

1 **Memory effects on greenhouse gas emissions (CO₂, N₂O and CH₄) following**
2 **grassland restoration?**

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28 **Abstract**

29 A five-year greenhouse gas (GHG) exchange study of the three major gas species (CO₂, CH₄
30 and N₂O) from an intensively managed permanent grassland in Switzerland is presented.
31 Measurements comprise two years (2010/2011) of manual static chamber measurements of
32 CH₄ and N₂O, five years of continuous eddy covariance (EC) measurements (CO₂/H₂O – 2010-
33 2014) and three years (2012-2014) of EC measurement of CH₄ and N₂O. Intensive grassland

34 management included both regular and sporadic management activities. Regular management
35 practices encompassed mowing (3-5 cuts per year) with subsequent organic fertilizer
36 amendments and occasional grazing whereas sporadic management activities comprised
37 grazing or similar activities. The primary objective of our measurements was to compare pre-
38 ploughing to post-ploughing GHG exchange and to identify potential memory effects of such
39 a substantial disturbance on GHG exchange and carbon (C) and nitrogen (N) gains/losses. In
40 order to include measurements carried out with different observation techniques, we tested two
41 different measurement techniques jointly in 2013, namely the manual static chamber approach
42 and the eddy covariance technique for N₂O, to quantify the GHG exchange from the observed
43 grassland site.

44 Our results showed that there were no memory effects on N₂O and CH₄ emissions after
45 ploughing, whereas the CO₂ uptake of the site considerably increased when compared to post-
46 restoration years. In detail, we observed large losses of CO₂ and N₂O during the year of
47 restoration. In contrast, the grassland acted as a carbon sink under usual management, i.e. the
48 time periods (2010-2011 and 2013-2014). Enhanced emissions/emission peaks of N₂O (defined
49 as exceeding background emissions $0.21 \pm 0.55 \text{ nmol m}^{-2} \text{ s}^{-1}$ (SE = 0.02) for at least two
50 sequential days and the seven-day moving average exceeding background emissions) were
51 observed for almost seven continuous months after restoration as well as following organic
52 fertilizer applications during all years. Net ecosystem exchange of CO₂ (NEE_{CO₂}) showed a
53 common pattern of increased uptake of CO₂ in spring and reduced uptake in late fall. NEE_{CO₂}
54 dropped to zero and became positive after each harvest event. Methane (CH₄) exchange
55 fluctuated around zero during all years. Overall, CH₄ exchange was of negligible importance
56 for both, the GHG budget as well as for the carbon budget of the site.

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

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68 **1 Introduction**

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

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

92 Up to date the majority of greenhouse gas exchange research has focused on CO_2 , with less
93 focus on the other two important GHGs N_2O and CH_4 , even though an increased interest in
94 these other gas species has become visible in recent years (Ammann et al., 2020; Ball et al.,
95 1999; Cowan et al., 2016; Krol et al., 2016; Kroon et al., 2007, 2010; Necpálová et al., 2013;
96 Rutledge et al., 2017). The existing exceptions are often referred to as “high-flux” ecosystems,
97 namely wetlands and livestock production system in terms of CH_4 (Baldocchi et al., 2012;
98 Felber et al., 2015; Laubach et al., 2016; Teh et al., 2011) and agricultural ecosystems such as
99 bioenergy system with considerable N_2O emissions (Cowan et al., 2016; Fuchs et al., 2018;
100 Krol et al., 2016; Skiba et al., 1996, 2013; Wecking et al., 2020; Zenone et al., 2016; Zona et

101 al., 2013). Agricultural ecosystems and specifically grazed systems are characterized by GHG
102 emissions caused through anthropogenic activities. These activities lead to changes in GHG
103 emission patterns and include harvests, amendments of fertilizer and/or pesticides and less
104 frequently occurring ploughing, harrowing and re-sowing events. While ploughing has been
105 shown to lead to considerable short-term emissions of CO₂ and N₂O (Buchen et al., 2017;
106 Cowan et al., 2016; Hörtnagl et al., 2018; MacKenzie et al., 1997; Merbold et al., 2014;
107 Rutledge et al., 2017; Vellinga et al., 2004), regular harvests have been shown to lead to
108 increased CO₂ uptake (Zeeman et al., 2010) and grazing leads to large CH₄ emissions (Dengel
109 et al., 2011; Felber et al., 2015). Other studies showed contrary results with reduced N₂O
110 emissions following ploughing of a drained grassland when compared to a fallow in Canada
111 (MacDonald et al., 2011).

112 Still, the full range of management activities occurring in intensively managed grasslands and
113 their respective impact on GHG exchange has not been investigated in detail. In a recent
114 synthesis including grasslands located along an altitudinal gradient in Central Europe, Hörtnagl
115 et al. (2018) highlighted the most important abiotic drivers of CO₂ (light, water availability and
116 temperature), CH₄ (soil water content, temperature and grazing) and N₂O exchange (water
117 filled pore space and soil temperature). The study by Hörtnagl et al. (2018) further elaborated
118 the variation in management intensity and related variations in GHG exchange across sites,
119 stressing the need for more case studies based on continuous GHG observations to improve
120 existing knowledge and close remaining knowledge gaps. To complete the picture on factors
121 driving ecosystem GHG exchange, irregular occurring events such as dry spells or
122 extraordinary wet periods can further lead to enhanced or reduced GHG emissions (Chen et al.,
123 2016; Hartmann and Niklaus, 2012; Hopkins and Del Prado, 2007; Mudge et al., 2011; Wolf
124 et al., 2013).

125 While drought has been shown to reduce CO₂ uptake in forests (Ciais et al., 2005) whereas
126 dry spells did not affect CO₂ uptake in grasslands (Wolf et al., 2013), flooding leads primarily
127 to enhanced CH₄ emissions (Knox et al., 2015) and large precipitation events can lead to
128 plumes of N₂O (Fuchs et al., 2018; Zona et al., 2013) similar to freeze-thaw events (Butterbach-
129 Bahl et al., 2011; Matzner and Borken, 2008) to name only some examples. Consequently,
130 understanding both, anthropogenic impacts such as management besides environmental
131 impacts on ecosystem GHG exchange, are crucially important to suggest appropriate climate
132 change mitigation as well as adaptation strategies for future land management with ongoing
133 climate change.

134 Different measurement techniques to quantify the net GHG exchange in ecosystems are known
135 and the most common approaches are either GHG chamber measurements or the eddy
136 covariance (EC) technique. Static manual chamber measurements have been used for more
137 than a century to quantify CO₂ emissions (Lundegardh, 1927) and their application has further
138 been expanded during the last decades to quantify losses of the three major GHGs, CO₂, N₂O
139 and CH₄ from soils (Imer et al., 2013; Pavelka et al., 2018a; Pumpanen et al., 2004; Rochette
140 et al., 1997). Even though more complex in technology and assumptions made before carrying
141 out measurements, the eddy covariance (EC) technique has become a valuable tool to derive
142 ecosystem integrated CO₂ and H₂O_{vapour} exchange across the globe (Balocchi, 2014; Eugster
143 and Merbold, 2015). The technique has been further extended to continuous measurements of
144 CH₄ and N₂O with the development of easy field-deployable fast-response analyzers during
145 the last decade (Brümmer et al., 2017; Felber et al., 2015; Kroon et al., 2007; Nemitz et al.,
146 2018a; Wecking et al., 2020). Each of the two approaches has its strengths and weaknesses and
147 it is beyond the scope of this study to discuss each of them in detail. However, we refer to a set
148 of reference papers highlighting the advantages and disadvantages of each technique separately
149 (chambers: (Ambus et al., 1993; Brümmer et al., 2017; Pavelka et al., 2018a); eddy covariance:
150 (Balocchi, 2014; Denmead, 2008; Eugster and Merbold, 2015; Nemitz et al., 2018)).

151 The overall objective of this study was to investigate the net GHG exchange (CO₂, CH₄ and
152 N₂O) before and after grassland restoration and thus fill existing knowledge gaps caused by
153 limited amounts of available GHG exchange data from intensively managed grasslands. The
154 specific goals were: (i) to assess pre- and post-ploughing GHG exchange in a permanent
155 grassland in central Switzerland accounting for changes in GHG exchange following frequent
156 management activities; (ii) to compare two different measurement techniques, namely eddy
157 covariance and static greenhouse gas flux chambers to quantify the GHG exchange in a
158 business-as-usual year; and (iii) to provide a five year GHG budget of the site and quantify
159 losses/gains of C and N. Based on our results we provide suggestions for future research
160 approaches to further understand ecosystem GHG exchange, to mitigate GHG emissions and
161 to ensure nutrient retention at the site for sustainable production from permanent grasslands in
162 the future.

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168 **2 Material and Methods**

169 **2.1 Study site**

170 The Chamau grassland site (Fluxnet identifier - CH-Cha) is located in the pre-alpine lowlands
171 of Switzerland at an altitude of 400 m a.s.l. (47°12' 37"N, 8°24'38"E) and characterized by
172 intensive management (Zeeman et al., 2010). The site is divided into two parcels (Parcel A and
173 B) with occasionally slightly different management regimes [see also *Fuchs et al., 2018*]. Mean
174 annual temperature (MAT) is 9.1 °C, and mean annual precipitation (MAP) is 1151 mm. The
175 soil type is a Cambisol with a pH ranging between 5 and 6, a bulk density between 0.9 and 1.3
176 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
177 species composition consists of Italian ryegrass (*Lolium multiflorum*) and white clover
178 (*Trifolium repens* L.). For more details of the site we refer to Zeeman et al., (2010).

179 CH-Cha is intensively managed, with activities being either recurrent – referred to as
180 usual/regular - or sporadic. Usual management refers to regular mowing and subsequent
181 organic fertilizer application in form of liquid slurry (up to 7 times per year). In addition, the
182 site is occasionally grazed by sheep and cattle for few days in early spring and/or fall (H.-R.
183 Wettstein personal communication, Table S1). Sporadic activities aim at maintaining the
184 typical fodder species composition and comprise reseeding, herbicide and pesticide application
185 or irregular ploughing and harrowing on an approximately decadal timescale (Merbold et al.,
186 2014). By such activity, mice are eradicated and a high-quality sward for fodder production is
187 re-established following weed contamination. Specific information on management activity
188 (timing, type of management, amount of biomass harvested) were reported by the farmers on
189 site (Table S1). Additionally, representative samples of organic fertilizer were collected shortly
190 before fertilizer application events and sent to a central laboratory for nutrient content analysis
191 (Labor fuer Boden- und Umweltanalytik, Eric Schweizer AG, Thun, Switzerland). Harvest
192 estimates were compared to estimates based on destructive sampling of randomly chosen plots
193 (n = 10) in the years 2010, 2011, 2013 and 2014. The amount of harvested biomass in the year
194 2012 was based on a calibration of the values presented by the farmer in comparison to the on-
195 site destructive harvests in previous and following years (Table S1).

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197 **2.2 Eddy covariance flux measurements**

198 **2.2.1 Eddy covariance setup**

199 The specific site characteristics with two prevailing wind directions (North-northwest and
200 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 $\text{H}_2\text{O}_{\text{vapour}}$ 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 $\text{H}_2\text{O}_{\text{vapour}}$. 3D wind components (u , v , w), CO_2 and $\text{H}_2\text{O}_{\text{vapour}}$ 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 $0.3\mu\text{m}$ 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} , $\mu\text{mol m}^{-2} \text{s}^{-1}$) 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

235 mean wind streamlines so that the vertical wind vector $\dot{w} = 0$. Turbulent departures were
236 calculated by Reynolds (block) averaging of 30 min data blocks. Frequency response
237 corrections were applied to raw fluxes, accounting for high-pass and low-pass filtering for the
238 CO₂ signal based on the open-path IRGA as well as for the closed-path CH₄ and N₂O data
239 (Fratini et al., 2014). All fluxes were calculated using the software *EddyPro* (version 6.0, LiCor
240 Biosciences, Lincoln, NE, USA) (Fratini and Mauder, 2014).

241 The quality of half-hourly raw time series was assessed during flux calculations following
242 (Vickers and Mahrt, 1997). Raw data were rejected if (a) spikes accounted for more than 1 %
243 of the time series, (b) more than 10 % of available data points were significantly different from
244 the overall trend in the 30 min time period, (c) raw data values were outside a plausible range
245 ($\pm 50 \mu\text{mol m}^{-2} \text{s}^{-1}$ for CO₂, $\pm 300 \text{ nmol m}^{-2} \text{s}^{-1}$ for N₂O and $\pm 1 \mu\text{mol m}^{-2} \text{s}^{-1}$ for CH₄) and (d)
246 window dirtiness of the IRGA sensor exceeded 80 %. Only raw data that passed all quality
247 tests were used for flux calculations.

248 Half-hourly flux data were rejected if (e) fluxes were outside a physically plausible range (ie.
249 $\pm 50 \mu\text{mol m}^{-2} \text{s}^{-1}$ for CO₂) (f) the steady state test exceeded 30 % and (g) the developed
250 turbulent conditions test exceeded 30 % (Foken et al., 2006). Between 1st January 2010 and
251 31st December 2014 64572 (88% of all possible data) 30-min flux values were calculated for
252 CO₂, of which 42865 (57.8%) passed all quality tests and were used for analyses in the present
253 study (Table 1). The amount of available flux values for N₂O and CH₄ were less, since we were
254 only capable to continuously measure both gases from 2012 onwards (Table 1). Flux values in
255 this manuscript are given as number of moles of matter/mass per ground surface area and unit
256 time. Negative fluxes represent a flux of a specific gas species from the atmosphere into the
257 ecosystem, whereas positive fluxes represent a net loss from the system.

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259 **2.3 Static greenhouse gas flux chambers**

260 *2.3.1 Manual static GHG chamber setup*

261 Static manual opaque GHG chambers were installed within the footprint of the site to measure
262 soil fluxes in 2010 and 2011 (n = 16) as well as during summer 2013 (n = 10). The chambers
263 were made of polyvinyl chloride tubes with a diameter of 0.3 m (Imer et al., 2013). The average
264 headspace height was $0.136 \text{ m} \pm 0.015 \text{ m}$ and average insertion depth of the collars into the
265 soil was $0.08 \text{ m} \pm 0.05 \text{ m}$. During sampling days with vegetation larger than 0.3 m inside the
266 chamber, collar extensions (0.45 m) were used (2013 only). Chamber lids were equipped with
267 reflective aluminium foil to minimize heating inside the chamber during the period of actual
268 measurement. Spacing between the chambers was approximately seven m and an equal number

269 of chambers were installed in each parcel. For further details we refer to Imer et al. (2013).
270 Chamber measurements were carried out on a weekly basis during the growing season in all
271 three years (2010, 2011 and 2013), and at least once a month during the winter season in 2010
272 and 2011. More frequent measurements of N₂O emissions (every day) were performed
273 following fertilization events in 2013 for seven consecutive days after each event. Besides this,
274 an intensive measurement campaign lasting 48 hours (two-hour measurement interval) was
275 carried out in September 2010.

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277 *2.3.2 GHG concentrations measurements*

278 During each chamber closure four gas samples were taken, one immediately after closure and
279 then in approximately ten-minute time increments. With this approach, we guaranteed that the
280 chambers were closed no longer than 40 minutes to avoid potential saturation effects. Syringes
281 (60 ml volume) were inserted into the chambers lid septa to take the gas samples. The collected
282 air sample was injected into pre-evacuated 12 ml vials (Labco Limited, Buckinghamshire, UK)
283 in the next step. Prior to the second, third and fourth sampling of each chamber, the air in
284 chamber headspace was circulated with the syringe volume of air from the chamber headspace
285 to minimize effects of built-up concentration gradients inside the chamber.

286 Gas samples were analyzed for their respective CO₂, CH₄ and N₂O concentrations in the lab as
287 soon as possible after sample collection and not stored for more than a few days. Gas sample
288 analysis was performed with a gas chromatograph (Agilent 6890 equipped with a flame
289 ionization detector, a methanizer - Agilent Technologies Inc., Santa Clara, USA - and an
290 electron capture detector – SRI Instruments Europe GmbH, 53604 Bad Honnef, Germany) as
291 described by Hartmann and Niklaus (2012).

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293 *2.3.3 GHG chamber flux calculations and quality control*

294 GHG fluxes were calculated based on the rate of gas concentration change inside the chamber
295 headspace. Data processing, which included flux calculation and quality checks, was carried
296 out with the statistical software R (R Development Core Team, 2010). Thereby the rate of
297 change was calculated by the slope of the linear regression of gas concentration over time. Flux
298 calculation was based on the common equation containing GHG concentration (c in nmol mol⁻¹
299 for CH₄ and N₂O), time (t in seconds), atmospheric pressure (p in Pa), the headspace volume
300 (V in m⁻³), the universal gas constant (R = 8.3145 m⁻³ Pa K⁻¹ mol⁻¹), ambient air temperature
301 (Ta in K) and the surface area enclosed by the chamber (A in m⁻²) (equation 1 in Imer et al.
302 (2013)).

303 Flux quality criteria were based on the fit of the linear regression. If the correlation coefficient
304 of the linear regression (r^2) was < 0.8 the actual flux value was rejected from the subsequent
305 data analysis (see Imer et al. 2013 for further details on data quality control, and Table 1).
306 Furthermore, if the slope between the 1st and 2nd GHG concentration measurement deviated
307 considerably from the following concentrations we omitted the first value and calculated the
308 flux based on three instead of four samples. Mean chamber GHG fluxes were then calculated
309 as the arithmetic mean of all available individual chamber fluxes for each date. A total of 35
310 GHG flux calculations (CH₄ and N₂O) were available for the years 2010 and 2011. Another 52
311 N₂O flux values were available for the five-month peak-growing season in 2013.

312

313 *2.4 Gapfilling and annual sums of CO₂, CH₄, and N₂O*

314 To date a common strategy to fill gaps in EC data of CH₄ and N₂O has not been agreed on. The
315 commonly used methods are simple linear approaches (Mishurov and Kiely, 2011) or the
316 application of more sophisticated tools such as artificial neural networks (Dengel et al., 2011).
317 The difficulty of finding an adequate gap-filling strategy results from the fact that emission
318 pulses of either N₂O or CH₄ remain challenging to predict. Similarly, different measurement
319 approaches – i.e. low temporal resolution manual GHG chambers compared to high temporal
320 resolution eddy covariance measurements - need different gap-filling approaches (Mishurov
321 and Kiely, 2011; Nemitz et al., 2018). In order to keep the gap-filling methods as simple and
322 reliable as possible, we used a running median (30 and 60 days for eddy covariance based and
323 chamber N₂O fluxes, respectively). A similar approach was recently chosen by Hörtnagl et al.
324 (2018) due to its reduced sensitivity to peaks in the N₂O exchange data. The approach was
325 particularly chosen as it minimizes the bias occurring from linear gap filling or simply using
326 an overall average value. While the gapfilling approach may be of less importance for EC flux
327 measurements with its high temporal data availability, it is the more important for less
328 frequently available GHG fluxes derived via manual chambers. Given the occurrence of
329 sporadic N₂O peaks which occur mostly in relation to management activities and last for few
330 hours/days only as well as the labour needed to carry out GHG chambers measurements,
331 researchers commonly aim at having weekly or biweekly flux data (i.e. Imer et al. 2013). The
332 respective sampling design is commonly designed to capture potential N₂O flux peaks as well
333 as some background values (Mishurov and Kiely, 2011). If one then uses either a linear
334 interpolation or an overall average value, one can derive a budget which is than a likely
335 overestimation of the annual flux budget caused by the few flux peaks observed in such
336 managed systems. The same bias is likely to occur if just flux averages are used since few very

337 high emission peaks will affect such an average. Thus, and in order to simulate N₂O emission
338 peaks more reliably, we have chosen the approach as taken by Hörtnagl et al. (2018).

339 In contrast to CH₄ and N₂O various well-established approaches to fill CO₂ flux data exist
340 (Moffat et al., 2007). Here, we filled gaps in CO₂ exchange data following the marginal
341 distribution sampling method (Reichstein et al., 2005) which was implemented in the R
342 package REddyProc (<https://r-forge.r-project.org/projects/reddyproc/>).

343 Calculation of the global warming potential (GWP) given in CO₂-equivalents followed the
344 recommendations given in the 5th Assessment Report of the Intergovernmental Panel on
345 Climate Change (IPCC), with CH₄ having a 28 and N₂O a 265 times greater GWP than CO₂
346 on a per mass basis over a time horizon of 100 years (Stocker et al., 2013).

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348 *2.5 Meteorological and phenological data*

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

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371 **3 Results**

372 *3.1 General site conditions*

373 The Chamau study site (CH-Cha) experienced meteorological conditions typical for the site
374 during the five-year observation period. Summer precipitation commonly exceeded winter
375 precipitation (Figure 1a). A spring drought was recorded from March till May 2011 (Wolf et
376 al., 2013), leading to considerably lower soil water content than in previous and following years
377 (Figure 1a). Average daily air temperatures rose up to 26.7 °C (27th July 2013) during summer
378 and average daily temperature in winter dropped as low as -12.7 °C (6th February 2012, Figure
379 1b) with soil temperature following in a damped pattern (Figure 1b). Average daily
380 photosynthetic photon flux density did not differ considerably over the five-year observation
381 period (Figure 1c). The site rarely experienced snow cover during winter (Figure 1b).

382 The complexity in management activities becomes apparent when comparing business as usual
383 years (e.g. 2011) with the restoration year (2012, Figure 2a and b), highlighting the importance
384 of grassland restoration to maintain productivity yields. Prior to 2012 an obvious decline in
385 productivity with larger C and N inputs was found compared to the outputs in the years after
386 restoration (2013 and 2014, Figure 2a and b).

387

388 *3.2 EC N₂O fluxes vs. chamber derived N₂O fluxes*

389 In 2013, we had the chance of comparing N₂O fluxes measured with two considerably different
390 GHG measurement techniques, namely eddy covariance and static chambers. The chambers
391 (n=10) were installed within the EC footprint. Our results reveal a similar temporal pattern,
392 with increased N₂O losses being captured by both methodologies following fertilizer
393 application. However, we could not identify a consistent bias of either technique (Figure 3a).
394 Direct comparison of both measurements revealed a reasonable correlation (slope m = 0.61, r²
395 = 0.4) and larger variation between both techniques with increasing flux values (Figure 3b).

396

397 *3.3 Temporal variation of GHG exchange*

398 Fluxes of CO₂ and N₂O showed considerable variation between and within years. This variation
399 primarily occurs due to management activities and seasonal changes in meteorological
400 variables (Figures 1 and 4). In contrast, methane fluxes did not show a distinct seasonal pattern.

401

402

403

404 *CO₂ exchange*
405 In pre-ploughing years (2010 and 2011), the Chamau site showed 60 % lower CO₂ uptake
406 compared to the post-ploughing years (2013 and 2014, Table 2). All four non-ploughing years
407 revealed largest CO₂ uptake rates in late spring (daily averaged peak uptake rates were >10
408 µmol CO₂ m⁻² s⁻¹, March and April, Figure 4a). Besides the seasonal effects a clear impact of
409 harvest events could be identified, with abrupt changes from net uptake of CO₂ to either
410 reduced uptake or net loss of CO₂ (light blue arrows indicate harvest event, Figure 4a). A
411 similar but less pronounced effect was found following grazing periods (light and dark brown
412 arrow, Figure 4a). A complete switch from net uptake to net CO₂ release was observed during
413 the first three months of 2012, after ploughing and during re-cultivation of the grassland. In
414 this specific year, the site only experienced snow cover for few days (Figure 1c) and
415 temperatures below 5 °C occurred more regularly than in all other years (Figure 1 b). Seasonal
416 CO₂ exchange was characterized by net release of CO₂ in winter (DJF), highest CO₂ uptake
417 rates were observed in spring (MAM), constant uptake rates during summer (JJA) which
418 however were lower than those measured in spring, and very low net release of CO₂ in fall
419 (Table 3). Average winter CO₂ exchange for the five-year observation period (gap-filled 30
420 min data) was $0.28 \pm 5.68 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ (SE = 0.04, Table 3). The restoration year 2012
421 showed a slightly different pattern with relatively large CO₂ release in winter and spring and
422 considerably lower uptake rates in summer. The years before the restoration (2010 and 2011)
423 were characterized by smaller net uptake rates during spring and summer when compared to
424 the post-ploughing years (2013 and 2014). Additionally, winter fluxes in 2010 and 2011 were
425 positive (net release of CO₂), while winter fluxes in the years 2013 and 2014 were showing a
426 small but consistent net uptake of CO₂ (Figure 4a, Table 3).

427
428 *CH₄ exchange*
429 The individual static chamber measurements (2011&2011) were often below the detection
430 limit and fluctuated around zero similar to the eddy covariance measurements (Figure 4b). Any
431 methane peaks expected due to freezing and thawing in late winter and early spring were not
432 observed. Also, commonly reported net emissions of methane during grazing of animals were
433 not seen (Figure 4b). Seasonal differences of methane exchange did not show a clear pattern
434 (Table 3). A comparison of methane fluxes obtained by both, static GHG chambers and EC
435 measurements as done for N₂O (see next paragraph) could not be performed due to a
436 malfunction of the respective detector in the gas chromatograph.

437

438 *N₂O exchange*

439 N₂O exchange was low during the majority of the days over the five-year observation period,
440 fluctuating around zero (Figure 4c). However, clear peaks in N₂O emissions were observed
441 following fertilization events or periods with high rainfall after a dry period in summer (i.e.
442 summer 2013 and 2014, Figures 3a and 4c). While event driven N₂O emissions were commonly
443 on the order of 4 to 8 nmol N₂O m⁻² s⁻¹ (Figure 4c), N₂O emissions following ploughing and
444 subsequent re-sowing of the grassland in 2012 lead to up to three times as high N₂O emissions
445 (Figure 4c, year 2012, see also Merbold et al. (2014)). Similar to methane, enhanced N₂O
446 emissions in late winter or early spring as reported by other studies could not be identified
447 (Figure 4c).

448 Background N₂O fluxes were estimated by analysing all high temporal resolution flux data but
449 excluding the restoration year 2012 and all values one week after a management event. Daily
450 average background fluxes were 0.21 ± 0.55 nmol m⁻² s⁻¹ (SE = 0.02). Differences in N₂O
451 exchange over the course of individual years became obvious when splitting the dataset into
452 the four seasons (winter – DJF, spring – MAM, summer – JJA and fall – SON). In contrast to
453 CO₂ exchange that showed large net uptake rates in spring, N₂O emissions were largest during
454 summer (JJA) and lowest in winter (DJF). As highlighted for the other gases, the year of
455 grassland restoration showed a completely different picture (Table 3).

456

457 *3.4 Annual sums and Global Warming Potential (GWP) of CO₂, CH₄ and N₂O*

458 Annual sums showed a net uptake of CO₂ during the two pre-ploughing years
459 (-695 g CO₂ m⁻² yr⁻¹ and -978 g CO₂ m⁻² yr⁻¹ in 2010 and 2011 respectively). Up to three times
460 of this net uptake was reached in 2013 and 2014, the two post-ploughing years (-2046 g CO₂
461 m⁻² yr⁻¹ and -2751 g CO₂ m⁻² yr⁻¹, Table 2). In contrast, the ploughing year 2011 was
462 characterized by a net release of CO₂ (1447 g CO₂ m⁻² yr⁻¹).

463 Methane budgets for the years 2010 and 2011 were not be calculated as many of the available
464 measurements were below the limit of detection. For the years 2012 – 2014, the annual methane
465 budget showed a minor release of 26.8 – 55.2 g CH₄ m⁻² yr⁻¹.

466 The Chamau site was characterized by a net release of nitrous oxide over the five-year study
467 period. While annual average N₂O emissions ranging between 0.34 and 1.17 g N₂O m⁻² yr⁻¹ in
468 the non-ploughing years, the site emitted 4.36 g N₂O m⁻² yr⁻¹ in 2012. **As an important note,**
469 **due to the limited data availability for the years 2010 and 2011, the budgets of those years are**
470 **likely incomplete.**

471 The global warming potential (GWP), expressed as the yearly cumulative sum of all gases after
472 their conversion to CO₂-equivalents, was negative during all years (between -387 and -2577
473 CO₂-eq. m⁻²) except for the ploughing year 2012 (+2629 CO₂-eq. m⁻²).

474 Overall, CO₂ exchange contributed more than 90% to the total GHG balance in 2011, 2013 and
475 2014. Clearly, CH₄ exchange was of minimal importance for the GHG budget (Table 2). In
476 2010, the contribution of CO₂ to the site's GHG budget was almost 70%, and N₂O contributed
477 about 30%. Only in 2012, the year of restoration, CO₂ and N₂O exchange contributed almost
478 equally to the site's overall GHG budget (55.1% and 43.9%, respectively).

479

480 *3.5. Carbon gains/losses of the Chamau site between 2010 and 2014*

481 The Chamau site assimilated on average $-441 \pm 260 \text{ g CO}_2\text{-C m}^{-2} \text{ yr}^{-1}$ ($4410 \text{ kg C ha}^{-1} \text{ yr}^{-1}$)
482 during the “business as usual” years (2010 and 2011 as well as 2013 and 2014). During the
483 restoration year the site lost $395 \text{ g CO}_2\text{-C m}^{-2}$ ($3950 \text{ kg C ha}^{-1}$) (Table 2). Carbon losses (and/or
484 gains) from methane were $< 1 \text{ g CH}_4\text{-C m}^{-2}$ during all five years.

485 Carbon was gained in both parcels during the pre-ploughing years (Table 4). Considerable net
486 losses of carbon were calculated for the ploughing year. In contrast, the post-ploughing years
487 were again recognized as years with large net gains in carbon. Over the observation period of
488 5 years, the Chamau grassland gained approximately 4 t C ha^{-1} , excluding losses via leaching
489 and deposition of C in form of dust.

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505 **4 Discussion**

506 The five-year measurement period is representative for other similarly managed grassland
507 ecosystems in Switzerland. Climate conditions were similar to the long-term average as
508 described in Wolf et al. (2013). Management activities, such as harvests and subsequent
509 fertilizer applications, were driven by overall weather conditions, (i.e. 2013 late spring, Figure
510 2a and b).

511

512 *4.1 Technical and methodological aspects of the study*

513 Different techniques are currently applied to measure GHG fluxes from a variety of ecosystems
514 (Denmead, 2008), each having its advantages and disadvantages or being chosen for a specific
515 purpose or reason. A common approach to study individual processes or time periods
516 contributing to specific greenhouse gas emissions is to measure with GHG chambers on the
517 plot scale (Pavelka et al., 2018). Chamber methods have been widely used to derive annual
518 GHG and nutrient budgets (Barton et al., 2015; Butterbach-Bahl et al., 2013). Critical
519 assessments of the suitability and associated uncertainty in chamber derived GHG budgets in
520 relation to sampling frequency have been published by Barton et al. (2013). Existing studies
521 have not only compared the two measurement techniques employed in this study (manual
522 chambers and eddy covariance) in grasslands before, but also estimated annual emissions based
523 on differing methodologies (Flechard et al., 2007; Jones et al., 2017). Additional confidence in
524 our approach was obtained from the N₂O emissions during the summer period 2013, where
525 both measurement techniques ran in parallel (Figure 3a and b). Annual budgets derived by
526 applying similar gap-filling approaches to the individual datasets led to comparable results
527 (Table 2).

528 We calculated detection limits for the individual GHGs from our manual chambers following
529 (Parkin et al., 2012). Detection limits were $0.34 \pm 0.26 \text{ nmol m}^{-2} \text{ s}^{-1}$, $0.05 \pm 0.02 \text{ nmol m}^{-2} \text{ s}^{-1}$,
530 and $0.06 \pm 0.06 \mu\text{mol m}^{-2} \text{ s}^{-1}$ for CH₄, N₂O and CO₂, respectively. Following this, methane flux
531 measurements frequently were below this limit of detection, hence we did not calculate
532 methane budgets for 2010 and 2011. The flux values measured with the EC technique between
533 2012 and 2014 compare well to similar measurements made by Felber et al. (2016) in an
534 intensively managed grassland in Western Switzerland. The observed values have been
535 identified to represent the soil methane exchange in EC measured fluxes (Felber et al. 2016).
536 N₂O fluxes in contrast were much better constrained by both methods due to clear N₂O sources
537 (i.e. fertilizer amendments) and better sensitivity of the instruments used by both techniques

538 for N₂O as compared to CH₄. Background N₂O emissions as observed in this study (0.21 ±
539 0.55 nmol m⁻² s⁻¹ (SE = 0.02)) compare well to estimates suggested by Rafique et al., (2011)
540 whom suggest an annual background N₂O losses of 1.8 kg N₂O-N for a grazed pasture (i.e.
541 0.20 nmol m⁻² s⁻¹).

542

543 *4.2 Annual GHG and C and N gains/losses*

544 Net carbon losses and gains estimated for the CH-Cha site between 2010 and 2015 were in
545 general within the range of values estimated by Zeeman et al., (2010) for the years 2006 and
546 2007. The slightly higher losses observed prior to ploughing may result from reduced
547 productivity of the sward. This becomes particularly visible when compared to the net
548 ecosystem exchange (NEE) of CO₂ values for the years after restoration. Losses via leaching
549 have previously been estimated to be of minor importance at this site (Zeeman et al., 2010) and
550 were therefore not considered in this study. Considerably higher C gains during post-ploughing
551 years were caused by enhanced plant growth in spring and summer. Restoration is primarily
552 done to eradicate weeds and rodents, favouring biomass productivity of the fodder grass
553 composition. Other grasslands in Central Europe, i.e. sites in Austria, France and Germany,
554 showed similar values for net ecosystem exchange (Hörtnagl et al., 2018). Still, total C budgets
555 as presented here are subject to considerable uncertainty which is strongly depending on
556 assumptions made for gap-filling etc. (Foken et al., 2004). Nevertheless, the values reported
557 here show the overall trend on C uptake/release of the site and clearly exceed the uncertainty
558 of ± 50 g C per year for eddy covariance studies as suggested by Baldocchi (2003).

559 Methane was of negligible importance for the C budget of this site. We did not observe distinct
560 peaks in CH₄ emissions in relation to grazing which is primarily due to the low grazing pressure
561 at CH-Cha. Studies carried out on pastures in Scotland, Mongolia, France and Western
562 Switzerland have shown that grazing can largely contribute to ecosystem-scale methane fluxes,
563 in particular if ruminants such as cattle are populating the EC footprint (Dengel et al., 2011;
564 Felber et al., 2015; Schönbach et al., 2012). If we included an approximation of methane
565 emissions of cattle which we may have missed in the EC flux measurements, we would have
566 to add **3.67 g CH₄-C m⁻² y⁻¹** to the current value of 1.48 g CH₄-C m⁻² in 2014 (Table 2). This
567 value is based on the average methane emissions of 404 g CH₄ head⁻¹ d⁻¹ stated in Felber et al.
568 (2016) and linking this to the average stocking density (4.04 head ha⁻¹) on the Chamua site
569 and the stocking duration (30 days in 2014). Still, the GHG budget as well as the C budget of
570 the site would not be altered.

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

575 Nitrogen inputs and losses via N_2O varied largely between the years before and after ploughing.
576 While the site was characterized by large N amendments prior to ploughing and with reduced
577 harvest, the picture was completely the opposite during the years after ploughing, with
578 considerably less N inputs compared to the nitrogen removed from the field via harvests.
579 Farmers aim every year at having a balanced N budget (fertilizer inputs = nutrients removed
580 from the field). Pasture degradation is the main motivation for enhanced fertilizer inputs in
581 order to stabilize forage productivity. Similarly, regular restoration of permanent pastures is
582 absolutely necessary (Cowan et al., 2016). So far, we identified only one study that investigated
583 the net effects on the overall GHG exchange following grassland restoration (Drewer et al.,
584 2017).

585

586 **5 Conclusion**

587 This study in combination with an overview of available datasets on grassland restoration and
588 their consequences on GHG budgets highlights the overall need of additional observational
589 data. While restoration changed the previous C sink to a C source at the Chamau site, the wider
590 implication in terms of the GWP of the site when including other GHGs have long-term
591 consequences (i.e. in mitigation assessments). Furthermore, this study showed the large
592 variations in N inputs and N outputs from this grassland and the difficulty farmers face when
593 aiming for balanced N budgets in the field. Still, the current study focused on GHGs only and
594 can thus not constrain the N budget but assess the losses of N via N_2O . Losses in form of NH_3 ,
595 N_2 and NO_x will have to be quantified to fully assess N budgets besides the overall fact that
596 GHG data following grassland restoration remain largely limited to investigate long-term
597 consequences. Fortunately, these are likely to become available in the near future by the
598 establishment of environmental research infrastructures (i.e. ICOS in Europe, NEON in the
599 USA or TERN in Australia) that aim at standardized, high quality and high temporal resolution
600 trace gas observation of major ecosystems, including permanent grasslands. With these
601 additional data, another major constraint of producing defensible GHG and nutrient budgets,
602 namely gap-filling procedures, will likely be overcome. New and existing data can be used to
603 derive reliable functional relations and artificial neural networks (ANNs) at field to ecosystem

604 scale that are capable of reproducing in-situ measured data. Once this step is achieved, both
605 the available data as well the functional relations can be used to improve, to train and to validate
606 existing biogeochemical process models (Fuchs et al., 2020). Subsequently, reliable
607 projections on both nutrient and GHG budgets at the ecosystem scale that are driven by
608 anthropogenic management as well as climatic variability become reality.

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

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619 **6 Tables and Figures**

620

621 **Table 1:** Table 1: Data availability of GHG fluxes measured over the five-year observation
622 period. Values are given as all data possible, raw processed values and high quality (HQ) data
623 or for the chamber flux data if above the detection limit, which were then used in the analysis.
624 Grey shaded areas represent time period where both methods (EC and static chambers) were
625 used simultaneously to estimate F_{N2O} . Static chamber flux data are further marked in *italic* font.

626

627 **Table 2:** Annual average CO_2 , CH_4 and N_2O fluxes and annual sums for the three GHGs as
628 well as carbon and nitrogen gain/losses per gas species. GWP were calculated for a 100-year
629 time horizon and based on the most recent numbers provided by IPCC (Stocker et al., 2013).
630 Annual budgets were derived from either gap-filled manual chamber (MC) or eddy covariance
631 (EC) measurements. n.c. stands for not calculated. Numbers in *italic* for N_2O in the years
632 2010/2011 are likely incomplete due to limited data availability. Sign convention: positive
633 values denote export/release, negative values import/uptake.

634

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

640

641 **Table 4:** Table 4: Carbon and nitrogen gains/losses through fertilization, harvest and GHGs
642 for the Chamau (CH-Cha) site in 2010- 2014. Values are given in $kg\ ha^{-1}$. Gains are indicated
643 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 ($\text{mg CO}_2\text{-C m}^{-2} \text{ h}^{-1}$, $\mu\text{g CH}_4\text{-C m}^{-2} \text{ h}^{-1}$ and $\mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$). 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, $\text{m}^3 \text{ m}^{-3}$) measured at 5 cm soil depth; (b) daily averaged air temperature ($^{\circ}\text{C}$), daily averaged soil temperature ($^{\circ}\text{C}$) and days with snow cover (horizontal bars); (c) daily averaged photosynthetic photon flux density (PPFD, $\mu\text{mol m}^{-2} \text{ s}^{-1}$). 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 N₂O 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), "P_h" = pesticide (herbicide) application). (b) 1:1 comparison between chamber based and eddy covariance based N₂O 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.

694
695 **Figure 4:** Temporal dynamics of gap-filled (except methane in 2010/2011) daily averaged
696 greenhouse gas (GHG) fluxes (white circles): a) (CO₂ exchange in $\mu\text{mol m}^{-2} \text{s}^{-1}$;
697 b) CH₄ exchange in $\text{nmol m}^{-2} \text{s}^{-1}$ and (c) N₂O exchange in $\text{nmol m}^{-2} \text{s}^{-1}$. Coloured circles indicate
698 manual chamber measurements. While both GHGs, CH₄ and N₂O were measured in 2010 and
699 2011 (blue circles), N₂O only was measured in 2013 (light blue circles). The grey dashed lines
700 indicate the beginning of a new year. Same color coding as used in Figure 3 a was used to
701 highlight management activities. Sign convention: positive values denote export/release,
702 negative values import/uptake. Grey lines behind the circles indicate standard deviation.
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741

742 **8 Author contributions**

743 LM and LH designed the study and wrote the first manuscript version. LM, CD, WE KF and
744 BW collected the data in the field. LH further provided the code for flux processing. All authors
745 revised and commented on the manuscript.

746

747 **9 Author contributions**

748 All flux and meta data are openly available via Fluxnet. The flux processing code is available
749 via the Grassland Sciences Group at ETH Zurich. Greenhouse gas chamber data is available
750 via Imer et al. 2013.

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