**Response to reviewers on our manuscript "Memory effects on greenhouse gas emissions (CO2, N2O and CH4) following grassland restoration?" by Lutz Merbold et al.**

We thank both reviewers for their critical assessment and provide a revised manuscript addressing the reviewer's comments. Throughout the following document, the reviewer's comment is stated first, followed by our response in *italic* font. We further attach a clean revised manuscript and a track changed version for ease of the editor and reviewers.

Reviewer #1:
The study presented here title "Memory effects on greenhouse gas emissions (CO2, N2O and CH4) following grassland renovation?" presents trace gas measurements from 5 years of a grazed and harvest pasture in Switzerland including a pasture restoration event. In general, this is a well written and worthwhile study. Few studies report all greenhouse gases, and even fewer for multiple years and covering infrequent management activities. I believe this to be of publication quality following consideration of my commentary below. I have separated my comments into major, moderate and minor/technical concerns based on importance and impact to the manuscript as I see it. I believe these can be dealt with by the authors and would further enhance the manuscript.

*We thank the reviewer for this positive assessment and share the opinion of few studies reporting on multiple years of GHG exchange measurements of the three GHGs covering specific management activities.*

Major concerns
1. CH4 fluxes: I have major concerns with the usage of the CH4 fluxes as presented in this manuscript. Firstly, while the authors present a comparison of N2O chamber and eddy covariance data (Figure 3), they do not for CH4. I believe this is likely as the comparison does not suggest any 1:1 relationship (based on my interpretation of Figure 4b). The authors then use this chamber data to derive annual CH4 fluxes for the years without EC data and assume to be comparable with the EC derived annual fluxes. From the data presented, I see no evidence to believe this to be the case (unlike N2O). Given the two chamber years suggest a small uptake of CH4, while the last three a release of CH4 coinciding with a difference in measurement methodology, I question whether the authors really believe these years are comparable. While the authors discuss these methodology differences in detail in the discussion section, and overall the contribution of CH4 to the GHG budget is small, I believe further attention needs to be given to this, and ideally the equivalent plot to figure 3b is presented for CH4. Based on the timing of management events (pasture restoration) and change in measurement methodology it could be easily interpreted as pasture restoration changes grassland CH4 exchange from an uptake to release.

*These are indeed relevant points and surely, we do not want to give the impression that pasture restoration changes grassland CH4 exchange from an uptake to release as this can not be proven by the data presented in this study (see following response). We had preferred to show a similar comparison as given for N2O, however the methane concentrations measurements were not reliable in 2013 due to a flame ionization detector (FID) malfunction in the gas chromatograph.*
*Overall, we also did not expect to find a similar relation between the methane flux measurements obtained by eddy covariance and chambers caused by the small magnitude of the fluxes measured. As stated in the original manuscript "*We calculated detection limits for the individual GHGs from our manual chambers following (Parkin et al., 2012). Detection limits were 0.34 ± 0.26 nmol $m^{-2}$ $s^{-1}$, 0.05 ± 0.02 nmol $m^{-2}$ $s^{-1}$, and 0.06 ± 0.06 µmol $m^{-2}$ $s^{-1}$ for $CH_4$, $N_2O$ and $CO_2$, respectively, clearly indicating that methane fluxes measured by GHG chambers in 2010/2011 were on average -0.16 ± 0.16 nmol $CH_4$ $m^{-2}$ $s^{-1}$, (see Table 2) and thus below the actual detection limit."
*However, we did compare our eddy covariance methane flux values (methane fluxes fluctuating around 0 with an overall range of -40 up to +40 nmol CH4 m-2 s-1 (Figure 4 b)) with the values reported by* (Felber et al., 2015) *from a similar grassland system in Western Switzerland.* (Felber et al., 2015) *have shown that such values measured by the EC technique represent a soil signal (Figure 6 in* Felber et al. 2015*).*

*Following this, we agree that we should not have computed annual sums for the years 2010/2011 for methane and will remove these in the revised manuscript. We will only present the gap-filled numbers for methane for 2012 -2014 and show the actual measurements derived with GHG flux chambers for the years 2010/2011 only (Figure 4b).*
*Overall, we would like to point out again that methane fluxes are of minor importance for the carbon and greenhouse gas budget of the site under the current management (see also our response to the second concern as well as the concern made by reviewer #2 on the influence of grazing animals on methane fluxes).*

2. The impact of grazing needs further consideration. While harvesting is more common in this study, the impact of grazing needs further clarification and/or modification of the presented results. Firstly, it is unclear to me how the grazing off-take was estimated (please clarify), and whether the deposition of excreta C was included in the C balances. While I'm not familiar with sheep grazing, at least for cattle this can be in the order of one-third of consumption, and therefore not an insignificant component (especially for 2014, Parcel A with 1769.9 kg C ha-1 of grazing removal according to table S1) and requiring acknowledgement of how this is currently dealt with, or included in the C balance (e.g. Table 2).
Furthermore, the authors state they did not detect any CH4 release with grazing (lines 432-433). Using the example of Parcel A in 2014, which was primarily grazed by cattle, and assuming _3% was converted and released as CH4 (e.g. Felber et al. (2016)), 53.1 kg C ha-1 would have been emitted from the grazers as CH4, which when converted to g CO2-eq m-2 calculated to 240 g CO2-eq m-2 or much larger than the 55 g CO2-eq m-2 reported in table. If this was not detected, then I suggest the authors reconsider how grazing related CH4 is dealt with in this manuscript given they are reporting ecosystem scale GHG budgets.

*Indeed, methane emissions from grazing animals need to be considered in annual budgets of methane and carbon. We argue that these are already accounted for in our data, since our observation boundary is the ecosystem and thus, we only include CH4 from animals when these are on the field. Grazing intensity was extremely low and only lasted for few days in the specific years (2010, 2011, 2014). Also, most of the grazing were sheep, and cattle were only present in 2014 in Parcel A for less than four weeks in total at an average stocking rate of 4.04 heads per hectare. Thus, the reviewer's statement that Parcel A was primarily grazed by cattle in 2014 is incorrect.*

*Furthermore, we are aware of the 3% assumption and while this approach could be taken, we were not able to follow the numbers presented by the reviewer. Possibly some additional explanation could be provided on how the values given were derived.*

*At the same time, we propose another approximation for methane emissions from enteric fermentation from cattle as follows and in relation to the study by* Felber et al. (2016). Felber et al. (2016) *reported an average of 404 g CH4 per head per day in a table summarizing several studies. Taking this average value and given the cattle occupied Parcel A (2.2 ha) for about four weeks with an average stocking rate of 4.04 heads per hectare (average of 12.5 and 5.3 for 2.2 hectares) our calculations are as follows.*

*Emissions for enteric methane = 404 g CH4/head/day * 4.04 head/ha * 30 days / 1000 to derive kg)*

*The total CH4 emissions calculated are thus 48.96 kg CH4 per ha (4.89 g CH4 m-2). When we convert this to C, we derive emissions of 4.07 kg CH4-C per ha (0.40 g CH4-C m-2). This would be the value we expect also to see with the EC flux tower under perfect conditions with a non-movable point source. Unfortunately, such perfect conditions are not the reality and we may not have captured all of these emissions due to shifts in wind direction, changes in turbulence as well as the actual animal movement out of the fetch. Also, as indicated by Felber et al. (2016) distance from the cow to the EC tower determines how much methane one measures with the EC tower. Moreover, 4.07 kg CH4-C ha-1 (0.40 g CH4-C m-2) are of minimal influence for both the C budget as well as for the GHG budget of the site (see Table 4). In order to clarify this point, we add this information on the issue of grazing in the revised manuscript.*

*Grazing removal was quantified experimentally by having areas in both parcels from which the animals were excluded. At the end of each grazing period, the grass in the enclosures was cut similar to the approach taken when estimating harvests with subsequent laboratory analysis for C and N. Grazing is included in the harvest in Table 4, as this is a removal of biomass from the system. The return of nutrients via excreta ((approx. 32% C,* (Felber et al., 2016)*) resembles a recycling of nutrients within the systems and associated GHG emissions would be included in the EC measurements. Following our previous argument of the very low stocking density, this is unlikely to have considerable effects to the results of the study.*

Moderate Concerns

3. The focus (or perhaps title?) of this manuscript needs sharpening. The title indicates a focus on pasture restoration which is matched by the abstract, yet much attention is given to methodological considerations. Specific goal (ii) states "briefly compare two different measurement techniques" however the first two-thirds of the discussion (i.e. not briefly) comments on this aspect! While important and noteworthy, either change the title/abstract, or return the primary focus of the discussion to management effects. Additionally, goal (iii) is not really explored in this manuscript – perhaps combine with goal (i)?

*Thank you for this suggestion. In the revised manuscript we combined the goals (i) and (iii) and shortened the discussion on the methodological aspects while giving more attention to the primary goal of the study. As a consequence, the former objective (iv) has now become the new objective (iii) (see the version of the manuscript with track changes).*

4. Providing a partial N budget provides little useful information. Including individual components is beneficial, but to sum them up as an incomplete "budget" is not. If the authors choose to retain the N budget, please include some further context including some ballpark estimates of the remaining components to aid interpretation.

*We agree that particularly in terms of N providing the partial budget is not as good as providing a full N budget. At the same time, we avoided after careful consideration, to provide a N budget with ballmark estimates as some fluxes would be largely uncertain due to little data availability from such systems (ie nitrate leaching) or overall limited data availability across agricultural systems (ie losses in form of NOx and N2). Yet we are aware that losses of nitrogen via ie NH3, N2, NOx can be much larger than the losses via N2O. Consequently, we rephrased the respective objective (previously iv now iii) to "*(iii) to provide a GHG budget *of the site". We further changed the wording from C and N budgets to C and N gains and losses with the losses we specifically refer to losses of N via $N_2O$.*

5. While N2O flux gap filling is difficult, the use of running medians may be problematic, and especially for gaps occurring during pulse emissions (e.g. the restoration period/fertiliser applications). The authors should comment on limitations of this approach, especially in the absence of any uncertainties (which I accept is rarely done in N2O flux studies so do not see them as a requirement here).

*This is a very relevant point made by the reviewer. The method chosen here, follows the approach taken by Hoertnagl et al. (2018), whom identified the running median being the most appropriate method to use if either large amounts of original data are available (ie as provided by the EC method) and/or if it is likely that the majority of N2O pulses have been covered by ie chamber measurements. Certainly, there are other options to fill N2O flux measurements and these were highlighted for instance in Nemitz et al. (2019) or Mishurov and Kiely (2011). Particularly, Nemitz et al. (2019) suggests linear interpolation for short gaps and daily averages to fill other gaps. For very long gaps more sophisticated and complex approaches such as machine learning tools are suggested.*
*Given that we aimed at deriving an annual budget which is relatively conservative we chose the running median approach. First of all, this way we are less likely to overestimate N2O emissions compared to ie the daily average approach. Linear interpolation would also have led to an overestimation of N2O*

*emissions particularly for the years 2010 and 2011 with few data points. Certainly, we see the lowest influence of gap filling errors for the years with EC measurements, whereas there may be a larger bias for the year with chamber measurements. Based on our 5-year observation period that indicated N2O emissions peaks during the growing season only and following fertilization events primarily (except 2012), we are confident that we covered the majority of these peaks during the years 2010 and 2011 when only chamber measurements were available. Thus, we decided to remain with the chosen approach as we do not think it is beneficial to state values which are likely to be more biased than the chosen approach.*

Minor/Technical Concerns

Lines 33-34: grazing is listed as both a regular and sporadic management activity. Please clarify which it is.

*We apologize for the mislead in wording and will rephrase as follows: "Grazing is a typical management activity in such intensive grassland. At our site, we observe grazing with either sheep or cattle for few days at the beginning or end of most years."*

Line 37: Missing the word "out" (or similar) after "carried".

*Done*

Lines 86-89: Why did you hypothesis continuous losses of CO2? Several studies (e.g. (Rutledge et al., 2017; Ammann et al., 2020, etc) show CO2 uptake in restoration and later years.

*Thank you for pointing this out. Actually, we had the hypothesis of increased CO2 uptake already in the manuscript (L. 89-90). We reworded these lines as follows: Prior to our measurements we hypothesized short-term losses of $CO_2$ after restoration and more continuous losses of primarily $N_2O$ following dramatic managements events such as ploughing occurring at irregular time intervals. We further hypothesized an increased carbon uptake strength compared to the pre-ploughing years.*

Lines 89-90: If you expect CO2 losses (as per the above point), why would you expect a C gain? Please adjust this and align with the previous sentence to clarify your hypothesis.

*See our comment to the previous remark made by the reviewer.*

Line 108: Do you mean CH4 emissions from the land or the grazers? In fact, this point needs clarity throughout the manuscript – are the grazers included within the system boundary, and therefore their emissions?

*We actually refer to both, land emissions/uptake as well as CH4 emissions from grazers. In terms of system boundaries, these are set to the ecosystem here, thus we account for the GHG emissions made by grazers (CH4 from enteric fermentation, as well as CH4 and N2O from excreta) for the years 2012-2014. Given that stocking rate was low and the actual time of grazing short we expected little effects of grazing on the budget while still aiming at being inclusive as we wanted to include all the management activities occurring in this field. We further included the offtake due to grazing in the budget calculations. The recycling of nutrients from grazing animals and their deposits is included in the eddy covariance measurements. While this may not be the case for 2011/2012. Given the small stocking rate and as explained before this is likely of minor importance and surely will not change the results.*

Lines 123-127: this sentence is very clunky – suggest reviewing.

*We are not sure what the reviewer refers to here as these are two sentences in the original manuscript. However, in order to increase the flow of reading the suggested lines will be adjusted as follows in the revised manuscript. "The study by Hörtnagl et al. (2018) further elaborated the variability in management intensity and related variations in GHG exchange across sites, stressing the need for more* *case studies based on continuous GHG observations to improve existing knowledge and close remaining knowledge gaps. To complete the picture on factors impacting ecosystem GHG exchange, irregular occurring events such as dry spells or extraordinary wet periods can further lead to enhanced or reduced GHG emissions (Chen et al., 2016; Hartmann and Niklaus, 2012; Hopkins and Del Prado, 2007; Mudge et al., 2011; Wolf et al., 2013)"*

Line 130: "adaptations" should be "adaptation" (no "s").

*Done*

Line 137: "respectively" is not needed – please delete.

*Done*

Lines 232-234: If an LI-7500 (rather than LI-7500A) was the self-heating correction applied?

*That was an oversight and we added the A. The correction was applied.*

Lines 241-249: It was unclear to me what QA/QC procedures were applied to the raw (10/20Hz) and which to the 30-minute data. I suggest improving the clarity here.

*We rephrased this section by clearly distinguishing between raw data and raw time series (high frequency) and specifically state when we refer to 30-minute data.*

Line 248: what was considered the physically plausible range? Please include this information.

*Done*

Line 280: Order of words: "no longer closed" should be "closed no longer".

*Done*

Line 314: Remove the word "Up"

*Done*

Line 413: Insert the word "and" between "(Figure 1c)" and "temperatures".

*Done*

Lines 477-478: I think the before and after restoration periods should be separated. I don't believe averaging the two periods to be fair as part of the purpose of restoration is to improve growth, and therefore modification of CO2 exchange should also be expected.

*This may be a misunderstanding. We clearly differentiate between periods as indicated in the original manuscript under sections 3.3. CO2 exchange and N2O exchange as well as under section 3.4.*

Line 480: According to Table 2, CH4 emissions for 2013 and 2014 were actually >1 – please correct.

*This is correct for the years 2012, 2013 and 2014 and the values seen are very similar to values reported by* Felber et al. (2015)*. Given the magnitude of the other GHG fluxes, methane remains a minor contribution to the GWP budgets.*

Line 538: Correct the format of the reference

*Done*

Line 579-580: Are you referring to the measured CO2 exchange to be +/- 50 g C m-2 y-1, or the uncertainty? This sentence is very unclear as no uncertainty has been presented, so please clarify.

*This refers to the statement made by Baldocchi et al. (2003), who stated that annual numbers presented from EC measurements can vary by as much as +/- 50 g C m-2 y-1. Thus, we want to encourage that this is an uncertainty anyone should keep in mind when evaluating annual budgets derived by the EC technique.*

Table 1: I find the "max data availability" columns repetitive – perhaps just a single column of this data?

*Good point, thank you! We removed the repetitive statement of numbers in the revised manuscript and also removed the columns presenting the water fluxes as these are not referred to in the manuscript.*

Table 4: I suspect the labelling of Parcels A and B for both fertilizer and harvest are not correct. As written, fertilizer was only applied to Parcel A, and Harvest to Parcel B. Please correct is appropriate.

*This is actually only an incorrect labelling and should refer to harvest for Parcel A and B as well as fertilizer for Parcel A and B. This has been corrected.*

References
Ammann, C., Neftel, A., Jocher, M., Fuhrer, J., Leifeld, J., 2020. Effect of management and weather variations on the greenhouse gas budget of two grasslands during a 10- year experiment. Agric. Ecosyst. Environ. 292.
Felber, R., Bretscher, D., Münger, A., Neftel, A., Ammann, C., 2016. Determination of the carbon budget of a pasture: effect of system boundaries and flux uncertainties. Biogeosciences 13, 2959-2969.
Rutledge, S., Wall, A.M., Mudge, P.L., Troughton, B., Campbell, D.I., Pronger, J., Joshi, C., Schipper, L.A., 2017. The carbon balance of temperate grasslands part II: The impact of pasture renewal via direct drilling. Agric. Ecosyst. Environ. 239, 132-142.

*We thank the reviewer for pointing us towards these references and we refer to these in the revised version of the manuscript.*

Reviewer #2:

We would like to thank reviewer #2 for the overall positive evaluation and for providing feedback on the points that the reviewer encourages to be addressed. Our responses to the questions/concerns are given in italic font.

The manuscript "Memory effects on greenhouse gas emissions (CO2, N2O and CH4) following grassland restoration?" by Merbold et al. is a well written longterm study of GHGs from a grazed grassland system in Switzerland. The team have used a mixture of measurement methods over a 5 year period to get a very good picture of a full GHG budget for the field. This is a very valuable study as such longterm observations are rare and it answers some questions that are not well studied.
I found the manuscript interesting to read, and it was written to a very high standard and I do believe that it should be published after some amendments.
I do have some comments that I feel should be addressed by the authors that I believe would improve the quality and usefulness of the study for others. Although these comments are numerous and not entirely simple to address, if the authors can amend their study to incorporate them I feel the work would benefit greatly.

*Thank you for the positive evaluation and we suggest ways forward point by point below.*

A large assumption made by the study is that the eddy covariance measurements are entirely truthful of the conditions in the field. It has been observed in the past that long-term carbon budgets derived from eddy covariance can be biased due to assumptions made by the method. Often negative carbon fluxes are reported in similar systems, however, when investigating deep soil cores there was found to be no significant difference in C content of the soil (see Jones et al., doi:10.5194/bg-14-2069-2017 for one such study). The manuscript does not provide evidence of the C stock in the soil beyond the Eddy C measurements to back up the evidence which would have made it a much more significant study. This does not invalidate the study by any means, but without clarification of potential uncertainties, it increases the danger that the study provides "concrete" evidence of mitigation methods (i.e. grazing animals is a carbon sink) that has been used recently by advocates of the meat industry to justify the long-term environmental aspects of livestock farming. I would advise a short message of discussion to highlight that there is room for error in the measurements and that soil carbon was not measured to validate the measurements. Alternatively, if the soil measurements are there, please include them.

*Indeed, we agree that soil inventories should be linked to EC fluxes more often, particularly since EC measurements are often seen as entirely truthful. We are confident that the EC method is a valuable and powerful tool to investigate C fluxes at ecosystem scale – not necessarily the exact entire field as suggested in the literature ( Hill et al. 2016 https://doi.org/10.1111/gcb.13547). Yet, it allows to derive a general view on whether an ecosystem is likely to gain/loose carbon. We further agree that continuous flux measurements and thus budgets should be validated with other independent methods, ie a soil inventory. Yet, determining changes in soil C/N is similarly not trivial and takes considerable time as suggested by ie. Schrumpf et al. 2011 https://doi.org/10.5194/bg-8-1193-2011. Additional, within this specific project we were not able to carry out a resampling of the soils while further advocating for this in follow-up projects. Multiple approaches to estimate the uncertainty in EC flux measurements as well as in gap-filling methods are available (ie Post et al. 2015 https://doi.org/10.5194/bg-12-1205-2015, Vitale et al 2019 https://doi.org/10.1007/s00477-019-01664-4, Hollinger and Richardson 2005 https://doi.org/10.1093/treephys/25.7.873, Nicolini et al. 2018, https://doi.org/10.1016/j.agrformet.2017.09.025) pointing towards the reliability of EC measurements. As we primarily provide a GHG budget – after having revised the objectives – these numbers do not represent a full farm-scale assessment.*

I do not agree with the way that the N2O flux data has been handled in the study. N2O fluxes measured using chambers almost always follow a log-normal distribution in space, so any data analysis must take this into account when handling means and uncertainties. A simple arithmetic mean with associated uncertainty (not sure what the error bars on Fig 3 and 4 represent?) will not be an adequate way to represent this data (although commonly used wrongly in previous studies). This will result in a skewing of the data and large overestimates in minimum confidence intervals and underestimations of maximum confidence intervals. An example is when uncertainties of N2O cross the negative threshold when no observations of flux dip below zero. This is not a satisfactory way to present the data. I recommend using a more sophisticated analysis technique and showing 95% confidence intervals where possible for a thorough comparison of the measurement techniques.

*We thank the reviewer for the critical assessment. Our approach followed the method used by Hoertnagl et al. 2018 https://doi.org/10.1111/gcb.14079. Hoertnagl et al. (2018), whom identified the running median being the most appropriate method to use if either large amounts of original data are available (ie as provided by the EC method) and/or if it is likely that the majority of N2O pulses have been covered by ie chamber measurements. Certainly, there are other options to fill N2O flux measurements and these were highlighted for instance in Nemitz et al. (2019) or Mishurov and Kiely (2011). Particularly, Nemitz et al. (2019) suggests linear interpolation for short gaps and daily averages to fill other gaps. For very long gaps more sophisticated and complex approaches such as machine learning tools are suggested.*

*Given that we aimed at deriving an annual budget which is relatively conservative we chose the running median approach. First of all, this way we are less likely to overestimate N2O emissions compared to ie the daily average approach. Linear interpolation would also have led to an overestimation of N2O emissions particularly for the years 2010 and 2011 with few data points. Certainly, we see the lowest influence of gap filling errors for the years with EC measurements, whereas there may be a larger bias for the year with chamber measurements. Based on our 5-year observation period that indicated N2O emissions peaks during the growing season only and following fertilization events primarily (except 2012), we are confident that we covered the majority of these peaks during the years 2010 and 2011 when only chamber measurements were available. Thus, we decided to remain with the chosen approach as we do not think it is beneficial to state values which are likely to be more biased than the chosen approach.*

L303: Due to the log-normal distribution of N2O emissions measured using chambers, most measurements will be very close to zero and ppb differences in gas samples will hover around detection limits of the analysis instrument. In such cases, the R2 value of the fits will be very low for many, but the regression between points will still be valid (effectively an average of the instrument noise with a slope near zero). By cutting data with R2 lower than 0.8 I assume that a very large number of small fluxes are removed from the dataset. If this is the case I would recommend a threshold on this QC method, or a more detailed explanation of what impact this had on the data in the text if this is not the case (as I read it, the method would likely contribute to a large bias is flux estimates).

*We implemented thorough QC criteria concerning the N2O flux calculations. All the details have been in detail provided in Imer et al. (2013), including the R2 threshold and how many data points were dismissed. Overall, the low fluxes being part of our observations were not being the limit of detection and have thus been included in this study.*

Uncertainties in cumulative emissions are not presented which makes it difficult to compare with other studies or what impact gap-filling and weather may have had on the study. This should be easily manageable for CO2 for which models exist, and probably for CH4 using simple gap-filling as it was found to be approximate zero throughout the study. I understand that there is no definitive way to gap-fill N2O, however a running median is not a statistically defensible way to “model” data. As a result no uncertainty will be calculated from this method. If the authors want to estimate uncertainties in cumulative N2O fluxes, they will have to develop a more sophisticated approach to gap-filling.

*We agree with the reviewer that there are different approaches to gapfill GHG flux data. Certainly, the gapfilling approaches for CO2 and CH4 are better developed than for N2O. The running median approach was chosen, following Hoertnagl et al. 2018 (see above) as this seemed at the time being the best possible way to fill N2O flux data gaps given the ecosystem observed.*

I feel a nitrogen budget without NH3, NOx and N2 is not very useful. Combined, these gases will likely contribute approx. 50% of nitrogen losses from the system. Perhaps a better way to confer N losses is to calculate the emission factors of the fertiliser applications, as that is a more generally used term for such activities in literature and is a better description of the presented results in the study.

*We are in full agreement with the reviewer that other N compounds build a large part of the N budget. We thus adjusted the manuscript to only show the GHG budget and avoid stating a full N budget as this could be only based on very rough estimates. We also decided to adjust the text and mention only C and N gains/losses.*

Is there a way to estimate the N content of the fodder/grass on the field before tillage to assess the emissions from the herbage being tilled into the soil?

*We have thought about this too when preparing the manuscript and realized that we had not taken such measurements. However, to our current knowledge the additional N being incorporated into soil during tillage should be very small due the very low vegetation height at this time of the year.*

Does the carbon budget take into account vehicle use? Is it insignificant or does tractor diesel have a role to play?

*The currently presented budget does not include C emissions from vehicle use for two reasons: (1) the hours farm vehicles are being used on this field are very limited over the course of the year given the small size of the fields (negligible). The negligibility of these emissions was further underlined (2) by a MSc thesis that investigated full farmgate budgets in the years prior this study.*

L225: Can you explain what you mean by an internal reference cell in the instrument for the QCLAS? To my knowledge, these cells are used to find absorption lines on the spectra and not for calibration as they leak over time. The QCLAS system typically does not require calibration as it operates on the principles that the absorption follows Hitran quantum mechanics laws.

*Thank you for this comment and this seems to be a misunderstanding of what we have written. We stated that the infrared gas analyser was calibrated regularly, while we also wrote that the QCLAS was fitted against an internal reference cell. In order to create better clarity we changed this sentence as follows: "The QCLAS did not need calibration due to its operating principles, and an internal reference cell (mini-QCL manual, Aerodyne Research Inc., Billerica, MA, USA) eased finding the absorption spectra after each restart of the analyzer."*

Some minor corrections
L283: I think there is a bit of wording here that is confusing. Flushing the chamber with the syringe isn't technically correct. I think it would be better to say that the syringe was used to pump the chamber to circulate the air to avoid the concentration gradients?

*Done*

L471: here the order of the sentences makes it sound like CH4 contributed to 70% of the budget. Please re-order.

*Done, we added "the contribution of* $CO_2$*"*

L606: Change highlight to highlights

*Done*

# Memory effects on greenhouse gas emissions ($CO_2$, $N_2O$ and $CH_4$) following grassland restoration?

Lutz Merbold[1,2*], Charlotte Decock[3+], Werner Eugster[1], Kathrin Fuchs[4], Benjamin Wolf[4], Nina Buchmann[1] and Lukas Hörtnagl[1]

[1] Department of Environmental Systems Science, Institute of Agricultural Sciences, Grassland Sciences Group, ETH Zurich, Universitätsstrasse 2, 8092 Zürich, Switzerland

[2] Mazingira Centre, International Livestock Research Institute (ILRI), Old Naivasha Road, PO Box 30709, 00100 Nairobi, Kenya

[3] Department of Environmental Systems Science, Institute of Agricultural Sciences, Sustainable Agro-ecosystem Group, ETH Zurich, Universitätsstrasse 2, 8092 Zürich, Switzerland, [+] now at: Department of Natural Resources Management and Environmental Sciences, California State University, San Luis Obispo, California, USA

[4] Institute for Meteorology and Climate Research (IMK-IFU), Karlsruhe Institute of Technology (KIT), Kreuzeckbahnstrasse 19, 82467 Garmisch-Partenkirchen, Germany

[*] corresponding author: lutz.merbold@gmail.com

**Keywords:** eddy covariance, global warming potential, manual static chamber, management, background greenhouse gas emissions, ploughing, fertilization

## Abstract

A five-year greenhouse gas (GHG) exchange study of the three major gas species ($CO_2$, $CH_4$ and $N_2O$) from an intensively managed permanent grassland in Switzerland is presented. Measurements comprise two years (2010/2011) of manual static chamber measurements of $CH_4$ and $N_2O$, five years of continuous eddy covariance (EC) measurements ($CO_2$/$H_2O$ – 2010-2014) and three years (2012-2014) of EC measurement of $CH_4$ and $N_2O$. Intensive grassland management included both regular and sporadic management activities. Regular management practices encompassed mowing (3-5 cuts per year) with subsequent organic fertilizer amendments and occasional grazing whereas sporadic management activities comprised

grazing or similar activities. The primary objective of our measurements was to compare pre-ploughing to post-ploughing GHG exchange and to identify potential memory effects of such a substantial disturbance on GHG exchange and carbon (C) and nitrogen (N) gains/losses. In order to include measurements carried out with different observation techniques, we tested two different measurement techniques jointly in 2013, namely the manual static chamber approach and the eddy covariance technique for $N_2O$, to quantify the GHG exchange from the observed grassland site.

Our results showed that there were no memory effects on $N_2O$ and $CH_4$ emissions after ploughing, whereas the $CO_2$ uptake of the site considerably increased when compared to post-restoration years. In detail, we observed large losses of $CO_2$ and $N_2O$ during the year of restoration. In contrast, the grassland acted as a carbon sink under usual management, i.e. the time periods (2010-2011 and 2013-2014). Enhanced emissions/emission peaks of $N_2O$ (defined as exceeding background emissions $0.21 \pm 0.55$ nmol m$^{-2}$ s$^{-1}$ (SE = 0.02) for at least two sequential days and the seven-day moving average exceeding background emissions) were observed for almost seven continuous months after restoration as well as following organic fertilizer applications during all years. Net ecosystem exchange of $CO_2$ ($NEE_{CO2}$) showed a common pattern of increased uptake of $CO_2$ in spring and reduced uptake in late fall. $NEE_{CO2}$ dropped to zero and became positive after each harvest event. Methane ($CH_4$) exchange fluctuated around zero during all years. Overall, $CH_4$ exchange was of negligible importance for both, the GHG budget as well as for the carbon budget of the site.

Our results stress the inclusion of grassland restoration events when providing cumulative sums of C sequestration potentials and/or global warming potentials (GWPs). Consequently, this study further highlights the need for continuous long-term GHG exchange observations as well as the implementation of our findings into biogeochemical process models to track potential GHG mitigation objectives as well as to predict future GHG emission scenarios reliably.

## 1 Introduction

Grassland ecosystems are commonly known for their provisioning of forage, either directly via grazing of animals on site, or indirectly by regular biomass harvest and preparation of silage or hay. Simultaneously, grasslands have further been acknowledged for their greenhouse gas (GHG) mitigation and soil carbon sequestration potential (Lal, 2004; Smith et al., 2008). However, greenhouse gas emissions from grasslands, particularly $N_2O$ and $CH_4$ have been shown to offset net carbon dioxide equivalent ($CO_2$-eq.) gains (Ammann et al., 2020; Dengel et al., 2011; Hörtnagl et al., 2018; Hörtnagl and Wohlfahrt, 2014; Merbold et al., 2014; Schulze et al., 2009). Still, datasets containing continuous measurements of all three major GHGs ($CO_2$, $CH_4$ and $N_2O$) in grassland ecosystems remain limited (Hörtnagl et al., 2018), include a single GHG only, or focus on specific management activities (Fuchs et al., 2018; Krol et al., 2016). At the same time such datasets are extremely valuable by providing key training datasets for biogeochemical process models (Fuchs et al., 2020a).

Here we investigate the GHG exchange of the three major trace gases ($CO_2$, $CH_4$ and $N_2O$) over five consecutive years in a typical managed grassland on the Swiss plateau. Our study includes the application of traditional GHG chamber measurements and state-of-the-art GHG concentration measurements with a quantum cascade laser absorption spectrometer and a sonic anemometer in an eddy covariance setup (Eugster and Merbold, 2015). Prior to our measurements we hypothesized short-term losses of $CO_2$ and more continuous losses of primarily $N_2O$ following dramatic managements events such as ploughing occurring at irregular time intervals. We further hypothesized an increased carbon uptake strength compared to the pre-ploughing years. Methane emissions were hypothesized to be of minor importance due to the limited time of grazing animals on site (Merbold et al., 2014).

Up to date the majority of greenhouse gas exchange research has focused on $CO_2$, with less focus on the other two important GHGs $N_2O$ and $CH_4$, even though an increased interest in these other gas species has become visible in recent years (Ammann et al., 2020; Ball et al., 1999; Cowan et al., 2016; Krol et al., 2016; Kroon et al., 2007, 2010; Necpálová et al., 2013; Rutledge et al., 2017). The existing exceptions are often referred to as "high-flux" ecosystems, namely wetlands and livestock production system in terms of $CH_4$ (Baldocchi et al., 2012; Felber et al., 2015; Laubach et al., 2016; Teh et al., 2011) and agricultural ecosystems such as bioenergy system with considerable $N_2O$ emissions (Cowan et al., 2016; Fuchs et al., 2018; Krol et al., 2016; Skiba et al., 1996, 2013; Wecking et al., 2020; Zenone et al., 2016; Zona et al., 2013). Agricultural ecosystems and specifically grazed systems are characterized by GHG emissions caused through anthropogenic activities. These activities lead to changes in GHG emission patterns and include harvests, amendments of fertilizer and/or pesticides and less frequently occurring ploughing, harrowing and re-sowing events. While ploughing has been shown to lead to considerable short-term emissions of $CO_2$ and $N_2O$ (Buchen et al., 2017; Cowan et al., 2016; Hörtnagl et al., 2018; MacKenzie et al., 1997; Merbold et al., 2014; Rutledge et al., 2017; Vellinga et al., 2004), regular harvests have been shown to lead to increased $CO_2$ uptake (Zeeman et al., 2010) and grazing leads to large $CH_4$ emissions (Dengel et al., 2011; Felber et al., 2015). Other studies showed contrary results with reduced $N_2O$ emissions following ploughing of a drained grassland when compared to a fallow in Canada (MacDonald et al., 2011).

Still, the full range of management activities occurring in intensively managed grasslands and their respective impact on GHG exchange has not been investigated in detail. In a recent synthesis including grasslands located along an altitudinal gradient in Central Europe, Hörtnagl et al. (2018) highlighted the most important abiotic drivers of $CO_2$ (light, water availability and temperature), $CH_4$ (soil water content, temperature and grazing) and $N_2O$ exchange (water filled pore space and soil temperature). The study by Hörtnagl et al. (2018) further elaborated the variation in management intensity and related variations in GHG exchange across sites, stressing the need for more case studies based on continuous GHG observations to improve existing knowledge and close remaining knowledge gaps. To complete the picture on factors driving ecosystem GHG exchange, irregular occurring events such as dry spells or extraordinary wet periods can further lead to enhanced or reduced GHG emissions (Chen et al., 2016; Hartmann and Niklaus, 2012; Hopkins and Del Prado, 2007; Mudge et al., 2011; Wolf et al., 2013).

While drought has been shown to reduce $CO_2$ uptake in forests (Ciais et al., 2005) whereas dry spells did not affect $CO_2$ uptake in grasslands (Wolf et al., 2013), flooding leads primarily to enhanced $CH_4$ emissions (Knox et al., 2015) and large precipitation events can lead to plumes of $N_2O$ (Fuchs et al., 2018; Zona et al., 2013) similar to freeze-thaw events (Butterbach-Bahl et al., 2011; Matzner and Borken, 2008) to name only some examples. Consequently, understanding both, anthropogenic impacts such as management besides environmental impacts on ecosystem GHG exchange, are crucially important to suggest appropriate climate change mitigation as well as adaptation strategies for future land management with ongoing climate change.

Different measurement techniques to quantify the net GHG exchange in ecosystems are known and the most common approaches are either GHG chamber measurements or the eddy covariance (EC) technique. Static manual chamber measurements have been used for more than a century to quantify $CO_2$ emissions (Lundegardh, 1927) and their application has further been expanded during the last decades to quantify losses of the three major GHGs, $CO_2$, $N_2O$ and $CH_4$ from soils (Imer et al., 2013; Pavelka et al., 2018a; Pumpanen et al., 2004; Rochette et al., 1997). Even though more complex in technology and assumptions made before carrying out measurements, the eddy covariance (EC) technique has become a valuable tool to derive ecosystem integrated $CO_2$ and $H_2O_{vapour}$ exchange across the globe (Baldocchi, 2014; Eugster and Merbold, 2015). The technique has been further extended to continuous measurements of $CH_4$ and $N_2O$ with the development of easy field-deployable fast-response analyzers during the last decade (Brümmer et al., 2017; Felber et al., 2015; Kroon et al., 2007; Nemitz et al., 2018a; Wecking et al., 2020). Each of the two approaches has its strengths and weaknesses and it is beyond the scope of this study to discuss each of them in detail. However, we refer to a set of reference papers highlighting the advantages and disadvantages of each technique separately (chambers: (Ambus et al., 1993; Brümmer et al., 2017; Pavelka et al., 2018a); eddy covariance: (Baldocchi, 2014; Denmead, 2008; Eugster and Merbold, 2015; Nemitz et al., 2018).

The overall objective of this study was to investigate the net GHG exchange ($CO_2$, $CH_4$ and $N_2O$) before and after grassland restoration and thus fill existing knowledge gaps caused by limited amounts of available GHG exchange data from intensively managed grasslands. The specific goals were: (i) to assess pre- and post-ploughing GHG exchange in a permanent grassland in central Switzerland accounting for changes in GHG exchange following frequent management activities; (ii) to compare two different measurement techniques, namely eddy covariance and static greenhouse gas flux chambers to quantify the GHG exchange in a business-as-usual year; and (iii) to provide a five year GHG budget of the site and quantify losses/gains of C and N. Based on our results we provide suggestions for future research approaches to further understand ecosystem GHG exchange, to mitigate GHG emissions and to ensure nutrient retention at the site for sustainable production from permanent grasslands in the future.

# 2 Material and Methods

## 2.1 Study site

The Chamau grassland site (Fluxnet identifier - CH-Cha) is located in the pre-alpine lowlands of Switzerland at an altitude of 400 m a.s.l. (47°12′ 37″N, 8°24′38″E) and characterized by intensive management (Zeeman et al., 2010). The site is divided into two parcels (Parcel A and B) with occasionally slightly different management regimes [see also *Fuchs et al., 2018*]. Mean annual temperature (MAT) is 9.1 °C, and mean annual precipitation (MAP) is 1151 mm. The soil type is a Cambisol with a pH ranging between 5 and 6, a bulk density between 0.9 and 1.3 kg $m^{-3}$ and a carbon stock of 55.5–69.4 t C $ha^{-1}$ in the upper 20 cm of the soil. The common species composition consists of Italian ryegrass (*Lolium multiflorum*) and white clover (*Trifolium repens L.*). For more details of the site we refer to Zeeman et al., (2010).

CH-Cha is intensively managed, with activities being either recurrent – referred to as usual/regular - or sporadic. Usual management refers to regular mowing and subsequent organic fertilizer application in form of liquid slurry (up to 7 times per year). In addition, the site is occasionally grazed by sheep and cattle for few days in early spring and/or fall (H.-R. Wettstein personal communication, Table S1). Sporadic activities aim at maintaining the typical fodder species composition and comprise reseeding, herbicide and pesticide application or irregular ploughing and harrowing on an approximately decadal timescale (Merbold et al., 2014). By such activity, mice are eradicated and a high-quality sward for fodder production is re-established following weed contamination. Specific information on management activity (timing, type of management, amount of biomass harvested) were reported by the farmers on site (Table S1). Additionally, representative samples of organic fertilizer were collected shortly before fertilizer application events and sent to a central laboratory for nutrient content analysis (Labor fuer Boden- und Umweltanalytik, Eric Schweizer AG, Thun, Switzerland). Harvest estimates were compared to estimates based on destructive sampling of randomly chosen plots (n = 10) in the years 2010, 2011, 2013 and 2014. The amount of harvested biomass in the year 2012 was based on a calibration of the values presented by the farmer in comparison to the on-site destructive harvests in previous and following years (Table S1).

## 2.2 Eddy covariance flux measurements

### *2.2.1 Eddy covariance setup*

The specific site characteristics with two prevailing wind directions (North-northwest and South-south east) allows continuous observations of both management parcels. It is noteworthy, that the separation of the two parcels is done exactly at the location of the tower. See Zeeman et al. (2010) and Fuchs et al. (2018) for further details. The eddy covariance setup consisted of a three-dimensional sonic anemometer (2.4 m height, Solent R3, Gill Instruments, Lymington, UK), an open-path infrared gas analyzer (IRGA, LI-7500A, LiCor Biosciences, Lincoln, NE, USA) to measure the concentrations of $CO_2$ and $H_2O_{vapour}$ and a recently developed continuous-wave quantum cascade laser absorption spectrometer (mini-QCLAS - $CH_4$, $N_2O$, $H_2O$ configuration, Aerodyne Research Inc., Billerica, MA, USA) to measure the concentrations of $CH_4$, $N_2O$, and $H_2O_{vapour}$. 3D wind components (u, v, w), $CO_2$ and $H_2O_{vapour}$ concentration data from the IRGA were collected at a 20 Hz time interval, whereas concentrations of $CH_4$ and $N_2O$ were collected at a 10 Hz rate from the QCLAS. The QCLAS provided the dry mole fraction for both trace gases ($CH_4$ and $N_2O$), and data were transferred to the data acquisition system (MOXA embedded Linux computer, Moxa, Brea, CA, USA) via an RS-232 serial data link and merged with the sonic anemometer and IRGA data streams in near-real time (Eugster and Plüss, 2010). Important to note is that the QCLAS was stored in a temperature-controlled box (temperature variation during the course of a single day was reduced to $< 2$ K) and located approximately 4 meters away from the EC tower to avoid long tubing. Total tube length from the inlet near the sonic anemometer to the measurement cell was 6.5 m. The inlet consisted of a coarse sinter filter (common fuel filter used in model cars) and a fine vortex filter (mesh size 0.3µm and a water trap) installed directly before the QCLAS. Filters were changed monthly or if the cell pressure in the laser dropped by more than 2 torr. Flow rate of approximately 15 l $min^{-1}$ was achieved with a large vacuum pump (BOC Edwards XDS-35i, USA and TriScoll 600, Varian Inc., USA – the latter was used during maintenance of the Edwards pump). The pumps were maintained annually and replaced twice due to malfunction during the observation period. The infrared gas analyzer was calibrated to known concentrations of $CO_2$ and $H_2O$ each year. The QCLAS did not need calibration due to its operating principles, and an internal reference cell (mini-QCL manual, Aerodyne Research Inc., Billerica, MA, USA) eased finding the absorption spectra after each restart of the analyzer.

*2.2.2 Eddy covariance flux processing, post-processing and quality control*

Raw fluxes of $CO_2$, $CH_4$, $N_2O$ ($F_{GHG}$, µmol $m^{-2}$ $s^{-1}$) were calculated as the covariance between turbulent fluctuations of the vertical wind speed and the trace gas species mixing ratio, respectively (Baldocchi, 2003; Eugster and Merbold, 2015). Open-path infrared gas analyzer (IRGA) $CO_2$ measurements were corrected for water vapor transfer effects (Webb et al., 1980). A 2-dimensional coordinate rotation was performed to align the coordinate system with the mean wind streamlines so that the vertical wind vector $\acute{w} = 0$. Turbulent departures were calculated by Reynolds (block) averaging of 30 min data blocks. Frequency response corrections were applied to raw fluxes, accounting for high-pass and low-pass filtering for the $CO_2$ signal based on the open-path IRGA as well as for the closed-path $CH_4$ and $N_2O$ data (Fratini et al., 2014). All fluxes were calculated using the software *EddyPro* (version 6.0, LiCor Biosciences, Lincoln, NE, USA) (Fratini and Mauder, 2014).

The quality of half-hourly raw time series was assessed during flux calculations following (Vickers and Mahrt, 1997). Raw data were rejected if (a) spikes accounted for more than 1 % of the time series, (b) more than 10 % of available data points were significantly different from the overall trend in the 30 min time period, (c) raw data values were outside a plausible range (± 50 µmol $m^{-2}$ $s^{-1}$ for $CO_2$, ± 300 nmol $m^{-2}$ $s^{-1}$ for $N_2O$ and ± 1 µmol $m^{-2}$ $s^{-1}$ for $CH_4$) and (d) window dirtiness of the IRGA sensor exceeded 80 %. Only raw data that passed all quality tests were used for flux calculations.

Half-hourly flux data were rejected if (e) fluxes were outside a physically plausible range (ie. +/- 50 µmol $m^{-2}$ $s^{-1}$ for $CO_2$) (f) the steady state test exceeded 30 % and (g) the developed turbulent conditions test exceeded 30 % (Foken et al., 2006). Between 1st January 2010 and 31st December 2014 64572 (88% of all possible data) 30-min flux values were calculated for $CO_2$, of which 42865 (57.8%) passed all quality tests and were used for analyses in the present study (Table 1). The amount of available flux values for $N_2O$ and $CH_4$ were less, since we were only capable to continuously measure both gases from 2012 onwards (Table 1). Flux values in this manuscript are given as number of moles of matter/mass per ground surface area and unit time. Negative fluxes represent a flux of a specific gas species from the atmosphere into the ecosystem, whereas positive fluxes represent a net loss from the system.

## 2.3 Static greenhouse gas flux chambers

### *2.3.1 Manual static GHG chamber setup*

Static manual opaque GHG chambers were installed within the footprint of the site to measure soil fluxes in 2010 and 2011 (n =16) as well as during summer 2013 (n = 10). The chambers were made of polyvinyl chloride tubes with a diameter of 0.3 m (Imer et al., 2013). The average headspace height was 0.136 m ± 0.015 m and average insertion depth of the collars into the soil was 0.08 m ± 0.05 m. During sampling days with vegetation larger than 0.3 m inside the chamber, collar extensions (0.45 m) were used (2013 only). Chamber lids were equipped with reflective aluminium foil to minimize heating inside the chamber during the period of actual measurement. Spacing between the chambers was approximately seven m and an equal number of chambers were installed in each parcel. For further details we refer to Imer et al. (2013). Chamber measurements were carried out on a weekly basis during the growing season in all three years (2010, 2011 and 2013), and at least once a month during the winter season in 2010 and 2011. More frequent measurements of $N_2O$ emissions (every day) were performed following fertilization events in 2013 for seven consecutive days after each event. Besides this, an intensive measurement campaign lasting 48 hours (two-hour measurement interval) was carried out in September 2010.

*2.3.2 GHG concentrations measurements*

During each chamber closure four gas samples were taken, one immediately after closure and then in approximately ten-minute time increments. With this approach, we guaranteed that the chambers were closed no longer than 40 minutes to avoid potential saturation effects. Syringes (60 ml volume) were inserted into the chambers lid septa to take the gas samples. The collected air sample was injected into pre-evacuated 12 ml vials (Labco Limited, Buckinghamshire, UK) in the next step. Prior to the second, third and fourth sampling of each chamber, the air in chamber headspace was circulated 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 respective $CO_2$, $CH_4$ and $N_2O$ concentrations in the lab as soon as possible after sample collection and not stored for more than a few days. Gas sample analysis was performed with a gas chromatograph (Agilent 6890 equipped with a flame ionization detector, a methanizer - Agilent Technologies Inc., Santa Clara, USA - and an electron capture detector – SRI Instruments Europe GmbH, 53604 Bad Honnef, Germany) as described by Hartmann and Niklaus (2012).

*2.3.3 GHG chamber flux calculations and quality control*

GHG fluxes were calculated based on the rate of gas concentration change inside the chamber headspace. Data processing, which included flux calculation and quality checks, was carried out with the statistical software R (R Development Core Team, 2010). Thereby the rate of change was calculated by the slope of the linear regression of gas concentration over time. Flux calculation was based on the common equation containing GHG concentration (c in nmol $mol^{-1}$ for $CH_4$ and $N_2O$), time (t in seconds), atmospheric pressure (p in Pa), the headspace volume (V in $m^{-3}$), the universal gas constant (R = 8.3145 $m^{-3}$ Pa $K^{-1}$ $mol^{-1}$), ambient air temperature (Ta in K) and the surface area enclosed by the chamber (A in $m^{-2}$) (equation 1 in Imer et al. (2013)).

Flux quality criteria were based on the fit of the linear regression. If the correlation coefficient of the linear regression ($r^2$) was < 0.8 the actual flux value was rejected from the subsequent data analysis. Furthermore, if the slope between the 1st and 2nd GHG concentration measurement deviated considerably from the following concentrations we omitted the first value and calculated the flux based on three instead of four samples. Mean chamber GHG fluxes were then calculated as the arithmetic mean of all available individual chamber fluxes for each date. A total of 60 GHG flux calculations ($CH_4$ and $N_2O$) were available for the years 2010 and 2011. Another 52 $N_2O$ flux values were available for the five-month peak-growing season in 2013.

*2.4 Gapfilling and annual sums of $CO_2$, $CH_4$, and $N_2O$*

To date a common strategy to fill gaps in EC data of $CH_4$ and $N_2O$ has not been agreed on. The commonly used methods are simple linear approaches (Mishurov and Kiely, 2011) or the application of more sophisticated tools such as artificial neural networks (Dengel et al., 2011). The difficulty of finding an adequate gap-filling strategy results from the fact that emission pulses of either $N_2O$ or $CH_4$ remain challenging to predict. Similarly, different measurement approaches – i.e. low temporal resolution manual GHG chambers compared to high temporal resolution eddy covariance measurements - need different gap-filling approaches (Mishurov and Kiely, 2011; Nemitz et al., 2018). In order to keep the gap-filling methods as simple and reliable as possible, we used a running median (30 and 60 days for eddy covariance based and chamber $N_2O$ fluxes, respectively). A similar approach was recently chosen by Hörtnagl et al. (2018) due to its sensitivity to peaks in the $N_2O$ exchange data. The approach was particularly chosen as it minimizes the bias occuring from linear gap filling or simply using an overall average value. While the gapfilling approach may be of less importance for EC flux measurements with its high temporal data availability, it is the more important for less frequently available GHG fluxes derived via manual chambers. Given the occurrence of sporadic $N_2O$ peaks which occur mostly in relation to management activities and last for few hours/days only as well as the labour needed to carry out GHG chambers measurements, researchers commonly aim at having weekly or biweekly flux data (i.e. Imer et al. 2013). The respective sampling design is commonly designed to capture potential $N_2O$ flux peaks as well as some background values (Mishurov and Kiely, 2011). If one then uses either a linear interpolation or an overall average value, one can derive a budget which is than a likely overestimation of the annual flux budget caused by the few flux peaks observed in such managed systems. The same bias is likely to occur if just flux averages are used since few very high emission peaks will affect such an average. Thus, and in order to simulate $N_2O$ emission peaks more reliably, we have chosen the approach as taken by Hörtnagl et al. (2018).

In contrast to $CH_4$ and $N_2O$ various well-established approaches to fill $CO_2$ flux data exist (Moffat et al., 2007). Here, we filled gaps in $CO_2$ exchange data following the marginal distribution sampling method (Reichstein et al., 2005) which was implemented in the R package REddyProc (https://r-forge.r-project.org/projects/reddyproc/).

Calculation of the global warming potential (GWP) given in $CO_2$-equivalents followed the recommendations given in the 5th Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), with $CH_4$ having a 28 and $N_2O$ a 265 times greater GWP than $CO_2$ on a per mass basis over a time horizon of 100 years (Stocker et al., 2013).

### *2.5 Meteorological and phenological data*

Flux measurements were accompanied by standard meteorological measurements. These included observations of soil temperature (depths of 0.01, 0.02, 0.05, 0.10, and 0.15 m, TL107 sensors, Markasub AG, Olten, Switzerland), soil moisture (depths of 0.02 and 0.15 m, ML2x sensors, Delta-T Devices Ltd., Cambridge, UK) and air temperature (2 m height, Hydroclip S3 sensor, Rotronic AG, Switzerland). Furthermore, we measured the radiation balance including short-wave incoming and outgoing radiation, long-wave incoming and outgoing radiation (CNR1 sensor with ventilated Markasub housing, Kipp and Zonen, Delft, the Netherlands) as well as photosynthetically active radiation at 2 m height (PARlite sensor, Kipp and Zonen, Delft, the Netherlands). All data were stored as 30 min averages on a datalogger in a climate-controlled box on site (CR10X, Campbell Scienctific, Logan, UT, USA).

# 3 Results

*3.1 General site conditions*

The Chamau study site (CH-Cha) experienced meteorological conditions typical for the site during the five-year observation period. Summer precipitation commonly exceeded winter precipitation (Figure 1a). A spring drought was recorded from March till May 2011 (Wolf et al., 2013), leading to considerably lower soil water content than in previous and following years (Figure 1a). Average daily air temperatures rose up to 26.7 °C (27$^{th}$ July 2013) during summer and average daily temperature in winter dropped as low as -12.7 °C (6$^{th}$ February 2012, Figure 1b) with soil temperature following in a dampened pattern (Figure 1b). Average daily photosynthetic photon flux density did not differ considerably over the five-year observation period (Figure 1c). The site rarely experienced snow cover during winter (Figure 1b).
The complexity in management activities becomes apparent when comparing business as usual years (e.g. 2011) with the restoration year (2012, Figure 2a and b), highlighting the importance of grassland restoration to maintain productivity yields. Prior to 2012 an obvious decline in productivity with larger C and N inputs was found compared to the outputs in the years after restoration (2013 and 2014, Figure 2a and b).

*3.2 EC $N_2O$ fluxes vs. chamber derived $N_2O$ fluxes*

In 2013, we had the chance of comparing $N_2O$ fluxes measured with two considerably different GHG measurement techniques, namely eddy covariance and static chambers. The chambers (n=10) were installed within the EC footprint. Our results reveal a similar temporal pattern, with increased $N_2O$ losses being captured by both methodologies following fertilizer application. However, we could not identify a consistent bias of either technique (Figure 3a). Direct comparison of both measurements revealed a reasonable correlation (slope m = 0.61, $r^2$ = 0.4) and larger variation between both techniques with increasing flux values (Figure 3b).

*3.3 Temporal variation of GHG exchange*

Fluxes of $CO_2$ and $N_2O$ showed considerable variation between and within years. This variation primarily occurs due to management activities and seasonal changes in meteorological variables (Figures 1 and 4). In contrast, methane fluxes did not show a distinct seasonal pattern.

*$CO_2$ exchange*

In pre-ploughing years (2010 and 2011), the Chamau site showed 60 % lower $CO_2$ uptake compared to the post-ploughing years (2013 and 2014, Table 2). All four non-ploughing years revealed largest $CO_2$ uptake rates in late spring (daily averaged peak uptake rates were >10 µmol $CO_2$ $m^{-2}$ $s^{-1}$, March and April, Figure 4a). Besides the seasonal effects a clear impact of harvest events could be identified, with abrupt changes from net uptake of $CO_2$ to either reduced uptake or net loss of $CO_2$ (light blue arrows indicate harvest event, Figure 4a). A similar but less pronounced effect was found following grazing periods (light and dark brown arrow, Figure 4a). A complete switch from net uptake to net $CO_2$ release was observed during the first three months of 2012, after ploughing and during re-cultivation of the grassland. In this specific year, the site only experienced snow cover for few days (Figure 1c) and temperatures below 5 °C occurred more regularly than in all other years (Figure 1 b). Seasonal $CO_2$ exchange was characterized by net release of $CO_2$ in winter (DJF), highest $CO_2$ uptake rates were observed in spring (MAM), constant uptake rates during summer (JJA) which however were lower than those measured in spring, and very low net release of $CO_2$ in fall (Table 3). Average winter $CO_2$ exchange for the five-year observation period (gap-filled 30 min data) was 0.28 ± 5.68 µmol $CO_2$ $m^{-2}$ $s^{-1}$ (SE = 0.04, Table 3). The restoration year 2012 showed a slightly different pattern with relatively large $CO_2$ release in winter and spring and considerably lower uptake rates in summer. The years before the restoration (2010 and 2011) were characterized by smaller net uptake rates during spring and summer when compared to the post-ploughing years (2013 and 2014). Additionally, winter fluxes in 2010 and 2011 were positive (net release of $CO_2$), while winter fluxes in the years 2013 and 2014 were showing a small but consistent net uptake of $CO_2$ (Figure 4a, Table 3).

*$CH_4$ exchange*

The individual static chamber measurements (2011&2011) were often below the detection limit and fluctuated around zero similar to the eddy covariance measurements (Figure 4b). Any methane peaks expected due to freezing and thawing in late winter and early spring were not observed. Also, commonly reported net emissions of methane during grazing of animals were not seen (Figure 4b). Seasonal differences of methane exchange did not show a clear pattern (Table 3). A comparison of methane fluxes obtained by both, static GHG chambers and EC measurements as done for $N_2O$ (see next paragraph) could not be performed due to a malfunction of the respective detector in the gas chromatograph.

*$N_2O$ exchange*

$N_2O$ exchange was low during the majority of the days over the five-year observation period, fluctuating around zero (Figure 4c). However, clear peaks in $N_2O$ emissions were observed following fertilization events or periods with high rainfall after a dry period in summer (i.e. summer 2013 and 2014, Figures 3a and 4c). While event driven $N_2O$ emissions were commonly on the order of 4 to 8 nmol $N_2O$ $m^{-2}$ $s^{-1}$ (Figure 4c), $N_2O$ emissions following ploughing and subsequent re-sowing of the grassland in 2012 lead to up to three times as high $N_2O$ emissions (Figure 4c, year 2012, see also Merbold et al. (2014)). Similar to methane, enhanced $N_2O$ emissions in late winter or early spring as reported by other studies could not be identified (Figure 4c).

Background $N_2O$ fluxes were estimated by analysing all high temporal resolution flux data but excluding the restoration year 2012 and all values one week after a management event. Daily average background fluxes were 0.21 ± 0.55 nmol $m^{-2}$ $s^{-1}$ (SE = 0.02). Differences in $N_2O$ exchange over the course of individual years became obvious when splitting the dataset into the four seasons (winter – DJF, spring – MAM, summer – JJA and fall – SON). In contrast to $CO_2$ exchange that showed large net uptake rates in spring, $N_2O$ emissions were largest during summer (JJA) and lowest in winter (DJF). As highlighted for the other gases, the year of grassland restoration showed a completely different picture (Table 3).

*3.4 Annual sums and Global Warming Potential (GWP) of $CO_2$, $CH_4$ and $N_2O$*

Annual sums showed a net uptake of $CO_2$ during the two pre-ploughing years (-695 g $CO_2$ $m^{-2}$ $yr^{-1}$ and -978 g $CO_2$ $m^{-2}$ $yr^{-1}$ in 2010 and 2011 respectively). Up to three times of this net uptake was reached in 2013 and 2014, the two post-ploughing years (-2046 g $CO_2$ $m^{-2}$ $yr^{-1}$ and -2751 g $CO_2$ $m^{-2}$ $yr^{-1}$, Table 2). In contrast, the ploughing year 2011 was characterized by a net release of $CO_2$ (1447 g $CO_2$ $m^{-2}$ $yr^{-1}$).

Methane budgets for the years 2010 and 2011 were not be calculated as many of the available measurements were below the limit of detection. For the years 2012 – 2014, the annual methane budget showed a minor release of 26.8 – 55.2 g $CH_4$ $m^{-2}$ $yr^{-1}$.

The Chamau site was characterized by a net release of nitrous oxide over the five-year study period. While annual average $N_2O$ emissions ranging between 0.34 and 1.17 g $N_2O$ $m^{-2}$ $yr^{-1}$ in the non-ploughing years, the site emitted 4.36 g $N_2O$ $m^{-2}$ $yr^{-1}$ in 2012.

The global warming potential (GWP), expressed as the yearly cumulative sum of all gases after their conversion to $CO_2$-equivalents, was negative during all years (between -387 and -2577 $CO_2$-eq. $m^{-2}$) except for the ploughing year 2012 (+2629 $CO_2$-eq. $m^{-2}$).

Overall, $CO_2$ exchange contributed more than 90% to the total GHG balance in 2011, 2013 and 2014. Clearly, $CH_4$ exchange was of minimal importance for the GHG budget (Table 2). In 2010, the contribution of $CO_2$ to the site's GHG budget was almost 70%, and $N_2O$ contributed about 30%. Only in 2012, the year of restoration, $CO_2$ and $N_2O$ exchange contributed almost equally to the site's overall GHG budget (55.1% and 43.9%, respectively).

*3.5. Carbon gains/losses of the Chamau site between 2010 and 2014*

The Chamau site assimilated on average -441 ± 260 g $CO_2$-C $m^{-2}$ $yr^{-1}$ (4410 kg C $ha^{-1}$ $yr^{-1}$) during the “business as usual” years (2010 and 2011 as well as 2013 and 2014). During the restoration year the site lost 395 g $CO_2$-C $m^{-2}$ (3950 kg C $ha^{-1}$) (Table 2). Carbon losses (and/or gains) from methane were < 1 g $CH_4$-C $m^{-2}$ during all five years.

Carbon was gained in both parcels during the pre-ploughing years (Table 4). Considerable net losses of carbon were calculated for the ploughing year. In contrast, the post-ploughing years were again recognized as years with large net gains in carbon. Over the observation period of 5 years, the Chamau grassland gained approximately 4 t C $ha^{-1}$, excluding losses via leaching and deposition of C in form of dust.

# 4 Discussion

The five-year measurement period is representative for other similarly managed grassland ecosystems in Switzerland. Climate conditions were similar to the long-term average as described in Wolf et al. (2013). Management activities, such as harvests and subsequent fertilizer applications, were driven by overall weather conditions, (i.e. 2013 late spring, Figure 2a and b).

## *4.1 Technical and methodological aspects of the study*

Different techniques are currently applied to measure GHG fluxes from a variety of ecosystems (Denmead, 2008), each having its advantages and disadvantages or being chosen for a specific purpose or reason. A common approach to study individual processes or time periods contributing to specific greenhouse gas emissions is to measure with GHG chambers on the plot scale (Pavelka et al., 2018). Chamber methods have been widely used to derive annual GHG and nutrient budgets (Barton et al., 2015; Butterbach-Bahl et al., 2013). Critical assessments of the suitability and associated uncertainty in chamber derived GHG budgets in relation to sampling frequency have been published by Barton et al. (2013). Existing studies have not only compared the two measurement techniques employed in this study (manual chambers and eddy covariance) in grasslands before, but also estimated annual emissions based on differing methodologies (Flechard et al., 2007; Jones et al., 2017). Additional confidence in our approach was obtained from the $N_2O$ emissions during the summer period 2013, where both measurement techniques ran in parallel (Figure 3a and b). Annual budgets derived by applying similar gap-filling approaches to the individual datasets led to comparable results (Table 2).

We calculated detection limits for the individual GHGs from our manual chambers following (Parkin et al., 2012). Detection limits were 0.34 ± 0.26 nmol $m^{-2}$ $s^{-1}$, 0.05 ± 0.02 nmol $m^{-2}$ $s^{-1}$, and 0.06 ± 0.06 µmol $m^{-2}$ $s^{-1}$ for $CH_4$, $N_2O$ and $CO_2$, respectively. Following this, methane flux measurements frequently were below this limit of detection, hence we did not calculate methane budgets for 2010 and 2011. The flux values measured with the EC technique between 2012 and 2014 compare well to similar measurements made by Felber et al. (2016) in an intensively managed grassland in Western Switzerland. The observed values have been identified to represent the soil methane exchange in EC measured fluxes (Felber et al. 2016).

$N_2O$ fluxes in contrast were much better constrained by both methods due to clear $N_2O$ sources (i.e. fertilizer amendments) and better sensitivity of the instruments used by both techniques for $N_2O$ as compared to $CH_4$. Background $N_2O$ emissions as observed in this study (0.21 ± 0.55 nmol $m^{-2}$ $s^{-1}$ (SE = 0.02)) compare well to estimates suggested by Rafique et al., (2011) whom suggest an annual background $N_2O$ losses of 1.8 kg $N_2O$-N for a grazed pasture (i.e. 0.20 nmol $m^{-2}$ $s^{-1}$).

### *4.2 Annual GHG and C and N gains/losses*

Net carbon losses and gains estimated for the CH-Cha site between 2010 and 2015 were in general within the range of values estimated by Zeeman et al., (2010) for the years 2006 and 2007. The slightly higher losses observed prior to ploughing may result from reduced productivity of the sward. This becomes particularly visible when compared to the net ecosystem exchange (NEE) of $CO_2$ values for the years after restoration. Losses via leaching have previously been estimated to be of minor importance at this site (Zeeman et al., 2010) and were therefore not considered in this study. Considerably higher C gains during post-ploughing years were caused be enhanced plant growth in spring and summer. Restoration is primarily done to eradicate weeds and rodents, favouring biomass productivity of the fodder grass composition. Other grasslands in Central Europe, i.e. sites in Austria, France and Germany, showed similar values for net ecosystem exchange (Hörtnagl et al., 2018). Still, total C budgets as presented here are subject to considerable uncertainty which is strongly depending on assumptions made for gap-filling etc. (Foken et al., 2004). Nevertheless, the values reported here show the overall trend on C uptake/release of the site and clearly exceed the uncertainty of ± 50 g C per year for eddy covariance studies as suggested by Baldocchi (2003).

Methane was of negligible importance for the C budget of this site. We did not observe distinct peaks in $CH_4$ emissions in relation to grazing which is primarily due to the low grazing pressure at CH-Cha. Studies carried out on pastures in Scotland, Mongolia, France and Western Switzerland have shown that grazing can largely contribute to ecosystem-scale methane fluxes, in particular if ruminants such as cattle are populating the EC footprint (Dengel et al., 2011; Felber et al., 2015; Schönbach et al., 2012). If we included an approximation of methane emissions of cattle which we may have missed in the EC flux measurements, we would have to add 0.407 g $CH_4$-C $m^{-2}$ $y^{-1}$ to the current value of 1.48 g $CH_4$-C $m^{-2}$ in 2014 (Table 2). This value is based on the average methane emissions of 404 g $CH_4$ $head^{-1}$ $d^{-1}$ stated in Felber et al. (2016) and linking this to the average stocking density (4.04 head ha-1) on the Chamua site and the stocking duration (30 days in 2014). Still, the GHG budget as well as the C budget of the site would not be altered.

The nitrous oxide budget reported for the years without ploughing in this study coincides with values reported for other grasslands in Europe, ranging from moist to dry climates and lower to higher elevations in Austria and Switzerland (Cowan et al., 2016; Hörtnagl et al., 2018; Imer et al., 2013; Skiba et al., 2013).

Nitrogen inputs and losses via $N_2O$ varied largely between the years before and after ploughing. While the site was characterized by large N amendments prior to ploughing and with reduced harvest, the picture was completely the opposite during the years after ploughing, with considerably less N inputs compared to the nitrogen removed from the field via harvests. Farmers aim every year at having a balanced N budget (fertilizer inputs = nutrients removed from the field). Pasture degradation is the main motivation for enhanced fertilizer inputs in order to stabilize forage productivity. Similarly, regular restoration of permanent pastures is absolutely necessary (Cowan et al., 2016). So far, we identified only one study that investigated the net effects on the overall GHG exchange following grassland restoration (Drewer et al., 2017).

## 5 Conclusion

This study in combination with an overview of available datasets on grassland restoration and their consequences on GHG budgets highlights the overall need of additional observational data. While restoration changed the previous C sink to a C source at the Chamau site, the wider implication in terms of the GWP of the site when including other GHGs have long-term consequences (i.e. in mitigation assessments). Furthermore, this study showed the large variations in N inputs and N outputs from this grassland and the difficulty farmers face when aiming for balanced N budgets in the field. Still, the current study focused on GHGs only and can thus not constrain the N budget but assess the losses of N via $N_2O$. Losses in form of $NH_3$, $N_2$ and $NO_x$ will have to be quantified to fully assess N budgets besides the overall fact that GHG data following grassland restoration remain largely limited to investigate long-term consequences.

Fortunately, these are likely to become available in the near future by the establishment of environmental research infrastructures (i.e. ICOS in Europe, NEON in the USA or TERN in Australia) that aim at standardized, high quality and high temporal resolution trace gas observation of major ecosystems, including permanent grasslands. With these additional data, another major constraint of producing defensible GHG and nutrient budgets, namely gap-filling procedures, will likely be overcome. New and existing data can be used to derive reliable functional relations and artificial neural networks (ANNs) at field to ecosystem scale that are capable of reproducing in-situ measured data. Once this step is achieved, both the available data as well the functional relations can be used to improve, to train and to validate existing biogeochemical process models (Fuchs et al., 2020). Subsequently, reliable projections on both nutrient and GHG budgets at the ecosystem scale that are driven by anthropogenic management as well as climatic variability become reality.

The study stresses the necessity of including management activities occurring at low frequency such as ploughing in GHG and nutrient budget estimates. Only then, the effect of potential best-bet climate change mitigation options can be thoroughly quantified. The next steps in GHG observations from grassland must not only focus on observing business as usual activities, but also aim at testing the just mentioned best-bet mitigation options jointly in the field while simultaneously in combination with existing biogeochemical process models.

# 6 Tables and Figures

Table 1: Data availability of GHG fluxes measured over the five-year observation period. Values are given as all data possible, raw processed values and high quality (HQ) data, which were then used in the analysis. High quality data are data with a quality flag "0" and "1" from the Eddypro output only. Grey shaded areas represent time period where both methods (EC and static chambers) were used simultaneously to estimate $FN_2O$. Static chamber flux data are highlighted in *italic* font.

**Table 2:** Table 2: Annual average $CO_2$, $CH_4$ and $N_2O$ fluxes and annual sums for the three GHGs as well as carbon and nitrogen gain/losses per gas species. GWP were calculated for a 100-year time horizon and based on the most recent numbers provided by IPCC (Stocker et al., 2013). Annual budgets were derived from either gap-filled manual chamber (MC) or eddy covariance (EC) measurements. n.c. stands for not calculated. Sign convention: positive values denote export/release, negative values import/uptake.

**Table 3:** Average GHG flux rates per season: winter (DJF), spring (MAM), summer (JJA) and fall (SON). Values are based on gap-filled data to avoid bias from missing nighttime data (predominantly relevant for $CO_2$). Data are only presented when continuous measurements (eddy covariance data) were available. Sign convention: positive values denote export/release, negative values import/uptake.

**Table 4:** Table 4: Carbon and nitrogen gains/losses through fertilization, harvest and GHGs for the Chamau (CH-Cha) site in 2010- 2014. Values are given in kg $ha^{-1}$. Gains are indicated

with "-" and losses/exports are indicated with "+". While management information was available for both parcels (A and B), flux measurements are an integrate of both parcels. n.c. = not calculated

**Table 5:** Existing studies investigating the GHG exchange over pastures following ploughing. Results presented show the flux magnitude following ploughing and are rounded values of the individual presented in the papers. Values were converted to similar units (mg $CO_2$-C m$^{-2}$ h$^{-1}$, µg $CH_4$-C m$^{-2}$ h$^{-1}$ and µg $N_2O$-N m$^{-2}$ h$^{-1}$). Based on Web of Knowledge search July 15th 2017 with the search terms "grassland", "pasture", "greenhouse gas", "ploughing" and/or "tillage". Only two studies representing conversion from pasture to cropland or other systems were included in this table.

**Table S1:** Detailed management information for the two parcels under investigation at the Chamau research station. Data are based on fieldbooks provided by the farm personnel as well as in-situ measurements. Organic fertilizer samples were sent to a central laboratory for nutrient content analysis (Labor fuer Boden- und Umweltanalytik, Eric Schweizer AG, Thun, Switzerland). Destructive harvests (n = 10) of biomass were carried out in the years 2010, 2011, 2013 and 2014. Harvest estimates are based on values derived from the in-situ measurements and data provided by the farm personnel. Detailed information on the grazing regime was furthermore provided by the farm personnel in hand-written form (not shown).

**Figure 1:** Weather conditions during the years 2010 – 2014. Weather data were measured with our meteorological sensors installed on site. (a) Daily sum of precipitation (mm) and soil water content (SWC, blue line, m$^3$ m$^{-3}$) measured at 5 cm soil depth; (b) daily averaged air temperature (°C), daily averaged soil temperature (grey line, °C) and days with snow cover (horizontal bars); (c) daily averaged photosynthetic photon flux density (PPFD, µmol m$^{-2}$ s$^{-1}$). Days with snow cover were identified with albedo calculations. Days with albedo > 0.45 were identified as days with either snow or hoarfrost cover.

**Figure 2:** Management activities for both parcels (A and B in panels (a) and (b), respectively) on the CH-Cha site. Overall management varied particularly in 2010 between both parcels, whereas similar management took place between 2011 and 2014. Arrow direction indicates whether carbon (C in kg ha$^{-1}$) and/or nitrogen (N in kg ha$^{-1}$) were amended to, or exported from the site ("$F_o$" and "$F_{o*}$"- organic fertilizers, slurry/manure (red); "$F_m$" - mineral fertilizer (light orange); "H" - harvest (light blue); "$G_s$" and "$G_c$" - grazing with sheep/cows (light/dark brown). Other colored arrows visualize any other management activities such as pesticide application ("$P_h$"- herbicide (light pink); "$P_m$"- molluscicide (dark pink); "T"- tillage (black), "R"- rolling (light grey) and "S"- sowing (dark grey) which occurred predominantly in 2010 (parcel B) and 2012 (parcels A and B). Carbon imports and exports are indicated by black and grey bars. Thereby black indicated the start of the specific management activities and grey the duration (e.g. during grazing, "$G_s$"). Green colors indicate nitrogen amendments or losses, with dark green visualizing the start of the activity and light green colors indicating the duration. Sign convention: positive values denote export/release, negative values import/uptake.

**Figure 3:** (a) Temporal dynamics of $N_2O$ fluxes measured with the eddy covariance (white circles) and manual greenhouse gas chambers (black circles measured in 2013) – grey lines indicate standard deviation. Arrows indicate management events ("H" = harvest, "$F_o$" = organic fertilizer application (slurry), "Ph" = pesticide (herbicide) application). (b) 1:1 comparison between chamber based and eddy covariance based $N_2O$ fluxes in 2013. The

dashed line represents the 1:1 line. ($y = mx + c$, $r^2 = 0.4$, $m = 0.61$, $c = 0.17$, $p < 0.0001$). Sign convention: positive values denote export/release, negative values import/uptake.

**Figure 4:** Temporal dynamics of gap-filled (except methane in 2010/2011) daily averaged greenhouse gas (GHG) fluxes (white circles): a) ($CO_2$ exchange in µmol $m^{-2}$ $s^{-1}$; (b) $CH_4$ exchange in nmol $m^{-2}$ $s^{-1}$ and (c) $N_2O$ exchange in nmol $m^{-2}$ $s^{-1}$. Coloured circles indicate manual chamber measurements. While both GHGs, $CH_4$ and $N_2O$ were measured in 2010 and 2011 (blue cirlces), $N_2O$ only was measured in 2013 (light blue circles). The grey dashed lines indicate the beginning of a new year. Same color coding as used in Figure 3 a was used to highlight management activities. Sign convention: positive values denote export/release, negative values import/uptake. Grey lines behind the circles indicate standard deviation.

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