Temporal and spatial variations of soil CO2, CH4 and N2O fluxes at three differently managed grasslands
Temporal and spatial variations of soil CO$_2$, CH$_4$ and N$_2$O fluxes at three differently managed grasslands

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Abstract. A profound understanding of temporal and spatial variabilities of soil carbon dioxide (CO$_2$), methane (CH$_4$) and nitrous oxide (N$_2$O) fluxes between terrestrial ecosystems and the atmosphere is needed to reliably quantify these fluxes and to develop future mitigation strategies. For managed grassland ecosystems, temporal and spatial variabilities of these three soil greenhouse gas (GHG) fluxes occur due to changes in environmental drivers as well as fertilizer applications, harvests and grazing. To assess how such changes affect soil GHG fluxes at Swiss grassland sites, we studied three sites along an altitudinal gradient that corresponds to a management gradient: from 400 m a.s.l. (intensively managed) to 1000 m a.s.l. (moderately intensive managed) to 2000 m a.s.l. (extensively managed). The alpine grassland was included to study both effects of extensive management on CH$_4$ and N$_2$O fluxes and the different climate regime occurring at this altitude. Temporal and spatial variabilities of soil GHG fluxes and environmental drivers on various timescales were determined along transects of 16 static soil chambers at each site. All three grasslands were N$_2$O sources, with mean annual soil fluxes ranging from 0.15 to 1.28 nmol m$^{-2}$ s$^{-1}$. Contrastingly, all sites were weak CH$_4$ sinks, with soil uptake rates ranging from $-0.56$ to $-0.15$ nmol m$^{-2}$ s$^{-1}$. Mean annual soil and plant respiration losses of CO$_2$, measured with opaque chambers, ranged from 5.2 to 6.5 µmol m$^{-2}$ s$^{-1}$. While the environmental drivers and their respective explanatory power for soil N$_2$O emissions differed considerably among the three grasslands (adjusted $r^2$ ranging from 0.19 to 0.42), CH$_4$ and CO$_2$ soil fluxes were much better constrained (adjusted $r^2$ ranging from 0.46 to 0.80) by soil water content and air temperature, respectively. Throughout the year, spatial heterogeneity was particularly high for soil N$_2$O and CH$_4$ fluxes. We found permanent hot spots for soil N$_2$O emissions as well as locations of permanently lower soil CH$_4$ uptake rates at the extensively managed alpine site. Including hot spots was essential to obtain a representative mean soil flux for the respective ecosystem. At the intensively managed grassland, management effects clearly dominated over effects of environmental drivers on soil N$_2$O fluxes. For CO$_2$ and CH$_4$, the importance of management effects did depend on the status of the vegetation (LAI).

1 Introduction

About 10% of the fossil fuel emissions that originate from EU-25 countries have been shown to be absorbed by terrestrial ecosystems (Janssens et al., 2003; Schulze et al., 2009). While most of the atmospheric CO$_2$ is sequestered by forests, grasslands are a small net sink for atmospheric CO$_2$. Croplands are reported to be CO$_2$ neutral, and managed peatlands act as net sources for atmospheric CO$_2$ (Ciais et al., 2010b). However, the net terrestrial sink for atmospheric CO$_2$ is almost counterbalanced by CH$_4$ and N$_2$O emissions from agriculture (Ciais et al., 2010a). Despite the relatively small atmospheric concentrations of CH$_4$ and N$_2$O, these two greenhouse gases (GHG) account for 26% of the global warming effect due to their high global warming potential (GWP) on a per-mass basis over a 100 yr time horizon (IPCC, 2007). Thus, understanding temporal and spatial variabilities of soil CO$_2$, CH$_4$ and N$_2$O fluxes between terrestrial ecosystems and the atmosphere is of key importance to reliably estimate these GHG fluxes and to develop mitigation strategies. While Vleeschouwers and Verhagen (2002) and Janssens et al. (2003) reported that the GWP of European...
grasslands is still highly uncertain, Soussana et al. (2007) and Schulze et al. (2009) estimated that European grasslands had negative GWPs (including vertical and lateral fluxes of the three GHGs: e.g., fertilizer input, harvested biomass, animal emissions). In particular, all these investigated European grasslands were net sinks for CO₂ on an annual timescale. Soil fluxes of CH₄ and N₂O, expressed in CO₂ equivalents, reduced the net CO₂ sink capacity, but were too small to change the overall effect into a positive GWP (Soussana et al., 2007; Schulze et al., 2009). These two studies, however, did not consider grasslands at higher elevations, with cooler and wetter climatic conditions that may subsequently lead to higher soil CH₄ and N₂O emissions. Mountain regions are characterized by more orographic precipitation under seasonal and annual timescales; and (3) to identify the spatial variations within sites.

2 Materials and methods

2.1 Site description

The three-stage farming system that is typical for many parts of the Swiss Alps (Ehlers and Kreutzmann, 2000) was represented exemplarily by the three ETH research sites Chamau (CHA), Frübül (FRU) and Alp Weissenstein (AWS). The lowland site is represented by Chamau, situated north of Lake Zug in the pre-alpine lowlands of Switzerland at 400 m a.s.l. (47°12′37″ N, 8°24′38″ E), with a mean annual temperature of 9.1 °C and 1151 mm of precipitation (Sieber et al., 2011; Finger et al., 2013). The dominating soil type at CHA is a Cambisol (pH = 5) with bulk density ranging between 0.9 and 1.2 kg m⁻³ in the uppermost 20 cm. The grass mixture consists of two dominating species, i.e., Italian ryegrass (Lolium multiflorum Lam.) and white clover (Trifolium repens L.). In 2010 and 2011, the pastures were used for forage production (Sautier, 2007; Zeeman et al., 2010). The site is intensively managed, with 5 to 10 management events per calendar year, including harvests and subsequent slurry applications. The exact number of management events per calendar year naturally depends on seasonal weather conditions.

The so-called Maiensaess site (early season mountain rangeland) is represented by Frübül, situated at the Zugerberg mountain ridge, east of Lake Zug (47°6′57″ N, 8°32′16″ E) at 1000 m a.s.l., with a mean annual temperature of 6.1 °C and 1682 mm of precipitation (Sieber et al., 2011; Finger et al., 2013). At FRU, the dominating soil type is a Gleysol (pH = 4.5), with bulk density ranging between 1.3 and 1.4 kg m⁻³ (uppermost 20 cm). The grass mixture is more diverse than at CHA, consisting of ryegrass (Lolium multiflorum Lam.), meadow foxtail (Alopecurus pratensis L.), cocksfoot grass (Dactylis glomerata L.), dandelion (Taraxacum officinale), buttercup (Ranunculus L.) and white clover (Trifolium repens L.) (Sautier, 2007; Zeeman et al., 2010). The pastures at FRU are used for cattle grazing in late spring (May) and early fall (September). The management includes two to four harvests and/or fertilization events (slurry/manure) per calendar year, depending on local weather conditions and resulting vegetation growth.

The alpine site is represented by Alp Weissenstein, situated near the Albula pass at 2000 m a.s.l. in the canton Grisons (46°34′59″ N, 9°47′25″ E), with a mean annual temperature of −1.4 °C and 877 mm of precipitation (Sieber et al., 2011; Finger et al., 2013; Michna et al., 2013). The soils at AWS are slightly humus to humus sandy loam (pH = 5.6), with comparable bulk density in the uppermost 20 cm as reported for FRU (1.3 to 1.4 kg m⁻³). The grass composition is classified as Deschampsio cespitosae–Poetum

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alpinae, with red fescue (*Festuca rubra*), alpine cat’s tail (*Phleum rhaeticum*), white clover (*Trifolium repens* L.) and dandelion (*Taraxacum ofcinale*) (Sautier, 2007; Zeeman et al., 2010). AWS represents the summer rangelands for cattle grazing without harvests in normal years. Manure is applied to the pastures at the end of the grazing season, typically in the second half of September. The geographic location of the three sites as well as the site setups are shown in Fig. 1, while dates for harvests and fertilization events are shown in Table 1.

### 2.2 Experimental setup

From June 2010 to June 2011, we measured soil fluxes of CO$_2$, CH$_4$ and N$_2$O using opaque static soil chambers at all three sites. The diameter of the polyvinyl chloride chambers was 0.3 m, the average headspace height 0.136 m (±0.015 m) and average insertion depth of the collars was 0.08 m (±0.05 m). On sampling campaigns with vegetation inside the chamber > 0.20 m, collar extensions (0.45 m) were used. Chamber lids were equipped with reflective aluminum foil to minimize heating inside the chamber during the period of actual measurement. At each site, transects of 16 soil chamber collars were installed in May 2010 (Fig. 1). Spacing between the chambers was 7 m at CHA and FRU, and 5 m at AWS. At FRU, two transects were established, one consisting of 12 chambers and a second consisting of four chambers (Fig. 1c). This was done to adopt the sampling approach to the usual management regime, which is coerced by two differently managed field parcels. Spacing was chosen so that the transects would fit into the previously calculated footprint of the eddy covariance towers at the respective sites (Zeeman et al., 2010), as well as to cover the topographic differences at each site. At FRU and AWS, chamber collars were fenced to avoid trampling and/or removal by the cattle. Sampling of the chambers was performed on a weekly basis during the growing season, and at least once a month during the winter season (except for AWS, which is inaccessible in winter). The vegetation inside the chamber collars was manually harvested at the times of regular management activities, i.e., harvests and grazing.

Additionally, we measured diel patterns of soil CH$_4$ and N$_2$O fluxes in an intensive observation campaign in September 2010. During this campaign, soil CH$_4$ and N$_2$O fluxes were measured over 48 consecutive hours at all three sites simultaneously, at intervals of two hours. During this 48 h intensive observation campaign (21–23 September 2010), 75 mean chamber fluxes of CH$_4$ and N$_2$O were obtained at the three sites ($N_{CHA} = 25$; $N_{FRU} = 25$; $N_{AWS} = 25$).
2.3 Data acquisition and processing

2.3.1 Flux sampling and calculations

GHG fluxes were calculated based on the rate of changing gas concentrations inside the chamber headspace. After closing the chambers, four samples were taken, one immediately after closure and then at 10 to 13 min increments so that the chamber was no longer closed than 40 min. This closing time was sufficiently short to avoid saturation effects inside the chamber headspace. We inserted 60 mL syringes into the chambers lid septums to take the gas samples, and then injected the gas into pre-evacuated 12 mL vials (Labco Limited, Buckinghamshire, UK). Prior to the second, third and fourth sampling of each chamber, the chamber headspace was flushed with the syringe volume of air from the chamber headspace to minimize effects of built-up concentration gradients inside the chamber. Gas samples were analyzed for their CO2, CH4 and N2O concentrations using gas chromatography (Agilent 6890 gas chromatograph equipped with a flame ionization detector, a methanizer and an electron capture detector, Agilent Technologies Inc., Santa Clara, USA) as described by Hartmann et al. (2011).

Data processing, which included flux calculation and quality checks, was carried out with the R statistical software (R Development Core Team, 2010). The rate of change was calculated by the slope of the linear regression between gas concentration and time. Fluxes were always small enough ($r^2 \geq 0.8$) that no saturation in measured concentration data could be detected that would be indicative of saturation effects inside the chamber. We used the following equation to derive the flux estimate $F_{GHG}$:

$$F_{GHG} = \frac{\delta c}{\delta t} \cdot \frac{p \cdot V}{R \cdot T \cdot A},$$

where $c$ is the respective GHG concentration ($\mu$mol mol$^{-1}$ for CO2; nmol mol$^{-1}$ for CH4 and N2O), $t$ is given in seconds, atmospheric pressure ($p$) in Pa, the headspace volume ($V$) in m$^3$, the universal gas constant ($R$) is 8.3145 m$^3$ Pa K$^{-1}$ mol$^{-1}$, $T$ is ambient air temperature ($K$) and $A$ is the surface area enclosed by the chamber (m$^2$). Individual chamber fluxes were only computed if the linear regression for each individual GHG yielded a $r^2 \geq 0.8$. If the slope between the first and second concentration obviously deviated from the one of the remaining three concentration measurements, then we omitted it and calculated the flux from the remaining three. Chambers for which the rate of change of CO2 was negative were also discarded, as photosynthesis is assumed to be zero inside the opaque chamber. The mean chamber flux was then calculated as the arithmetic mean of all available individual chamber fluxes for each date and site.

A total of 81 sampling campaigns were performed between June 2010 and June 2011 (NCHA = 35; NFRU = 32; NAWS = 17), resulting in an equivalent number of mean chamber fluxes of N2O, CH4 and CO2. We follow the micrometeorological convention, where positive fluxes are directed to the atmosphere, and negative fluxes to the ecosystem.

2.3.2 Ancillary measurements

At each field site, the following environmental variables were recorded in the center of the transects (eddy covariance towers, Fig. 1) as 10 min averages: air temperature ($T_a$) at 2 m (HydroClip S3, Rotronic AG, Baselland, CH), soil temperature ($T_s$) at $-0.02$ m (TL107 sensors, Markapub AG, Olten, CH), volumetric soil water content (SWC) at $-0.05$ m (ML2x, Delta-T Devices Ltd, Cambridge, UK), and photosynthetic active radiation (PAR) at 2 m (PARile, Kipp and Zonen B.V, Delft, The Netherlands). Leaf area index (LAI) of the vegetation outside the chambers was measured at each flux sampling campaign (LAI-2000, Licor, Lincoln,
Fig. 2. Annual courses of $N_2O$, $CH_4$ and $CO_2$ fluxes at Chamaun (CHA), Früblü (FRU) and Alp Weissenstein (AWS). The black lines indicate the mean soil flux of the respective greenhouse gas (GHG), and the gray-shaded areas indicate the 95% confidence intervals. Dashed, vertical lines indicate fertilizer applications. Black boxes for FRU and CHA denote periods of permanent snow cover. Note the different scaling for $N_2O$ fluxes at the three sites. Fluxes of $N_2O$ and $CH_4$ are given in nmol m$^{-2}$ s$^{-1}$ and $CO_2$ in µmol m$^{-2}$ s$^{-1}$.

USA), i.e., every week during the growing period and at least monthly during winter (when there was no snow cover). LAI measurements represent averages of 12 measurements along the chamber transects.

2.3.3 Statistics

For each site, mean chamber GHG fluxes were used to establish functional relationships with environmental drivers on annual timescales, using stepwise multiple linear regression models for annual timescales. A standardized principal component analysis (PCA) was performed prior to the multiple linear regressions to minimize potential artifacts from collinearities between environmental drivers. Since $CH_4$ fluxes at CHA and FRU showed exponential relationships with soil water content, data were log-transformed for the respective multiple linear models. The relative importance of each driver within the regression model was estimated using hierarchical partitioning (Chevan and Sutherland, 1991). The uncertainty of the calculated relative importance was assessed using parametric bootstrapping methods (Efron, 1979).

To investigate spatial variability of GHG fluxes, a one-way analysis of variance (ANOVA) was performed. The chamber number along the transect (Fig. 1) was used as a predictor, and all valid annual flux values per chamber were included. To assess the statistical significance of pairwise comparisons of ANOVA results, the Tukey honestly significant differences (HSD) test was chosen.

3 Results

3.1 Temporal variation of GHG fluxes

3.1.1 Seasonal flux patterns and management

Soil fluxes of $N_2O$ and $CO_2$ showed a commonly known seasonal pattern, with highest emission rates during the summer months (Fig. 2). Soil emissions of $CH_4$ were mostly observed during winters, whereas uptake rates were prevailing in summers.

At all three sites, soil $N_2O$ fluxes were mostly positive, indicating a source for $N_2O$. Occasional uptake was observed for mean and individual chamber fluxes, as indicated by the 95% confidence interval in Fig. 2. Highest $N_2O$ efflux rates were observed at CHA (intensively managed) shortly after slurry applications. These peak fluxes exceeded maximum $N_2O$ fluxes at FRU (moderately managed) and AWS (extensively managed) by a factor of 2 and higher. Emission factors at the intensively at CHA were on average $3.1 \pm 3.6\%$. At FRU, emission factors were $0.8 \pm 0.8\%$. At AWS, peak emissions occurred around the manure application and biomass removal, which was done manually during the period of grazing. Mean $N_2O$ soil efflux over the 12-month
study period was highest at CHA with 1.28 nmol m\(^{-2}\) s\(^{-1}\), and lowest at FRU with 0.15 nmol m\(^{-2}\) s\(^{-1}\). At AWS, an average efflux of 0.23 nmol m\(^{-2}\) s\(^{-1}\) was observed. The low mean N\(_2\)O efflux at FRU is likely due to low emission rates during winter, which were missing from the AWS data. Hence mean emissions were larger at AWS than at FRU.

At CHA and FRU, both positive and negative methane fluxes were observed, yet uptake dominated at both sites. During eight sampling campaigns, which represented 20% of all campaigns, CHA was a source of CH\(_4\). FRU was a source during five sampling dates, representing 12.5% of all campaigns, while AWS acted as a sink for atmospheric CH\(_4\) throughout the measurement period. However, no measurements are available for the winter period at this site, due to the inaccessibility of the site. The response of CH\(_4\) fluxes to fertilization, i.e., reduced uptake rates, was not as distinct as for N\(_2\)O fluxes. Over the 12-month study period, mean uptake rates for CH\(_4\) were \(-0.15\), \(-0.22\) and \(-0.56\) nmol m\(^{-2}\) s\(^{-1}\) at CHA, FRU and AWS, respectively.

Temporal flux patterns of CO\(_2\) were comparable at all three sites. The annual range of flux magnitudes was slightly smaller at AWS than at CHA and FRU. Respiration increased after fertilization at all three sites, yet the magnitude of this response was variable (Fig. 2). Over the 12-month study period, mean respiration rates were 6.5 and 6.3 µmol m\(^{-2}\) s\(^{-1}\) at CHA and FRU, respectively. At AWS, respiration rates were slightly smaller, with an overall mean flux of 5.2 µmol m\(^{-2}\) s\(^{-1}\) (growing season average).

3.1.2 Seasonal response of fluxes to environmental drivers

With the PCA, we were able to identify similarities among potential driver variables (\(T_a\), \(T_s\), SWC, PAR and LAI) and to reduce the set of driver variables for exploring functional relationships to those drivers that (a) are as independent from each other as possible, and (b) that are of similar relevance at all three elevations, such that functional relationships built with the selected drivers can be compared among sites. Nitrogen inputs in the form of slurry/manure applications were not considered for the PCA, as only six and three data points would have been available at CHA and FRU, respectively, and LAI can already be seen as a proxy for management activity. At CHA and FRU, \(T_a\) and \(T_s\) had similar loadings (Fig. 3) because both followed a similar annual cycle. Since \(T_a\) generally had the higher explanatory power than \(T_s\), we selected \(T_a\). The first and second principal components further indicated that PAR and SWC may be treated as one variable at CHA and FRU (Fig. 3), where variations in PAR were of opposite sign compared to variations in SWC, simply indicating that episodic increases in SWC after rain events at the lower two sites coincided with periods of high cloudiness that reduced PAR over several days. In contrast, under fair weather conditions, the typical diurnal cycle of PAR was likely linked to a similar cycle of SWC in the opposite

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**Fig. 3.** Biplots of the first two components of a standardized principal component analysis of environmental variables for the three sites Chamau (CHA), Frübül (FRU) and Alp Weissenstein (AWS). Environmental variables measured during sampling campaigns within the 12-month study period were considered (numbers 1 to maximum 34 at CHA). \(T_a\) = air temperature; \(T_s\) = soil temperature; SWC = soil water content; LAI = leaf area index; and PAR = photosynthetic active radiation.
direction. Since our selection of $T_a$ already represented atmospheric conditions at a site, we selected SWC instead of PAR as the second variable. The third variable selected via PCA was LAI, which was almost independent of SWC at all three sites (Fig. 3), and hence was expected to add explanatory value to a functional model where the three compartments atmosphere ($T_a$), soil (SWC) and vegetation (LAI) were represented (Fig. 4).

The explanatory power of the multiple linear model with regard to the temporal variation of $N_2O$ fluxes varied considerably at the three sites, with adjusted $r^2$ values ranging from 0.19 to 0.42. The relative importance (RI) of each selected driver, i.e., the contribution to the overall variance of flux variability explained by the model, was not consistent among sites (Fig. 4). At CHA, the model was able to explain 42 % of the total variance inherent in annual soil $N_2O$ flux data, with LAI and $T_a$ being the most important explanatory variables ($RI_{LAI} = 45.4 \%$; $RI_{T_a} = 38.9 \%$). SWC had much less influence on the $N_2O$ efflux, with a RI of 16.1 %. At FRU, $T_a$ was clearly the most important driver, with a RI of 84.7 %, followed by SWC with a RI of 14 %. LAI was of minor importance, with a RI of 1.3 %. In total, only 19 % of the variance in soil $N_2O$ fluxes was explained by the model at FRU. At AWS, 34 % of the variance was explained. Here, SWC was the most powerful explanatory variable, with a RI of 54.7 %, followed by LAI (RI = 43.7 %). $T_a$ had almost no impact on the variability of the $N_2O$ flux at AWS (RI = 1.7 %) (Fig. 4).

The variation of explanatory power among the driver variables within the multiple linear model for the prediction of CH$_4$ fluxes was more pronounced than that for $N_2O$ fluxes. However, soil CH$_4$ fluxes were better constrained by the set of drivers, with adjusted $r^2$ values ranging from 0.46 to 0.83. Again, the RI of the drivers was not consistent among sites (Fig. 4). At CHA, the model explained 46 % of the total variance inherent in annual soil CH$_4$ flux data, with LAI and SWC being the most influential variables ($RI_{LAI} = 45.4 \%$; $RI_{SWC} = 35.4 \%$), followed by $T_a$ with a RI of 19.2 %. At FRU, 72 % of the total variance was explained, with SWC being the most important variable, exhibiting a RI of 89 %. At AWS, 83 % of variance was explained by the model, of which 82.7 % were due to changes in SWC. At both sites, FRU and AWS, the variables LAI and $T_a$ had minor influences on soil CH$_4$ fluxes.

The explanatory power of the multiple linear model for soil CO$_2$ fluxes was almost the same at CHA and FRU, with 80 % total explained variance. At AWS, the model was still able to explain 47 %. At CHA and FRU, $T_a$ was the most influential variable, with RI values of 71 and 81 %, respectively. Yet, at CHA, LAI had a considerable influence on the temporal variability of CO$_2$ fluxes, with a RI of 21.8 %. At FRU, the contribution of seasonal changes in LAI was less important (RI = 9.1 %). At AWS, $T_a$ was the most important variable in the model similar to CHA and FRU (RI of 55.7 %), followed by SWC with a RI of 30.9 % (Fig. 4).

3.1.3 Diel variation of $N_2O$ and CH$_4$ fluxes

Mean chamber efflux rates of $N_2O$ and mean chamber uptake rates of CH$_4$ were observed during the intensive observation campaign at all sites in September 2010 (Fig. 5). The diel $N_2O$ flux magnitudes along the elevational transect yielded a different ranking among sites than that observed at the annual scale. During the intensive observation campaign, highest emissions of $N_2O$ were measured at AWS, with an average flux of 0.54 nmol m$^{-2}$ s$^{-1}$, followed by CHA with 0.21 and FRU with 0.15 nmol m$^{-2}$ s$^{-1}$. Diel variations of soil $N_2O$ fluxes were clearly found only at FRU, where high emission rates were observed during the day, and smaller emissions during nights (Fig. 5). $T_a$ was a good predictor for $N_2O$ efflux rates at the diel timescale at CHA and FRU, explaining

![Fig. 4. Explanatory power of driver variables for GHG fluxes on the annual timescale at Chamau (CHA), Frübul (FRU) and Alp Weissenstein (AWS), with $T_a$ as air temperature, SWC as soil water content and LAI as leaf area index. Contribution (relative importance) to the overall variance explained is given in %. Error bars indicate 95% confidence intervals as determined from bootstrapping ($N_{runs} = 1000$). $r^2$ values represent overall model performance. Significance levels of each driver can be found in Table 2. The upper panel shows the model performance at all three sites for $N_2O$ fluxes, the middle panel for CH$_4$ fluxes and the lower panel for CO$_2$ fluxes.](image-url)
54 and 59 % of the variance, respectively. In contrast, N₂O emissions did not significantly correlate with Tₐ at the diel scale (nor at the annual scale) at AWS (Fig. 4).

Highest mean uptake rates of CH₄ were measured at AWS with −0.47 nmol m⁻² s⁻¹, followed by CHA and FRU with −0.31 and −0.16 nmol m⁻² s⁻¹ (Fig. 5). Although considerable variation in soil CH₄ flux rates was visible at CHA and FRU, no obvious diel trend was identified. At AWS, CH₄ uptake rates were almost constant, with only very little temporal variation. Hence, regression analysis to determine flux drivers was not successful. At CHA, 13 % of the variance in CH₄ uptake rates could be explained by Tₐ, while no significant relationship could be established at FRU and AWS. SWC variations, important at the annual scale, were affected only slightly during the 48 h intensive observation campaign, and were therefore omitted for developing any explanatory power at the diel scale.

3.2 Spatially invariant hot spots of GHG fluxes on the seasonal scale

Spatial variability of annually averaged soil N₂O fluxes, i.e., the flux variation among chambers along the transect, was highest at CHA (Fig. 6), in contrast to the spatial variation seen over 48 h. The one-way ANOVA with chamber as a factor yielded a p value of 0.57 for soil N₂O fluxes, indicating that all chambers showed high variation during the 12 months of measurements. At AWS, chambers one to three showed a wider range of annual average N₂O efflux rates relative to the other chambers along the transect. Yet, due to some very high flux estimates, the ANOVA yielded a p value of 0.52, indicating no significant differences among individual chambers. This suggested that the spatial variability of soil N₂O fluxes was not larger than the temporal variability of the fluxes measured at AWS. Spatial variations of annual average N₂O fluxes at FRU were negligible (Fig. 6), and in a similar magnitude as at the diel scale (not shown).

Spatially invariant annual averages of soil CH₄ fluxes were found at CHA and FRU (Fig. 6). At AWS, we observed a spot of significantly (p = 0.02) lower CH₄ uptakes rates around chambers two and three (Fig. 6).

4 Discussion

It was not a priori expected that the three grasslands at different elevations, and thus management intensities, were all weak net sinks for CH₄ due to abundant precipitation in mountainous areas, whereas the tight coupling between soil N₂O efflux rates and fertilization confirmed earlier studies. The alpine grassland (AWS), characterized by sufficient rainwater supply and hypothesized to be a net source of CH₄, acted as a net sink of methane, given the fact that the long winter period could not be included in our measurement regime. In addition, soil N₂O as well as CH₄ fluxes were highly variable in time and space, supporting previous studies (e.g., Folorunso and Rolston, 1984; Mosier et al., 1991; Velthof et al., 2000).

Fig. 5. Diel courses of N₂O (top) and CH₄ (bottom) fluxes at Chamau (CHA), Früebül (FRU) and Alp Weissenstein (AWS). The gray-shaded areas indicate the 95 % confidence intervals of mean chamber fluxes (black line). Measurements started at noon (12:00) on September 21 and ended at noon (12:00) on 23 September.
4.1 Seasonal GHG fluxes and drivers of their temporal variability

4.1.1 N$_2$O fluxes

Our measurements give strong evidence that managed grasslands are a constant source of N$_2$O, as hardly any uptake was observed (Ryden, 1981; Wagner-Riddle et al., 1997; Glatzel and Stahr, 2001; Neftel et al., 2007). The small number of negative soil N$_2$O fluxes (uptake) observed was evenly distributed throughout the 12-month study period. As expected, the intensively managed site, CHA, was the strongest source of N$_2$O. While total N addition (per application) at CHA was comparable to FRU, mean annual emissions were more than 8 times higher than those at FRU, leading to much higher emission factors at CHA. FRU was characterized by the lowest mean annual N$_2$O emissions. Emission factors at the intensively managed grassland (CHA) were on average 3.1 ± 3.6 %, and therefore considerably higher than the IPCC (2007) default value of 1.25 % (without grazing). At FRU (moderately managed), emission factors were only 0.8 ± 0.8 %.

Our findings also underline the challenge of predicting N$_2$O fluxes from grasslands. N$_2$O fluxes were weakly constrained by a set of three environmental variables. Their influence on the flux varied temporally and from site to site, indicating the importance of additional factors (e.g., land management, fertilization). Yet, our results are in agreement with previous studies, which identified air temperature and soil water content as influential variables (e.g., Wang et al., 2005; Liebig et al., 2010; Schaufler et al., 2010). The high relative importance of LAI at CHA (intensively managed) can be explained by the fact that step changes in LAI due to regular harvests during the growing season reflect subsequent slurry applications at CHA (N addition usually within five days after the harvest). This finding underlines the importance of a realistic management consideration when predicting N$_2$O fluxes. This was corroborated in Fig. 7, which shows the response of the three GHG fluxes to their primary environmental driver (at two LAI classes) on the annual scale (Table 2). For N$_2$O we found that $T_a$ had no significant influence on flux magnitudes at constant LAI ranges. Thus, management (with LAI as proxy) had a larger effect on N$_2$O fluxes than the environment (with $T_a$ as proxy).

4.1.2 CH$_4$ fluxes

Our results showed consistently small CH$_4$ sinks at all three sites, which is in agreement with other studies on temperate grasslands at low altitudes (Mosier et al., 1997; Liebig et al., 2010). Positive soil CH$_4$ fluxes mostly occurred during periods with high SWC and/or after fertilization (Lessard et al., 1997), and hence were observed most frequently at the intensively and moderately managed site, CHA and FRU. For FRU, our results are in contrast to the results presented by Hartmann et al. (2011), who exclusively measured uptake of

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Fig. 6. Box plots depicting spatial gradients of mean annual N$_2$O and CH$_4$ chamber fluxes (1–16 on x axis) at Chamau (CHA), Frühül (FRU) and Alp Weissenstein (AWS). The $p$ values refer to the ANOVA results with chamber as a factor. Values below 0.05 indicate significant differences among mean chamber fluxes over the 12-month study period.
Table 2. Multiple linear model equations for annual GHG fluxes at the three sites Chamau (CHA), Frübühl (FRU) and Alp Weissenstein (AWS); p values are given for the individual drivers (Ta = air temperature; SWC = soil water content; LAI = leaf area index).

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<th>Equation</th>
<th>CHA</th>
<th>FRU</th>
<th>AWS</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>N2O</td>
<td>[ N_2O = 6e^{-6} \pm 2e^{-6} \cdot T_a + 8e^{-4} \pm 3e^{-4} \cdot \text{SWC} - 2e^{-5} \pm 6e^{-6} \cdot \text{LAI} ] [ \alpha = 0.002; \beta = 0.014; \gamma = 0.002 ]</td>
<td>[ N_2O = 7e^{-7} \pm 9e^{-5} \cdot T_a + 3e^{-5} \pm 2e^{-5} \cdot \text{SWC} - 1e^{-7} \pm 1e^{-6} \cdot \text{LAI} ] [ \alpha = 0.038; \beta = 0.222; \gamma = 0.900 ]</td>
<td>[ N_2O = 0.01 \pm 0.02 \cdot T_a + 1.79 \pm 1.17 \cdot \text{SWC} - 0.19 \pm 0.12 \cdot \text{LAI} ] [ \alpha = 0.590; \beta = 0.160; \gamma = 0.160 ]</td>
</tr>
<tr>
<td></td>
<td>log(CH4)</td>
<td>[ \log(CH_4) = -0.01 \pm 0.02 \cdot T_a + 7.14 \pm 2.77 \cdot \text{SWC} - 0.17 \pm 0.05 \cdot \text{LAI} ] [ \alpha = 0.751; \beta = 0.016; \gamma = 0.003 ]</td>
<td>[ \log(CH_4) = 8e^{-5} \pm 0.01 \cdot T_a + 4.36 \pm 0.68 \cdot \text{SWC} + 0.01 \pm 0.03 \cdot \text{LAI} ] [ \alpha = 0.990; \beta &lt; 0.001; \gamma = 0.860 ]</td>
<td>[ \log(CH_4) = 3e^{-3} \pm 7e^{-3} \cdot T_a + 2.45 \pm 0.39 \cdot \text{SWC} - 0.09 \pm 0.04 \cdot \text{LAI} ] [ \alpha = 0.990; \beta &lt; 0.001; \gamma = 0.860 ]</td>
</tr>
<tr>
<td></td>
<td>CO2</td>
<td>[ CO_2 = 0.38 \pm 0.05 \cdot T_a + 24.24 \pm 8.43 \cdot \text{SWC} + 0.45 \pm 0.16 \cdot \text{LAI} ] [ \alpha &lt; 0.001; \beta = 0.024; \gamma = 0.009 ]</td>
<td>[ CO_2 = 0.39 \pm 0.05 \cdot T_a + 0.47 \pm 4.11 \cdot \text{SWC} + 0.20 \pm 0.17 \cdot \text{LAI} ] [ \alpha &lt; 0.001; \beta = 0.910; \gamma = 0.250 ]</td>
<td>[ CO_2 = 0.26 \pm 0.07 \cdot T_a + 7.96 \pm 3.71 \cdot \text{SWC} + 0.24 \pm 0.39 \cdot \text{LAI} ] [ \alpha = 0.004; \beta = 0.057; \gamma = 0.557 ]</td>
</tr>
</tbody>
</table>
CH₄ at FRU in the years 2007, 2008 and 2009. However, their measurements were taken in generally drier soils, supporting the idea of regulating effects of SWC on CH₄ fluxes (RI of 89%). At the extensively managed AWS site, both Hartmann et al. (2011) and this study exclusively observed uptake of CH₄.

Annually, soil CH₄ fluxes were well predictable by SWC (Liebig et al., 2010; Schrier-Uijl et al., 2010; Hartmann et al., 2011). Furthermore, at the intensively managed site (CHA), LAI had comparable explanatory power to SWC. As mentioned before, step changes in LAI during the growing period reflect management activities (fertilization at low LAI). Although ammonium-based fertilizers (e.g., organic fertilizers) can inhibit CH₄ uptake (oxidation) by methanotrophs (Willison et al., 1995; Stiehl-Braun et al., 2011), SWC still had a highly significant impact on CH₄ fluxes at LAI < 1 (Fig. 7), indicating dominant environmental drivers.

### 4.1.3 CO₂ fluxes

Opaque soil chambers were used to exclusively measure respiratory fluxes of CO₂, which were in the expected range from close to zero in winter up to 15 µmol m⁻² s⁻¹ during summer, similar to, e.g., Myklebust et al. (2008). At FRU, chamber measurements agreed well with eddy covariance-based respiration data, simultaneously recorded at all three sites (Fig. 8). At CHA, eddy-covariance-based fluxes were systematically higher, which is likely due to the different scale of both measurement techniques (Wang et al., 2009). Except for one outlier of eddy-covariance-based soil respiration, chamber measurements agreed well with eddy covariance data at AWS. This outlier on August 25 may be an artifact of the gap-filling procedure for eddy covariance respiration data. Besides the the expected importance of Tₜ and SWC for CO₂ efflux rates, LAI was once more a reasonable predictor (RI of 21.8%) at the intensively managed grassland (CHA). Keeping LAI constant at < 1 (Fig. 7) revealed a still highly significant effect of Tₜ on soil CO₂ fluxes, supporting the notion that management impacts on soil flux magnitudes of CO₂ are rather small compared to environmental drivers (Peng et al., 2011).

### 4.2 Diel variation of N₂O and CH₄ fluxes

In contrast to what we learned on the annual scale, highest emissions of N₂O of all three grasslands were observed at AWS, being twice the observed seasonal mean. As manure was applied to AWS pastures 10 days prior to sampling, we likely captured a “hot” moment, supporting our conclusion that management impacts are dominating soil N₂O flux variability. Already studies by Christensen (1983) and Flechard et al. (2007) have shown that peak emissions of N₂O appeared lagged to manure application (8 to 12 days).

In the literature, there is no consistent picture regarding the presence of significant diel patterns of N₂O and CH₄ fluxes (Christensen, 1983; Skiba et al., 1996; Maljanen et al., 2002; Duan et al., 2005; Hendriks et al., 2008; Baldocchi et al., 2012). Duan et al. (2005) suggested that the type of ecosystem might have an influence on the presence of diel soil N₂O and CH₄ flux variations. However, our study exclusively investigated grasslands, and we found significant diel variations of N₂O fluxes for one out of three sites (FRU). At CHA, changes in air temperature affected soil N₂O fluxes less strongly, and thus no significant diel patterns in N₂O fluxes were observed. This might be attributable to the fact...
that the last slurry application at CHA was more than four weeks prior to the intensive campaign (25.08.2010; slurry 30 kg N ha$^{-1}$), and that the vegetation composition at FRU exhibits a larger fraction of legumes. This suggests that not only ecosystem type, but also site specifics (e.g., management intensity or vegetation composition) might influence diel variations in soil N$_2$O fluxes.

In contrast to what we observed on the annual scale (Table 2), CH$_4$ fluxes were not constrained by changes in SWC over the course of 48 h, probably because changes were too small (decrease of less than 2% vol) to significantly affect the magnitude of CH$_4$ fluxes.

4.3 Spatial patterns and autocorrelation

Working with soil chambers requires information on the spatial distribution of GHG fluxes at ecosystem scale to design appropriate experiments and to be able to correct mean soil fluxes for potential biases. In relation to the magnitude of mean chamber fluxes of N$_2$O and CH$_4$, individual chamber fluxes are highly variable in space (e.g., Mattheus et al., 1980; Folorunso and Rolston, 1984; Mosier et al., 1991), as they are largely determined by small-scale biochemical processes (Ambus and Christensen, 1994; Dalal and Allen, 2008).

We observed that out of our three sites, only AWS exhibited permanent spots where CH$_4$ uptake was significantly smaller than at the rest of the transect. This corresponded well with local microtopographical conditions, i.e., the inclination of the terrain (Fig. 9). Chambers placed in terrain with greater inclination systematically exhibited lower SWC values (data not shown). This in turn corresponded well with what was observed at the annual scale, where lower CH$_4$ uptake rates correlated significantly with higher SWC. At CHA and FRU, we observed that flux magnitudes varied spatially; spots of reduced CH$_4$ uptake were, however, not permanent. Thus, the situation at CHA and FRU represents the ideal case when trying to sample a representative mean of an ecosystem. Omitting permanent hot spots may lead to a systematic bias in GHG flux budgets. In our case, omitting chambers one to four at AWS would have lead to an underestimation of annual CH$_4$ uptake of roughly 5% and an overestimation of annual N$_2$O emissions of 56% (both regarding annual budgets). Thus, all aspects of exposition and slope should be covered when assessing flux estimates of CH$_4$ and N$_2$O in sloping terrain.

5 Conclusions

Highest mean annual emissions of N$_2$O were observed at the intensively managed site, whereas highest uptake rates of CH$_4$ were measured at the extensively managed site. This clearly illustrates the impact that management intensity has on the magnitude of soil N$_2$O and CH$_4$ fluxes in grasslands. This clearly illustrates that management acts as a major driver of CH$_4$ and N$_2$O fluxes in grasslands in addition to the commonly shown influences of environmental variables.

We identified the known set of drivers for fluxes of CO$_2$, CH$_4$ and N$_2$O fluxes (T$_a$ and SWC for N$_2$O and CH$_4$, respectively). At the intensively managed site (CHA), LAI proved to be a good proxy for management influence on fluxes of all three GHGs.

Spatial variability, especially of soil CH$_4$ and N$_2$O fluxes was as high as expected. Permanent spots with lower CH$_4$ uptake coincided with smaller inclination of the terrain on which chambers were placed. Thus, on sloping terrain, mean chamber fluxes of CH$_4$ should be estimated from an ensemble that is (a) sufficient in size, (b) represents the common species composition including hot spots occurring due to grazing and (c) representative for the terrain of the site. This is important since SWC is one of the major environmental drivers of CH$_4$ exchange. For soil fluxes of N$_2$O, we suggest the use of portable chambers in conjunction with recently developed laser spectrometers allowing for much shorter sampling times and therefore sampling of additional hot spots occurring during grazing and hot moments after fertilization.
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References


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