig. 6. Grazing cattle on Skjern Meadows promote the spreading of soft rush plants which enable methane to bypass the aerated top soil through their aerenchyma. The upper photos show the view from the instrument mast towards the meadows in late summer, first in an undisturbed status (2009) and then after a long grazing period (2010). The lower photo illustrates that the soft rush plants are spared by the cattle.