Methane and nitrous oxide exchange over a managed hay meadow

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Abstract

The methane (CH_4) and nitrous oxide (N_2O) exchange of a temperate mountain grassland near Neustift, Austria, was measured during 2010 – 2012 over a time period of 22 months using the eddy covariance method. Exchange rates of both compounds at the site were low, with 97 % of all half-hourly CH_4 and N_2O fluxes ranging between ±200 and ±50 ng m^{-2} s^{-1}, respectively. The meadow acted as a sink for both compounds during certain time periods, but was a clear source of CH_4 and N_2O on an annual time scale. Therefore, both gases contributed to an increase of the global warming potential (GWP), effectively reducing the sink strength in terms of CO_2-equivalents of the investigated grassland site. In 2011, our best guess estimate showed a net GHG sink of -32 g CO_2-equ. m^{-2} yr^{-1} for the meadow, whereby 55 % of the CO_2 sink strength of -71 g CO_2 m^{-2} yr^{-1} was offset by CH_4 / N_2O emissions of 7 / 32 g CO_2-equ. m^{-2} yr^{-1}. When all data were pooled, the ancillary parameters explained 27 / 42 % of observed CH_4 / N_2O flux variability, and up to 62 / 76 % on shorter time scales in-between management dates. In case of N_2O fluxes, we found highest emissions at intermediate soil water contents and at soil temperatures close to zero or above 14 °C.

In comparison to CO_2, H_2O and energy fluxes, the interpretation of CH_4 and N_2O exchange was challenging due to footprint heterogeneity regarding their sources and sinks, uncertainties regarding post-processing and quality control. Our results emphasize that CH_4 and N_2O fluxes

over supposedly well-aerated and moderately fertilized soils cannot be neglected when evaluating the GHG impact of temperate managed grasslands.

1 Introduction

Methane (CH_4) and nitrous oxide (N_2O) are the most important anthropogenic greenhouse gases (GHG) after carbon dioxide (CO_2). Due to their long atmospheric lifetimes of approx. 9 and 131 years (Prather et al., 2012), respectively, both compounds are well-mixed in the atmosphere and can influence atmospheric chemistry directly and indirectly. The emission or deposition strength of terrestrial ecosystems is possibly influenced by climate change, which may trigger important feedbacks to the global climate system (Xu-Ri et al., 2012).

Methane has a major influence on climate and chemistry of the atmosphere (Crutzen and Lelieveld, 2001; Khalil et al., 2007). CH_4 can react with hydroxyl radicals, resulting in a reduction of the oxidizing capacity of the atmosphere and the production of ozone (O_3) in the troposphere. Methane can influence the lifetime or production of other atmospheric constituents like stratospheric water vapor and CO_2 (Boucher et al., 2009; Collins et al., 2010; Shindell et al., 2009). Its global warming potential over a 100 year lifespan (GWP) and on a per molecule basis is 25 times that of CO_2 (Forster et al., 2007) or higher when the production of CO_2 from CH_4 oxidation is taken into account (Boucher et al., 2009).

The main portion of global CH_4 originates from single-celled archaea (methanogens) found in anaerobic microsites in the soil, in water-saturated zones rich in carbon and in the digestive systems of ruminants (Baldocchi et al., 2012; Whalen, 2005). CH_4 is also emitted from organic waste deposits, e.g. manure, and from thermogenic and pyrogenic sources (Kirschke et al., 2013). Highest emissions were previously reported from regions with intensive agriculture and animal husbandry (Schulze et al., 2009). Atmospheric CH_4 increased significantly since the industrial revolution until the end of the 1990s, remained constant for nearly a decade and again began to increase after 2007 (Bousquet et al., 2011; Dlugokencky et al., 2009; Nisbet et al., 2014).

The main sink of methane is through its reaction with the hydroxyl radical OH in the troposphere (Ehhalt and Heidt, 1973). Other, minor sinks are methanotrophic bacteria in aerated soils and reactions with atmospheric constituents in the stratosphere and the marine boundary layer (Allan et al., 2007; Cicerone and Oremland, 1988). Previous studies reported reduced CH_4 deposition in a forest and in a temperate grassland due to elevated CO_2 (Dubbs and Whalen, 2010; Ineson et al., 1998; Phillips et al., 2001) and increased CH_4 uptake due to warming in a temperate forest and several subarctic ecosystems (Peterjohn et al., 1994; Sjogersten and Wookey, 2002).

Nitrous oxide can deplete O_3 in the upper and increase O_3 in the lower regions of the stratosphere (Revell et al., 2012). It can therefore influence tropospheric chemistry by increasing the stratosphere-troposphere exchange of O_3 and odd nitrogen species, and by increasing OH formation (Prather and Hsu, 2010). Similar to CH_4, N_2O has a high warming potential, 298 times that of CO_2 over a 100 year lifespan (Forster et al., 2007). The dominant source of N_2O is microbial production through nitrification and denitrification processes in soils, which is fueled by accelerated use of nitrogen fertilizers in agriculture (Davidson, 2009; Fowler et al., 2009). As a consequence of fertilization agricultural soils are unlikely to act as a sink for N_2O (Syakila and Kroeze, 2011).

The production of N_2O by bacteria in soils is controlled by a number of factors, for example soil water content, temperature and labile carbon availability (Barnard et al., 2005; Holtan-Hartwig et al., 2002; Xu-Ri and Prentice, 2008). Food production was described as the largest single source of N_2O (Syakila and Kroeze, 2011), while photolysis and oxidation reactions in the stratosphere are the main processes involved in N_2O depletion (Prather et al., 2012).

Denitrification is an anaerobic process (Zumft and Kroneck, 2007) that is likely exclusively responsible for N_2O uptake in the soil (Vieten et al., 2008). On a global scale, the uptake of N_2O by soils may be limited (Chapuis-Lardy et al., 2007). Schlesinger (2013) estimated that the global N_2O sink in soils is not more than 2 % of current estimated sources in the atmosphere. Deposition fluxes to the soil were reported before, e.g. for grasslands, forests, low-nitrogen soils, wetlands and peatlands (Dijkstra et al., 2013; Flechard et al., 2005; Goldberg and Gebauer, 2009a, 2009b; Schlesinger, 2013; Syakila et al., 2010; Wu et al., 2013).

Over managed grasslands, CH_4 and N_2O fluxes are characterized by high spatial and temporal variability (Baldocchi et al., 2012; Imer et al., 2013), with emissions of both compounds greatly influenced by land use, management events and animal husbandry. As a consequence, long-term year-round GHG measurements are indispensable when it comes to assessing the effectiveness and feasibility of GHG mitigation strategies.

In this work we present long-term eddy covariance CH_4 and N_2O fluxes above a temperate mountain grassland near Neustift, Austria. To this end we investigated 22 months of diurnal, seasonal and interannual exchange rates of both compounds at ecosystem scale and in relation to biotic and abiotic drivers under *in situ* conditions.

The objective of this study is to (1) quantify eddy covariance CH_4 and N_2O fluxes, (2) couple exchange patterns to independent driving variables, (3) determine the annual total GHG balance and (4) compare our findings to previous results from chamber and eddy covariance measurements at ecosystem scale and from laboratory measurements. In line with these objectives and based on earlier studies we hypothesized for both compounds that (1) the investigated grassland, due to generally well-aerated soils and modest fertilizer input, is characterized by low fluxes and (2) exchange patterns are predominantly driven by soil parameters. In addition we assumed that (3) despite their low fluxes, CH_4 and N_2O exchange significantly contribute to the GHG balance of the meadow.

The study site Neustift, a managed temperate mountain grassland in Austria that is cut three times per year for hay production, was selected because it has been the focus of numerous studies over the last ten years and is therefore well described in terms of management effects, net ecosystem CO_2, H_2O, energy (Brilli et al., 2011; Hammerle et al., 2008; Wohlfahrt et al., 2008b) and VOC exchange (Bamberger et al., 2010, 2011; Brilli et al., 2012; Hörtnagl et al., 2011, 2014; Müller et al., 2010; Ruuskanen et al., 2011).

2 Methods

2.1 Site description

The study site is an intensively managed meadow in the middle of the flat valley bottom of the Stubai Valley in the Austrian Alps, in proximity of the village of Neustift (47°70’ N, 11°19’ E) at an elevation of 970 m a.s.l. The climate is humid continental with alpine influences, with an average annual temperature of 6.5 °C, the average annual precipitation amounts to 852 mm. The fetch is homogeneous up to 300m to the north-northeast (the dominant daytime wind direction) and 900m to the south-south-west (nighttime) of the instrument tower, parallel to the Valley’s orientation. Typically, higher wind speeds and unstable conditions result in a smaller footprint during daytime than during nighttime, where the footprint of the site is larger due to the stable stratification of the atmosphere (Bamberger et al., 2010). The vegetation of the meadow is dominated by a few graminoids (*Dactylis glomerata*, *Festuca pratensis*, *Phleum pratensis*, *Trisetum flavescens*) and forbs (*Ranunculus* *acris*, *Taraxacum officinale*, *Trifolium repens*, *Trifolium pratense*, *Carum carvi*), while the slopes of the surrounding mountains are covered mainly by coniferous forest. The soil was classified as a Fluvisol (FAO classification) and is approximately 1 m deep, with a thin organic layer (0.001 m), followed by an A horizon that extends down to 0.02 m and a B horizon, best described as a sandy loam. The organic volume fraction of the A horizon is approximately 14 %.

Measurements of CH_4 and N_2O for this work were conducted from 13 April 2010 – 29 February 2012 (684 days). In each year, the meadow was cut three times, with the 1st cut on 5 / 6 June in 2010 / 2011, respectively, the 2nd cut on 31 July / 1 August and the 3rd cut on 20 / 26 September. In addition, the meadow was fertilized by manure spreading between 18 – 22 October in 2010 and on 18 and 19 October in 2011. The meadow was snow-covered from 1 January – 28 February 2010, from 26 November 2010 – 10 March 2011 and from 7 December 2011 – 24 March 2012, resulting in a total of 246 snow days for this analysis. During the measurement campaign, no cows were present on the meadow.

2.2 Eddy covariance measurements

The net ecosystem exchange for CH_4 and N_2O was calculated by combining the 20 Hz three-dimensional wind speeds quantified by a sonic anemometer (R3IA, Gill Instruments, Lymington, UK) at a height of 2.5 m above ground with the simultaneously detected volume mixing ratios (VMRs) of CH_4 and N_2O, which were both measured by a commercially available continuous-wave quantum cascade laser (QCL; CWQC-TILDAS-76-D, Aerodyne, USA). Fluxes were then calculated using the virtual disjunct eddy covariance method (vDEC) method proposed by Karl et al. (2002), which is based on the eddy covariance (EC) method (Baldocchi et al., 1988; McMillen, 1988). The intake tube for the QCL was mounted at 0.2 m below the sonic anemometer and displaced laterally perpendicular to the predominating wind direction in order to minimize flux loss due to vertical and longitudinal sensor separation (Massman, 2000). Sample air was drawn from the inlet through a filter (1-2 µm, PTFE) and heated (35 °C) PFA Teflon tubing (1/4'' inner diameter) of 12 m length to the QCL at a flow rate of around 8 SLPM (standard liter per minute; air volume normalized to standard temperature and pressure conditions: 273K, 1013 hPa). Sonic anemometer data were stored to the hard drive of a personal computer (PC) using the *Eddymeas* software (O. Kolle, Max

Planck Institute for Biogeochemistry, Jena, Germany). More details regarding the CO_2, H_2O and energy flux measurements are given in Wohlfahrt et al. (2008) and Hammerle et al. (2008).

2.3 QCL setup

Ambient air was analyzed for CH_4, N_2O and H_2O at time resolutions of 10 Hz (13 March – 16 August 2010), 5 Hz (16 – 24 August 2010) and 2 Hz (26 August 2010 – 29 February 2012). The QCL and associated hardware (vacuum pump and thermo cube) were housed in a climate-controlled instrument hut next to the field site. During the last five minutes of every half-hour, CH_4- and N_2O-free air and air with known, close-to-ambient, VMRs were switched into the sampling line to determine zero and span of the QCL, respectively. The QCL was operated at a pressure of 4 kPa using a built-in pressure controller and temperature of the optical bench and housing controlled to 35°C. The importance of a temperature controlled environment was previously pointed out by Kroon et al. (2007). Fitting of absorption spectra, storing of calculated VMRs, switching of zero/calibration valves, control of pressure lock and other system controls were realized by the TDLWintel software (Aerodyne, USA) run on a PC synchronized with the main PC collecting anemometer data using the NTP software (Meinberg, Germany).

2.4 Despiking

Similar to observations by Baldocchi et al. (2012) for methane, we experienced elevated VMRs of both compounds, but especially CH_4, at night. We attributed these increased VMRs to atmospheric phenomena in the calm and stable nocturnal boundary layer rather than to elevated biogenic emissions. Therefore, VMRs of both compounds were subjected to a rigorous outlier removal routine before entering flux calculations (Fig. 1a). The despiking method in this study is based on a median filter that runs through each half-hourly VMR time series data point by data point. In comparison to the arithmetic mean, the median value of a time series is relatively insensitive to outlier values. For each 30 min period, (1) a smoothed time series of the original VMR time series was created. This was done by replacing each original data point with the median value of a moving time window of ± 500 values around the respective VMR value. In order to enable the calculation of median values also for data points at the start and end of the measured time series, the first and last 500 values were copied and repeated at the start and end of the smoothed time series, respectively. (2) Each data point in the smoothed time series was then subtracted from the respective measured data point, generating a time series of differences between the two data matrices. (3) When the difference exceeded the empirically determined outlier threshold of 100 ppb, the data point in the measured time series was marked as an outlier. This outlier threshold was tailored to the CH_4 variability, but worked also well for removing extreme values in the N_2O time series. (4) The arithmetic mean without these outliers was then calculated and used to (5) replace outliers in the respective half-hourly time series. As turbulent fluctuations for final flux computations are calculated using block averaging, the contribution of these substituted data points to resulting half-hourly fluxes is minor. To better account for natural variability in the time series, three different runs with varying window sizes (± 500, 250, 150 values) and outlier thresholds (100, 80, 60 ppb) were performed for each 30 min period.

During daytime / nighttime, at least one outlier was removed in 30 % / 66 % of half-hourly CH_4, but only in 1 % / 1 % of all recorded N_2O VMRs.

2.5 Flux calculations

Half-hourly fluxes of CH_4 (F_{CH4}) and N_2O (F_{N2O}) were then calculated using the virtual disjunct eddy covariance (vDEC) method (Karl et al., 2002) as the covariance between turbulent fluctuations of the vertical wind speed and the VMRs derived from Reynolds averaging of 30 min blocks of data. The time lag between the high-resolution wind data and the disjunct QCL time series was removed using a homemade program, resulting in a subsample of the wind data corresponding to the sampling rate of the QCL. In the same step, CH_4 and N_2O fluxes were corrected for the effect of air density fluctuations and laser band-broadening following *Neftel et al.* (2010), using the QCL H_2O VMR. It was shown previously that flux estimates using the vDEC method are characterized by a larger random uncertainty compared to the true EC, but are unbiased (Hörtnagl et al., 2010). The tubing induced delay time between the wind and the QCL concentration time series was determined in a procedure comprising multiple steps. First, the correlation coefficient between the H_2O time series measured concurrently by the QCL and a closed-path infrared gas analyzer (Li-7000, LiCor, USA), the data of which were acquired together with the sonic anemometer wind data, was optimized to remove potential time differences between the two PCs caused by deviating internal clocks, effectively adjusting the starting points of the two time series. Due to generally low values of F_{CH4} and F_{N2O} at our study site, the determination of lag times between the CH_4 / N_2O time series and the wind data was difficult, but worked well between the QCL H_2O signal and the wind data. Therefore, secondly, the time delay between the wind components and the QCL H_2O was determined by identifying the maximum/minimum of the cross-correlation function in a time window of +/- 7 s. The frequency distribution of this search revealed a peak around 2 s. Thirdly, a second time window of +/- 2 s (daytime) and +/- 5 s (nighttime) was then applied around this peak and used for the final lag search between the CH_4 / N_2O signal and the vertical wind velocity.

Final fluxes were then calculated using the post-processing software *EdiRe* (University of Edinburgh). Frequency response corrections were applied to raw fluxes of both compounds, accounting for high-pass (block averaging, finite impulse response filter) and low-pass (lateral sensor separation, dynamic frequency response, scalar and vector path averaging, frequency response mismatch and the attenuation of concentration fluctuations down the sampling tube) filtering according to *Massman* (2000), using a site-specific cospectral reference model (Wohlfahrt et al., 2005a). The importance of correcting CH_4 and N_2O fluxes for high frequency losses was shown previously (Kroon et al., 2010c). The high pass, non-recursive, finite impulse response (FIR) filter was applied digitally to account for an overestimation of the flux contributions of low-frequency eddies. Best results were achieved by applying the FIR filter using a Hamming window, whereby time constants of respectively 50 and 100s for CH_4 and N_2O sufficiently filtered out unwanted flux contributions at frequencies < 0.05 Hz (Fig. 1b). Missing low-frequencies were then back-corrected based on the site-specific reference model co-spectrum (Wohlfahrt et al., 2005b). Exchange rates of CH_4 and N_2O calculated with these settings represent our final best guess fluxes that were used for all analyses in this manuscript.

Two days in April 2011 are used to exemplify the effect of different FIR filters, applied to the CH_4 and N_2O time series, on the resulting flux estimates (Fig. 1b). The largest difference between unfiltered and filtered data as well as between the different filter time constants was found during nighttime. In contrast, during turbulent conditions e.g. around noon, fluxes calculated with different time constants exhibited exchange patterns of comparable magnitude

(Fig. 1b, left panels). FIR filtering had a larger effect on CH_4 than on N_2O fluxes. As an example, over the course of one day unfiltered CH_4 exchange rates fluctuated between -217 and 780 ng m^{-2} s^{-1} (average: 4 ±260 ng m^{-2} s^{-1}), while best guess fluxes ranged between -96 and 87 ng m^{-2} s^{-1} after FIR filtering (-7 ±51). Similarly, unfiltered N_2O fluxes were between -38 – 146 ng m^{-2} s^{-1} (11 ±46), with best guess fluxes of -33 – 18 ng m^{-2} s^{-1} (-5 ±15). Cospectral analyses revealed that lower frequencies of the CH_4 and N_2O fluxes were overrepresented compared to the sensible heat flux (Fig. 1b, right panels).

In total, 28891 raw flux values were calculated for CH_4 / N_2O, which corresponds to a data coverage of 88 % over the whole measurement period between 13 March 2010 and 29 February 2012. Flux results of each FIR run required separate quality control. When applying a FIR filter with a time constant of 50 s / 100 s / 150 s to the data, 57 / 55 / 55 % of all raw CH_4 fluxes and 66 / 64 / 63 % of all raw N_2O fluxes passed all quality tests, respectively. However, only 28 % and 39 % of all raw CH_4 and N_2O fluxes, respectively, passed all tests when no FIR filter was used in the flux calculations. Only data that passed all quality tests in a respective scenario were used in the present study. All fluxes in this manuscript are expressed as molecular mass per unit time and ground surface area.

In order to calculate the annual balance of CH_4 and N_2O in 2011, the respective quality-controlled half-hourly flux dataset was gap-filled. Gaps less than or equal to two hours were filled by linear interpolation. For the filling of larger gaps a lookup table was generated, using flux data in a time window of 14 days around the missing flux value and T_{soil} bin widths of 1°C. If no lookup table could be generated, e.g. no flux data were available within the time window, the mean diurnal variation (±14 days) was used to fill the gap. For the calculation of the annual GWP of the meadow in Neustift, CH_4 and N_2O fluxes were converted to CO_2-equivalents using the respective compound warming potential as given by Forster et al. (2007).

Instrumentation, data treatment and quality control of CO_2, sensible and latent heat fluxes have been described at length by Wohlfahrt et al. (2008) and Hammerle et al. (2008).

2.6 Quality control

Half-hourly methane and nitrous oxide fluxes were excluded from the analysis if (i) the deviation of the integral similarity characteristics was larger than 60 % (Foken and Wichura, 1996), (ii) the maximum of the footprint function (Hsieh et al., 2000) was outside the boundaries of the meadow, (iii) fluxes were outside a specific range (F_{CH4}: +/- 800 ng m^{-2} s^{-1}, F_{N2O}: +/-220), (iv) half-hourly VMRs were outside a specific range (CH_4: 1800 – 3500 ppb, N_2O: 280 – 450 ppb), (v) the stationarity test for the respective flux exceeded 60 % (Foken and Wichura, 1996), (vi) the third rotation angle exceeded 10° (McMillen, 1988), (vii) the number of half-hourly VMR values was below 3000 or (viii) more than 20% of data were classified as spikes in any half-hourly period.

2.7 Ancillary data

Major environmental parameters were measured continuously at the field site, including air temperature (T_{air}), soil temperature (T_{soil}) at 0.05 m depth (TCAV thermocouple, Campbell Scientific, Logan, UT, USA), volumetric soil water content (SWC) (ML2x, Delta-T Devices, Cambridge, UK), soil heat flux (SHF) quantified by means of heat flux plates (3 replicates at 0.05 m depth, corrected for the change in heat storage above that depth; HFP01, Hukseflux, Delft, Netherlands), total photosynthetically active radiation (PAR) (BF3H, Delta-T, Cambridge, UK) and precipitation (52202, R. M. Young, Traverse City, MI, USA). All data were collected continuously by a data logger (CR10X, Campbell Scientific, Logan, UT, USA). The green plant area index (GAI) was assessed (i) in a destructive fashion by harvesting the plant matter of square plots (0.09 m^2, 3-5 replicates) and subsequent plant area determination (Li-3100, LiCor, Lincoln, NE, USA) and (ii) from measurements of canopy height which was related to destructively measured GAI (Wohlfahrt et al., 2008b). Continuous time series of the GAI were derived by fitting appropriate empirical functions to measured data separately for each growing phase before and after cutting events. A more detailed list of all auxiliary parameters measured at this site is given by Wohlfahrt et al. (2008b) and Hammerle et al. (2008).

2.8 Statistical Analyses

Statistical analyses were done using *Statistica 9* (StatSoft, Inc.), *SigmaPlot 12.5* (Systat Software, Inc.) and *Excel 2010* (Microsoft, Inc.). The natural logarithm (ln) of the observed daily average CH_4 and N_2O fluxes was calculated and used in the simple (SLR) and multiple linear regression (MLR) analyses as the dependent variable. The *partial correlation* in the MLR analysis gives the correlation between two variables after controlling for the effect of all other variables in the equation. To determine significant differences between daily average group means in a repeated measures analysis of variance (ANOVA) setting, the Unequal N HSD *post hoc* test, a modification of the Tukey's HSD test, was used. For statistical analyses, only days or half-hours where all parameters were available were included. In case of ancillary data, the daily average of the respective parameter was calculated when at least 40 half-hours of data were present for the respective day. In comparison, fewer values were available for CH_4 / N_2O fluxes and VMRs due to the strict quality criteria. For CH_4 / N_2O data, the daily average was regarded as representative for the day when at least 14 half-hours were available after quality control. In total 91 and 95 % of the presented CH_4 and N_2O daily average values, respectively, were calculated from at least 20 half-hourly values. Using daily average values of CH_4 and N_2O fluxes in the statistical analyses as opposed to 30 min flux averages reduces random uncertainty (Kroon et al., 2010a).

3 Results

Daily average values of F_{CH4} / F_{N2O} were calculated for 567 / 574 out of 684 days, respectively (Fig. 2). While fluxes of both compounds fluctuated around zero towards the end of the vegetation period and during snow cover, net emission and deposition on a daily basis occurred for both compounds during certain time periods. Daily net uptake (negative sign) was recorded on 162 / 203 days, whereby time periods characterized by clear deposition were found especially for N_2O, for example some weeks after snowmelt in spring 2011 (Fig. 2). Highest daily average emissions for both compounds were found around the 2nd cutting of the meadow at the end of July 2010 (123.5 / 33.4 ng m^{-2} s^{-1}). CH_4 VMRs were highest during snow cover and lowest during periods of strong growth (Fig. 2). We attribute the sudden drop of N_2O concentration values around the 1st cut in 2010 to a problem with the zero-calibration of the QCL. Over all two years, the median VMR was 2.02 / 0.32 ppm for CH_4 / N_2O, respectively, the median flux amounted to 9.6 / 0.9 ng m^{-2} s^{-1} (Fig. 2).

Daily average PAR was found between approx. 40 µmol m^{-2} s^{-1} in winter and 674 µmol m^{-2} s^{-1} in summer, with a median value of 215 µmol m^{-2} s^{-1}. In 2010, the yearly average T_{air} at the field site of 6.1 °C was colder than the long-term average (2001 – 2007) of 6.7 °C, while 2011 was warmer than average (7.1 °C). During this study, the maximum daily average T_{air} was

22.7 °C in July 2010, the minimum of -17.3 °C was recorded in February 2012 (Fig. 2). T_{soil} was similar in both years, about 8.5 °C on average and values just above 0 °C when snow covered the ground. SWC was highest immediately after snow melt, with a maximum daily average value of 0.44 m^3 m^{-3} at the end of February 2010, and lowest in May 2011 after a period of only little precipitation (0.08 m^3 m^{-3}). In 2011, SWC was generally low (0.25 m^3 m^{-3} averaged over the growing season) and significantly lower ($p < 0.001$) than in 2010 (0.32 m^3 m^{-3}). Over the duration of the flux measurements, precipitation was detected on 262 days, amounting to 525 and 537 mm in 2010 and 2011, respectively, and 46 mm over the first two months in 2012 (Fig. 2). Relative air humidity (RHA) was around 80 % on average over the whole measurement campaign, with minima below 50 % in June 2010 (Fig. 2). In 2010 and 2011, highest VPD values of more than 1 kPa were recorded during the warmer months between the end of May and August. GAI was below 1 m^2 m^{-2} right after snow melt, reached maximum values of up to 8 m^2 m^{-2} right before the 1st cut and was then reduced to below 1.5 m^2 m^{-2} as a consequence of the cutting. GAI maxima before the 2nd and 3rd cut were lower compared to the 1st cut. Towards the end of the year after the 3rd cut, GAI first increased and later decreased due to vegetation regrowth and senescence, respectively (Fig. 2).

The meadow was a source for CO_2 during snow cover and became a net sink for CO_2 some weeks after snowmelt and until the 1st cut (Fig. 3). The cutting event turned the meadow into a CO_2 source for about two weeks before it again became a net sink. This behavior recurred after the 2nd and 3rd cut, however the CO_2 uptake after the last cutting was less pronounced than after the previous cuttings. More information about CO_2 fluxes at the site was given by Wohlfahrt et al. (2008).

Fluxes of both CH_4 and N_2O showed high variability on a half-hourly time scale, especially during the first two months of the measurements and during the night (Fig. 3). However, 97 % of all half-hourly CH_4 and N_2O fluxes during the vegetation period were found between ±200 and ±50 ng m^{-2} s^{-1}, respectively. During snow-free conditions and including only days not influenced by management events, the average CH_4 / N_2O flux was found at 14.0 ±80.7 / 2.6 ±21.6 ng m^{-2} s^{-1}, respectively (Fig. 3). Compared to these undisturbed conditions, average fluxes were higher on days where the meadow was influenced by cutting events (17.5 ±83.7 / 4.8 ±20.7 ng m^{-2} s^{-1}) and lower on days characterized by snow cover (2.1 ±82.8 / 0.9 ±20.7). The day of manure spreading and the two days thereafter were covered by our measurements only in October 2011. On the day of fertilization and two days later, average N_2O fluxes were elevated (3.5 ±17.2 ng m^{-2} s^{-1}) when compared to the rest of the same month (1.8 ±13.6), while CH_4 fluxes remained virtually unaffected (24.7 ±91.0 vs. 27.0 ±88.9). In total, emission fluxes were observed in 56 / 57 % of all recorded CH_4 / N_2O half hour periods (Fig. 3).

Average diurnal cycles of F_{CH4} and F_{N2O} were often characterized by high variability with large fluctuations around zero, but followed a clear diurnal cycle during certain time periods (Fig. 4). Methane fluxes showed weak diurnal cycles after snowmelt and before the 2nd cut in 2011, with peak average uptake rates of -31.0 ±41.4 ng m^{-2} s^{-1} around noon. The uptake of CH_4 before the 1st cut coincided with strong N_2O deposition during daytime, with average peak rates of up to -12.3 ±23.8 ng m^{-2} s^{-1} in the early afternoon. While CH_4 fluxes continued to exhibit a very similar deposition pattern up until the 2nd cut, N_2O fluxes switched in sign and showed a clear diurnal cycle of constant emission during daytime, up to 15.4 ±18.9 ng m^{-2} s^{-1} on average just before noon. The N_2O flux pattern after the 1st and before the 2nd cut was very similar in both years, whereby peak emission rates in 2010 occurred earlier in the day (Fig. 4). In contrast to CH_4 fluxes, which showed no clear diurnal pattern after the 2nd cut in both years, the meadow constantly emitted N_2O during daytime and before the 3rd cut in 2011, on average up to 26.8 ±23.3 ng m^{-2} s^{-1} around noon, while during daytime after the 3rd cut in 2010 N_2O was transported to the meadow, peak deposition amounted to -7.5 ±8.8 ng m^{-2} s^{-1} on average. During snow cover, fluxes of both compounds fluctuated around zero (Fig. 4).

When all data were pooled, a MLR analysis explained 27 / 42 % of the variability in daily average $\ln(F_{CH4})$ / $\ln(F_{N2O})$ during snow-free conditions (Table 1). Over all years, the partial correlation (PC) of the net ecosystem exchange of CO_2 (NEE) and T_{air} with $\ln(F_{CH4})$ was high and positive in sign, while SHF was negatively correlated with $\ln(CH_4)$; all three PCs were highly significant ($p < 0.001$). During shorter time periods in-between, before and after cutting events in single years the chosen set of parameters explained between 23 and 62 % of the observed flux variability, with r^2 being highly significant only once, namely in a period of high CH_4 uptake before the 1st cut 2011, with NEE and H as the dominant regressors (Table 1). Explaining the $\ln(F_{CH4})$ variance during the same time periods but using data of both years worked best during the vegetation period until the 2nd cut, and again after the 3rd cut until snow cover, explaining up to 40 % of observed ln transformed CH_4 fluxes. The PC of SHF and NEE were significant during the early vegetation period and towards the end of the year, respectively. LE was a significant regressor towards the end of the vegetation period and during snow cover (Table 1). We expanded on these findings by performing a forward step-wise MLR analysis using the same data, effectively reducing the number of variables in the regression equation but yielding similar results. In this analysis NEE, SHF, T_{air} and VPD were identified as the most significant regressors (all $p < 0.05$), explaining 25 % of the observed $\ln(F_{CH4})$ variability over all years excluding snow periods (data not shown). The SLR analysis found highly significant positive correlations for NEE and RHA, and highly significant negative correlations for LE, H, and PAR (Table 1).

Generally, the MLR analysis resulted in r^2 being considerably higher for $\ln(F_{N2O})$ than for $\ln(F_{CH4})$ (Table 1). The partial correlations were highly significant for multiple regressors. A positive PC was found for the ecosystem fluxes NEE and LE, and in addition for RHA, T_{air} and N_2O VMR, while significant negative PCs were found for SWC, H and T_{soil}. All regressors combined were able to explain between 55 and 76 % of the $\ln(F_{N2O})$ variance during shorter time periods in single years, with the exception of the time period before the 1^{st} cut 2010 when r^2 was found to be statistically not significant (Table 1). The chosen set of parameters performed well with pooled data during the same time periods and especially after the 1^{st} cut, explaining between 66 and 73 % of observed daily average $\ln(F_{N2O})$ values. SWC was the most dominant regressor towards the end of the year, featuring a highly significant, negative PC (Table 1). Similarly, T_{soil} was an important parameter in the MLR analysis after the 1^{st} cut, being first positively, later negatively correlated with ln transformed N_2O exchange. Seven parameters were highly significant ($p < 0.001$) in a forward step-wise MLR analysis and explained 41 % of the $\ln(F_{N2O})$ variance during snow-free conditions, with T_{air}, N_2O VMR, RH, NEE and LE being positively correlated, SWC and H negatively (data not shown). In a simple linear regression eight out of 11 parameters were significantly correlated with the $\ln(F_{N2O})$, with T_{air} and T_{soil} as the highest positively and SWC as the highest negatively correlated regressors, respectively (Table 1).

A closer look at the two most prominent soil related regressors, T_{soil} and SWC, and $\ln(F_{N2O})$ under snow-free, undisturbed conditions revealed a clear pattern. Daily average N_2O exchange showed a bell-shaped relationship with SWC with highest emissions during periods of intermediate soil water content (Fig. 5, top panel). Even clearer was the correlation between T_{soil} and N_2O flux: days with a daily average T_{soil} above 14 °C showed an almost consistent net emission of N_2O. This was also observed for days where T_{soil} was close to zero, whereas N_2O exchange fluctuated around zero with no clear pattern between 0 and 14 °C (Fig. 5, middle panel). Taking both SWC and T_{soil} into account, days characterized by low to intermediate SWC with T_{soil} close to 0 °C or above 14 °C generally resulted in a net emission of N_2O, while deposition was mainly observed during cool conditions with high SWC (Fig. 5, lower panel). In contrast to N_2O, comparably clear exchange patterns were not found for CH_4 fluxes.

On a daily average time scale, a repeated-measures ANOVA revealed statistically significant differences among environmental conditions on days with net uptake (group f-), net emission (f+) or close-to-zero exchange (f0) of CH_4 and N_2O (Table 2). In case of CH_4, T_{air} was significantly colder on low-flux days than on emission and deposition days. Generally, environmental conditions were most different between high deposition days and days resulting in emission or close-to-zero exchange of CH_4 (Table 2). In group f-, the ecosystem fluxes LE and H, SHF, PAR, VPD and RHA were all significantly higher compared to f+ and f0, while also the net uptake of CO_2 was larger. Although results were less clear for N_2O fluxes, the meadow tended to act neither as a source or sink on days when air and soil temperatures as well as LE were low (Table 2). In addition, SWC was significantly lower in f+, while H was significantly higher on deposition days.

Cumulative fluxes for 2011 resulted in a net CO_2-uptake of -70.5 g CO_2 m^{-2} (Fig. 6). CH_4 and N_2O fluxes were converted to CO_2-equivalents, with cumulative fluxes being calculated for each of the different FIR filter time constants. In 2011, the meadow acted as a source for both compounds. When no FIR filter was applied, i.e. the overestimation of the low frequency eddy flux contribution was not corrected for, cumulative methane fluxes amounted to an emission of 54.5 g CO_2-equ. m^{-2}. With FIR filters of varying time constants, cumulative fluxes were considerably lower, in the range of 6.8 – 19.3 g CO_2-equ. m^{-2}, whereby the lower number was obtained using a FIR filter time constant of 50 s and constitutes our best guess estimate. Results were very similar for N_2O, the cumulative fluxes of which resulted in a net emission of 97.9 g CO_2-equ. m^{-2} without FIR filter, and 25.2 – 39.8 g CO_2-equ. m^{-2} using filters with different time constants. In case of N_2O, a time constant of 100 s was considered to give the most representative flux results, yielding 32.0 g CO_2-equ. m^{-2} over the whole year (Fig. 6).

The total GHG budget can be calculated by summing up the different cumulative contributions of CO_2, CH_4 and N_2O. Based on the best guess estimates, the meadow acted as a GHG sink (-31.7 g CO_2-equ. m^{-2}) in 2011. However, when no FIR filter was applied to neither CH_4 nor N_2O data, the sum of the two compound fluxes more than compensated for the sink effect of CO_2, turning the meadow into a GHG source (81.9 g CO_2-equ. m^{-2}; Fig. 6).

4 Discussion

4.1 Methane

It was shown recently that plants do not contain a known biochemical pathway to synthesize methane (Nisbet et al., 2009), a finding that contradicts observations of methane emissions from terrestrial plants under aerobic conditions in an earlier study (Keppler et al., 2006). Methane emissions from plant tissue may be due to the transpiration of water that contains dissolved CH_4 or due to the abiotic breakdown of plant material as a consequence of high UV stress conditions (Nisbet et al., 2009), but the contribution of terrestrial plants to the global methane emission is considered to be small (Dueck et al., 2007). Based on these earlier findings it is feasible to regard observed eddy covariance emission fluxes in this study as a direct (methanogen microorganisms) or indirect (transpiration of soil CH_4) consequence of processes in the soil, an important player in the global methane cycle (Kirschke et al., 2013; Smith et al., 2000).

Therefore, one might expect clear relationships between soil environmental parameters such as temperature or moisture and CH_4 exchange, which were also reported by other studies (Dijkstra et al., 2013; Hartmann et al., 2010; Imer et al., 2013; Jackowicz-Korczyński et al., 2010; Kroon et al., 2010b ; Liebig et al., 2009; Rinne et al., 2007; Schrier-Uijl et al., 2010). However, when all data were pooled no clear correlation between soil parameters and eddy covariance CH_4 exchange at the grassland site in Neustift was observed. Although the explanatory power of T_{soil} in the MLR was relatively high and significant between the 1st and 2nd cutting of the meadow in 2011 – a period when small quantities of CH_4 were taken up by the meadow around noon – no consistent relationship between soil parameters and the CH_4 flux was observed (Table 1). SHF was significantly higher on days with net deposition compared to zero-flux and net emission days (Table 2), which might be an indication of soil processes as possible drivers for observed exchange patterns. The partial correlations of SWC with CH_4 exchange, however, were statistically not significant throughout the measurement campaign and close to zero when all data were pooled (Table 1). This is in contrast to chamber studies that identified soil moisture as a key driver for methane exchange (e.g. Dijkstra et al., 2013b).

One explanation for this lack of correlation between soil parameters and methane fluxes might be that half-hourly eddy covariance fluxes represent an integral signal, averaged over 30 minutes over a possibly heterogeneous area of methane sources and covering both "hot spots" of high methane emission and areas of relatively high uptake within the same flux footprint (Baldocchi et al., 2012). Therefore, SWC may be high in certain patches of the meadow and create environmental conditions conducive for methanogenic microorganisms, but low in other microsites across the grassland. Half-hourly fluxes reflect this heterogeneity across the footprint to a varying degree, mainly depending on wind direction, wind speed and atmospheric stability. In addition, the direct effect of certain drivers on CH_4 exchange may smear out at ecosystem scale, especially if associated fluxes are generally low. Recently Yvon-Durocher et al. (2014) found an average temperature dependence of CH_4 emissions from aquatic, wetland and rice-paddy ecosystems similar to that of CH_4 production derived from pure cultures of methanogens and anaerobic microbial communities in the laboratory. No such relationship was found in the present study, which may be a direct consequence of a heterogeneous footprint with regards to CH_4 sources and generally low CH_4 fluxes at the measurement site in Neustift.

The observation of weak CH_4 uptake around noon between March and July 2011 (Figure 2) is most likely a consequence of methanotrophic microorganisms in the soil, a process enhanced by increased soil temperature. However, it is difficult to observe this temperature dependence at ecosystem scale, as the whole footprint regardless of emission / deposition hot spots is sampled. In addition, it was shown that both methanotrophic and methanogenic activity in the soil are temperature dependent (von Fischer and Hedin, 2007; Yavitt et al., 1995), whereby the latter tends to be more responsive to temperature (Topp and Pattey, 1997). Imer et al. (2013) reported nearly consistent methane uptake throughout the year except for winter at three different grassland sites along an altitudinal and management gradient using static chambers, with flux rates of generally below 10 ng m^{-2} s^{-1}. Three pastures investigated by Liebig et al. (2009) were identified as minor CH_4 sinks.

Daily average CH_4 emissions in this study generally ranged ranged up to 100 ng m^{-2} s^{-1} and were relatively similar to eddy covariance results over a drained and grazed peatland pasture during dry periods, when fluxes were often below 160 ng m^{-2} s^{-1} (Fig. 2; Baldocchi et al., 2012). However, the maximum CH_4 flux and concentration of more than 5700 ng m^{-2} s^{-1} and 3500 ppb, respectively, at the peatland site were much higher than the 128 ng m^{-2} s^{-1} and 2300 ppb recorded at Neustift. Higher maximum methane fluxes were also observed by Schrier-Uijl et al. (2010) over a grass ecosystem on peat (1604 ng m^{-2} s^{-1}).

In comparison to CO_2 and energy fluxes, there are only few long-term EC methane exchange studies. However, year-round measurements are indispensable for accurately estimating the CH_4 budget of an ecosystem. Baldocchi et al. (2012) give a three-year mean annual methane efflux at a peatland pasture of 11.6 ±9.0 g m^{-2} yr^{-1} without any discrimination for cattle or elongated footprints during the night, and 3.6 ±1.9 g m^{-2} yr^{-1} when only daytime data representing the well-drained portion of the pasture, additionally filtered for favorable wind directions and the presence of cows, were used. This latter number is relatively similar to the methane efflux of 2.1 g m^{-2} yr^{-1} in Neustift in 2011. In comparison, Hendriks et al. (2007) reported 14.2 +/-26.1 g m^{-2} yr^{-1} from the relatively dry portions of an abandoned peat meadow using chamber measurements, and 42.5 ±27.7 g m^{-2} yr^{-1} when the whole meadow, including water-saturated land and ditches, was considered. Mander et al. (2010) conducted a literature survey and reported median fluxes of 0.16 g m^{-2} yr^{-1} for fertilized grasslands on hydromorphic soils in Estonia, similar to Neustift (0.27 g m^{-2} yr^{-1}). Methane emissions reported by Merbold et al. (2014) from a grassland after restoration where one order of magnitude higher (3.6 g m^{-2} yr^{-1}). Using eddy covariance measurements, methane emissions between 24 – 29 g m^{-2} yr^{-1} were reported from a subarctic peatland (Jackowicz-Korczyński et al., 2010), 12.6 g m^{-2} yr^{-1} from a boreal fen (Rinne et al., 2007) and 16.5 g m^{-2} yr^{-1} from a managed fen meadow (Kroon et al., 2010b).

Baldocchi et al. (2012) reported mean diurnal patterns characterized by lowest methane efflux densities during midday and elevated methane emission throughout the night, a pattern very similar to Neustift during certain time periods, e.g. between the 1st – 2nd cut 2010 (Fig. 4). We mainly attributed this observation to meteorological factors, i.e. intermittent exchange during calm and stable nighttime conditions, which was also the reasoning behind the outlier handling in our despiking procedure (Fig. 1a). Another reason might be the preferential sampling of an elevated methane source in combination with a larger nighttime footprint as described by Baldocchi et al. (2012). It is possible that methane emissions from a small stream and adjacent wet patches of the meadow, that are normally not part of the footprint, have contributed disproportionally to observed methane emissions. Unfortunately we lack detailed high-resolution spatial data (e.g. vegetation, soil) about small areas and patches within the sampled flux footprint in Neustift, which would be required for a meaningful footprint analysis. Therefore, we are currently not able to further discuss potential emission hotspots, their impact on calculated CH_4 balances and the problem of possibly preferential sampling within this manuscript. Hot spot footprint analysis merits its own research and would provide important insights in how to interpret eddy covariance flux data.

Several studies reported that 81 – 90 % of the total annual methane emission occurred during the snow free period or between spring – autumn (Jackowicz-Korczyński et al., 2010; Rinne et al., 2007), which is very similar to Neustift in 2011, where 84 % of the yearly net CH_4 emission occurred during snow free conditions.

4.2 Nitrous oxide

Despite occasional uptake, the meadow was a source of N_2O, in accordance with previous studies over managed grasslands. Half-hourly emission rates of N_2O, mostly below 50 ng N_2O m^{-2} s^{-1}, were similar to exchange rates reported by Neftel et al. (2010) for an experimental farm site and Imer et al. (2013) from a mountain rangeland. N_2O fluxes in 2011 amounted to an emission of 107 mg m^{-2} yr^{-1}. For comparison, Mander et al. (2010) reported approx. 94 and 723 mg m^{-2} yr^{-1} for unfertilized and fertilized grasslands, respectively. Considerably higher emissions were found by Kroon et al. (2010b) for a managed fen meadow (2.4 g N_2O m^{-2} yr^{-1}), and by Merbold et al. (2014) for a grassland after restoration (4.6 g m^{-2} yr^{-1}).

Many of the observations made for CH_4 were also valid for N_2O, with generally low fluxes, a possibly heterogeneous flux footprint with respect to emission / deposition hot spots and soil processes as the driving force behind N_2O exchange patterns. In contrast to CH_4 exchange, N_2O fluxes on a daily scale could be well explained by environmental parameters during specific time periods. The important role of temperature in soil processes was shown previously, as N mineralization, nitrification, denitrification and N_2O emissions all increase with temperature (Barnard et al., 2005), while reduced soil moisture as a result of high air temperatures and increased plant transpiration can decrease N_2O emissions (Li et al., 1992). These findings are comparable to observations in the present study, where N_2O exchange tended to emission during warm and relatively dry soil conditions (Figure 5, lower panel).

N_2O consumption in the soil occurs when N_2O reduction exceeds N_2O production (Chapuis-Lardy et al., 2007). Soil water is probably the key driver regulating N_2O consumption in soils, as it can act as a temporary storage body that entraps N_2O, effectively hindering its diffusion from the soil matrix to the surface. As a consequence, the time for potential reduction of N_2O to N_2 through anaerobic denitrification is increased (Clough et al., 2005). This can result in a low N_2O / N_2 ratio during wet conditions, which favors N_2O consumption (Ruser et al., 2006; Wu et al., 2013). These observations agree with our findings at ecosystem scale. When all data were pooled, N_2O uptake was highest during relatively wet conditions (Figure 5, top panel) and SWC was significantly lower on days with clear net emission of N_2O (Table 2). The latter finding is further highlighted by a clear positive correlation between daily average $\ln(F_{N2O})$ and T_{soil} in the soil temperature range 12-16 °C as long as SWC was low (data not shown).

In October 2011, manure application resulted in a pulse of N_2O emission one day later, after which fluxes rapidly decreased and reached pre-fertilization rates two days after manure spreading. Similar behavior of N_2O fluxes returning to background levels within 2-6 days after fertilization has been observed by Jones et al. (2011) for a Scottish grassland and Neftel et al. (2010) for an experimental farm site. Pulses of N_2O emissions after fertilizer application were also described in other studies (e.g. Granli and Bockman, 1994; Jones et al., 2011) and might be the result of animal manure – the most concentrated form of anthropogenic N input (Davidson, 2009) – directly fueling nitrifying and denitrifying bacteria in the soil, which are most active when N is abundant (Firestone and Davidson, 1989). Over the weeks following fertilization, N_2O emissions increased with air temperature, which is in-line with the temperature dependence of the involved processes. We observed a sharp increase of N_2O emissions once the daily average air temperature fell below the freezing point, approx. four weeks after manure spreading in November 2011. During this time period the meadow remained snow-free, with soil temperatures close to 0°C. The combination of reduced plant metabolism (low nitrate demand by plants) and prior manure spreading could result in an abundance of soil NO_{3-} at the end of the vegetation period. Wertz et al. (2013) showed that denitrification can still occur at very low temperatures and even below the freezing point when NO_{3-} and C are present. The observation of high N_2O emissions from frozen or nearly frozen soil was also made by earlier studies (Röver et al., 1998; Teepe et al., 2001).

Production and subsequent emission of N_2O remained high after the beginning of the snow cover in December 2011. Zhu et al. (2005) described a similar situation where microbial activity in the soil of a lowland tundra did not cease during snow cover and N_2O continuously diffused to the atmosphere through the snowpack. In Neustift, high N_2O emissions were not observed one year earlier during similar conditions.

4.3 Global warming potential

The availability of year-round data allows for the calculation of a yearly GWP balance over a specific ecosystem. In this study, year-round CH_4, N_2O and CO_2 flux data were available for 2011. When expressing the net exchange of the three compounds in terms of CO_2-equivalents and adding up these different contributions, the resulting GWP of the meadow in Neustift was -32 g CO_2-equ. m^{-2} yr^{-1} in 2011, whereby a yearly NEE of -71 g CO_2 m^{-2} yr^{-1} was offset by CH_4 and N_2O emissions of 7 and 32 g CO_2-equ. m^{-2} yr^{-1}, an offset of approx. 55%.

Liebig et al. (2009) investigated three years of CH_4 / N_2O static chamber fluxes, soil organic carbon change, CO_2 emissions associated with N fertilizer production and CH_4 emission from enteric fermentation for three grazing management systems. The resulting net GWP between -78 – 40 g CO_2-equ. m^{-2} yr^{-1} is similar to results in this study. Hendriks et al. (2007) reported -86 g CO_2-equ. m^{-2} yr^{-1} from an abandoned peat meadow. Merbold et al. (2014) give the full GHG flux budget of an intensively managed grassland after restoration, including ploughing. GHG emissions reported in their study were much higher than in Neustift, amounting to 2.9 kg CO_2-equ. m^{-2}, and relatively similar to the balance of 1.6 kg CO_2-equ. m^{-2} found by Kroon et al. (2010b) for a managed fen meadow. Zona et al. (2013) reported a GHG balance of -260 g CO_2-equ. m^{-2} yr^{-1} for a poplar plantation in 2011, taking into account CO_2 fluxes of -351 g CO_2-equ. m^{-2} yr^{-1}, and CH_4 and N_2O fluxes of 49 and 42 g CO_2-equ. m^{-2} yr^{-1}, respectively, with CH_4 and N_2O offsetting the NEE sink by 26 %. Soussana et al. (2007) investigated the GHG budget of nine European grassland sites over two years, covering a major climatic gradient and a wide range of management regimes. On average, the investigated grassland plots were a net sink of -879 g CO_2 m^{-2} yr^{-1}, and a net source of 117 and 51 g CO_2-equ. m^{-2} yr^{-1} for CH_4 and N_2O, respectively, with emissions of the latter two compounds resulting in a 19 % offset of the NEE sink activity. Tian et al. (2014) reported offset ratios of 73 % for the whole North American continent, with the grassland GWP being nearly neutral.

Rinne et al. (2007) reported a GWP balance of +108 g CO_2-equ. m^{-2} when taking into account CO_2 and CH_4 fluxes from a boreal fen, with respective fluxes amounting to -156 and +264 g CO_2-equ. m^{-2}. Although the GWP calculated from CO_2 and CH_4 fluxes was much lower in Neustift (-64 g CO_2-equ. m^{-2}), the situation was similar in that the carbon uptake of the meadow through CO_2 was partially offset by carbon loss through CH_4 emission. The number for Neustift may change drastically on a year-to-year basis, as the meadow can act both as a source and sink of CO_2 (Wohlfahrt et al., 2008a), while it is supposedly a constant source of CH_4. Dijkstra et al. (2013) used static chambers to calculate the GWP for five years of CO_2 and CH_4 data in a semiarid grassland, ranging between -3 and -6 g CO_2-equ. m^{-2}.

5 Conclusion

The grassland site in Neustift is characterized by low fluxes of CH_4 and N_2O. Although the meadow can act as a source and sink for both compounds during certain time periods, it is a clear source of CH_4 and N_2O on an annual time scale. As a consequence, both gases contribute to an increase of the GWP, effectively reducing the sink strength in terms of CO_2-equivalents.

Our analyses showed that daily average N_2O exchange during most of the vegetation period can be well explained with simultaneously recorded ancillary data, especially in the time period after the 1st cut in June up until snow cover towards the end of the year. In contrast, modeling daily average exchange with the same ancillary data worked considerably worse for CH_4, a finding that suggests the possibility of a more heterogeneous footprint in regard to methane sources and sinks. For both compounds it was not possible to single out one driving variable as the most important one, which is to be expected due to the nature of the eddy covariance flux signal in combination with generally low CH_4 and N_2O fluxes at the investigated grassland site.

In comparison to CO_2, H_2O and energy fluxes, the interpretation of CH_4 and N_2O exchange is challenging due to uncertainties regarding post-processing, quality control and footprint heterogeneity. Knowledge about emission and deposition hotspots within the footprint area would allow for a more comprehensive interpretation of the bulk EC flux. Additional information about GHG producing and consuming patches within the flux footprint could be achieved for example via chamber measurements, another possibility would be to perform a detailed statistical analysis of EC fluxes and underlying footprint information in combination with detailed spatial data of the sampled area.

We conclude that CH_4 and N_2O fluxes over supposedly well-aerated and moderately fertilized soils cannot be neglected when evaluating the GHG impact of temperate managed grasslands. Both compounds can significantly influence the GWP balance of a meadow and be determining if a grassland is acting as a source or sink of CO_2-equivalents. In order to reliably assess GHG budgets on a local and global scale, long-term measurements of CH_4 and N_2O fluxes in combination with CO_2 exchange are necessary, especially over ecosystems that are normally characterized by low GHG fluxes. In addition, we recommend to carefully check flux results and underlying cospectra for an overestimation in the low spectral range and correct for this effect if necessary.

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Table 1. Partial correlations of a multiple linear regression analysis and correlation coefficients (r) of a simple linear regression analysis using daily average values of ln transformed CH_4 (F_{CH4}) and N_2O (F_{N2O}) flux rates as dependent variables and air temperature (T_{air}), soil temperature (T_{soil}) and soil water content (SWC) in 5 cm depth, soil heat flux (SHF), net ecosystem CO_2 exchange (NEE), latent (LE) and sensible (H) heat flux, photosynthetically active radiation (PAR), vapor pressure deficit (VPD), relative air humidity (RHA) and CH_4 / N_2O volume mixing ratios (VMR) as independent variables. Management events were excluded from the analysis. Bold numbers highlight $p < 0.05$, except bold underlined numbers resulted in $p < 0.001$. Results shown for the "vegetation period" do not include time periods with snow cover on the meadow.

	MULTIPLE LINEAR REGRESSION partial correlations														**SIMPLE LINEAR REGRESSION r**
	vegetation period	**snow melt – 1st cut**			**1st cut – 2nd cut**			**2nd cut – 3rd cut**			**3rd cut – snow cover**			**snow cover**	**vegetation period**
	2010-11	**2010**	**2011**	**2010-2011**	**2010**	**2011**	**2010-2011**	**2010**	**2011**	**2010-2011**	**2010**	**2011**	**2010-2011**	**2010-2012**	**2010-2011**
ln(F_{CH4})															
T_{air}	**0.19**	0.07	0.07	**0.25**	-0.02	-0.35	0.11	-0.05	0.32	0.02	0.20	0.01	-0.02	0.17	**0.13**
T_{soil}	-0.04	0.10	-0.07	-0.16	-0.13	**0.57**	0.10	-0.08	-0.12	-0.09	0.01	0.15	0.11	-0.11	**0.16**
SWC	0.07	0.06	-0.24	0.04	-0.20	-0.13	0.13	0.03	0.33	0.05	0.33	-0.05	-0.09	-0.13	**0.10**
SHF	**-0.22**	-0.14	-0.14	**-0.26**	0.12	0.22	-0.16	0.02	-0.28	0.01	-0.29	-0.08	-0.10	0.04	-0.09
NEE	**0.20**	0.12	**0.38**	0.18	0.24	0.10	0.19	-0.04	0.05	0.01	0.33	0.18	**0.32**	0.20	**0.30**
LE	-0.05	-0.16	-0.17	-0.12	0.09	-0.23	-0.05	-0.21	0.10	-0.17	0.20	**0.48**	**0.44**	**0.28**	**-0.19**
H	-0.06	-0.08	**-0.38**	-0.13	-0.25	-0.03	-0.09	-0.08	0.13	0.10	0.01	-0.33	-0.18	-0.09	**-0.19**
PAR	0.10	0.23	0.16	**0.25**	-0.08	-0.16	0.00	0.25	-0.20	0.06	-0.17	-0.07	-0.13	0.00	**-0.19**
VPD	-0.07	0.08	0.02	-0.09	-0.01	0.10	-0.01	0.20	-0.26	0.19	-0.11	-0.16	-0.08	-0.12	-0.09
RHA	0.03	0.12	0.07	0.06	0.12	0.05	0.05	0.30	-0.28	0.21	-0.31	0.03	0.02	-0.08	**0.23**
CH_4 VMR	0.01	0.08	0.00	0.02	0.15	**0.39**	0.06	**-0.35**	0.11	-0.15	0.35	-0.12	-0.11	0.01	0.02
multiple r^2	**0.27**	0.31	**0.54**	**0.20**	**0.43**	**0.62**	**0.36**	0.41	0.23	0.18	**0.55**	**0.53**	**0.40**	0.22	
N	356	47	67	114	50	36	86	44	40	84	35	37	72	82	365-397
ln(F_{N2O})															
T_{air}	**0.14**	-0.04	**0.27**	0.03	0.28	-0.06	0.03	0.10	0.14	0.21	0.03	0.05	0.05	0.17	**0.29**
T_{soil}	**-0.11**	0.09	-0.16	0.06	-0.16	0.22	**0.30**	-0.07	-0.06	**-0.27**	-0.22	-0.18	**-0.33**	-0.12	**0.24**
SWC	**-0.24**	-0.13	-0.15	**-0.24**	-0.18	-0.27	-0.21	**-0.38**	-0.31	**-0.51**	0.01	**-0.45**	**-0.47**	-0.05	**-0.33**
SHF	-0.02	0.03	-0.23	0.02	-0.26	0.16	-0.11	-0.12	-0.11	-0.14	**0.42**	0.19	0.24	-0.10	**0.15**
NEE	**0.23**	-0.16	**0.31**	0.13	0.10	-0.10	0.10	**0.35**	0.24	**0.32**	0.32	-0.12	0.09	0.02	0.00
LE	**0.19**	-0.16	-0.11	-0.08	-0.10	0.07	-0.03	0.04	0.19	0.11	**0.41**	0.05	0.17	0.18	**0.22**
H	**-0.14**	-0.24	-0.16	**-0.23**	-0.08	-0.20	-0.14	**0.45**	0.00	0.18	0.22	-0.10	0.13	**-0.25**	**-0.20**
PAR	-0.02	0.21	0.21	0.13	**0.37**	-0.03	0.22	-0.22	0.18	0.02	-0.37	-0.04	**-0.32**	0.06	0.05
VPD	0.01	-0.24	-0.11	-0.11	0.09	-0.04	0.13	0.26	0.05	**0.23**	**-0.47**	-0.03	-0.17	-0.10	**0.16**
RHA	**0.24**	-0.21	0.18	-0.01	**0.45**	0.03	**0.33**	**0.37**	0.23	**0.37**	**-0.60**	0.07	-0.13	0.00	0.08
N_2O VMR	**0.25**	-0.05	-0.11	0.02	**0.39**	0.28	**0.26**	-0.15	0.11	-0.13	-0.26	-0.06	-0.04	**0.39**	**0.17**
multiple r^2	**0.42**	0.19	**0.55**	**0.26**	**0.76**	**0.73**	**0.66**	**0.72**	**0.56**	**0.68**	**0.73**	**0.68**	**0.73**	**0.44**	
N	360	49	67	116	50	36	86	44	41	85	36	37	73	83	369-401

Table 2. Daily average means in three different groups of daily net CH_4 / N_2O exchange. Significant differences between group means were determined in a repeated measures ANOVA setting, using the Unequal N HSD *post hoc* test. Group labels to the right of a given group mean show to which flux group the respective value was significantly different. Bold numbers mark group means that were significantly different from one other group, except bold underlined numbers denote group means that were significantly different from both other groups. f+… daily average CH_4 / N_2O emission fluxes > 3 / 0.4 ng m^{-2} s^{-1}, f0…fluxes between 3 / 0.4 and -3 / -0.4 ng m^{-2} s^{-1}, f-…deposition fluxes < -3 / -0.4 ng m^{-2} s^{-1}.

	Unit	**Mean values, standard deviations and significant differences**					
compound		**CH_4**			**N_2O**		
flux class		**f+**	**f-**	**f0**	**f+**	**f-**	**f0**
T_{air}	°C	**9.2 ±7.0 f0**	**9.6 ±6.0 f0**	**5.9 ±8.5 f+,f-**	**10.1 ±7.5 f0**	**8.4 ±5.1 f0**	**4.2 ±7.3 f+,f-**
T_{soil}	°C	10.8 ±6.5	10.9 ±5.3	8.2 ±6.7	**11.5 ±6.8 f0**	**10.1 ±4.8 f0**	**6.5 ±5.5 f+,f-**
SWC	m^3 m^{-3}	0.29 ±0.06	0.28 ±0.06	0.29 ±0.09	**0.27 ±0.07 f-,f0**	**0.31 ±0.05 f+**	**0.32 ±0.06 f+**
SHF	W m^{-2}	**1.0 ±6.9 f-**	**3.9 ±6.6 f+,f0**	**0.5 ±6.4 f-**	**2.2 ±7.3 f0**	1.4 ±6.0	**-1.5 ±6.1 f+**
NEE	µg CO_2 m^{-2} s^{-1}	**-70 ±224 f-**	**-220 ±229 f+**	-119 ±220	-106 ±246	-141 ±220	-40 ±172
LE	W m^{-2}	**55 ±53 f-**	**86 ±58 f+,f0**	**50 ±57 f-**	**67 ±64 f0**	**64 ±44 f0**	**30 ±37 f+,f-**
H	W m^{-2}	**6.9 ±22.0 f-**	**20.2 ±22.7 f+,f0**	**8.8 ±19.8 f-**	**7.5 ±22.8 f-**	**16.7 ±21.9 f+,f0**	**4.4 ±16.8 f-**
PAR	µmol m^{-2} s^{-1}	**271 ±158 f-**	**372 ±169 f+,f0**	**250 ±168 f-**	293 ±180	**314 ±149 f0**	**217 ±139 f-**
VPD	kPa	**0.33 ±0.28 f-**	**0.42 ±0.26 f+,f0**	**0.28 ±0.29 f-**	0.36 ±0.30	0.35 ±0.25	0.23 ±0.22
RHA	%	**81 ±10 f-**	**75 ±10 f+,f0**	**82 ±11 f-**	**81 ±10 f-**	**77 ±11 f+**	82 ±12
VMR	ppb	2014 ±59	2004 ±53	2021 ±60	**319 ±6 f-**	**317 ±4 f+**	319 ±4
N	days	294	96	48	261	138	44

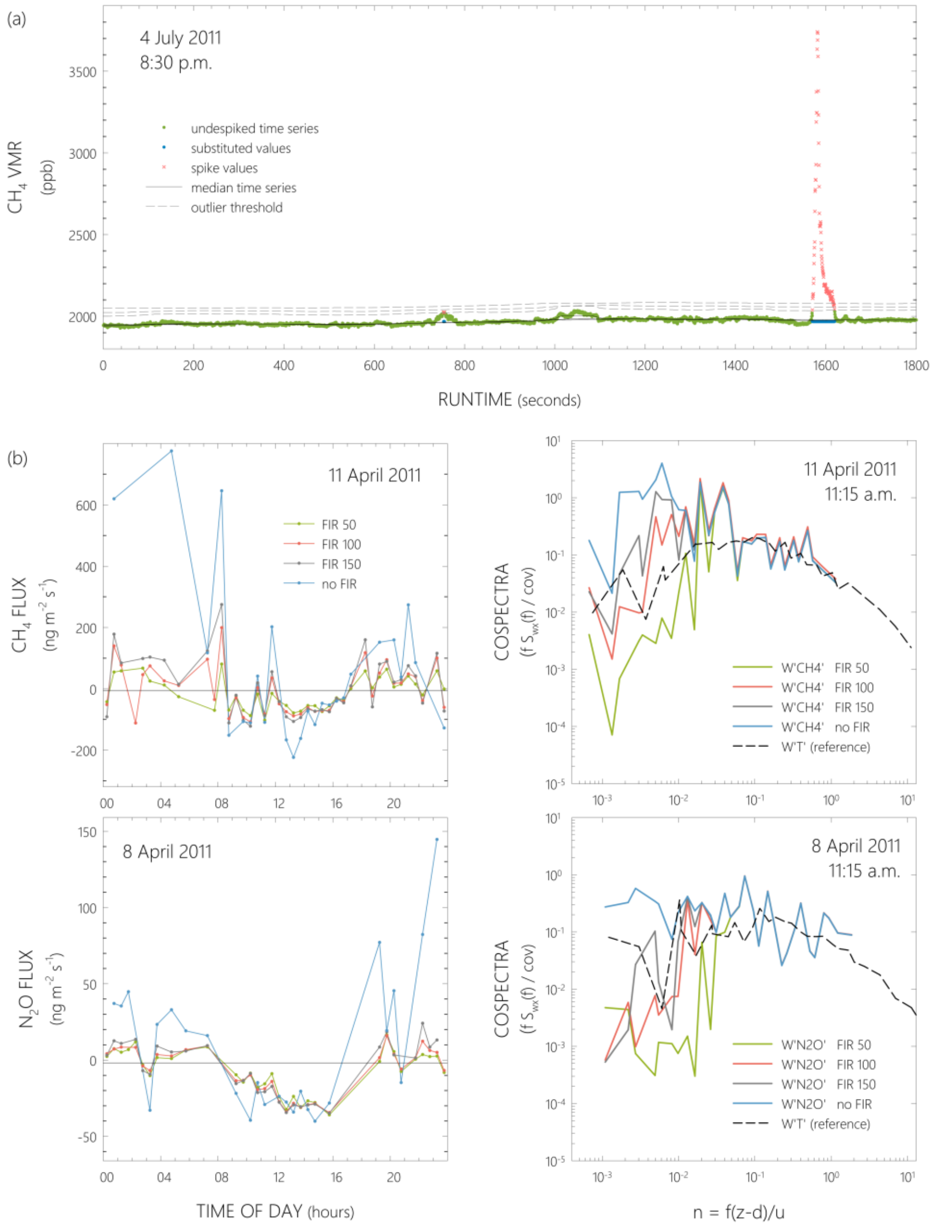

Figure 1. (a) Despiking example of 2 Hz methane VMRs using median filters. (b) Diurnal courses (left panels) and normalized co-spectra (right panels) illustrating the effect of high-pass filtering CH_4 (upper panels) and N_2O (lower panels) time series with a non-recursive finite impulse response (FIR) filter with different time constants (50, 100 und 150 s). Sensible heat cospectra are shown in the right panels for reference.

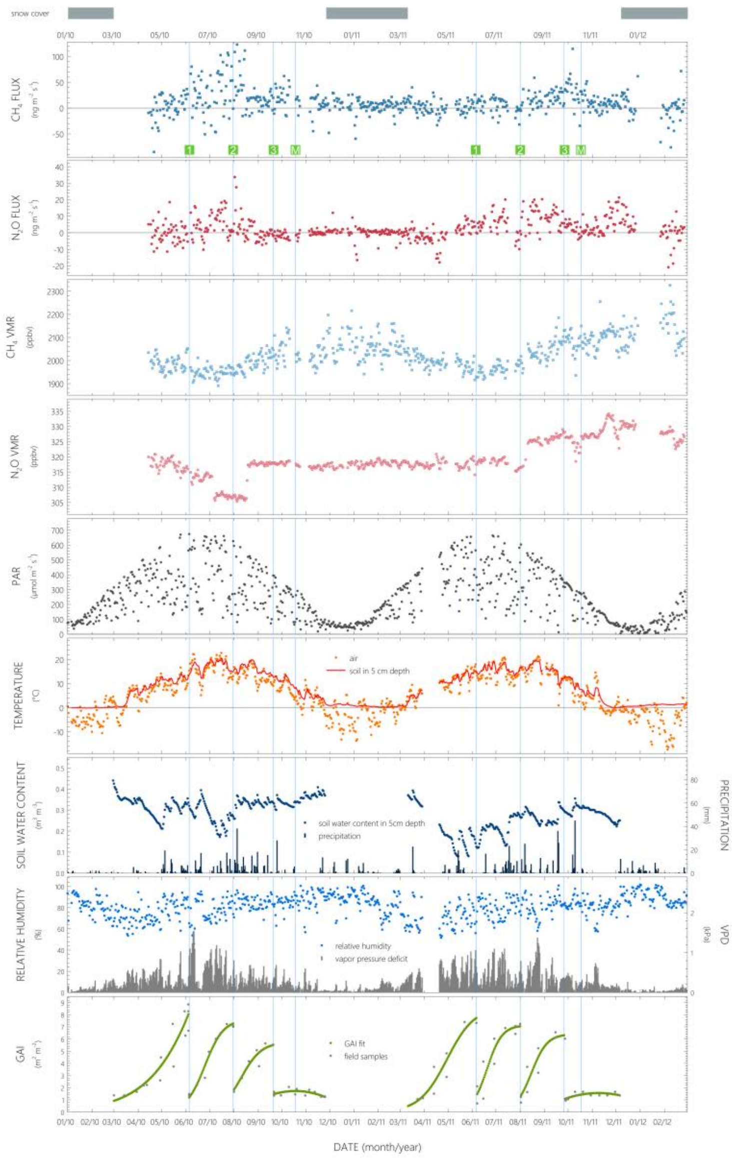

Figure 2. Daily average CH_4 and N_2O fluxes and volume mixing ratios (VMR), photosynthetically active radiation (PAR), air temperature, soil temperature at 5 cm depth, soil water content at 5 cm depth, relative air humidity, vapour pressure deficit, green plant area index (GAI) and daily sums of precipitation over 22 months of measurements between April 2010 and February 2012. Vertical lines show management dates, numbers 1, 2 and 3 in green squares indicate the 1st, 2nd and 3rd cutting of the meadow, respectively, while M denotes manure spreading.

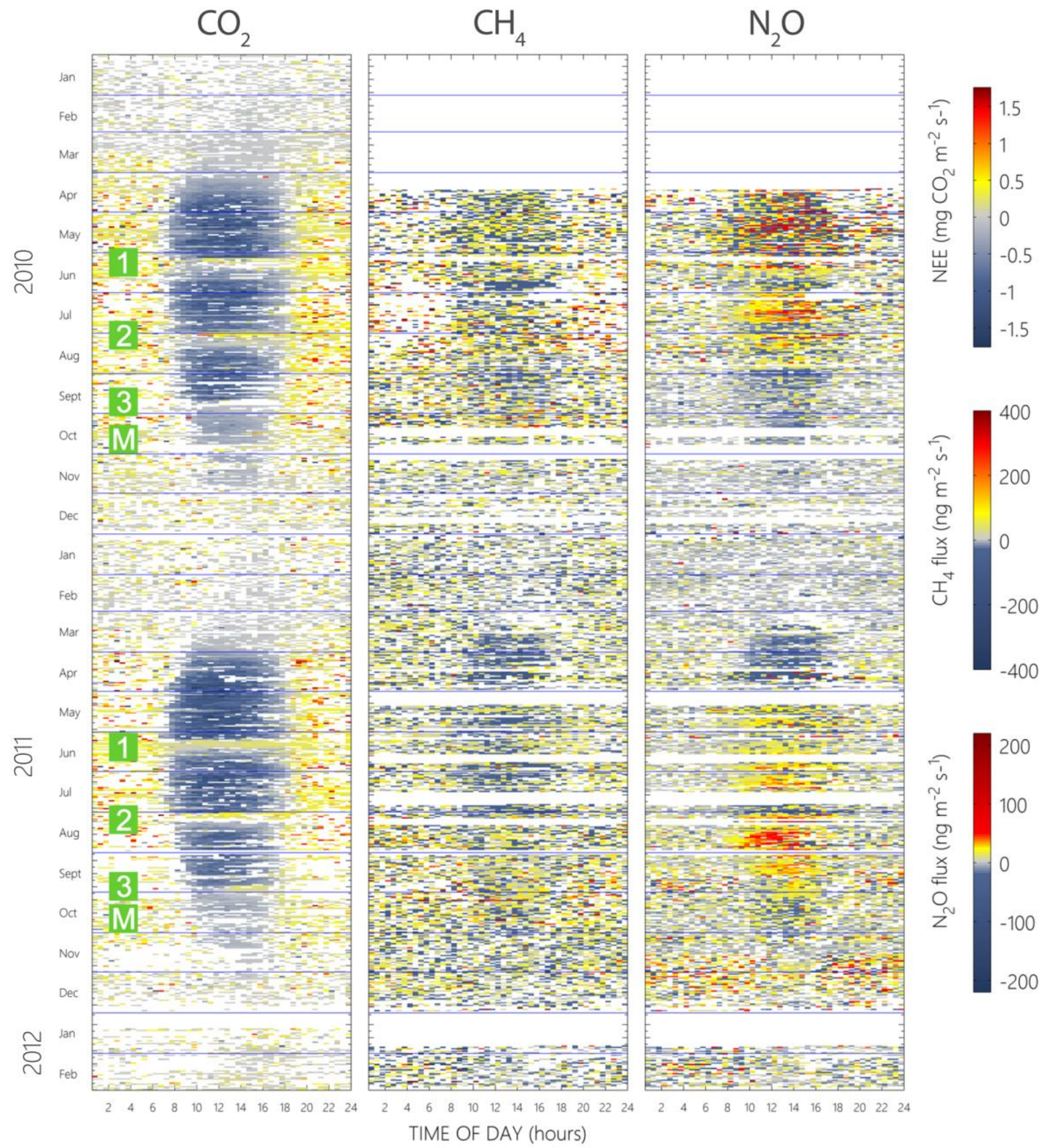

Figure 3. Half-hourly CO_2, CH_4 and N_2O fluxes over two years of GHG flux measurements. Numbers 1, 2 and 3 in green squares indicate the 1st, 2nd and 3rd cutting of the meadow, respectively, while M denotes manure spreading. Horizontal blue lines show the start and end of months. White color marks missing data.

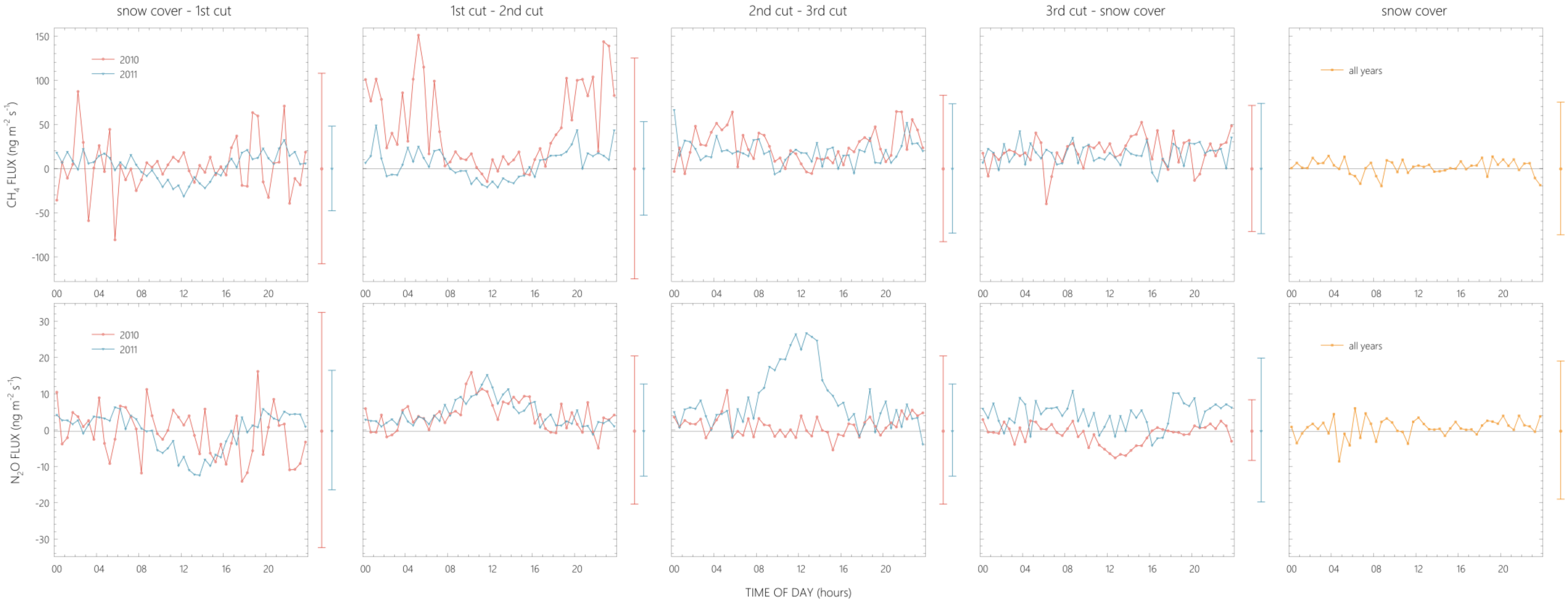

Figure 4. Diurnal cycles of CH_4 and N_2O fluxes during different time periods in 2010 and 2011. Whiskers to the right of each plot show the average standard deviation during the respective time period. Management data were excluded from the analysis.

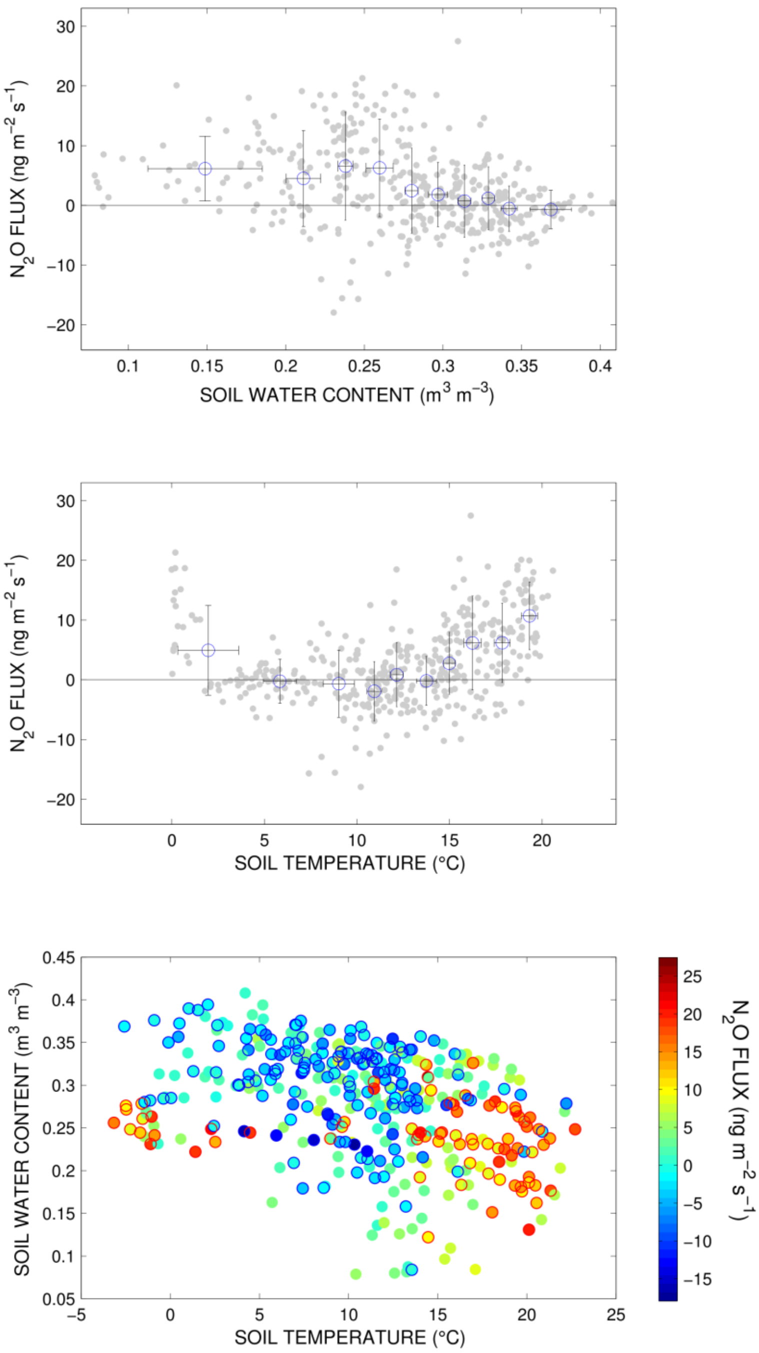

Figure 5. N_2O daily average fluxes (grey dots) *versus* soil water content and soil temperature. Blue circles in the upper two panels show bin averages (40 days per bin), with error bars representing the standard deviation within each bin. In the lower panel, fluxes < 0 ng m^{-2} s^{-1} are circled in blue, fluxes > 9 ng m^{-2} s^{-1} are circled in red. Management events were excluded from the analysis.

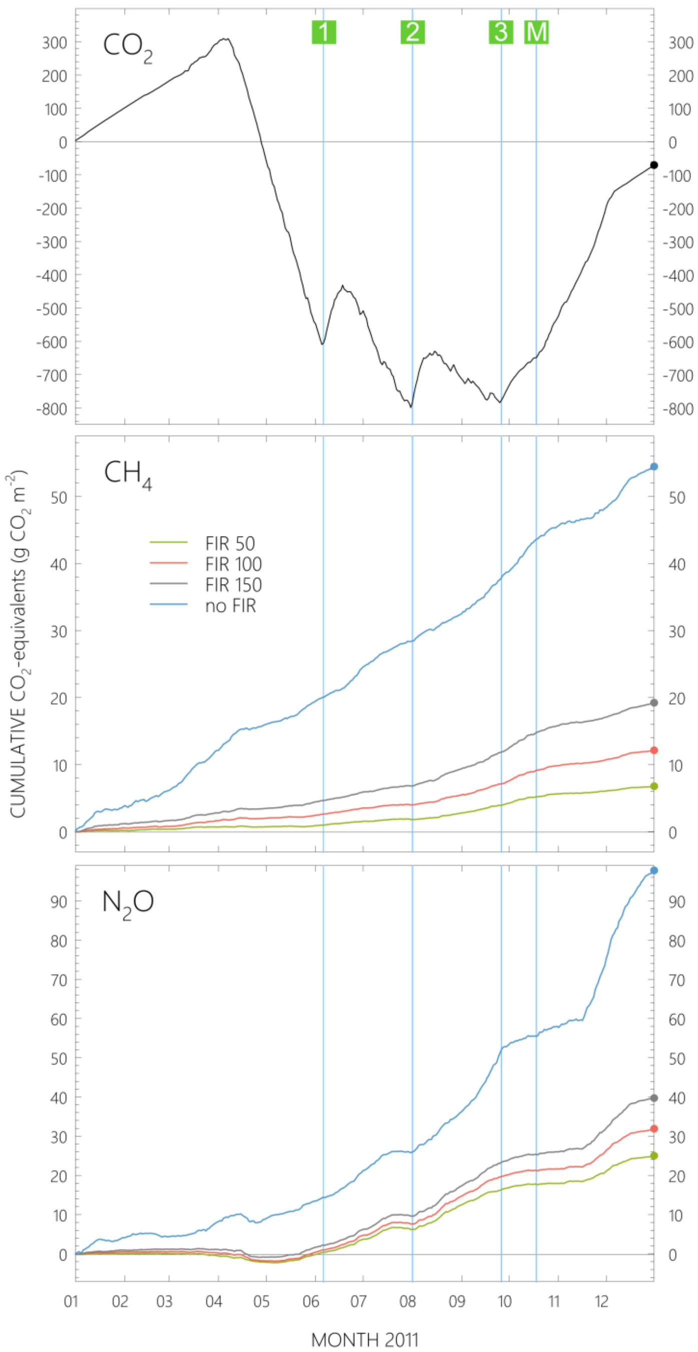

Figure 6. Cumulative GHG fluxes in 2011 expressed as CO_2-equivalents. The effect of the finite impulse response (FIR) filter with different time constants is shown for CH_4 and N_2O budgets. Vertical lines show management dates, numbers 1, 2 and 3 in green squares indicate the 1st, 2nd and 3rd cutting of the meadow, respectively, while M denotes manure spreading.