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Title: The use of biogeochemical models to evaluate mitigation of  
greenhouse gas emissions from managed grasslands

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Keywords: GHG emission intensity, livestock density; nitrogen  
fertilization, process-based model, sensitivity analysis

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Abstract: Simulation models quantify the impacts on carbon (C) and nitrogen (N) cycling in grassland systems caused by changes in management practices. To support agricultural policies, it is however important to contrast the responses of alternative models, which can differ greatly in their treatment of key processes and in their response to management. We applied eight biogeochemical models at five grassland sites (in France, New Zealand, Switzerland, United Kingdom and United States) to compare the sensitivity of modelled C and N fluxes to changes in the density of grazing animals (from 100% to 50% of the original livestock densities), also in combination with decreasing N fertilisation levels (reduced to zero from the initial levels). Simulated multi-model median values indicated that input reduction would lead to an increase in the C sink strength (negative net ecosystem C exchange) in intensive grazing systems:  $-64 \pm 74$  g C m<sup>-2</sup> yr<sup>-1</sup> (animal density reduction) and  $-81 \pm 74$  g C m<sup>-2</sup> yr<sup>-1</sup> (N and animal density reduction), against the baseline of  $-30.5 \pm 69.5$  g C m<sup>-2</sup> yr<sup>-1</sup> (LSU [livestock units]  $\geq 0.76$  ha<sup>-1</sup> yr<sup>-1</sup>). Simulations also indicated a strong effect of N fertilizer reduction on N fluxes, e.g. N<sub>2</sub>O-N emissions decreased from  $0.34 \pm 0.22$  (baseline) to  $0.1 \pm 0.05$  g N m<sup>-2</sup> yr<sup>-1</sup> (no N fertilization). Simulated decline in grazing intensity had only limited impact on the N balance. The simulated pattern of enteric methane emissions was dominated by high model-to-model variability. The reduction in simulated offtake (animal intake + cut biomass) led to a doubling in net primary production per animal (increased by  $11.6 \pm 8.1$  t C LSU<sup>-1</sup> yr<sup>-1</sup> across sites). The highest N<sub>2</sub>O-N intensities (N<sub>2</sub>O-N / offtake) were simulated at mown and extensively grazed arid sites. We show the possibility of using grassland models to determine sound mitigation practices while quantifying the uncertainties associated with the simulated outputs.

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23

24 *Keywords:* GHG emission intensity, livestock density; nitrogen fertilization, process-based  
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## 26 **1. Introduction**

27 Finding solutions to emerging ecological and societal challenges (climate change, food  
28 security, ecosystem sustainability) requires improved knowledge of the underlying processes  
29 affecting carbon-nitrogen (C-N) pools and fluxes in agricultural systems (West et al., 2002;  
30 Giardina et al., 2014; Campbell and Paustian, 2015). Grassland ecosystems have a potentially  
31 important role to play in meeting the challenge of climate change because they can act as a  
32 source or sink for atmospheric carbon dioxide (CO<sub>2</sub>) (Smith et al., 2008; Oates and Jackson,  
33 2014) and are a source of non-CO<sub>2</sub> greenhouse gases (GHG) such as nitrous oxide (N<sub>2</sub>O) and  
34 methane (CH<sub>4</sub>). Importantly, these GHG emissions can be manipulated by management such  
35 as the method of grazing and the fertiliser regime (Soussana et al., 2004; Herrero et al., 2016).  
36 Several grassland experiments have addressed the role of management on the short-term GHG  
37 balance and global warming potential (e.g. Allard et al., 2007; Soussana et al., 2007; Hörtnagl  
38 et al., 2018). However, direct measurement of C-N balances should be supplemented by the  
39 use of simulation models, to support the implementation of effective practices and policies in  
40 agriculture, e.g. to mitigate GHG emissions (Rosenzweig et al., 2014; Elliott et al., 2015;  
41 Folberth et al., 2016). Biogeochemical process models address many of the complex  
42 interactions of weather, soil, vegetation and management practices (Bondeau et al., 1999;  
43 Churkina et al., 1999; Huntzinger et al., 2012; Warszawski et al., 2014; Chang et al., 2015)  
44 and can do so over long time intervals that are not feasible with experimentation. Existing  
45 modelling studies have focused on the determination of the C source and sink activity of  
46 grasslands (Soussana et al., 2010). Grassland models have been shown to provide adequate  
47 accuracy in representing observed yield and GHG emissions across a wide range of  
48 environments and management intensities (e.g. White et al., 2008; Chang et al., 2013; Graux  
49 et al., 2013; Ben Touhami and Bellocchi, 2015; Ma et al., 2015; Senapati et al., 2016;  
50 Ehrhardt et al., 2018).

51 Models can thus be beneficial for decision makers and farmers because they can be  
52 used to explore the productivity and environmental performances of specific systems as a  
53 consequence of changed management. However, the effect of management on C and N fluxes  
54 in agriculturally managed permanent grasslands (not re-sown more frequently than every five  
55 years, which is the focus of this study) is often uncertain (Schulze et al., 2009; Ciais et al.,  
56 2010), and such uncertainties are reflected in the outputs of the models used to simulate  
57 responses to management (Sándor et al., 2017). Grasslands are highly complex ecosystems  
58 and their behaviour is affected by multifaceted interactions of management drivers with water  
59 and nutrient availability, soil physics, and vegetation dynamics (Rees et al. 2013; Soussana et  
60 al., 2013). The dynamic grassland simulation models developed since the 1990s (e.g.  
61 Challinor et al., 2013; Snow et al., 2014; Calanca et al., 2016; Jones et al., 2017a) differ  
62 greatly in their treatment of key processes, and hence in their response to environmental and  
63 management conditions (Brilli et al., 2017). A thorough assessment of the variation in the  
64 response, or sensitivity, of different grassland models to management factors can be critical in  
65 determining to what extent simulated responses may vary depending on the model used. From  
66 a policy perspective, it is critically important to identify the extent to which management  
67 interventions influence C-N fluxes (including productivity) prior to promoting policies that  
68 alter farming practices. If the impact of a given practice is uncertain, a sensitivity analysis can  
69 give information on the reliability of the models when representing C-N fluxes-management  
70 relationships under a variety of conditions. It is thus important to examine model behaviour  
71 under changed management in order to characterise the types of responses estimated, contrast  
72 the responses of different models and consider the reasons for these differences. In particular,  
73 hypotheses about the contribution of grassland management to GHG emissions can be tested  
74 *via* simulation models, which allow understanding, diagnosing and forecasting complex

75 interactions (Chen et al., 2008; Seijan et al., 2011; Graux et al., 2012; Sándor et al., 2017,  
76 2018).

77         Consequently, using five case studies, we tested the sensitivity of eight grassland  
78 models to gradients of management intensity that were selected for their potential to mitigate  
79 GHG emissions (e.g. Soussana et al., 2014; Abdalla et al., 2017). With the aim of increasing  
80 the reliability and confidence in simulated results, a multi-model ensemble approach was  
81 adopted to explore patterns of simulated C and N responses against imposed gradients of N  
82 fertilisation and animal stocking rate (to which grassland models are generally sensitive, after  
83 Brillì et al., 2017). For this study, we included a range of well-known grassland models, and  
84 used them to simulate biogeochemical and related outputs (productivity and energy  
85 measures). The wider ensemble analysis presented in Ehrhardt et al. (2018) forms the baseline  
86 for the work presented here, which analyses factors that may explain the major differences  
87 observed in model responses. We further explored to what extent multi-model ensembles can  
88 be used to help identify farming practices that reduce GHG emissions. While restricting the  
89 analysis to a limited set of management options, this study examines a wide range of output  
90 variables and thus provides a framework for assessing grassland performance where direct  
91 casual links with farming practices are not obvious, and changes in performance are difficult  
92 to measure. As a corollary outcome, viewing and interpreting a variety of model outputs lay  
93 ground for future model developments.

94

## 95 **2. Materials and Methods**

96         We refer to a sub-set of the grassland models described in Ehrhardt et al. (2018), in  
97 which models were initialized and calibrated using vegetation and soil variables, and surface-  
98 to-atmosphere fluxes at four sites worldwide. We used an ensemble of grassland models  
99 (Table 1) and compared their sensitivity to changes in management by comparing simulated

100 outputs against gradients of management practices. Model anonymity was maintained  
101 throughout the process and model results are presented without attributing them to specific  
102 models or modelling teams.

103

104 Table 1. The biogeochemical models used for testing the impact of grassland management  
 105 options.

Model / Version	Description / References
APSIM-SWIM v7.7	APSIM (The Agricultural Production Systems sIMulator; Holzworth et al., 2014) simulates several systems through the interaction among plants, animals, soil, climate and management. The model allows the analysis of the whole-farm system, including livestock, crop and pasture sequences and rotations.
APSIM-SoilWater v7.7	Users can select between two soil water models: the capacitance-based SoilWater (Probert et al., 1998) and SWIM, which is based on Richards' equation (Huth et al., 2012). The pasture model was that of Li et al. (2011).
DayCent v4.5 2006	DayCent is the daily time-step adaptation of the biogeochemical model CENTURY (Parton et al., 1998). It simulates biomass growth, soil C dynamics, N leaching, gaseous emissions (e.g. N <sub>2</sub> O, NO, N <sub>2</sub> , NH <sub>3</sub> , CH <sub>4</sub> and CO <sub>2</sub> ) and C fluxes (e.g. NPP, NEE) in croplands, grasslands, forests and savannahs, as affected by management practices (such as fertilization, tillage, pruning, cutting and grazing) and specific external disturbances (e.g. fires). Different versions of the model result in different parameter settings and a few variations in the model structure. DayCent v4.5 2006 applies grazing on a daily basis as linear impact on aboveground biomass and root/shoot ratio, with aboveground biomass removed as a percentage of total aboveground biomass.
DayCent v4.5 2010	DayCent v4.5 2010 and 2013 apply grazing on a daily basis with aboveground biomass removed as a percentage of total aboveground biomass rather than as continuous grazing. In DayCent models after 2013, water stress effect on biomass production differs from the previous versions.
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LPJmL v.3.5.003	LPJmL (Lund-Potsdam-Jena managed Land) explicitly simulates key ecosystem processes such as photosynthesis, plant and soil respiration, C allocation, evapotranspiration and phenology of nine plant functional types representing natural vegetation at the level of biomes (Sitch et al., 2003), and of 12 plant functional types (Bondeau et al., 2007; Rolinski et al., 2018).
PaSim	PaSim (Pasture Simulation model; Riedo et al., 1998; Calanca et al., 2007; Ma et al., 2015) is a process-based, grassland-specific ecosystem model that simulates grassland productivity and GHG emissions to the atmosphere. The model consists of sub-models for vegetation, grazing animals, microclimate, soil biology, soil physics and management.
SPACSYS v5.0	SPACSYS (Wu et al., 2007, 2015) is a multi-layer, field scale, weather-driven and daily-time-step dynamic simulation model. The current version includes a generic plant growth and development, C and N cycling, plus simulation of soil water that includes representation of water flow to field drains as well as downwards through the soil layers, together with a heat transfer component. The model simulates root architecture.

117 We present multi-model medians and box-plots, and focus on long-term averages for  
 118 the same four grassland sites (G1 to G4) described in Ehrhardt et al. (2018) plus an additional  
 119 site (G5) for which full calibration was only completed after the initial publication (Table 2).  
 120 Overall, there is a lack of case studies in Asia, Africa and South America (which would have  
 121 extended the comprehensiveness of the research), but sites from G1 to G5 are intended to  
 122 represent situations commonly encountered in temperate grasslands. While the choices made  
 123 are described in Ehrhardt et al. (2018), in summary it was thanks to international  
 124 collaborations that we could pool and share experimental data for five grassland sites (one  
 125 more than in Ehrhardt et al., 2018). These sites provided high- quality, previously published  
 126 data encompassing climate, soil, agricultural practices, and C and N fluxes.

117  
 118 Table 2. Selected grassland sites for the modelling exercise.

<b>General description</b>	<b>Grassland sites</b>				
Site code	G1	G2	G3	G4	G5
Country	United States	New Zealand	France	United Kingdom	Switzerland
Location	Mandan	Flockhouse	Laqueuille	Easter Bush	Chamau
Climate <sup>1</sup>	Dfb (humid continental)	Cfb (oceanic)	Cfb (oceanic)	Cfb (oceanic)	Cfb (oceanic)
Latitude	46.77	-40.20	45.64	55.52	47.20
Longitude	-100.89	175.30	2.74	-3.33	8.40
Elevation a.s.l. (m)	591	30	1040	190	393
Simulation period	2003-2006	1997-2008	2003-2012	2002-2010	2010-2013
Mean annual minimal air temperature (°C) <sup>2</sup>	0.0±1.0	9.1±0.5	4.0±0.6	4.6±0.9	4.4±0.6
Mean annual maximal air temperature (°C) <sup>2</sup>	11.9±1.3	17.6±0.6	11.0±0.8	11.4±0.8	14.7±0.8
Mean annual cumulated precipitation (mm) <sup>2</sup>	411±128	896±107	1047±144	961±142	1084±143
<b>Management</b>					
Type	Grazed	Grazed	Grazed	Grazed / mown	Grazed / mown
Animal type	Yearling steers	Non-lactating sheep	Heifers	Ewes, lambs, heifers and calves	Sheep
Mean annual number of grazing days <sup>3</sup>	107	22	163	162	14
Stocking rate (LSU ha <sup>-1</sup> yr <sup>-1</sup> ) <sup>3</sup>	0.08	1.29	1.34	0.76	0.10
Vegetation type	C3 grasses	C3 grasses,	C3 grasses,	C3 grasses	C3 grasses,



		legumes, forbs, C4 grasses	legumes, forbs		legumes
Mean annual number of cutting events <sup>3</sup>	0	0	0	0.9	6.3
Total annual N fertilisation (kg N ha <sup>-1</sup> yr <sup>-1</sup> ) <sup>3</sup>	0	0	210	220	230
<b>Soil properties</b>					
Soil type <sup>4</sup>	Calcic Siltic Chernozem	Mollic Umbrisol	Loamic Andosol	Eutric Cambisol	Gleysol
Maximum depth of the soil profile (m)	4	0.9	0.9	1.0	1.0
Number of documented layers	6	4	5	5	4
Soil texture: ~sand (%) <sup>5</sup>	29.7	93.1	24.6	22.9	57.2
~silt (%) <sup>5</sup>	51.0	3.8	55.5	19.0	28.9
~clay (%) <sup>5</sup>	19.5	3.1	21.8	58.1	14.0
Bulk density (gcm <sup>-3</sup> ) <sup>5</sup>	1.17	1.20	0.67	1.45	1.34
<b>References</b>	Liebig et al. (2006, 2010, 2013)	Newton et al. (2010, 2014)	Allard et al. (2007); Klumpp et al. (2011)	Skiba et al. (2013); Jones et al. (2017b)	Imer et al. (2013); Merbold et al. (2014)

119 <sup>1</sup>Köppen-Geiger climate classification (Kottek et al., 2006).

120 <sup>2</sup>Mean minimum and maximum air temperatures, and precipitation totals calculated over 30 years (1980-2009)  
121 using AgMERRA (<https://data.giss.nasa.gov/impacts/agmipcf/agmerra>) meteorological datasets (Ruane et al.,  
122 2015).

123 <sup>3</sup>Mean values over the simulation period. Grazing at G2 site was on a rotational basis, i.e. animals were brought  
124 in at intervals for short periods at a high stocking rate, while at all other sites grazing was by set-stocking, i.e.  
125 animals were maintained continuously on the pasture at a low stocking rate.

126 <sup>4</sup>World Reference Base for Soil Resources (FAO, 2014).

127 <sup>5</sup>Mean values across multiple layers.

128

129 To analyse the sensitivity of models with respect to changes in grassland management  
130 practices, viz. animal stocking density and N fertilisation, management scenarios were  
131 obtained by adjusting the observed baseline management (business-as-usual) for each site  
132 with systematic decrements over a range of values (Table 3). Sensitivity is defined as the  
133 proportional change in models outputs that results from a change in a given factor (here  
134 management practices).

135 In our study-sites, two major practices are responsible for C and N fluxes from  
136 grasslands: (1) vegetation removal and (2) fertilizer inputs. The harvesting of vegetation was  
137 predominantly controlled by grazing animals for the majority of sites. The exception was G5  
138 where the grazing was light and vegetation was predominantly removed by cutting.  
139 Accordingly, a reduction in grassland use was assessed by a limitation of livestock density,

140 either alone or together with reduction (down to cessation) of fertilizer N in N-fertilised sites  
 141 (G3, G4 and G5) (Table 3). The livestock unit (LSU) based on the grazing equivalent of one  
 142 adult cow was used to compare different animal types (yearling steers, non-lactating sheep,  
 143 ewes, lambs, heifers and calves).

144

145 Table 3. Design of management options (where 100% indicates the baseline - business-as-  
 146 usual - management scenario).

Action <sup>1</sup>	Sites	Description
Reduction of livestock density (LD)	G1, G2, G3, G4, G5	The livestock density in the pasture was decreased in five steps of 10% (from 100% down to 50% of the livestock density indicated by the default standard management)
Abbreviations <sup>2</sup> : LD <sub>90</sub> , LD <sub>80</sub> , LD <sub>70</sub> , LD <sub>60</sub> , LD <sub>50</sub>		
Reduction of livestock density (LD) and nitrogen (N) fertilizer	G3, G4, G5	The amount of mineral or slurry N added to the pasture was decreased in five steps of 20% (from 100% to 0% of the N amount indicated by the default standard management), whilst the livestock density in the pasture is decreased in five steps of 10% (from 100% down to 50% of the livestock density indicated by the default standard management)
Abbreviations <sup>2</sup> : LD <sub>90</sub> N <sub>80</sub> , LD <sub>80</sub> N <sub>60</sub> , LD <sub>70</sub> N <sub>40</sub> , LD <sub>60</sub> N <sub>20</sub> , LD <sub>50</sub> N <sub>0</sub>		

147 <sup>1</sup>When animal density was decreased, cutting events (if present) were left unaltered. When present,  
 148 supplementary feeding was proportionally reduced along with the animal density.

149 <sup>2</sup>Percent livestock density (90, 80, 70, 60, 50) or N fertilizer (80, 60, 40, 20, 0) against baseline (business-as-  
 150 usual). Without assessing all possible LD x N combinations, we focussed on reducing overall levels of  
 151 management intensity through reductions in N inputs and grazing levels (according to most agri-environment  
 152 schemes for grassland; Atkinson et al., 2005).

153

154 Impacts of the defined changes in management were calculated on the changes in a set  
 155 of output variables related to biomass production and C-N fluxes (Table 4). Fluxes of CO<sub>2</sub>  
 156 included emissions from ecosystem respiration (R<sub>ECO</sub>), respiration from plants (R<sub>PLANT</sub>), soil  
 157 (R<sub>SOIL</sub>) and grazing animals (R<sub>ANIMAL</sub>) as well as estimates of the plant production of organic  
 158 compounds from atmospheric CO<sub>2</sub> (GPP) and other system variables: Net Ecosystem  
 159 Exchange, NEE=R<sub>ECO</sub>-GPP; Net Primary Production, NPP=GPP-R<sub>PLANT</sub>; Net Biome

160 Production,  $NBP = -NEE + C$  losses through enteric  $CH_4$  emissions at pasture, forage harvests  
161 and milk production at pasture.

162 Methane released from soil and enteric fermentation in animals was included in the list  
163 of non- $CO_2$  fluxes, along with the gaseous N compounds emitted to the atmosphere:  $N_2$  (N  
164 gas),  $NO_x$  (N oxides: the sum of N monoxide, NO, and N dioxide,  $NO_2$ ),  $N_2O$  (nitrous oxide)  
165 and  $NH_3$  (ammonia). Nitrogen lost by nitrate ( $NO_3$ ) leaching was also examined. The  
166 biological information included productivity measures such as the plant biomass produced  
167 above – and below-ground (AB and BB), two outputs of agronomic interest – the plant  
168 biomass consumed by grazing animals (Intake) or otherwise harvested (HAB), and their sum  
169 (Offtake) – and the energy that ultimately is utilised by grazing animals ( $ME_{Offtake}$ : offtake  
170 metabolisable energy).

171 To estimate the amount of plant biomass available for feeding animals, the annual  
172 NPP values were normalized by animal stocking rates. We also expressed C and N fluxes  
173 relative to the overall productivity of the system, so that we could express the intensity of  
174 GHG emissions on the basis of productivity (i.e. g of emitted C or N per g C of harvested or  
175 per g C ingested dry matter, DM). This approach is similar to the concept of ‘yield-scaled  
176 emission’ or emissions intensity as defined by Van Groenigen et al. (2010) and has important  
177 policy significance and delivers results that are relevant to stakeholders (Venterea et al, 2011;  
178 Valin et al., 2013). For this purpose, three additional variables were analysed, representing the  
179 ratios of  $CO_2$ -C,  $N_2O$ -N and  $CH_4$ -C emissions to the total amount of C biomass (Offtake)  
180 consumed by animals (Intake) and harvested as fodder (HAB):  $Int_{CO_2-C} = -NEE/Offtake$ ,  $Int_{N_2O-N} =$   
181  $N_2O-N/Offtake$ ,  $Int_{CH_4-C} = CH_4-C/Offtake$ .

182

183 Table 4. Model outputs (annual cumulative) generated by each model (✓: available; NA: not  
 184 available) and assessed in the study. The identities of models were kept anonymous by using  
 185 the same model codes as in Ehrhardt et al. (2018).

Variable/Models			M05	M06	M07	M08	M16	M22	M24	M28
CO <sub>2</sub> fluxes	GPP (gross primary production): g C m <sup>-2</sup>		✓	✓	✓	✓	✓	✓	✓	✓
	NPP (net primary production): g C m <sup>-2</sup>		✓	✓	✓	✓	✓	✓	✓	✓
	NEE (net ecosystem exchange): g C m <sup>-2</sup>		✓	✓	✓	✓	✓	✓	✓	✓
	NBP (net biome production): g C m <sup>-2</sup>		NA	✓	NA	NA	✓	NA	✓	✓
	R <sub>ECO</sub> (ecosystem respiration): g C m <sup>-2</sup>		✓	✓	✓	✓	✓	✓	✓	✓
	R <sub>PLANT</sub> (plant respiration): g C m <sup>-2</sup>		✓	✓	✓	✓	✓	✓	✓	✓
	R <sub>SOIL</sub> (soil respiration): g C m <sup>-2</sup>		✓	✓	NA	✓	✓	✓	✓	✓
	R <sub>ANIMAL</sub> (animal respiration): g C m <sup>-2</sup>		NA	✓	NA	NA	✓	NA	✓	NA
Non CO <sub>2</sub> fluxes <sup>1</sup>	CH <sub>4</sub> emissions (methane) <sup>2</sup> : g C m <sup>-2</sup>		NA	✓	✓	✓	✓	NA	✓	NA
	N <sub>2</sub> O (nitrous oxide) emissions: g N m <sup>-2</sup>		✓	✓	✓	✓	✓	NA	✓	✓
	NH <sub>3</sub> (ammonia) emissions: g N m <sup>-2</sup>		NA	✓	NA	✓	✓	NA	✓	NA
	NO <sub>x</sub> (nitrogen oxides) emissions: g N m <sup>-2</sup>		NA	NA	✓	✓	NA	NA	NA	NA
	N <sub>2</sub> (nitrogen gas) emissions: g N m <sup>-2</sup>		NA	✓	✓	✓	NA	NA	✓	✓
	N (nitrogen) leaching: g N m <sup>-2</sup>		NA	✓	NA	✓	✓	NA	✓	✓
Productivity and energy	Aboveground (AB): g DM m <sup>-2</sup>	biomass	NA	✓	✓	✓	✓	✓	✓	✓
	Belowground (BB): g DM m <sup>-2</sup>	biomass	NA	✓	✓	✓	✓	NA	✓	✓

Harvested biomass (HAB): g DM m <sup>-2</sup>	NA	✓	✓	✓	✓	NA	✓	✓
Animal intake (Intake): g DM m <sup>-2</sup>	NA	✓	NA	✓	✓	✓	✓	✓
Metabolisable energy of Offtake (grazing plus harvesting) (ME <sub>Offtake</sub> ): MJ kg <sup>-1</sup> DM	NA	✓	NA	NA	NA	NA	✓	NA

186 <sup>1</sup>Fluxes are expressed in units of C (CH<sub>4</sub>-C) and N (N<sub>2</sub>O-N, etc.).

187 <sup>2</sup>CH<sub>4</sub> emissions include emissions from both animals (enteric) and their manure. The latter were estimated for M16, and the  
188 former were estimated for M06, M08 and M24; all estimations were based on Clark et al. (2003).

189

190 Following Sándor et al. (2016), we report the proportional change, named effect size  
191 of mean annual output variables from a change in each factor relative to the baseline  
192 management at each site.

193 The N<sub>2</sub>O-N emission factor (EF) for fertiliser was calculated as percent ratio of the  
194 total yearly N<sub>2</sub>O-N emissions over the amount of the annual N fertiliser. This simplified  
195 version of the N<sub>2</sub>O-N emission factor calculation does not take into account of background  
196 emissions because not all the models allowed for a consistent estimation of this component.  
197 For this reason, following the IPCC (2006) guidelines, 1 kg N<sub>2</sub>O-N ha<sup>-1</sup>yr<sup>-1</sup> background  
198 emission was subtracted from the simulated values.

199 The present study was based on yearly aggregated model outputs. R software (R Core  
200 Team, 2016) was used for statistical computing and visualization. Accounting for the different  
201 global warming potential (GWP) of CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O, total GHG balances were achieved  
202 by converting CH<sub>4</sub> and N<sub>2</sub>O emissions rates to CO<sub>2</sub> equivalents (CO<sub>2</sub>e) using the 100-year  
203 Global Warming Potential (GWP<sub>100</sub>) as established in national GHG inventories, i.e.: 1 kg  
204 N<sub>2</sub>O=298 kg CO<sub>2</sub>e, 1 kg CH<sub>4</sub>=25 kg CO<sub>2</sub>e (IPCC, 2006;  
205 <https://www.epa.gov/ghgemissions/understanding-global-warming-potentials>). Mass factors  
206 were also applied to model outputs, the latter being expressed in C and N units: 1 kg CH<sub>4</sub>-  
207 C=1.33 kg CH<sub>4</sub>, 1 kg N<sub>2</sub>O-N=1.57 kg N<sub>2</sub>O, 1 kg CO<sub>2</sub>-C=3.67 kg CO<sub>2</sub>.

208

### 209 3. Results and Discussion

210 Simulated results are presented and discussed separately, with selected graphs, for the  
211 following groups of variables: CO<sub>2</sub> fluxes, non-CO<sub>2</sub> fluxes, productivity and energy, and  
212 emission intensities. Additional results are provided in the supplementary material (Figs. A to  
213 S).

214

#### 215 3.1. CO<sub>2</sub> fluxes

216 In the baseline scenario, GPP showed a wide range of variations in multi-model  
217 medians (137.2-1732.4 g C m<sup>-2</sup>), while animal respiration (R<sub>ANIMAL</sub>) was the output with the  
218 least divergent results amongst the models (0.0-211.6 g C m<sup>-2</sup>; Fig. 1). For the R<sub>ANIMAL</sub>, model  
219 differences tended to be smaller at lower input levels, especially when animal density was  
220 reduced without reductions in N fertilisation (0.0–143.9 g C m<sup>-2</sup>). For plant respiration  
221 (R<sub>PLANT</sub>), an increase of model variability was associated with the reduced influence of the  
222 livestock. It is also interesting to note that net ecosystem exchange (NEE) values showed  
223 large variability among models with intermediate intensification levels, e.g. 70% reduction of  
224 LD and 40% reduction of N. The greatest variability was simulated under the mowing-  
225 dominated G5 site (Figs. A and B in supplementary material), but the model variability was  
226 also high under mowing and grazing combined management at G4 and under intensively  
227 grazed (by heifers) management at the G3 site. Reduction in N fertilization tended to decrease  
228 NEE variability at the G3 and G4 sites. In G5, the analysis of proportional changes indicated -  
229 with the combined reduction of N fertilisation and animal density - a clear linear decrease in  
230 NEE compared to the baseline (Fig. C in supplementary material). Since NEE is defined as  
231 the difference between ecosystem respiration (R<sub>ECO</sub>) and gross primary production (GPP), the  
232 variability of its basic components have an effect on the spread of NEE values. The ensemble  
233 uncertainty of GPP and R<sub>ECO</sub> were highest at G2, G3 and G4 sites, associated with the highest

234 animal densities (Table 2). This suggests that the intensification of grazing management tends  
235 to increase the variation of GPP and  $R_{\text{ECO}}$  estimates between models, with a smaller  
236 uncertainty envelope at the G1 and G5 sites, where altered animal density variation is very  
237 low ( $0.1\text{-}0.04 \text{ LSU ha}^{-1} \text{ yr}^{-1}$ ).

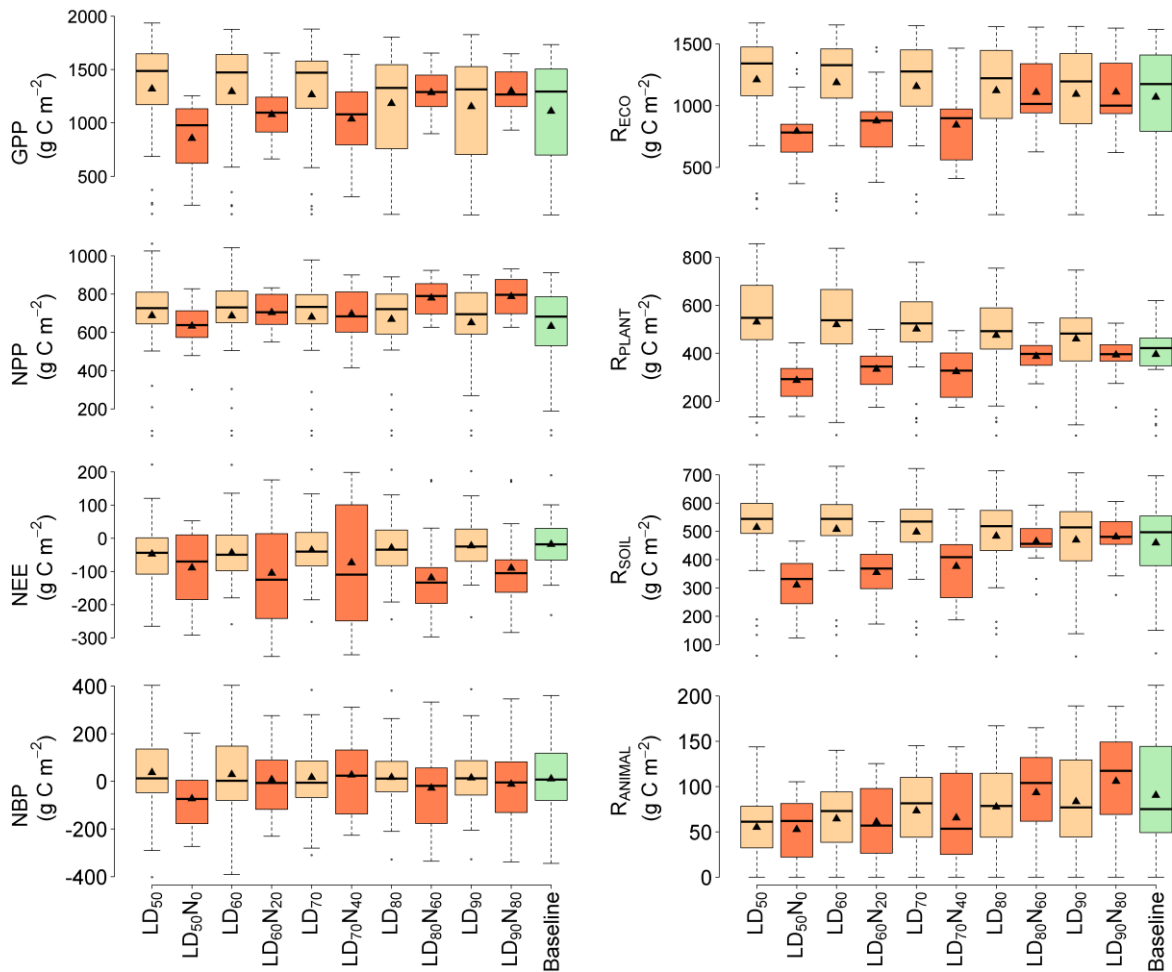
238 The five grassland sites showed different dynamics in C fluxes with respect to the  
239 simulated management options, with NEE varying between  $-231.3$  and  $+189.2 \text{ g C m}^{-2} \text{ yr}^{-1}$ ,  
240 considering all sites and simulation years. These results suggest higher NEE ( $-19.0 \pm 75.9 \text{ g}$   
241 and  $-47.6 \pm 89.8 \text{ C m}^{-2} \text{ yr}^{-1}$  for baseline and LD<sub>50</sub>, respectively), or lower C uptake, than  
242 Sousanna et al. (2007) concluded from nine European grassland sites equipped with eddy-  
243 covariance flux measurements, which showed an average net sink of atmospheric CO<sub>2</sub> with  
244 NEE of  $-240 \pm 70 \text{ g C m}^{-2} \text{ yr}^{-1}$  (which is in the range  $-486.3$  to  $24.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ , or  $-1783$  to  $-91$   
245  $\text{g CO}_2 \text{ m}^{-2} \text{ yr}^{-1}$ , provided by Hörtnagl et al., 2018 for managed grasslands in Central Europe).  
246 The site-by-site analysis (Fig. C in supplementary material) indicated, except at the G5 site,  
247 that C uptake was the dominant process. At the G5 site, an NEE of  $<0$  only occurred with  
248 LD<sub>70</sub>N<sub>40</sub> management options. At this site, the greatest model uncertainty in NEE values  
249 occurred with the LD<sub>70</sub>N<sub>40</sub> management option (Fig. 1), related to differently simulated CO<sub>2</sub>  
250 release and uptake processes. According to the median values, the G5 site reached the highest  
251 amount of C sequestration ( $\sim 113.7 \text{ g C m}^{-2} \text{ yr}^{-1}$ ) at LD<sub>50</sub>N<sub>0</sub>. The general observation from the  
252 modelled sites of increasingly negative NEE in response to increasing N inputs is consistent  
253 with experimental observations that report increases in the flow of C to the soil in response to  
254 long-term fertiliser N use (Ammann et al. 2007; Skiba et al. 2013). For the period during  
255 which the C is sequestered, it is removed from the atmosphere and does not contribute to  
256 global warming. This effect is simulated at both grazed (G1, G2 and G3) and grazing  
257 dominated (G4) sites, for all scenarios. Owing to the large quantity of harvested aboveground  
258 biomass ( $332.1 \pm 62.3 \text{ g DM m}^{-2} \text{ yr}^{-1}$  on average for the baseline scenario), the G5 site seems to

259 release C from the soil. However, the grazing removal ( $395.8 \pm 38 \text{ g DM m}^{-2} \text{ yr}^{-1}$  on average  
260 for the baseline scenario of G3 site) may drive less radical changes in the C balance. Overall,  
261 these simulation results are consistent with eddy-covariance measurement data (Senapati et  
262 al., 2014; Koncz et al., 2017), in which mown treatments were observed to release C, whilst  
263 grazed treatments acted as net C sinks. For instance, in Senapati et al. (2014) mown  
264 treatments had lower annual net C storage ( $22.7 \pm 32.3 \text{ g C m}^{-2} \text{ yr}^{-1}$ , net sink), related to hay  
265 removal, than grazed plots ( $140.9 \pm 69.9 \text{ g C m}^{-2} \text{ yr}^{-1}$ , net sink) - though the observed site  
266 (Lusignan, France) was recently converted from cropland to grassland, so would be expected  
267 to be increasing in soil C (Senapati et al., 2014).

268         Considering all the sites, the estimated average C exchange (net biome productivity,  
269 NBP, Fig. 1) ranged between -176.9 (sink) and +140.4  $\text{g C m}^{-2} \text{ yr}^{-1}$  (source), with its extremes  
270 at LD<sub>50</sub>N<sub>0</sub> and LD<sub>60</sub> management options, respectively. This high variability was caused by  
271 different management systems at G4 (grazed and mown) site (Figs. A and B in supplementary  
272 material), while the extensification combined with N reduction tended to increase C storage in  
273 some cases, e.g. at G5 site. Owing to the high organic C exports (from haycut and/or intensive  
274 cattle grazing: 1.34 and 1.21 LSU ha<sup>-1</sup> yr<sup>-1</sup>), which could be greater than C imports from  
275 manure and slurry, the soil processes would be dominated by C emissions at the G2 site  
276 (intensive scenarios). The N mitigation reduced the net biome production at G3 site even  
277 further.

278





279

280 Fig. 1. Changes in CO<sub>2</sub> fluxes (g C m<sup>-2</sup>) calculated over multiple years at five sites, for ten  
 281 altered management scenarios and the baseline (as in Table 3; LD: livestock density; N:  
 282 nitrogen). For each management level, triangles show the multi-model (as in Table 4) mean,  
 283 black lines show multi-model median. Boxes delimit the 25<sup>th</sup> and 75<sup>th</sup> percentiles. Whiskers  
 284 are 10<sup>th</sup> and 90<sup>th</sup> percentiles. Points indicate outliers.

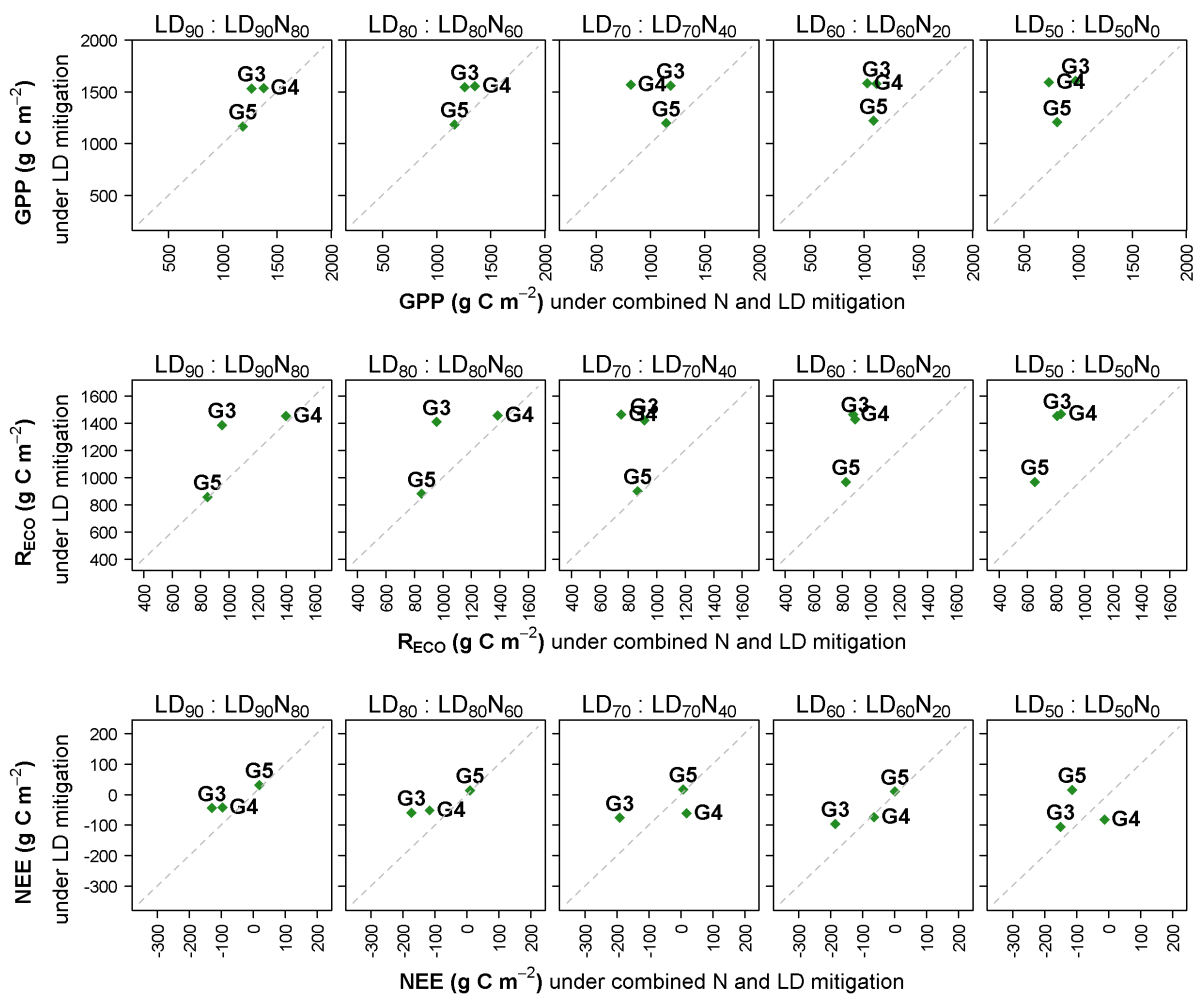
285

286 Ecosystem respiration (R<sub>ECO</sub>), together with R<sub>PLANT</sub> and R<sub>SOIL</sub>, showed a linear  
 287 decrease as LD and N levels simultaneously decreased, but tended to increase with a  
 288 reduction in animal density only (Fig. 1). Animal respiration (R<sub>ANIMAL</sub>) tended to decrease as  
 289 animal density decreased, though the multi-model median line (Fig. 1) was associated with  
 290 some uncertainty at the baseline and LD<sub>70</sub>N<sub>40</sub> options (i.e. 30% less animals and 60% less N  
 291 fertiliser). Site-by-site analyses showed (Figs. A and B in supplementary material) that the

292 greatest simulated  $R_{\text{ECO}}$  occurred with G3 and G4 grazing systems. Reduction in N fertilizer  
293 tended to decrease  $R_{\text{ECO}}$ , however the variability of soil respiration ( $R_{\text{SOIL}}$ ) increased the  
294 uncertainty at G4, particularly under sheep-heifer grazing. Based on model simulations, the  
295 main losses of  $\text{CO}_2$  at the G4 site were caused by  $R_{\text{SOIL}}$  and plant respiration ( $R_{\text{PLANT}}$ ).  
296 Simulated yearly  $R_{\text{ANIMAL}}$  values, and their proportional changes (Fig. D in supplementary  
297 material) with respect to the baseline management showed a distinct emission decrease with  
298 extensification (which is not the case with  $R_{\text{PLANT}}$  and  $R_{\text{SOIL}}$ , Figs. A and B in supplementary  
299 material), at a quasilinear rate (by  $y=-0.56+0.05x$ ,  $R^2\sim 1$  for LD changes and  $y=-0.62+0.05x$ ,  
300  $R^2=0.99$  for N and LD combined options at G3 site). The data also suggest that there were  
301 substantial differences among models in the estimated effects of altered management. These  
302 differences were amplified when N fertilization was decreased jointly with animal density  
303 (e.g. option  $\text{LD}_{70}\text{N}_{40}$  at G4 and G5; Fig. D in supplementary material), which suggests that  
304 interactions between the two factors may make a sizeable contribution to this variability in the  
305 response of different models.

306 Some relationships between model outputs and management inputs were apparent  
307 from an analysis of data at each site. GPP decreased strongly with stocking rates while  
308  $R_{\text{ANIMAL}}$  increased, with the exception of G2 site (Fig. E in supplementary material). The  
309 simulated outputs at this site may reflect different and non-linear responses of alternative  
310 models (Fig. F in supplementary material). Often, the offtake increased and NPP decreased  
311 with management intensification, albeit with large differences between models. With M06,  
312 the highest Offtake was simulated when animal density decreased by 30%, while this  
313 happened at 10% lower animal density with M24. M08 simulated the highest NPP value at  
314 30% less livestock density, while with M28 the highest NPP values were simulated at the  
315 most intensive management condition.

316 The influence of N fertilization was investigated at G3, G4 and G5 sites (Fig. 2) by  
 317 comparing the combined effect of livestock density reduction and N fertilisation reduction. In  
 318 terms of GPP (Fig. 2) and NPP (Fig. L in supplementary material), the simulations showed a  
 319 considerable decrease in GPP with >60% less applied N fertilizer at site G4. The  $R_{ECO}$  (Fig. 3,  
 320 middle panel) values also decreased with extensification, where the N reduction had a greater  
 321 effect at the G3 and G4 sites. Animal respiration was driven by livestock density (Fig. G in  
 322 supplementary material), while soil and plant respiration were mainly influenced by N inputs  
 323 (lower  $R_{PLANT}$  and  $R_{SOIL}$  with lower N inputs). In terms of NEE (Fig. 2, lower panel) and NBP  
 324 (Fig. G in supplementary material) the trend was less obvious, owing to differences in  
 325 management and site conditions.



329 Fig. 2. Comparison of the combined effect of livestock density reduction (LD from 90% down  
330 to 50% of initial density) and the N fertilisation reduction (N from 80% down to 0% of initial  
331 amount) at G3, G4 and G5 sites for gross primary production (GPP), ecosystem respiration  
332 ( $R_{ECO}$ ) and net ecosystem exchange (NEE) using the multi-model median.

333

334 The effectiveness of the different management strategies such as fertilizer amounts,  
335 different animal stocking rates, grazing alone or combined with mowing, was also influenced  
336 by site-specific soil (type and depth) and weather conditions (ie. Precipitation). If we are to  
337 distinguish between environmental from management effects, then precipitation patterns must  
338 be taken into account as it can also have an influence on the results of the CO<sub>2</sub> fluxes (e.g.  
339 Polley et al., 2010). In our simulation study, the amount of precipitation showed a positive  
340 correlation with  $R_{SOIL}$  and all the other investigated CO<sub>2</sub> fluxes (Fig. H in supplementary  
341 material) apart from NEE and NBP. The respiration outputs demonstrated higher sensitivity to  
342 N fertilization, than to percent livestock density changes. NEE values suggested greater  
343 respiration in very arid years (such as some years at G1 site, where annual mean precipitation  
344 was  $271 \pm 141$  mm), where the animal density reduction did not reduce the amount of CO<sub>2</sub>  
345 emissions. A recent review highlighted the particular sensitivity of warm and dry climates to  
346 change in stocking density where increased livestock density was associated with significantly  
347 lower rates of C sequestration (Abdalla et al., 2018). Gilmanov et al. (2007) pointed out that  
348 organic and semi-arid grassland sites have the potential to become C sources. With decreasing  
349 stocking rate, NEE tended to increase above 800 mm annual precipitation. In humid and very  
350 humid years, the soil may be saturated and anaerobic, and organic C decomposition can be  
351 slowed or stopped under these conditions (yet anaerobic decomposition of partly decomposed  
352 organic matter may emit methane, e.g. Bannert et al., 2012). The variability of NEE decreased

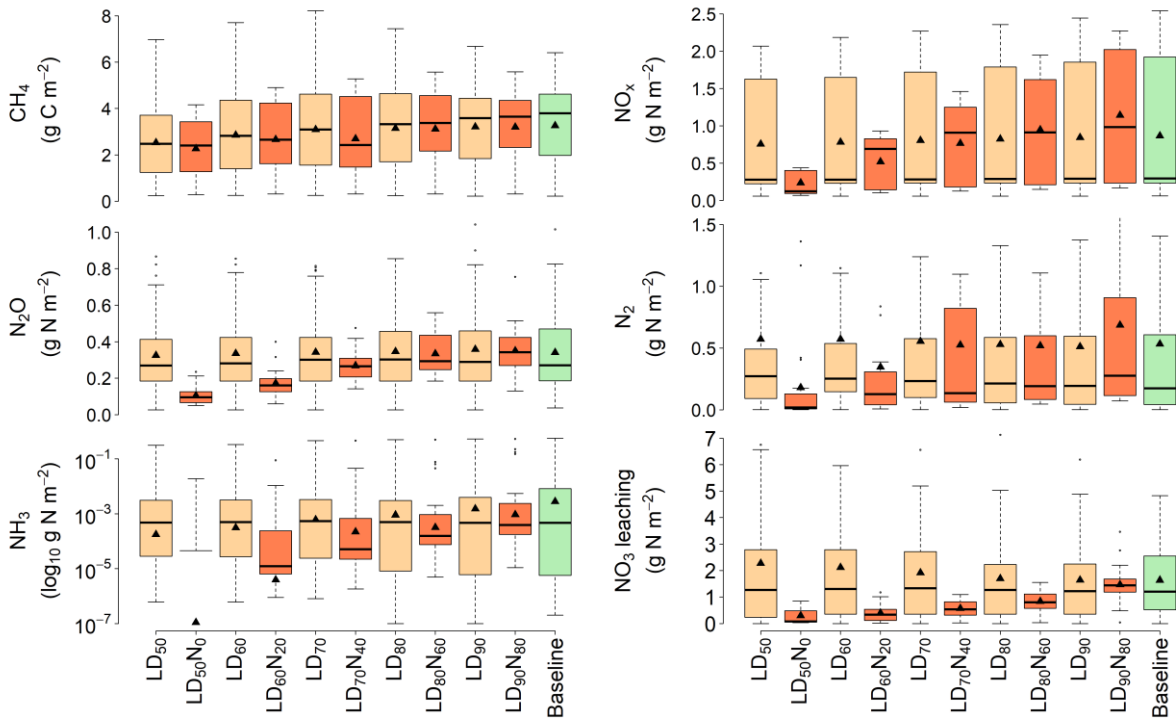
353 with N reduction, e.g. the most extensive treatment showed the smallest difference from zero  
354 (Fig. H in supplementary material).

355

### 356 3.2. *Non-CO<sub>2</sub> fluxes*

357 A recent IPCC report (2014) and other analyses (Schulze et al., 2009; Tubiello et al.,  
358 2014; Gerber et al., 2015; Hörtnagl et al., 2018) highlight the importance of the reduction of  
359 non-CO<sub>2</sub> GHGs, as an important element of agricultural mitigation strategies. Particularly  
360 methane and nitrous oxide cause longer-term warming effects than CO<sub>2</sub>. There were clear  
361 trends in model responses, indicating decreases in N<sub>2</sub>O-N, NH<sub>3</sub>-N, NO<sub>x</sub>-N, N<sub>2</sub> and NO<sub>3</sub>-N  
362 leaching with reductions in N fertilizer, while there were no clear trends in the response to  
363 animal livestock reduction (Fig. 3, all models and sites confounded). Only NO<sub>3</sub>-N leaching  
364 showed a distinctly linear decrease with simultaneous decreases of N fertiliser and livestock  
365 density, suggesting a close dependance of this output on N fertilization input. Methane (CH<sub>4</sub>-  
366 C) emissions tended to decrease with decreasing livestock density and also with reductions in  
367 N fertilisation.

368



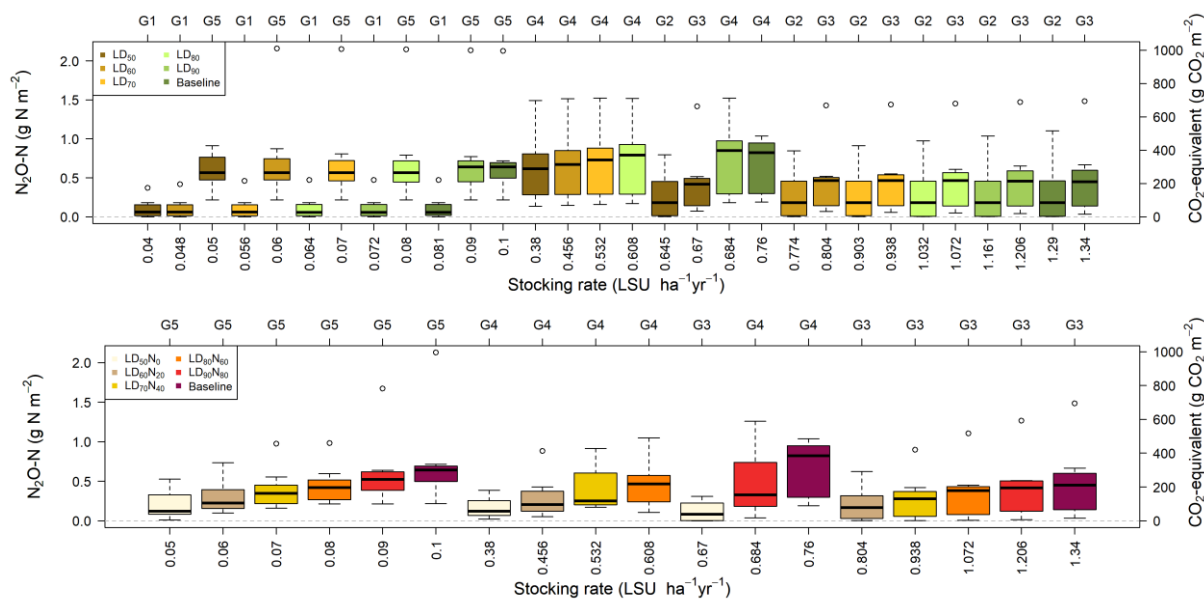
369  
 370 Fig. 3. Changes in non-CO<sub>2</sub> fluxes (g C m<sup>-2</sup>, g N m<sup>-2</sup>, log<sub>10</sub> g N m<sup>-2</sup>) calculated over multiple  
 371 years at five sites, for ten altered management scenarios and the baseline (as in Table 3). For  
 372 each management level, triangles show the multi-model (as in Table 4) mean, black lines  
 373 show multi-model median. Boxes delimit the 25<sup>th</sup> and 75<sup>th</sup> percentiles. Whiskers are 10<sup>th</sup> and  
 374 90<sup>th</sup> percentiles. Hollow circles indicate outliers (LD from 90% down to 50% of initial  
 375 density, N fertilisation from 80% to 0% of initial amount).

376  
 377 For N<sub>2</sub>O emissions, relative effect size analysis (Fig. I in supplementary material) and  
 378 simulated emissions (Fig. 4 and Fig. J in supplementary material, respectively) revealed linear  
 379 decreasing trends as both livestock density and N fertiliser were reduced (sites G3, G4 and  
 380 G5, Fig. 4, bottom). The greatest mitigation of N<sub>2</sub>O-N emission was obtained by reductions in  
 381 N fertilizer at the G4 and G5 sites, where the initial 0.51 and 0.63 g N<sub>2</sub>O-N m<sup>-2</sup> were reduced  
 382 to 0.11 and 0.16 g N<sub>2</sub>O-N m<sup>-2</sup>, respectively. In terms of total GHG emissions, using the 100-  
 383 year Global Warming Potential (GWP<sub>100</sub>), the N mitigation from baseline to zero would  
 384 reduce the multi-model median simulated N<sub>2</sub>O emissions by 135.7, 187.1 and 219.9 g CO<sub>2</sub>e

385  $\text{m}^{-2} \text{yr}^{-1}$  at G3, G4 and G5 sites, respectively. This corresponds to ~16–25% of C sink  
386 potential, reported by Soussana et al. (2007) across nine European grassland sites, but a larger  
387 percentage (~40–65%) of the sink potentials determined in this study.

388         The reduction of N fertilizer logically decreases the  $\text{N}_2\text{O}$ -N emissions, as reported here  
389 (Fig. J in supplementary material) and by experimental studies (Cardenas et al., 2010; Bell et  
390 al., 2016; Hörtnagl et al., 2018). Our results at the G3 site showed the same trend (Table 5),  
391 when the  $\text{N}_2\text{O}$ -N emissions are compared to the applied N fertilizer amounts, the estimated  
392 (simplified)  $\text{N}_2\text{O}$ -N emission factors (percent ratios of the total yearly  $\text{N}_2\text{O}$ -N emissions over  
393 the amount of annually applied N fertiliser, both in  $\text{kg N ha}^{-1}$ ). Our simulated results (varying  
394 between 1.0 and 3.5% across sites and treatments) are not far from the IPCC (2006) default  
395 EF for fertilizer N value, which is 1%. At G4 and G5 sites, the EF values tended to increase as  
396 grassland management received less N fertilizer (Table 5), which suggest some non-linear  
397 reduction of  $\text{N}_2\text{O}$  under reduced fertiliser supply, which can be explained by a decrease of  
398 plant N uptake with decreasing N fertilizer rate (e.g. Lü et al., 2014). Negative relationships  
399 between N use efficiency and soil N availability were observed in a variety of ecosystems,  
400 including grasslands (e.g. Yuan et al., 2006). Decreased N uptake from the soil and less  
401 efficient use of the N assimilated by plants leave more N available for microbes in the soil  
402 (which is the most important factor for  $\text{N}_2\text{O}$ -N emissions). Thus, the most intensive systems  
403 (G3, G4 and G5) had the highest  $\text{CO}_2\text{e}$  emission rates while the  $\text{N}_2\text{O}$ -N emission factors  
404 varied between the managements options. There was no trend in median values of simulated  
405  $\text{N}_2\text{O}$  emissions and LD levels, with the exception of G1 and G4 (Fig. I in supplementary  
406 material). At the G1 site, a slight decreasing trend was noticeable with decreasing grazing  
407 intensity, with increasingly diverging results among models as more extensive management  
408 was introduced.

409



410

411

412 Fig. 4. Relationship between nitrous oxide emissions (multi-year averages of seven models)  
 413 given in  $N_2O-N$  and  $CO_2e$  forms, and increasing animal stocking rate at all sites comparing  
 414 livestock density reductions (top graph) and livestock density and N fertilization reduction  
 415 (bottom graph), as in Table 3.

416

417 Table 5.  $N_2O-N$  emission factors: multi-model median at the three N fertilised sites (as in  
 418 Table 2). Grey cells indicate the lowest values.

Management options	G3	G4	G5
Baseline	1.3%	1.8%	2.3%
LD <sub>90</sub> N <sub>80</sub>	1.4%	1.2%	2.0%
LD <sub>80</sub> N <sub>60</sub>	1.6%	2.0%	1.9%
LD <sub>70</sub> N <sub>40</sub>	1.4%	2.2%	2.5%
LD <sub>60</sub> N <sub>20</sub>	1.0%	1.7%	3.5%
LD <sub>50</sub> N <sub>0</sub>	-	-	-

419

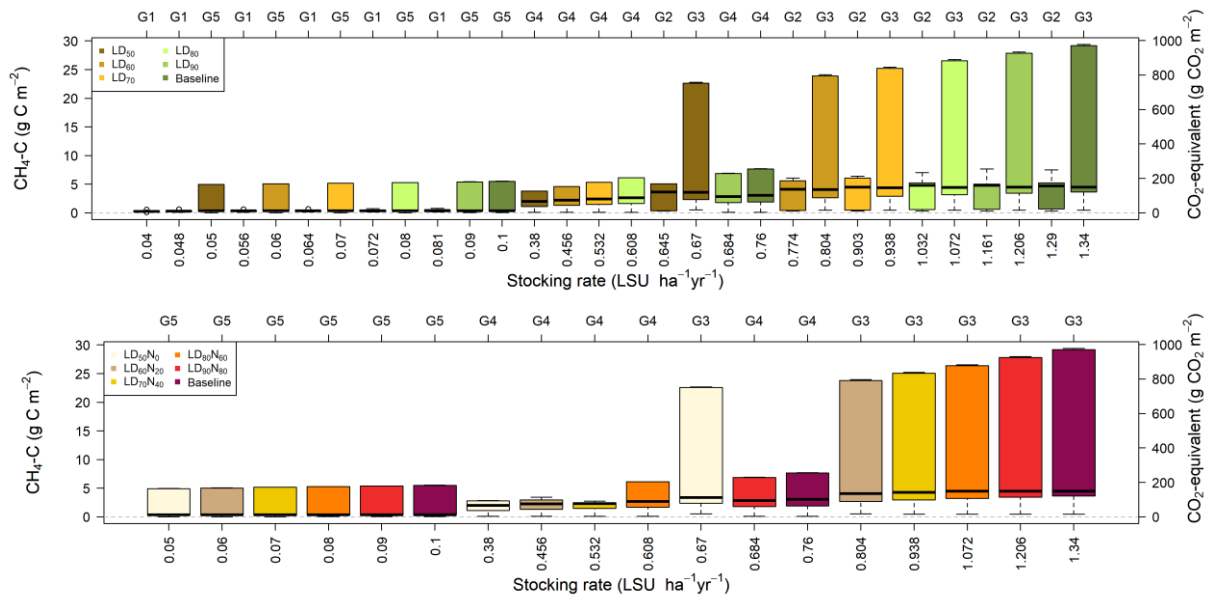
420 Overall, the different N fluxes (Fig. 4 and Fig. K in supplementary material) tended to  
 421 decrease with reduced N fertilization, mainly after a 60% reduction in the amount of N  
 422 applied in both grazed and combined (mown and grazing) systems.

423 Regarding  $CH_4-C$  emissions (Fig. 5), the G3 site had much higher model uncertainty  
 424 than other sites, mostly due to high estimates from M06 and M24. The simulated multi-model



425 median values were the highest, with lower livestock density. For example, the multi model  
426 multi-year average baseline simulation was  $4.6 \text{ g C m}^{-2}$ , which was reduced to  $3.6 \text{ g C m}^{-2}$   
427 with stocking rate reduction alone, and to  $3.4 \text{ g C m}^{-2}$  in combination with N reduction at the  
428 G3 site (Fig. 5). The main agricultural source of  $\text{CH}_4$  at the G3 site was the intensive heifer  
429 grazing system. Other sites were less affected because either they had grazing sheep (G2 and  
430 G5) or were less intensively grazed (low cattle density in G1, combined sheep and cattle  
431 grazing in G4).

432



433

434

435 Fig. 5. Relationship between methane emissions (multi-year averages of five models),  
 436 expressed as CH<sub>4</sub>-C and CO<sub>2</sub>e, and increasing animal stocking rate at all sites, comparing  
 437 livestock density reduction (top graph) and livestock and N fertilization reduction (bottom  
 438 graph) as given in Table 3.

439

440 Site-specific circumstances, mainly soil properties (particularly soil N availability),  
 441 and precipitation patterns, have considerable effect on the N balance of the grassland sites.  
 442 Our modelled outputs show that soils tended to release more N<sub>2</sub>O-N and CH<sub>4</sub>-C (Fig. L in  
 443 supplementary material) in humid and very humid years, while the net N emissions were  
 444 lower during drier years. The highest N<sub>2</sub>O-N emissions and also N<sub>2</sub> and NO<sub>x</sub> emissions (Fig.  
 445 L in supplementary material) were simulated when annual precipitation was around 1000 mm,  
 446 owing to the more available N, because high rainfall rates increase the rate of N transport to  
 447 deeper soil layers and increase nitrate leaching (Fig. L in supplementary material). Also, there  
 448 would be more anaerobic microsites leading to greater rates of denitrification in waterlogged  
 449 soils (Smith et al., 1998). Besides, higher NPP values (Fig. H in supplementary material) were  
 450 associated with elevated precipitation, thus higher organic N inputs to soil may contribute to  
 451 the larger N<sub>2</sub>O-N emissions. This indicates that the intensity of N losses tends to be associated

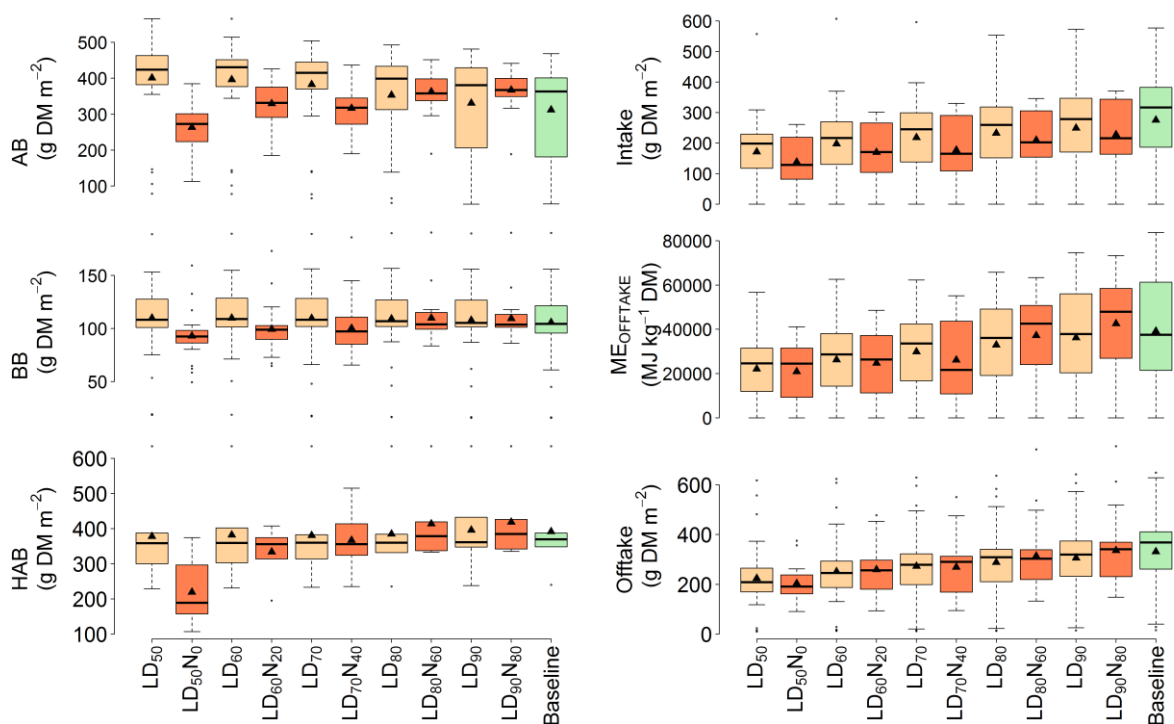
452 with annual precipitation levels, and N losses can be effectively mitigated by reducing  
 453 livestock density and/or N fertilisation rates. Our simulations indicate reductions in N  
 454 fertilization as the most effective option for mitigating non-CO<sub>2</sub> fluxes, mainly in humid  
 455 areas, an observation that is consistent with a number of previous studies (Bouwman et al.,  
 456 2002; Rees et al., 2013) but further studies are required considering the high variability of  
 457 model responses.

458

### 459 3.3. Productivity and energy outputs

460 Some decreasing trends with management extensification can be observed in the box-  
 461 plots of Fig. 6, e.g. for simulated Offtake and Intake, whilst aboveground and belowground  
 462 biomass increased with lower stocking rates, combining simulation results from five sites. In  
 463 terms of harvested aboveground biomass, a drop was simulated with no N fertilization (e.g.  
 464 baseline simulation: 421.7±118.6 g DM m<sup>-2</sup>, LD<sub>50</sub>N<sub>0</sub> management: 200.9±78.8 g DM m<sup>-2</sup>  
 465 across the multi-year site averages).

466



467

468 Fig. 6. Changes in productivity and energy outputs ( $\text{g DM m}^{-2}$ ,  $\text{MJ kg}^{-1} \text{DM}$ ) calculated over  
469 multiple years at five sites, for ten altered management scenarios and the baseline (as in Table  
470 3). For each management level, triangles show the multi-model (as in Table 4) mean, black  
471 lines show multi-model median. Boxes delimit the 25<sup>th</sup> and 75<sup>th</sup> percentiles. Whiskers are 10<sup>th</sup>  
472 and 90<sup>th</sup> percentiles. Points indicate outliers.

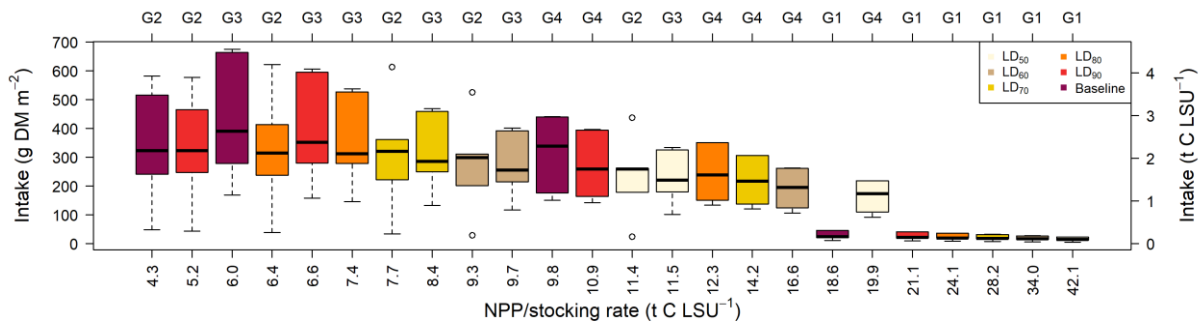
473

474 In fertilised sites, where LD levels have been assessed alone and in combination with  
475 N fertilisation levels, Fig. 6 and Fig. M in the supplementary material indicate that, overall,  
476 reducing N fertiliser rate will have more effect on aboveground biomass and biomass offtake  
477 than reducing livestock density, as data points relative to sites G3, G4 and G5 tend to stay  
478 below the 1:1 line. The effect of N fertiliser reduction starts becoming visible at G4 with 60%  
479 N reduction, while only with no N fertilisation is this effect visible at G5. However, livestock  
480 density reduction has a greater effect on animal intake, belowground biomass and  $\text{ME}_{\text{Offtake}}$   
481 (Fig. M in supplementary material). Animal intake decreases considerably with  
482 extensification (Fig. 6 and Fig. N in supplementary material).

483 The annual NPP values, normalized by animal stocking rates, are shown in Fig. 7 for  
484 each management option (Fig. 7). The G5 site (mowing dominated) was excluded from this  
485 analysis owing to the very low stocking rate practised at this site, thus a relationship was  
486 established of animal intake (not offtake) with NPP over stocking rate ratio. There was a  
487 decreasing trend in animal intake with extensification when animals have access to more  
488 biomass per head. This trend is also supported by the relationship between the minimum  
489 required amount of biomass per animal and the productivity of the sites under different  
490 management options. Using  $1.5 \text{ LSU ha}^{-1} \text{ yr}^{-1}$  as an overall reference estimate of potential  
491 ecological carrying capacity (e.g. UK Rural Payments Agency, 2003), which is equal to 1.5  
492 adult cattle on 1 ha pasture field, we see that around ~46% of the total biomass produced each

493 year is consumed by animals in the most intensive grazing systems (baseline) of G2 and G3  
 494 sites (Figure 7). With extensification, this ratio is reduced to ~23% at LD<sub>50</sub> (average of G2  
 495 and G3 sites).

496



497

498 Fig. 7. Relationship between animal intake (Intake) (multi-year averages of eight models,  
 499 expressed with two units) and the ratio of net primary production (NPP) (multi-year averages  
 500 of eight models) over animal stocking rate for different livestock densities (as in Table 3).

501

502 Overall, grassland productivity increases with annual precipitation levels, though  
 503 uncertainties can be large (e.g. seasonal waterlogging spells and heat waves may have  
 504 negative effects on grassland productivity also with rainfall >1000 mm yr<sup>-1</sup>), indicating higher  
 505 sensitivity to animal density reduction (which has some positive effects) than to N fertilization  
 506 reduction (with even greater negative effects) of ME<sub>Offtake</sub>, Intake, Offtake, AB and BB (Fig.  
 507 O in supplementary material).

508

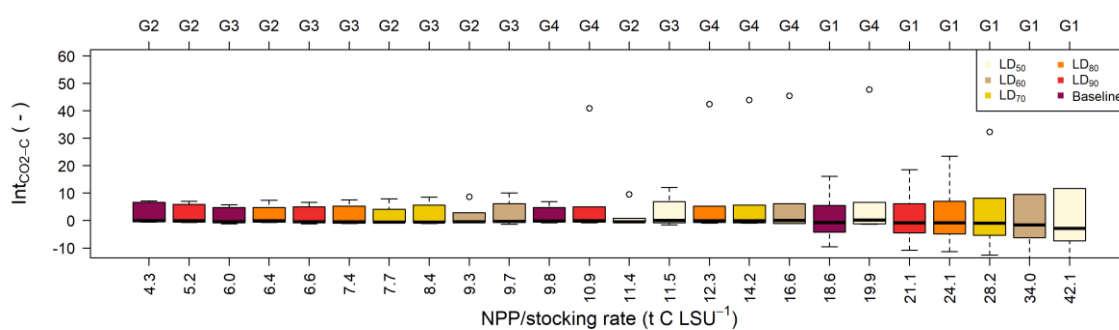
### 509 3.4. CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O intensities

510 Extensification, mostly through N input reduction, increased the variability of results  
 511 in the case of methane intensity. Overall, N<sub>2</sub>O-N intensity reduces with greater extensification  
 512 (Fig. P in supplementary material), mainly driven by N fertilizer reduction, while CO<sub>2</sub>-C  
 513 intensity shows different patterns at each site. In G4, in particular, CO<sub>2</sub>-C intensity is >1 when

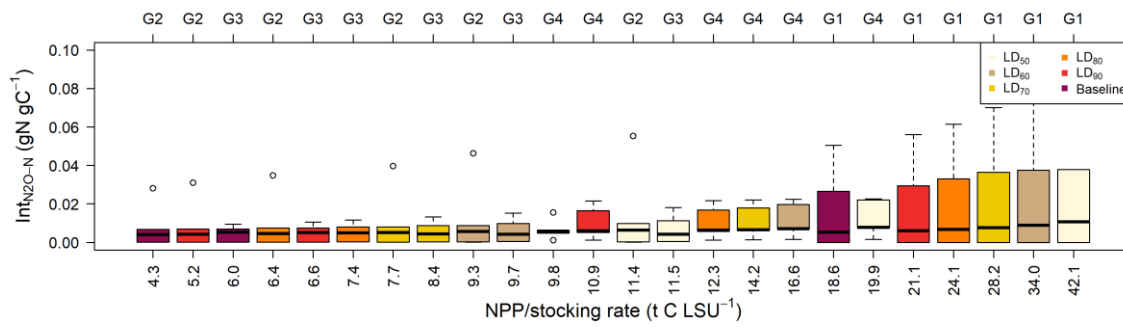
514 animal density was reduced by 60%. Multi-model median simulations suggest that reducing N  
 515 fertilization does not affect methane emissions. However, at the G3 site (grazed by heifers),  
 516 which shows the greatest intensities (Fig. Q in supplementary material), CH<sub>4</sub>-C emissions  
 517 increase with reducing animal density (with cattle having a larger area to graze, and thus more  
 518 biomass available for feeding).

519 For N<sub>2</sub>O-N intensity (Fig. R in supplementary material), model variability increased  
 520 with reductions in animal density at the G1 and G3 sites (cattle grazing systems). Under sheep  
 521 grazing (G2, G4, G5 sites), different models did not differ much in their output when reducing  
 522 animal density. The intensity of C (in the form of CO<sub>2</sub>-C and CH<sub>4</sub>-C) and N (in the form of  
 523 N<sub>2</sub>O-N) emissions with respect to biomass offtake did not change with extensification (Fig. P  
 524 in supplementary material). The plot of CH<sub>4</sub>-C intensity values against NPP-stocking rate  
 525 ratios (Fig. 8) show the extensification at G3 site increased the simulation uncertainty. For  
 526 sheep grazing systems, methane emissions did not vary greatly with management options.  
 527 However, CO<sub>2</sub>-C and N<sub>2</sub>O-N intensities, and their simulation uncertainties, increased with  
 528 extensification, when animals had more available biomass.

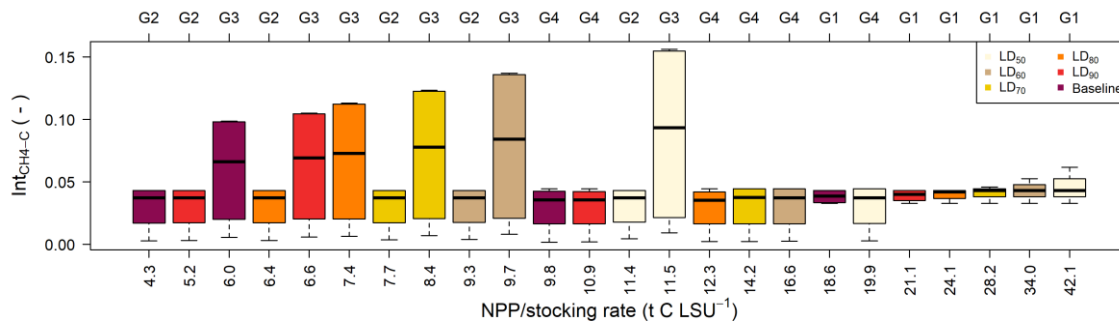
529



530



531

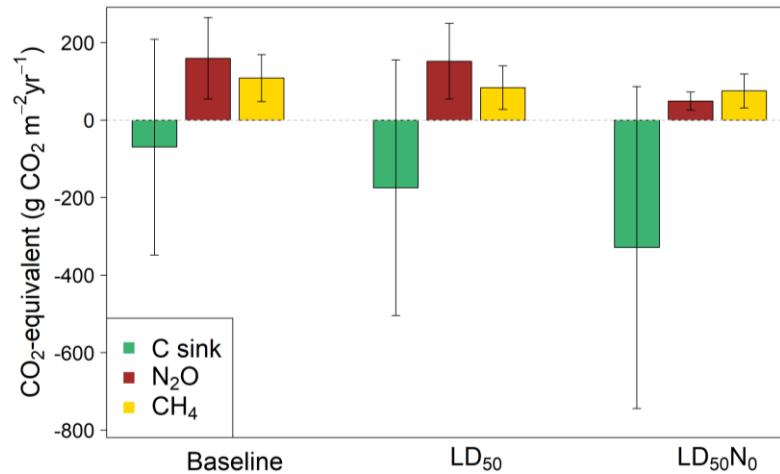


532

533 Fig. 8. Relationship between CO<sub>2</sub>-C, N<sub>2</sub>O-N and CH<sub>4</sub>-C intensity outputs (multi-year  
 534 averages of eight, seven and five models, respectively) and the ratio of net primary production  
 535 (NPP) output and animal stocking rate for different livestock densities (as in Table 3).

536

537 In relation to annual precipitation levels, CO<sub>2</sub>-C, CH<sub>4</sub>-C and N<sub>2</sub>O-N intensities (Fig. R  
 538 in supplementary material) showed different patterns for arid and humid conditions. In the  
 539 case of CO<sub>2</sub>-C intensity, C sequestration was moving around its equilibrium at humid  
 540 conditions, whilst for drier years it showed different patterns. N fertilization reduction may  
 541 increase C fixation, with its maximum at 30% less animal density and 60% less N  
 542 fertilization.



543

544 Fig. 9. Ensemble modelling of carbon sink (C sink), nitrous oxide emissions (N<sub>2</sub>O) and  
 545 methane emissions (CH<sub>4</sub>) in CO<sub>2</sub>eq form using multi-site averages of the multi-model median  
 546 of eight models under grassland mitigation options (as in Table 3).

547

548 Overall, the carbon sink increased with extensification (baseline: ~-70, LD<sub>50</sub>: ~-175 and  
 549 LD<sub>50</sub>N<sub>0</sub>: ~-329 CO<sub>2</sub>eq) (Fig.9), but N<sub>2</sub>O and CH<sub>4</sub> emissions decreased. Livestock density  
 550 reduction showed greater effect on CH<sub>4</sub>-CO<sub>2</sub>eq reduction (baseline: ~108, LD<sub>50</sub>: ~84 and  
 551 LD<sub>50</sub>N<sub>0</sub>: ~75 CO<sub>2</sub>eq), while N fertilization reduction considerably reduced the N<sub>2</sub>O emission  
 552 from ~160 and ~152 CO<sub>2</sub>eq (baseline and LD50, respectively) to ~49 CO<sub>2</sub>eq with no N  
 553 fertilisation.

554

#### 555 4. Summary and conclusions

556 This is the first multi-model study to simulate the effect on C and N fluxes of reduced  
 557 grazing intensity and N fertiliser inputs in multiple grassland systems across the globe. By  
 558 mobilizing a multi-model approach, it has provided an improved understanding of GHG flux  
 559 dynamics in pastures. This study confirms that grasslands (which have the advantage of  
 560 potentially acting as a C and N sink compared to many croplands) can be exploited for GHG



561 mitigation in beef and dairy production, because C and N sequestrations can, under some  
562 circumstances, offset GHG emissions.

563 Simulated C fluxes indicated that there may be some shift towards a C sink ( $NEE < 0$ )  
564 with decreasing inputs, though it depends on complex, multifaceted processes of C fixation  
565 (GPP) and release ( $R_{ECO}$ ) occurring in the ecosystem. This is especially true for G3 and G4  
566 sites, while grasslands managed with low animal densities may not support C sequestration  
567 under arid conditions or in the presence of high organic C exports from mowing. Simulated N  
568 outputs showed the considerable effect of N fertilizer reduction on C and N emissions, while  
569 changes in animal density only slightly affected the N balance. Both simulated  $CH_4$ -C and N  
570 emissions (including leaching) were, as expected, highly sensitive to precipitation levels, with  
571 higher values being seen under humid conditions (annual precipitation  $> 1000$  mm). This  
572 indicates the importance of considering climate patterns when determining budgets of C and  
573 N under varying management options. With the most intensively grazed systems, ~35-40% of  
574 the simulated net primary production was grazed by animals, with this ratio decreasing to  
575 ~13% with decreasing stocking rates. The greatest enteric  $CH_4$ -C intensities were estimated  
576 for intensive grazing systems, whilst the highest estimates of  $N_2O$ -N intensities were found  
577 for mown and extensively grazed arid systems. Considering the dynamic behaviour of  
578 grassland systems, the amounts of C and N sequestered or released are not the same each  
579 year. However, uncertainties in the year-to-year variations are not critical in this context as  
580 our focus was on capturing major trends and levels rather than modelling exact annual or  
581 seasonal fluxes.

582 While suggesting the possibility of using models to determine sound mitigation  
583 practices, the present study also showed limitations. Our findings are based purely on  
584 simulated data and lack evaluation against measured outputs (experimental trials that have  
585 appeared in the published literature give us, at a minimum, a hint at what comprehensive

586 assessment of multi-model ensemble would look like). Although the models used in this study  
587 are only a subset of the available grassland models, we think that the various model types and  
588 variants (and related parameterizations) evaluated here are reasonably representative of  
589 current approaches. Another study limitation is that grassland practices other than grazing  
590 density and N fertilisation were not assessed. For instance, an option that has not been  
591 accounted for in this study is increasing the proportion of legume species in the sward which  
592 can allow for reduced use of N fertiliser, and has the potential to mitigate GHG emissions  
593 (e.g. Lanigan et al., 2013; Fuchs et al., 2018). For an analysis of the mitigation potential of  
594 legumes, we refer readers to a parallel study underway as part of the Model4Pastures project  
595 ([https://www.faccejpi.com/Research-Themes-and-Achievements/GHG-Mitigation/multi-](https://www.faccejpi.com/Research-Themes-and-Achievements/GHG-Mitigation/multi-partners-call/Models4Pastures)  
596 [partners-call/Models4Pastures](https://www.faccejpi.com/Research-Themes-and-Achievements/GHG-Mitigation/multi-partners-call/Models4Pastures)). Other options, such as organic N fertilisation, nitrification  
597 inhibitors or supplemental feeding, which are common practices in grassland management,  
598 have been left out given that state-of-the-art models are not unambiguously sensitive to such  
599 management interventions (Brilli et al., 2017). These difficulties, and those associated with  
600 model-to-model variability, suggest that some development work would be sensible given the  
601 importance of grasslands in supporting the broader GHG emissions reduction agenda. Despite  
602 their limitations, biogeochemical models (which evolve with the progress of research) are  
603 today a valuable tool for evaluating alternative options for mitigation of GHG emissions  
604 through grassland management. It is still rare for results in support to management decisions  
605 to be reported by an assessment of uncertainty. Our results show the potential for associating  
606 quantification of uncertainties with the results of grassland modelling under alternative  
607 management.

608

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632

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