Soil greenhouse gas budget of two intensively managed grazing systems

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Abstract.

Increasing the soil carbon stock by optimizing the grassland management is seen as a potential cost-effective mitigation strategy to counteract greenhouse gas (GHG) emissions from pasture fields associated with the application of fertilizer or cattle excreta. This study presents results of GHG flux measurements in a paired field experiment of two neighboring grazing systems over a full year. Each pasture was grazed by 12 dairy cows in a rotational grazing management. Different feeding strategies (system G: grazing only; system M: grazing with supplement of maize silage) resulted in different nitrogen excretion rates. The field scale emissions of CO_2, methane (CH_4) and nitrous oxide (N_2O) were quantified using the eddy covariance (EC) technique on both parallel pasture fields excluding the direct emissions by the animals. Small-scale CH_4 emissions from excreta patches and background areas were also measured using the 'fast-box' chamber technique. The fast-box measurements dominated by dung patch emissions were up-scaled to the field and compared to the EC measurements revealing some discrepancies between the two measurement approaches. N_2O emissions resulted mainly from slurry and mineral fertilization (44 ± 17 %) while both animal excreta and background emissions contributed each about 28 % to the annual emissions. Total N_2O emissions amounted to 3.8 ± 1.1 kg N-N_2O ha^{-1} yr^{-1} and 3.7 ± 1.4 kg N-N_2O ha^{-1} yr^{-1} for the pasture fields M and G, respectively. In order to determine the net ecosystem carbon budget (NECB) of the pasture fields, additional non-gaseous carbon fluxes were either measured (harvest, slurry application) or estimated based on an animal feeding model (grazing intake, excreta). For the investigated year, the NECB (negative sign indicating sequestration) resulted in –103 ± 74 g C m^{-2} yr^{-1} and –63 ± 67 g C m^{-2} yr^{-1} for pastures M and G, respectively, indicating the tendency of carbon storage especially in the pasture M soil. However, the emissions of CH_4 and N_2O almost balanced the NECB resulting in non-significant (near-neutral) net GHG budgets for the pasture soils (field M: -144 ± 277 g CO_2 eq. m^{-2} yr^{-1}, field G: -9 ± 256 g CO_2 eq. m^{-2} yr^{-1}). Due to the paired plot design of the experiment, the difference between the two pasture fields was significant with pasture M showing a reduction in net GHG emissions by 135 ± 89 g CO_2 eq. m^{-2} yr^{-1} compared to pasture G.

Keywords: Greenhouse gas budget; N_2O; NECB; Eddy covariance; Grazing; dietary effect

1 **Introduction**

The agricultural sector is a main contributor to anthropogenic global greenhouse gas (GHG) emissions (14 %, IPCC, 2014) and grasslands occupy a large share of the agricultural land (Western Europe: 40 %; Switzerland: 72%, Peeters, 2004), which is usually used for dairy and meat production. An important GHG source on pastoral lands are nitrous oxide (N_2O) emissions resulting from fertilization (Jones et al., 2011) and animal excretion (Voglmeier et al., 2019). On the contrary, atmospheric carbon dioxide (CO_2) is predominantly recycled by the ecosystem with uptake by the plants through photosynthesis and release through respiration from soils and plants. The net source or sink effect for CO_2 depends on the change in soil carbon (C) storage, which is influenced by climatic conditions and agricultural management. An optimized grassland management is therefore regarded as a potential cost-effective mitigation strategy to reduce atmospheric CO_2 concentrations (Soussana et al., 2010) by enhancing the CO_2 sink strength and thereby increasing the soil carbon stock. The soil also can release or take up atmospheric methane (CH_4), which often depends on environmental conditions such as soil moisture (Schaufler et al., 2010). Emissions of CH_4 from pastures may also result from the decomposition of dung patches from the animals, but studies looking into this effect on real pasture systems are very rare. The annual CH_4 emissions from the pasture soil (including the emissions from dung patches) are usually rather small, but taking into account the strong global warming potential (GWP for an averaging period of 100 years including climate-carbon feedbacks) of CH_4 (34 CO_2-eq, IPCC, 2014) the effect on the GHG budget is still notable. N_2O has an even higher GWP of 298 CO_2-eq (IPCC, 2014). The magnitude of N_2O emissions depends on the nitrogen (N) input to the soil, and on grazed pastures the largest share of the emissions is typically determined by the N applied via fertilization and excreta of the grazing animals (Luo et al., 2017). There are studies determining N_2O emissions from grazed pastures (e.g. Cowan et al., 2015; Jones et al., 2011; Luo et al., 2010), but these studies mostly report only on the annual emissions without quantifying the contribution of the different emission sources from pastures (e.g. fertilizer vs. grazing induced emissions). Additionally, studies often only assessed N_2O emissions without considering the other relevant GHGs needed for the calculation of the net GHG budget (NGB, e.g. Felber et al., 2016) for the pasture systems. However, the climatic impact of different agricultural management practices can only be determined by calculating the NGB for such ecosystems.

The determination of the NGB of an agricultural ecosystem is rather complex. Beside the direct measurements of N_2O and CH_4, the carbon storage change must be quantified by the net ecosystem carbon budget (NECB) (Chapin et al., 2006; Rutledge et al., 2015). It is determined as the budget of all relevant carbon import and export processes. These typically include the net ecosystem exchange (NEE) of CO_2 with the atmosphere (incl. respiration and photosynthesis) and agricultural induced carbon fluxes like organic fertilizer input or harvest removal (Felber et al., 2016). The NGB of pasture systems have rarely been measured (e.g. Flechard et al., 2007; Jones et al., 2017; Soussana et al., 2007), but if measured different approaches with non-coherent spatial and temporal coverage were used for the different GHGs.

The NGB strongly depends on the applied management practice. In recent years, mitigation strategies to reduce the emissions of N_2O have been increasingly tested (Dijkstra et al., 2013; Voglmeier et al., 2019). Beside a reduction of the fertilizer N input, a N-optimized feeding strategy can lead to less N excreted by animals (Arriaga et al., 2010) and may consequently lead to less N_2O (Voglmeier et al., 2019) and ammonia emissions (Voglmeier et al., 2018). Such feeding strategies typically add forage with a low N content (e.g. maize) as a supplement to the N rich grass resulting in lower N in the excreta (mainly in the form of lower urine-N).

The gaseous emissions of pasture fields are typically quantified either with the eddy covariance (EC) method or with static chambers (Flechard et al., 2007). The EC method integrates over a larger domain and is ideal to measure field scale emissions and is commonly used to measure NEE. However, due to the area integration it lacks the ability to relate the measured fluxes to specific sources, which is especially important for N_2O and CH_4 on pastures (e.g. CH_4 emissions from dung patches, N_2O emission from urine patches). For such tasks, chamber measurements are optimal as they measure fluxes over a limited area (typically below 1 m^2) and can therefore be used to quantify the emissions of several source areas on a pasture system (e.g. dung / urine patches, background surfaces). The combined approach of EC and chamber measurements was already tested on grazing systems (Jones et al., 2011) and is regarded as the best approach to understand and quantify field scale as well as hot spot emissions of such grazing systems (Cowan et al., 2015).

The main goal of our experiment was to determine the full annual carbon and GHG budgets of two neighboring pasture soils. Moreover, we examined the influence of external feed supplements for the grazing cows on the GHG budget. This study builds upon previously published results for the same experiment on the grazing-related emissions of ammonia (Voglmeier et al., 2018) and N_2O (Voglmeier et al., 2019). They showed positive impacts of low-protein feed supplements to reduce excreta-related NH_3 and N_2O emissions. The GHG budgets in this study were mainly determined from field-scale EC flux measurements over an entire year. They were complemented with measurements of non-gaseous carbon fluxes. For the interpretation and source attribution of CH_4 and N_2O field-scale fluxes, small scale measurements by a 'fast-box' chamber of were temporarily performed on urine and dung patches.

2 Material and methods

2.1 Study site and experimental design

The experiment was conducted at the research farm Agroscope Posieux (642 a.s.l, 46°46´04´´N, 7°06´28´´E) in Switzerland in the canton of Fribourg. The study site was already described in detail in Voglmeier et al. (2018, 2019). Climate records show an annual average temperature of 8.7 °C and an annual rainfall of 1075 mm (MeteoSwiss, 2018). The vegetation consisted of a typical Swiss grass-clover mixture (dominated by 10 % to 50 % *Lolium perenne* and 7 % to 40 % *Trifolium repens,* value ranges represent the temporal variation during the grazing season) and the soil was classified as a stagnic Anthrosol with a loamy texture (about 20 % clay, 35 % silt and 45 % sand). The last renovation (ploughing and reseeding) of the field took place in 2007. Since then, the field was used as an intensive pasture for various livestock. Between 2007 and 2015 the pasture was fertilized with mineral fertilizer or animal manure in the order of 120 kg N ha^{-1} yr^{-1}, in addition to the excreta N by animals. During the experiment, 24 dairy cows grazed on a 5.5 ha pasture between mid of March and beginning of November. Of this area, 4.4 ha were reserved for the main experiment and 1.1 ha were optional areas when low grass growth conditions occurred on the studied pastures. The pasture was further divided into two separate neighbouring systems (Fig. 1). The diet of the 12 cows of each system differed in the energy to protein ratio. System G represented a full grazing regime with no additional forage. This diet resulted in a considerable N surplus mainly affecting the excreted urine N (Table 1). The cows of system M were, in addition to grazing, fed with maize silage (roughly 19 % of the animal dry matter intake DMI), which resulted in a demand optimized protein content in the diet (Arriaga et al., 2010; Yan et al., 2006). This feeding strategy resulted in about 14 % less N in the excreta.

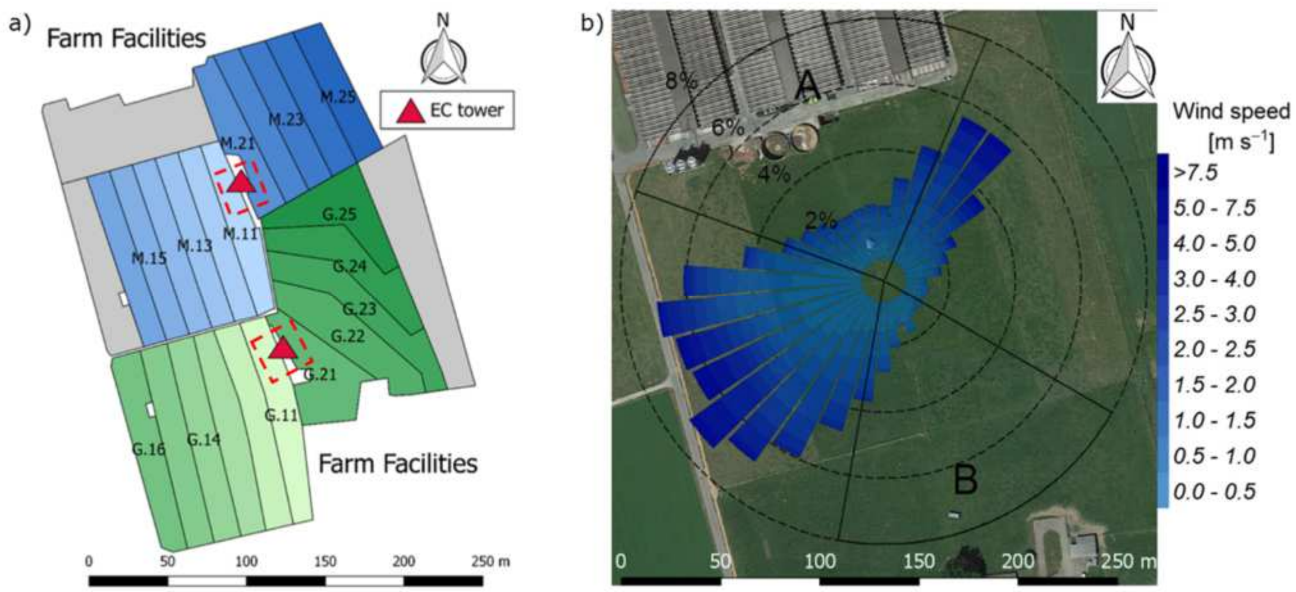

Figure 1: a): Measurement site with the pastures for the two herds (blue: grass diet with additional maize silage; green: grazing only diet; grey: optional pasture areas) and the division into the paddocks (M.11–M.16, M.21-M.25; G.11-G.16, G.21-G.25). The location of the two EC towers and the area of the chamber measurements (red dashed rectangles) are also shown. b) Distribution of wind speed and direction for the northern sonic anemometer for the period May – October 2016. The areas A and B indicate wind sectors from which advection from nearby farm building could occur. The wind distribution was overlaid on a Google Earth image of the experimental area (Map data: Google, DigitalGlobe) (from Voglmeier et al., 2019). The corresponding EC footprint climatology for the same pasture and experiment and for both fields is shown in Voglmeier et al. (2019).

Each pasture field was managed as a rotational grazing system with 11 paddocks. The size of the paddocks was adjusted to the additional fodder and resulted in paddock sizes of about 1700 m^2 in pasture field M and 2200 m^2 in pasture field G. In order to allow a comparison of the systems, the rotation on both pastures was managed synchronously (including similar barn times). Depending on the grass availability, a typical rotation period of about 20 days was obtained. Longer barn times occurred in June due to heavy rain events that prevented grazing on the pasture and in August/September due to high air temperatures and additional experiments of other research groups. Mainly in July/August dry weather conditions and subsequently low grass

growth required grazing on neighbouring pasture areas. Summarized, the cows spent approximately 86 full days on the main pasture area. Subtracting the time in the barn (e.g. for milking) resulted in about 65 days of effective pasture time.
In order to determine weather and soil conditions, the measurement site was equipped with an automated weather station (with data logger CR10X, Campbell Scientific Ltd., UK) at the northern field next to the EC system. A WXT520 (Vaisala, Vantaa, Finland) measured the wind speed, precipitation, temperature and barometric pressure, and global radiation was measured with a pyranometer (CNR1, Kipp&Zonen, Delft, Netherlands). Soil moisture and soil temperature were measured continuously with two repetitions on each pasture close to the EC towers with ML3 ThetaProbe (Delta-T Devices Ltd, UK) devices at depths of 5, 10, 20, and 40 cm.

2.2 Carbon budget concept

The NECB is a measure to assess if an ecosystem is a sink or source for atmospheric CO_2 by calculating the storage change of C in the soil and vegetation (Chapin et al., 2006; Rutledge et al., 2015). For agricultural ecosystems, C storage in plants is usually negligible on the annual or multi-annual scale due to the lack of continuous biomass accumulation. Also C losses due to leaching and erosion are assumed to be very small at the study site (Felber et al., 2016a). The calculation of the NECB of pastures depends on the choice of the system boundaries, i.e. cows can be either included or excluded. However, the two system boundary approaches for NECB should yield comparable results (Felber et al., 2016a). We chose the approach of excluding the cows from the system boundaries (Fig. 2) in order to make the results of this study easily comparable to other grassland sites where typically area-related concepts for the GHG budget are applied. Moreover, excluding the cows from the system boundary removes the need of detailed cow position information on the pasture for comparing the two fields and also makes an assessment of the cow emissions in the barn unnecessary. The NECB was thus calculated (Eq. 1) as the balance between animal related fluxes (excreta input and biomass removal through grazing), soil and vegetation related gaseous fluxes (NEE without cow respiration, CH_4 from / to soil) and external inputs / exports (fertilization, harvest). It has to be noted, that the soil CH_4 flux in the present concept also comprise the emission from deposited animal excreta (Felber et al., 2016a). Generally, throughout the manuscript the micrometeorological sign convention was applied, and thus negative values indicate an import (uptake) to the ecosystem.

$$\mathrm{NECB} = F_{\mathrm{C-CO2,past}} + F_{\mathrm{C-CH4,soil}} + F_{\mathrm{C-fertil}} + F_{\mathrm{C-grazing}} + F_{\mathrm{C-harvest}} + F_{\mathrm{C-excreta,past}} \qquad \text{(Eq. 1)}$$

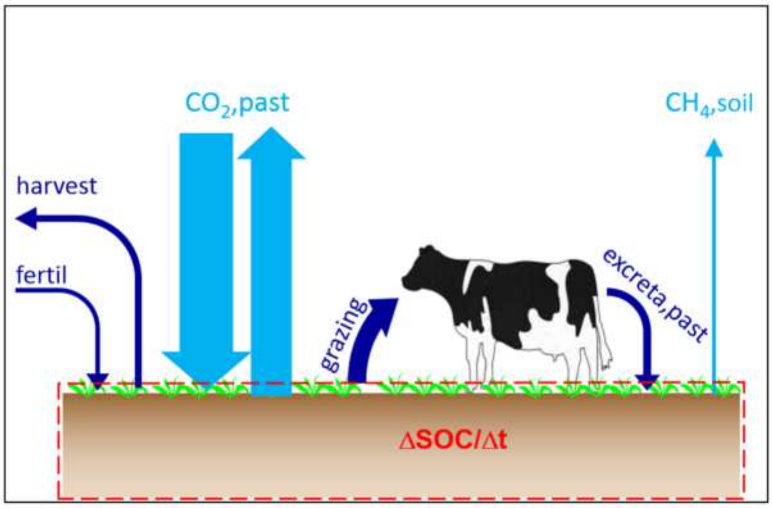

Figure 2: NECB approach with the cows outside the system boundaries (red box) and relevant carbon fluxes through the system boundaries (gaseous fluxes: light blue, liquid/solid fluxes: dark blue). The acronym 'past' refers to pasture as defined by the red box (without direct cow emissions), and thus CO_2, past is regarded as equivalent to NEE in this study. Figure was adapted from Felber et al. (2016a).

It has to be noted that the discussed C fluxes have different time periods when they contribute to the NECB. Soil and vegetation related fluxes (CO_2,past and CH_4,soil in Fig. 2) occur throughout the full year. Animal related fluxes occur only during the grazing days (Table 1, including the milking time in the barn) and the C import through excreta to the pasture is dependent on the effective time the cows spent on the pasture.

2.3 Area related fluxes

This section presents the methods used to quantify the components of the pasture GHG budgets and also covers the calculation of the uncertainty. The GHG budget (Sect. 2.6) was mainly determined from the field-scale EC flux measurements (Sect. 2.3.2) but also made use of a number of cow-related quantities (Sect. 2.4). For annual sums of the different fluxes, we estimated systematic and random errors. As the experimental setup was very similar on both pasture fields, we assumed that the systematic errors (e.g. due to limited fetch, high-frequency damping, low u_* related gaps and corresponding gap filling, etc.) were very similar for both fields. Thus, only the random errors were considered for comparing the annual budgets of the two pastures (Ammann et al., 2009). The combined error (random and systematic) is further on denoted as total uncertainty and represents the uncertainty of the individual systems and can be used to e.g. compare the results of this study with results from other sites.

2.3.1 Small scale flux measurements

For the interpretation and source attribution of the field-scale fluxes, small-scale fluxes of CH_4 and N_2O were temporarily measured on the pasture. In a first step, the intensive observation areas (red boxes in Fig. 1) were scanned for urine patches (using soil electric conductivity measurements; for details see Voglmeier et al., 2019) and dung patches as well as for excreta-free background areas shortly after grazing on the respective paddock. Small scale CH_4 and N_2O fluxes from these areas were continuously measured with the fast-box chamber (FB, Hensen et al., 2006) within a time period of 20 days after the end of grazing to capture temporal emission patterns. We used the same approach for flux calculation and flux quality control as described by Voglmeier et al. (2019), where the method (incl. excreta detection) and the FB itself is described in detail. In short, the light shielded 0.8 m x 0.8 m x 0.5 m box was connected via a 40 m tube to a fast-response quantum cascade laser spectrometer (QCL, QC-TILDAS, Aerodyne Research Inc.) that allowed the quantification of CH_4 and N_2O fluxes on the intensive observation areas of the two pastures. CO_2 was measured additionally inside the box with a GMP43 probe (Vaisala, FL). Concentrations of the various gases were recorded every three seconds for a time period of about 90 seconds and corresponding fluxes were calculated in a post-processing step from the concentration increase within the chamber headspace. The flux measurements were complemented with soil moisture, soil temperature, chamber temperature and soil electric conductivity measurements in order to relate the measured fluxes to potential driving parameters. In summary, 761 individual flux measurements remained after quality control. 391 (167 background, 135 urine, 94 dung) were taken on field M and 365 (169 background, 110 urine, 86 dung) on field G. These measurements were performed between beginning of July and end of October.

2.3.2 Field scale GHG flux measurements and annual integration

The field scale fluxes of CO_2, CH_4 and N_2O were determined using the EC technique. Two identical measurement systems were placed in the centre of the two pastures (see Fig. 1). Dry air mixing ratios of CH_4 and N_2O were measured with a closed path QCL system that analyzed air samples drawn through a 25 m PA tube (inner diameter 6 mm) by a vacuum pump (Bluffton Motor Works, flow rate ca. 13 l min^{-1}). This QCL system was also used for FB measurements (see Sect. 2.3.1). Thus, during time periods of FB measurements (up to 3 h) no flux measurement of CH_4 and N_2O with the EC technique was possible on the respective field. CO_2 concentrations were measured with an open-path CO_2 and water vapor (H_2O) gas analyser (LI-7500, LI-COR Inc., US). The EC data were sampled with a temporal resolution of 10 Hz. Flux calculation, quality control and flux selection procedures for N_2O as well as the footprint calculation with a backward Lagrangian stochastic dispersion model (bLS, Häni, 2017; Häni et al., 2018) are discussed in detail in Voglmeier et al. (2019). CH_4 and CO_2 flux calculations followed similar quality criteria, except that the threshold of the friction velocity (u_*) filter was set to 0.07 m s^{-1} for N_2O and CH_4, while it was set to a more restrictive value of 0.1 m s^{-1} for CO_2 (see Felber et al., 2016b). Those u_* thresholds were kept constant throughout the whole year.

The u_* filter removing time periods with low turbulence is particularly important for CO_2 fluxes to determine a reasonable annual NEE. In order to avoid a bias due to differing gaps in the CO_2 flux time series, all quality criteria had to be fulfilled for both pasture systems synchronously. Quality controlled CO_2 fluxes were additionally restricted to periods without cows in the footprint. We also rejected fluxes with a wind direction from sectors A and B in Fig. 1 in order to avoid a potential influence of the nearby farm facilities on the CO_2 flux. After this rigorous filtering, about 28 % of the data were kept for the NEE determination. Of these, about two thirds were measured during day time conditions (global radiation > 10 W m^{-2}). During the night, many EC measurements were discarded due to the typical calm nights in western Switzerland with low u_* values. Annual integrated NEE values were calculated following the pasture approach by Felber et al. (2016a), including gap filling by the downloadable R version of the REddyProc gapfilling tool (Wutzler et al., 2018). The EC flux footprint usually extended over several paddocks in different grass growth stage due to the rotational grazing. But since the individual grazing phases were very short (1-2 days), the grass height after grazing was not very low in order to prevent shortage of feed and to allow a fast regrowth. For the same site, Felber et al. (2016b) showed that the variation in the assimilation capacity (GPP with global radiation > 500 Wm^{-2}) due to grazing was low (in contrast to observations of Wall et al., 2019), while the effect of environmental driving factors was more pronounced. Thus the flux gap filling across individual paddocks did not seem problematic.

The total uncertainty of the annual NEE was calculated from three different contributions. Systematic uncertainties due to u_* filtering were quantified by varying the u_* threshold between 0.04 m s^{-1} and 0.16 m s^{-1} to simulate different gaps in the time series which resulted in an uncertainty of 22 g C m^2 yr^{-1} and 15 g C m^2 yr^{-1} for pasture fields M and G. Further systematic uncertainties due to gaps in general in the flux time series before gap filling (sampling uncertainty, 40 g C m^{-2} yr^{-1} for both fields), and the random uncertainty for both fields (14 g C m^{-2} yr^{-1}) were directly taken from Felber et al. (2016b) who quantified them for the same pasture in 2013 with a similar setup. The total field-specific uncertainty was finally calculated by combining the three uncertainties using Gaussian error propagation. For comparing the two neighbouring pastures, the random errors are more important because systematic uncertainties were assumed to be similar for both fields.

The field scale soil CH_4 fluxes were processed in a comparable way by only using fluxes without cows in the footprint, but in contrast to CO_2 and N_2O fluxes (Voglmeier et al., 2019), cases without a detectable peak in the cross-covariance function (i.e., flux below the detection limit) were also allowed, because they were expected to occur frequently for CH_4. This led to a data coverage of about 28 % and 20 % for fields M and G. A delayed installation of the QCL in field G, which was not operational before the beginning of April, led to the lower data coverage in field G. We assumed that the CH_4 and N_2O fluxes during the cold weather period before the installation of the QCL were identical on both fields. Thus, we gap filled the missing N_2O and CH_4 data of field G during this time period with data from field M. The absence of intensive grazing and fertilization before April justified this gap filling method as only small fluxes are expected for such conditions. To our knowledge no established gap filling method for pasture CH_4 fluxes exists and we also could not attribute the flux variations to certain environmental driving parameters (soil temperature/moisture, relative humidity, precipitation). Thus, we calculated the annual CH_4 emissions by using the medians of a moving 3-day time period, where at least 20 valid half-hourly measurements had to be available in each 3-day window to prevent outliers from dominating the annual emissions. Remaining gaps after the usage of the moving median filter were filled by using linear interpolation between available data points. The systematic uncertainty of the annual estimate was assessed by calculating the SE of varying annual estimates. These additional annual estimates were computed by varying the length of the used time period (between 1 and 7 days) and the number of required valid measurements (between 5 and 40). The random uncertainty of the annual emissions for both pastures was assessed by calculating the SE of the differences between the 3-day medians of the two fields, excluding the time period before the installation of the QCL in field G and the time period in June where the measurements between the two fields showed a larger systematic difference (Sect. 3.1). Using this method to calculate the random uncertainty was possible because the fluxes on the two fields were very similar for the remaining time period, and because the experimental setup (e.g. management, weather, and turbulence) was very identical.

The annual cumulative N_2O emissions for the two pastures were computed from gap filled EC flux time series. In order to gap fill the N_2O time series, we applied a lookup table (LUT) for time periods during grazing and without influence of fertilization events. The LUT is based on the preceding cumulative rainfall in the last 12 h, the soil temperature and on the footprint weighted averaged cow density on the single paddocks over the preceding 5 days (Voglmeier et al., 2019). The remaining time periods were gap filled using a running mean with a default window size of 6 hours, where the window size was increased if less than two valid measurements were available in the window. The delayed installation of the QCL in field G resulted in a additional uncertainty for annual cumulative N_2O emission of the pasture field G. The additional uncertainty was derived from the standard deviation of the cumulative fluxes during time periods where both fields reported measured fluxes (excluding fertilization periods). This is a rather conservative uncertainty estimation because fluxes during grazing periods are expected to behave more variably in comparison to cold season fluxes in the first part of the year. The total 2 SE uncertainty of the annual N_2O emission sums was finally calculated by combining the uncertainty from this gap-filling approach (only relevant for field G) with the systematic uncertainty for both fields calculated by Voglmeier et al. (2019) for the same experiment. This resulted in relative uncertainty values of 28% and 38% for pasture fields M and G, respectively. The random uncertainty for both fields was calculated from the inter-monthly variation of the cumulative fluxes between the two EC systems, as the annual cumulative emissions were by chance of similar magnitude (Sect. 3.3), and resulted in a random error of about 3 % for the annual sums.

The detection limits for the N_2O and CH_4 fluxes was derived from the variation (3 times the standard deviation) of the left and right side of the covariance functions of the half-hourly fluxes following Felber et al. (2016b), and resulted in lowest possible (5 % quantile) flux detection limits of 0.05 nmol m^{-2} s^{-1} and 0.8 nmol m^{-2} s^{-1} for N_2O and CH_4, respectively. Including environmental effects (e.g. instationarity) resulted in flux detection limits (median) of 0.15 nmol m^{-2} s^{-1} and 4.8 nmol m^{-2} s^{-1} for N_2O and CH_4.

2.3.3 Fertilizer applications and harvest

In 2016, the pasture was fertilized with ammonium nitrate (28 kg ha^{-1}, end of June), urea (42 kg N ha^{-1}, mid of August in plots X.11–X.16 and beginning of September in plots X.21–X.25, X indicating both pasture fields in Fig. 1) and cattle slurry from the nearby farm (45 m^3 ha^{-1}, 57 ± 6 kg N ha^{-1}, end of November). The C import by the slurry was determined as 460 kg C ha^{-1} with an associated uncertainty of 17 % following Ammann et al. (2009). The C import due to the urea fertilization was calculated from the C/N ratio in urea resulting in a comparably small value of 18 kg ha^{-1}.

Heavy rain events in June led to a break in the rotational grazing management and necessitated a grass cut on the entire pasture area between 22 and 27 June. The exported C content of the removed biomass was calculated by weighing the dried grass on transport trailers in combination with an analysis of the dry matter content of the grass. This resulted in a C export ($F_{C\text{-harvest}}$ in Eq. 1) of 693 kg C ha^{-1} and 719 kg C ha^{-1} for the pasture fields M and G, respectively. The associated uncertainty of 10 % is based on a study by Ammann et al. (2007) with a similar technique.

2.4 Animal C and N balance and grazing related fluxes

The animal related C fluxes $F_{C\text{-excreta,past}}$ and $F_{C\text{-grazing}}$ are directly related to the DMI of the cows. As DMI could not be measured directly on the pasture, we used a coupled C / N budget model for Swiss dairy cows which has been described already in Voglmeier et al. (2018). This model calculated the DMI (Table 1) of all cows in the experiment based on their daily measured energy corrected milk yield, the body weight gain and the lactation period. The C content of the grass was measured periodically (weekly to biweekly) on both pastures and resulted in 425 ± 21 g C (kg^{-1} DM) (2σ) and 425 ± 26 g C (kg^{-1} DM) for fields M and G, respectively. The maize supplement was measured three times during the grazing season and resulted in a C content of 430 ± 9 g C (kg^{-1} DM). Together with the calculated DMI, the diet and the number of full grazing days, $F_{C\text{-grazing}}$ was calculated as 358 ± 67 g C m^{-2} yr^{-1} and 337 ± 50 g C m^{-2} yr^{-1} for fields M and G, respectively. The excreta N was calculated by closing the N budget of the cows based on the measured N in the feed and in the milk, and the N storage due to body weight gain. The separation into urine and dung N were finally estimated using relationships from a synthesis by Bracher et al. (2011). Based on Voglmeier et al. (2018), an associated total uncertainty of 15 % was assumed for the DMI and the N calculations.

Table 1: Properties of the two pasture systems: average ± standard deviation of observed cow characteristics, feed protein content, and resulting DMI as well as urine N and dung N calculated by the Swiss dairy cow budget model for both systems. Values are calculated for the full grazing days. (ECM: energy corrected milk yield; DMI: dry matter intake)

Parameter (units)	System M	System G
Total grazing area (ha)	1.9	2.5
Number of cows	12	12
Full grazing days (d)	85	87
Effective grazing time (d)	64	66
Milk yield, ECM (kg cow^{-1} d^{-1})	25.1 ± 2.9	24.2 ± 3.7
Animal weight (kg)	633 ± 14	634 ± 10

DMI (kg cow^{-1} d^{-1})	19.2 ± 0.9	19.0 ± 1.3
Grazing intake (kg C cow^{-1} d^{-1})	6.6 ± 1.2	8.1 ± 1.2
Maize intake (kg C cow^{-1} d^{-1})	1.6 ± 0.2	
Urine (g N cow^{-1} d^{-1})	250 ± 54	314 ± 71
Dung (g N cow^{-1} d^{-1})	154 ± 9	155 ± 12

The C excretion rate of grazing cows is very difficult to measure directly, therefore it was estimated based on the digestibility of the forage and the calculated animal intake. The average organic matter digestibility of the grass was determined to be 72 % with an uncertainty of 10 %, based on the method by Tilley and Terry (1963). The maize digestibility was estimated as 75 % based on data by Baux (2013) who analysed 20 years of data from maize silage in Switzerland. However, Pettygrove et al. (2010) found that the typical organic matter content in dried excreted dung was around 50 %, which is higher compared to the roughly 43 % C content found in our analysis of the grass and maize samples. Therefore, the effective carbon digestibility of the feed was calculated as 67 % and 66 % and accordingly the animal C excretion rate as 33 ± 8 % and 34 ± 8 % of the respective DMI for systems M and G. $F_{C\text{-}excreta,\ past}$ was finally calculated from the animal excretion rate and the effective grazing time on the pasture (Table 1).

2.5 CH_4 and N_2O flux partitioning

Field scale CH_4 emissions of the soil, as measured by EC without cows in the footprint, were expected to result from combined emissions of excreta patches and exchange of background pasture surfaces. In order to disentangle these fluxes and estimate the contribution of the individual sources, the small-scale fluxes measured by the FB were used and up-scaled to the paddock and field scale and integrated for the time when FB measurements were taken (July-October). The up-scaling routine was based on Voglmeier et al. (2019) but adapted to the needs for CH_4 emissions. This allowed a comparison of the cumulative FB-derived emissions with the cumulative emissions using the EC measurements.

According to the IPCC methodology for N_2O emissions (IPCC, 2006), the annual N_2O emissions from grazed pastures are mainly a combination of fluxes resulting from N input by fertilization, excreta of grazing animals, and background emissions (from atmospheric deposition and plant residues). Emissions resulting from N fixation through clover are not considered in the IPCC methodology. In order to separate the measured N_2O emissions into the different emission sources, we applied the following approaches:

1. Background emissions were measured with the EC technique during time periods that were not influenced by either fertilization (see 2.) or grazing (e.g. in winter). Background emissions during the remaining time periods were calculated by using the background parametrization by Voglmeier et al. (2019) for the same experiment. This parametrization was derived by fast-box measurements during the summer months and only depends on soil moisture.

2. Slurry and mineral fertilizer emissions were quantified with the EC method during respective influenced time periods (0-15 d after mineral fertilizer application, 0-25 d after organic fertilizer application). Background emissions (see 1.) and excreta emissions from concurrent grazing events were subtracted from the EC measurements. The excreta emissions were quantified using already calculated, footprint-weighted excreta emissions by Voglmeier et al. (2019) for the same experiment.

3. Excreta emissions were quantified with the EC method during time periods with grazing but without influence of fertilizer (see 2.). Background emissions using the background-parametrization (see 1.) were subtracted from the EC measurements. During time periods with fertilizer influence (see 2.), the excreta emissions were estimated using the parametrization already used in 2.

2.6 Greenhouse gas budget

The calculation of the NGB of the pasture was performed with the same system boundary concept as the NECB (i.e. excluding the cows, see Fig. 2) based on the field scale measurements. For this purpose, the annual NECB, soil CH_4 flux and the N_2O emission were converted to CO_2 equivalents using the 100-year global warming potentials including climate-carbon feedbacks (GWP, γ) of the individual gases following IPCC (2014): γ_{CH4} = 34 g CO_2-eq. g^{-1} and γ_{N2O} = 298 g CO_2–eq. g^{-1}. To avoid double counting in the calculation of the NGB, the contribution of the soil CH_4 emission to the NECB was subtracted according to Eq. 2 where m_x indicates the molar or atomic mass of the respective molecule.

$$\mathrm{NGB} = \frac{m_{\mathrm{CO2}}}{m_{\mathrm{C}}}\left[\mathrm{NECB} - \frac{m_{\mathrm{C}}}{m_{\mathrm{CH4}}}F_{\mathrm{CH4}}\right] + \gamma_{\mathrm{N2O}}F_{\mathrm{N2O}} + \gamma_{\mathrm{CH4}}F_{\mathrm{CH4}} \qquad \text{(Eq. 2)}$$

3 Results and discussion

3.1 Methane emissions

The EC measurements yielded predominantly positive half-hourly CH4 fluxes. Negative fluxes were also found (about 22 %), however, almost all of these fluxes were associated with the application of the default lag time and were below the detection limit. The 3-day medians of the CH_4 fluxes were mostly in the range of 0 – 7 nmol m^{-2} s^{-1}, however, during a very wet period in June and beginning of July, when the soil was partly water-logged, the fluxes increased strongly (Fig. 3a). The cumulated annual CH_4 emissions were 1.3 ± 0.6 g CH_4-C m^{-2} yr^{-1} and 1.1 ± 0.6 g CH_4-C m^{-2} yr^{-1} for the pasture soil M and G, respectively (Fig. 3b). Both pasture soils were therefore a source of CH_4.

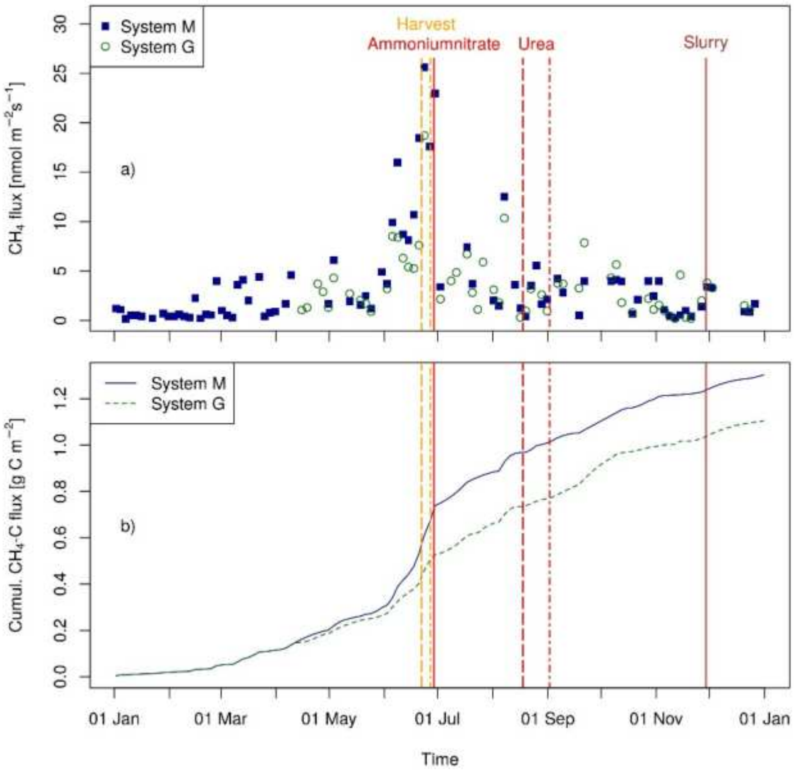

Figure 3: Time series of a) 3-day median CH_4 fluxes and b) cumulative CH_4 emissions for pasture soils M and G in 2016. The CH_4 analyser on field G was not operational before beginning of April, and thus the measured values of field M were used for the cumulative emissions of the pasture soil G. The vertical lines represent important management-related activities. Broken vertical lines indicate that harvest and urea application were split for the western (dashed line, X.11–X.16) and eastern (dash-dotted line, X.21–X.25) part of the pasture fields.

EC CH_4 fluxes are not ideal to estimate the individual emission contributions of the excreta and background surfaces. For this purpose, the FB measurements were used. Because no statistical differences between the patch-scale CH_4 fluxes of the pasture soils M and G were found ($p > 0.68$, 0.59, 0.17 for background, dung, and urine, respectively) they were combined for further analysis. We found that the background and urine fluxes were not statistically different from zero ($p > 0.37$ for both classes) and showed no temporal trend (Fig. 4). On the other hand, dung patch emissions showed a clear decay with excreta age. Highest emissions were typically measured directly after dung application on the pasture and they ceased after about 20 d, which is slightly longer than found by Holter (1997) who applied one kg of fresh dung and found periods with significant emissions of 10-18 days. The decay of the CH_4 dung emissions could be approximated with an exponential function (Eq. 3; $p < 0.001$; $a = 0.0023$ g C m^{-2} h^{-1}; Δt_{exc} excreta age in days) and resulted in an integrated CH_4 emission of about 0.3 g CH_4-C $patch^{-1}$.

$$F_{C,D} = a \cdot exp^{-0.192 \cdot \Delta t_{exr}} \qquad \text{(Eq. 3)}$$

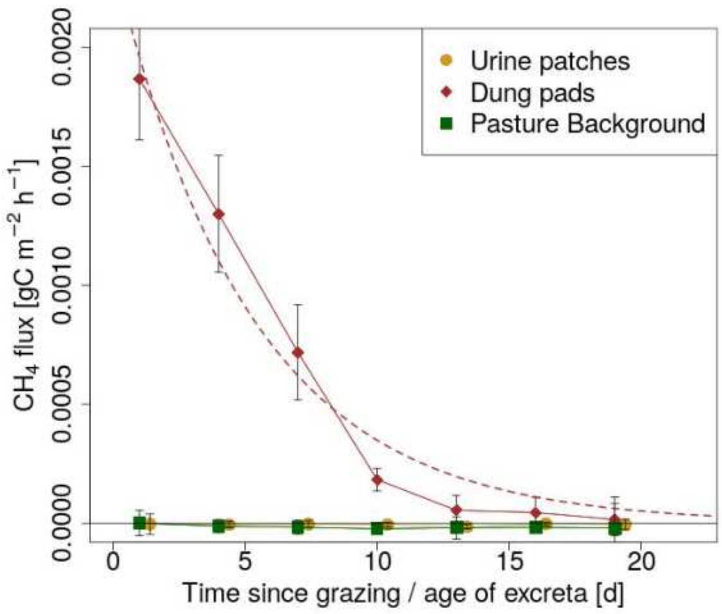

Figure 4: CH_4 fluxes measured with FB and plotted versus excreta age for urine and dung patches as well as pasture background areas. The fluxes were averaged over periods of three days and plotted together with the respective standard error. The dashed line shows the fitted curve for dung patch fluxes. The fluxes from urine patches were shifted slightly to the right for better readability.

To assess the quality and representativeness of the FB measurements, we up-scaled them to the paddock and field scale and compared the cumulative sum to the corresponding EC derived soil CH_4 emission for the time period were FB measurements were performed (beginning of July until end of October). Due to the previously described non-significant FB urine and background flux, we used the integrated dung emission in combination with the calculated number of dung patches (see Sect. 2.5.1, about 12.5 patches d^{-1} cow^{-1}) and the effective time on the pasture area to retrieve FB based paddock and field scale dung CH_4 emissions (see also comparable approach for N_2O emissions and urine patches in Voglmeier et al., 2019). In summary, the calculated cumulative FB emissions were 85 mg CH_4-C m^{-2} and 71 mg CH_4-C m^{-2} whereas the EC emissions accumulated to 459 ± 211 mg CH_4-C m^{-2} and 467 ± 257 mg CH_4-C m^{-2} for the pasture soils M and G, respectively. Unfortunately, we did not find a conclusive explanation for the large deviation between the FB and EC results. A bias due to environmental influences on the up-scaling procedure was analysed by relating the measured EC and FB fluxes to potential driving soil (soil moisture / temperature) and atmospheric (global radiation, air temperature) parameters. However, no clear relation was found unlike in other studies (Imer et al., 2013; Kroon et al., 2010). Yet, Hörtnagl and Wohlfahrt (2014) also did not find a relationship between EC measured soil CH_4 fluxes and soil parameters at a grassland site in Austria. The CH_4 emission per dung patch in our study was rather at the upper end of reported emissions (Flessa et al., 1996: 0.2 g CH_4-C $patch^{-1}$; Tully et al., 2017: 30-70 mg CH_4-C m^{-2}) and the calculated number of dung patches per cow (12.5) was also reasonable (Villettaz Robichaud et al., 2011, 9.8 - 15.4 defecations cow^{-1} d^{-1}). We assume that undetected soil moisture variations on the pasture might have led to larger CH_4 background fluxes (which were assumed to be zero for the up-scaling routine) and subsequently might have led (at least partly) to the difference.

While we found significant CH_4 emissions by both measurement approaches, chamber studies by other authors often showed relatively low or negative emissions, e.g. Skiba et al. (2013) found fluxes close to zero (4.5 µg CH_4-C m^{-2} h^{-1}) on an intensively sheep-grazed pasture in a temperate maritime climate with static chambers, however, without directly targeting dung patch emissions (chambers were moved every fortnight to allow free grazing). Imer et al. (2013) measured CH_4 emissions with static chambers on three managed grassland sites (two of them extensively grazed) in Switzerland and found typically CH_4 uptake at all three sites, but again without directly targeting dung patch emissions. In incubation experiments of soil samples from grassland sites in Hungary, United Kingdom and Germany, Schaufler et al. (2010) found flux rates of 0.5 ± 0.8 µg CH_4-C m^{-2} h^{-1}. While mown or extensively grazed grasslands can either be a small sink or source (Schaufler et al., 2010), the enhanced CH_4 emission of intensively grazed pastures may be explained by the effect of dung patch emissions and thus micrometeorological measurements seem better suited to quantify the field-scale CH_4 emissions. Thus, we used the EC derived CH_4 emissions (Fig. 3b) for the annual NECB (Sect. 3.2) and NGB (Sect. 3.4) calculation. The annual integrated EC CH_4 emissions in this study were very similar to the results by Felber et al. (2015) obtained previously at the same site with a different EC gas analyser. They were also in the same order of magnitude but somewhat lower compared to the results of Laubach et al. (2016) who measured soil CH_4 emissions of an irrigated, fertilised and rotationally grazed site in New Zealand with micrometeorological techniques (also excluding cows) and found emissions of about 3.4 g CH_4-C m^{-2} yr^{-1}.

3.2 Carbon budget

The carbon budget as described in Sect. 2.2 combines fluxes from measured and inferred components (Fig. 5, 6). As shown in Fig. 5, carbon is imported to the pasture by net CO_2 uptake of the growing grass (see NEE curve) until the harvest event in June. From July onwards, the grazing export dominated the NECB as the NEE stayed fairly constant further on, indicating that on average the grass growth was compensated by respiration processes. In both systems, the C uptake by the pasture was mainly balanced by harvest and grazing related fluxes (C import from excreta, export through grazing). The slurry application at the end of November resulted in a significant C import that contributed to the negative NECB results. The important role of slurry applications on the C budget has already been recognized in other studies (Ammann et al., 2007; Soussana et al., 2007). In summary, the NECB was -103 ± 74 g C m^{-2} yr^{-1} and -63 ± 67 g C m^{-2} yr^{-1} for pasture fields M and G, respectively (see also Fig. 6). For comparing the two pastures, only the random part of the measurement uncertainty has to be considered (field M: ±16.4 g C m^{-2} yr^{-1}, field G: ±15.9 g C m^{-2} yr^{-1}) which resulted in a significant difference of the NECB between the two pasture soils (40 ± 23 g C m^{-2} yr^{-1}). It has to be noted that potential (unknown) systematic effects, e.g. differences in the long-term history of the two pasture fields, may add additional uncertainty for the comparison. While the contributions to NECB are dominated by the NEE and grazing export (Fig. 6), also the higher excreta input density from the cows in pasture field M (due to the supplemental maize silage fed in the barn and the smaller pasture area) may have contributed to the stronger C sink. This effect is also visible in Fig. 6 and supported by the respective uncertainties, as the annual excreta imports (per area) exhibit the statistically most significant difference between the two fields (bold error bars). Thus, the study indicates that feed supplements from outside the grazed pastures can lead to an increased C import to the pasture through the excreted C, even though the absolute magnitude of the effect is rather small. This excreta effect was also found by other authors (e.g. Kirschbaum et al., 2017), however Wall et al. (2019) reported that supplementary feed does not always lead to higher C sequestration in an intensively grazed temperate grassland in New Zealand.

The calculated NECBs in this study are very similar to the average net C storage of -104 ± 73 g C m^{-2} yr^{-1} found by Soussana et al. (2007) for nine European grassland sites or the 3-year averaged NECB of -71 ± 77 g C m^{-2} yr^{-1} found by Wall et al. (2019) in New Zealand. Yet, our study only covered the results of a one-year campaign. Thus, they are not necessarily representative for the long-term behavior of the experimental site. Indeed, Felber et al. (2016a) found a NECB around zero at the same pasture in 2013.

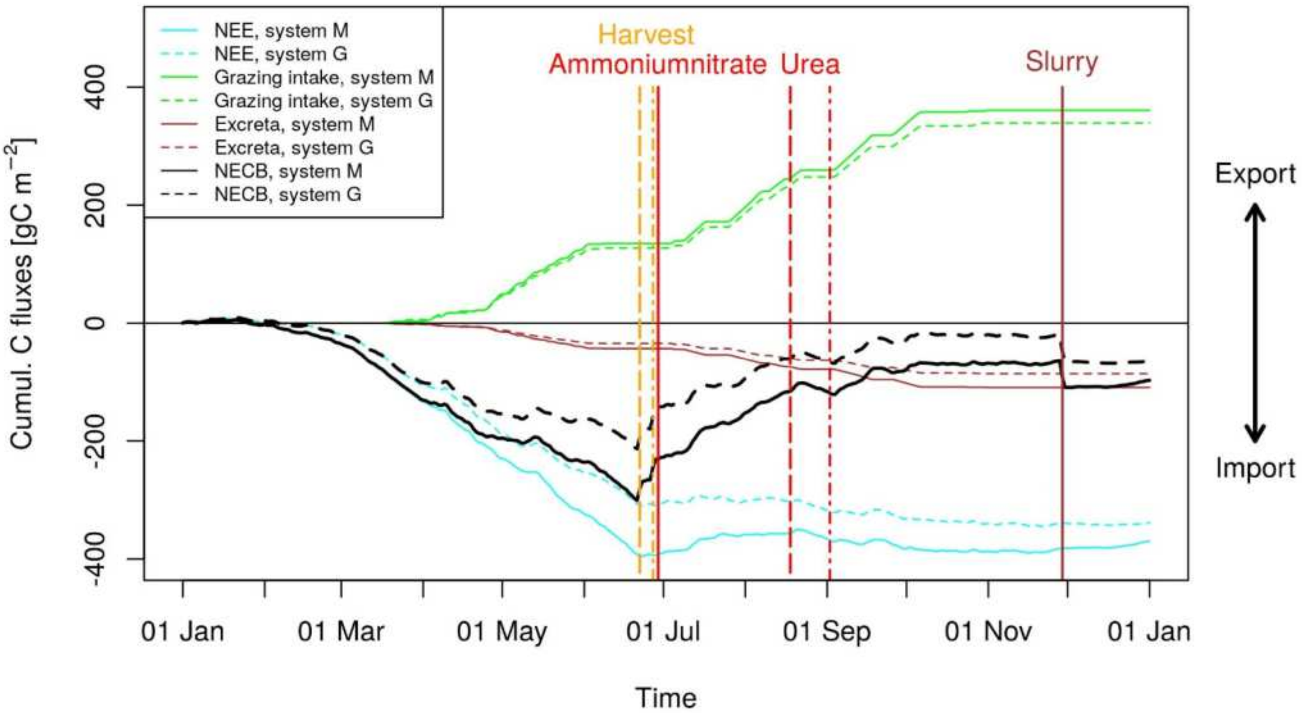

Figure 5: Time series of individual C fluxes contributing to the NECB for both pastures (solid lines: field M, dashed lines: field G). The vertical lines represent important management-related activities. Broken vertical lines indicate that harvest and urea application were split for the western (dashed line, X.11–X.16) and eastern (dash-dotted line, X.21–X.25) part of the pasture fields. Corresponding uncertainties are shown in Fig. 6.

The highest uncertainties were associated with grazing (Sect. 2.4.1) and the net CO_2 exchange of the pasture (Sect. 2.3.2). The grazing related uncertainty mainly reflects the uncertainty of the modeled DMI of the cows. A reduction of this uncertainty is hard to achieve, as direct high-quality measurements of the intake from the pasture are very difficult to obtain. The uncertainties of the CO_2 exchange mainly resulted from the gaps (e.g. due to u_* filtering) in the EC flux time series which required gap filling in order to calculate the annual NEE. These uncertainties are specific to the region of the experimental site (mainly due to low wind conditions) and hard to reduce.

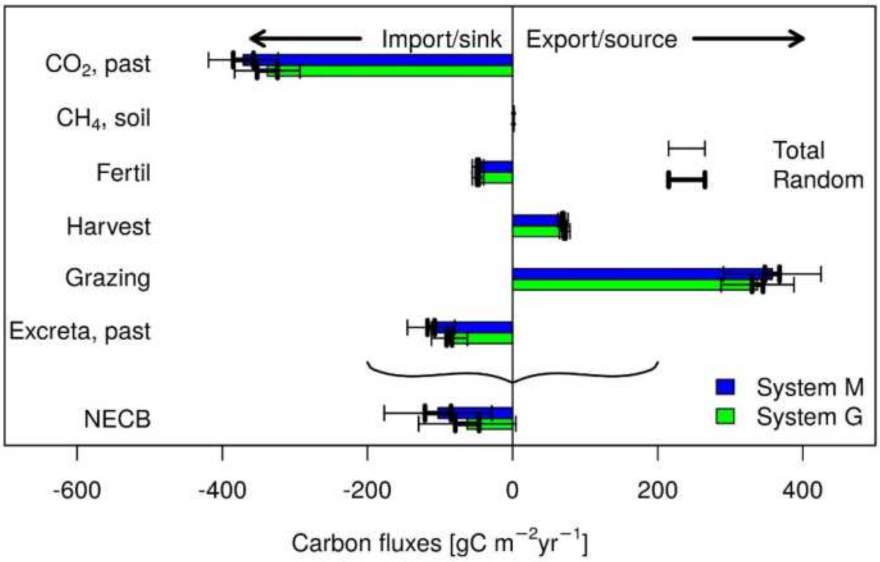

Figure 6: Net ecosystem carbon budget (NECB) and the contributing components for both pasture fields. The error bars represent the 95 % confidence range. The micrometeorological sign convention is used with negative values indicating imports to the field. 'Excreta, past' and 'Grazing' were modelled based on cow performance measurements (see Sect. 2.4), whereas the remaining values were derived from direct measurements. The acronym 'past' refers to pasture without direct emissions from the cows. Thin error bars represent the total absolute uncertainties (systematic and random) while the bold error bars indicate only the random uncertainties relevant for the field comparison.

3.3 Nitrous oxide emissions

The N_2O fluxes showed a strong temporal pattern (Fig. 7a) and were typically driven either by environmental or by management related parameters. The overall highest emissions were found after the fertilization with ammonium nitrate at the end of June, which took place directly after the harvest. It resulted in fluxes up to 29.0 g N_2O-N ha^{-1} h^{-1} and 24.3 g N_2O-N ha^{-1} h^{-1} for pasture fields M and G, respectively, and the main emission peak lasted around four days. The harvest event itself produced a small peak in emissions as well. Post-harvest N_2O emissions have likewise been observed by Rafique et al. (2012), and may be explained by increased rhizodeposition (Fuchs et al., 2018). We therefore assume that the high emission in June and beginning of July resulted from a combined effect of harvest, fertilization and grazing. A significant long-lasting emission peak was also observable after the fertilization with cattle slurry at the end of November. The low temperatures (around zero °C) might have led to a slower processing of the ammonium in the slurry which resulted in longer lasting emissions. Higher emissions due to freeze-thaw cycles are unlikely as the measured soil temperature at 2 cm depth always stayed above zero degrees during this period. The highest N_2O emissions during the grazing season (periods indicated in light green in Fig. 7a) were usually observed shortly (a few days) after a precipitation event in combination with concurrent grazing, when soil moisture measured at a depth of 5 cm increased from intermediate to high values (e.g. from VWC values around 0.35 to values above 0.45, peaks in May and beginning of August). However, during the very wet soil conditions beginning of June until end of June, when the soil was partly water-logged, N_2O fluxes were very low. Averaged over the whole grazing period, N_2O fluxes also increased with increasing soil temperatures and showed highest fluxes for soil temperatures (5 cm depth) above 15 °C. The grazing related emissions have already been discussed in more detail by Voglmeier et al. (2019), who also divided the emissions in urine and dung related contributions. They showed, that the emissions during the grazing season (excluding the fertilizer influenced time periods) were dominated by emissions from urine patches (about 57 %).

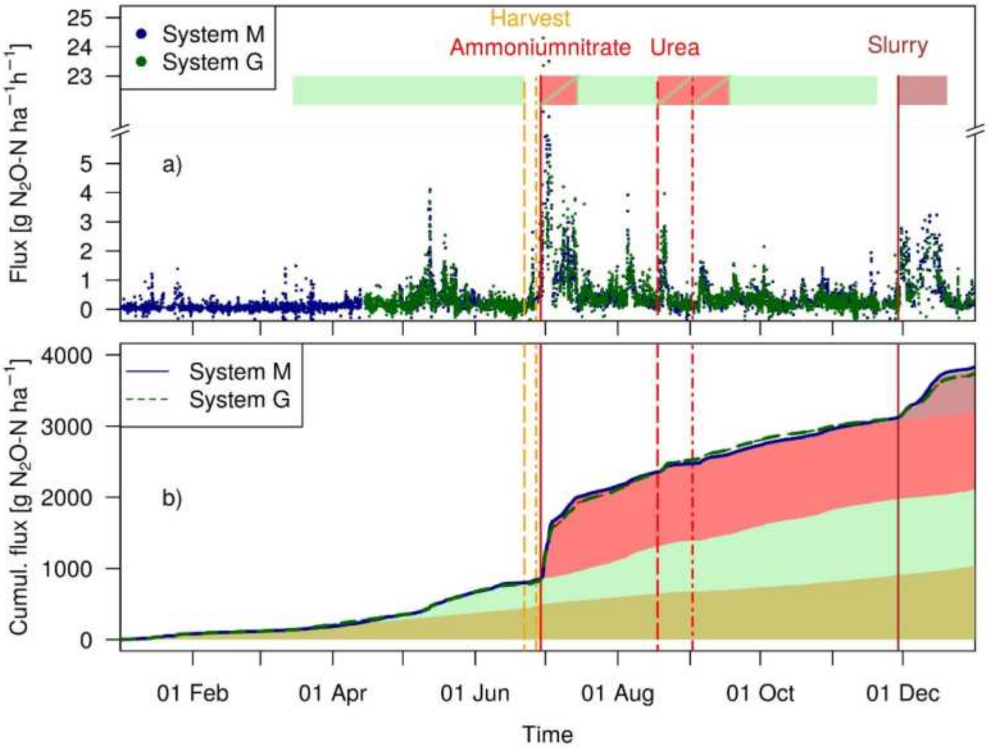

Figure 7: Time series of a) N_2O EC fluxes measured on field M (grazing with maize silage supplement) and G (grazing only diet). One value (29.0 g N_2O-N ha^{-1} h^{-1} on field M) not shown for better readability. The vertical lines represent important management-related activities. Broken vertical lines indicate that harvest and urea application were split for the western (dashed line, X.11–X.16) and eastern (dash-dotted line, X.21–X.25) part of the pasture fields. The shaded areas indicate time periods where N_2O emissions responded predominantly to specific management activities (green: grazing; red: mineral fertilizer application; brown: slurry application). The diagonal green lines in the red bars during the fertilizer applications in the summer month indicate a combined effect of significant emissions from grazing and fertilization. Due to a delayed installation of the QCL on field G no data before April were available. b) Cumulative emissions of the gap filled EC data of pasture fields M and G. As the N_2O analyser on field G was not operational before beginning of April, measured values of field M were used for the cumulative emissions of the pasture soil G. The stacked plot shows the emission contribution from background (olive), excreta (green), mineral fertilization (red) and slurry application (brown).

The cumulative emissions over the year were computed from the gap filled EC time series (see Sect. 2.3.2) and resulted in 3.8 ± 1.1 kg N_2O-N ha^{-1} yr^{-1} and 3.7 ± 1.4 kg N_2O-N ha^{-1} yr^{-1} for pasture fields M and G, respectively (Fig. 7b). This is lower than but not significantly different from the modelled estimate of 4.7 kg N_2O-N ha^{-1} (CI: 3.3 kg N_2O-N ha^{-1} – 9.4 kg N_2O-N ha^{-1}) by Felber et al. (2016a) for the same site and similar N input.

In order to evaluate the effect of the individual N_2O emission sources (Table 2), we partitioned the total emission according to the IPCC guidelines (see Sect. 2.5). Time periods with the main emission source attributed to fertiliser applications or grazing excreta are highlighted in Fig. 7a. In summary, the excreta related emissions contributed to about 28 ± 18 % and slurry and mineral fertilizer induced emission to about 44 ± 17 % of the total N_2O emissions. The remaining 28 ± 19 % of the emissions are regarded as background. These background emissions might be attributed to N imports by atmospheric deposition (about 25 kg ha^{-1} yr^{-1} in that area according to Purghart et al., 2014) and decomposing plant residues after harvest or grazing. In agroecosystems, the emissions can also originate from fertilization or grazing management effects in e.g. previous years (Bhandral et al., 2007; Bouwman, 1996; Gu et al., 2009).

Table 2: Cumulative annual N_2O emissions and partitioning into main emission sources (field M / field G*)* during the experimental year 2016. Uncertainty is given as 95 % confidence interval.

Source attribution	Emission field M (kg N_2O-N ha^{-1} yr^{-1})	Emission field G (kg N_2O-N ha^{-1} yr^{-1})
Background	1.04 ± 0.66	1.05 ± 0.69
Grazing excreta	1.07 ± 0.61	1.05 ± 0.69
Mineral fertilization	1.11 ± 0.41	1.09 ± 0.51
Slurry application	0.57 ± 0.18	0.47 ± 0.20
Total sum	3.8 ± 1.1	3.7 ± 1.4

This simple time related partitioning concept is not able to attribute the measured emissions to other management related N sources like NH_3 re-deposition from neighbouring pastures or from the nearby farm buildings. Nevertheless, the effect of such N inputs should be rather small compared to fertilizer and excreta input. Voglmeier et al. (2018) measured the NH_3 emissions originating from the animal excreta for the same experiment and found an EF in the range of 6.4 % to 8.7 % of the applied urine-N (see Table 1) for the pasture fields M and G, respectively. Since the surrounding fields were managed in a similar way and intensity like our experimental pastures, their grazing-related NH_3 emissions are assumed to be of comparable magnitude. The re-deposition of the emitted NH_3 to a pasture was modelled by Bell et al. (2017) who found re-deposition factors around 15 %. Accordingly, the additional N input to the pasture should have been in the range of 1 kg N ha^{-1} yr^{-1} and was therefore very small.

3.4 Greenhouse gas budget

In order to calculate the GHG budget, annual emission estimates of CH_4 fluxes are required. We decided to use the EC measurements instead of upscaled FB estimates for annual integration due to the larger spatial coverage and the longer time series. Whereas the soil CH_4 emissions were negligible for the NECB calculation, the high GWP of CH_4 made it a significant component for the NGB (Fig. 8). However, the N_2O emissions were the strongest source of GHG emissions (in CO_2 eq.) for both pasture fields and were about 3.3 ± 2.0 times higher than the contribution by the observed soil CH_4 emissions. The NGB was finally calculated according to Eq. 2 taking into account the emissions of CH_4 and N_2O and the calculated, for CH_4 corrected (see Eq. 2) NECB (Fig. 8). We found that the NECB of the two pasture soils counterbalanced the corresponding soil emissions of CH_4 and N_2O and resulted in not significantly different to zero NGBs of -144 ± 277 g CO_2 eq. m^{-2} yr^{-1} and -9 ± 256 g CO_2 eq. m^{-2} yr^{-1} for pasture soils M and G, respectively. The uncertainty of the NGB was dominated by the NECB uncertainty, which mainly resulted from the large compensating fluxes of grazing and NEE (Sect. 3.2) with their respective uncertainties. The significant NGB difference between both pastures (135 ± 89 g CO_2 eq. m^{-2} yr^{-1}, for uncertainty calculation see bold error bars in Fig. 8) resulted mainly from the difference in NECB. Thus, feeding supplements from outside the pasture field can improve the GHG budget. However, it has to be noted that supplementary feed from outside the system boundaries can potentially result in trade-offs (e.g. GHG emissions of maize production: Louro et al. ,2015; soil erosion: Brant et al., 2017). Moreover, on the farm or system level direct cow-related emissions of CH_4 can strongly offset potential C gains in the soil and can even lead to a GHG source of such pastoral ecosystems (Felber et al., 2016a; Jones et al., 2017; Soussana et al., 2007). These direct-cow related emissions were not considered in this study. Full life cycle assessments are typically required if environmental impacts at the farm level should be investigated e.g. where GHG emissions from manure storage and management have to be taken into account as well (Mc Geough et al., 2012; Nemecek and Ledgard, 2016).

The NGBs in the present study are similar to the results by Felber et al. (2016a) who calculated a non-significant GHG source for the same pasture in 2013, however with a different grazing management and only modelled N_2O emissions. Our study also shows the possibility to counteract the soil emissions of CH_4 and N_2O by increasing the C stock in the soil, similar to a study by Soussana et al. (2010).

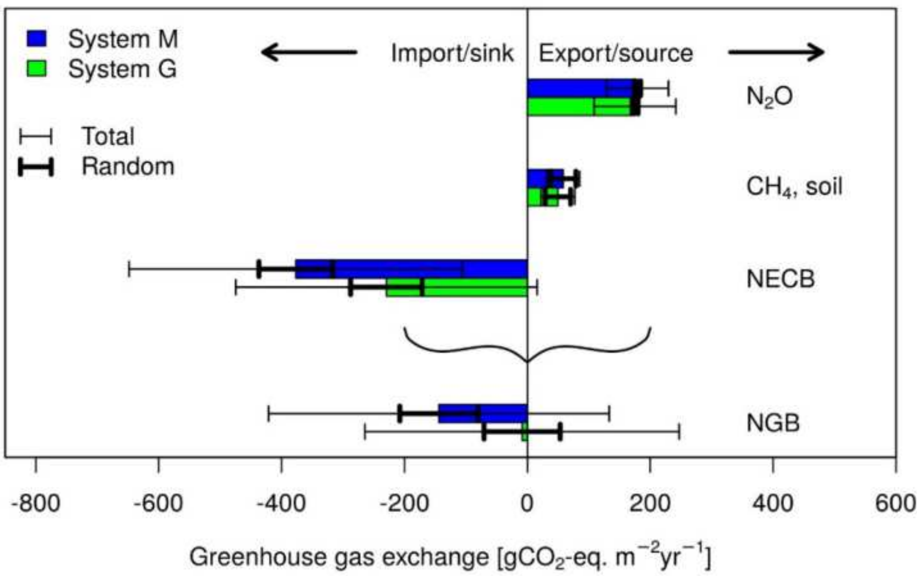

Figure 8: Net greenhouse gas balance NGB and GHG exchange fluxes of the two pasture fields as described in Sect. 2.6 (cows outside system boundaries). The micrometeorological sign convention is used, with negative values indicating a sink (flux from the atmosphere to the system). Please note that the NECB includes soil CH_4 fluxes. For calculation of NGB, this C source was subtracted from NECB according to Eq. 2. The error bars show the 95 % uncertainty. Thin error bars represent the total uncertainties (systematic and random) while the bold error bars indicate the random uncertainties relevant for the system comparison.

4 Conclusions

We found that soil CH_4 emissions had a significant impact on the soil GHG balance of the studied pastures due to the high GWP of CH_4. FB measurements suggested that the soil CH_4 flux during the grazing season was dominated by the emissions of dung patches but comparisons of up-scaled emissions with the EC derived emissions revealed unresolved discrepancies. This indicates the need for a higher sampling frequency of the FB measurements in combination with a higher spatial resolution to fully understand CH_4 emissions. The FB measurements were also used to disentangle the main emission sources for N_2O. On an annual basis, total N_2O emissions were dominated by emissions from slurry and mineral fertilization. The remaining emissions could be attributed equally to the background and excreta-related N_2O emissions from grazing. In summary, the management-related emissions resulting from grazing and fertilization contributed to about 72 % to the annual cumulative N_2O emissions. The remaining background N_2O emission could not be attributed to individual emissions sources. Continuous observations over multiple years with different management situations could help in optimizing the N_2O source attribution from pastures. The combination of FB and EC measurements allowed us to separate the annual emissions into individual emission sources and enabled a better understanding of the emission pattern of CH_4 and N_2O from the pasture. It is therefore suggested to combine small scale and field scale measurements on such ecosystems in further studies.

Our measurements showed that the emissions of N_2O and CH_4 were counteracted by C sequestration and that both pastures were statistically GHG neutral, yet with large absolute uncertainties. However, the setup with two independent neighboring EC systems considerably reduced the relevant uncertainty of the comparison between the pasture soils. This allowed us to hypothesize that the higher C import through excreta to field M as a result of the maize silage fed in the barn contributes positively to the C sink strength of the pasture field M and also contributed positively to an improved GHG budget. However, the results of this study are only based on a single year of measurements and potential differences on a multi-year scale (e.g. different N_2O emissions due to varying fertilizer / excreta N input, weather effect on grass growth and subsequent grazing intake / NEE) could not be quantified. Moreover, potential trade-offs associated with maize production (e.g. GHG emissions, soil erosion) were not taken into account in this study and thus full life cycle assessments are required to further analyze the environmental impacts of feed supplements in the forage.

Declarations of interest: none

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