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Title

Extensive grazing in contrast to mowing is climate-friendly based on the farm-scale greenhouse gas balance

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Highlights

- Systems based on extensive grazing act as net sink for greenhouse gases.
- Greenhouse gas balance of livestock farming highly depends on precipitation.
- Livestock farming could be a net sink for greenhouse gases under proper management regimes.

Abstract

Livestock is both threatened by and contributing to climate change. The contribution of livestock to climate change and greenhouse gas (GHG) emission greatly vary under different management regimes. A number of mitigation options comprise livestock management, although there are a lot of uncertainties as to which management regime to use for a given pedoclimatic and farming system. Therefore, we 1) tested if an extensive cattle livestock farm is a net sink or a net source for GHG (carbon–dioxide, CO_2; methane, CH_4; nitrous oxide N_2O) in Central–Eastern Europe, 2) compared the annual GHG balances between the grazed and mowed treatments of the farm 3) and investigated the role of climate variability in shaping these balances. Net ecosystem exchange of CO_2 (*NEE*) was measured with eddy covariance technique in both the grazed and mowed treatments. Estimations of lateral C fluxes were based on management data. Other GHG fluxes (CH_4, N_2O) were determined by chamber gas flux measurements technique (in case of soil) and IPCC guidelines (in case of manure decomposition and animal fermentation). Net greenhouse gas balance (*NGHG*) for the grazed treatment was 228±283 g CO_2 equivalent m^{-2} $year^{-1}$ (net sink) and –475±144 g CO_2 equiv. m^{-2} $year^{-1}$ (net source) for the mowed treatment. Net source activity at the mowed treatment was due to its higher herbage use intensity compared to the grazed treatment. At the farm scale the system was estimated to be a net sink for *NGHG* in a year with wet (135 g CO_2 equiv. m^{-2} $year^{-1}$), while a net source in years with dry soil moisture conditions (–267±214 g CO_2 equiv. m^{-2} $year^{-1}$). We conclude that under a temperate continental climate extended extensive grazing could serve as a potential mitigation of GHG in contrast to mowing. Our study highlights the fact that livestock farming could create a net sink for GHG under proper management regimes.

Keywords: Grassland management; Climate change mitigation; Carbon uptake; CH_4, N_2O, CO_2 fluxes

1. Introduction

Livestock is not only threatened by climate change (IPCC, 2013; Nardone et al., 2010), but it also contributes to it because the share of livestock sector in total anthropogenic greenhouse gas (GHG) emission is estimated to be between 10 to 25% (IPCC, 2007; Schwarzer, 2012; Gerber et al., 2013). Due to climate change the frequency of drought, heat waves and other extreme weather events (e.g. sudden rainfall) increased in temperate continental climate (Bartholy and Pongrácz, 2007; IPCC, 2013). Drought decreases the productivity of grasslands, which support livestock (Craine et al., 2012; Kanneganti and Kaffka, 1995; Thornton et al., 2014; Zhang et al., 2010) and heat stress lowers meat and milk yield of cattle (Gaughan 2012; Gauly et al., 2013; Nardone et al., 2010). Concurrently, livestock farming will need to supply an expected 20% increase in food demand between 2002 and 2050 under the threats of climate change (Steinfeld et al., 2006; Foley et al., 2011). Therefore, to maintain food security livestock farming has to adapt to climate change while reducing its GHG emissions (Smith et al., 2014). Decreasing GHG (carbon–dioxide, CO_2; methane, $CH_{4;,}$ and nitrous oxide, N_2O) emissions of livestock systems and increasing carbon (C) sequestration of grasslands could be achieved by the implementation of several management techniques (Bellarby et al., 2013; Herrero et al., 2016; Ripple et al., 2014; Smith et al., 2008; Soussana, 2008; Soussana et al., 2010). Management and climate variability have an integrated effect on shaping the GHG balances of grasslands and grassland-based farming systems.

Improper grazing management such as over or under grazing (Smith et al., 2008, Wang et al., 2011), degradation due to livestock expansion (Zhang et al., 2011) or intensification (Smith et al., 2008) led to a net loss of C from the ecosystem. On the other hand, improved grazing management (e.g. optimized grazing intensity, introduction of legumes, fertilization) of grasslands was found to increase C sequestration (Smith et al., 2008, Soussana et al. 2010, Oates and Jackson 2014). In general, grasslands were observed to be net sinks for CO_2 (Oliphant,

2012) but grazing was found to have a positive, negative or no impact on net ecosytem exchange (*NEE*) of grasslands (Luo et al. 2016). *NEE* was observed to vary between 2394 g CO_2 m^{-2} year^{-1} (net sink) and –1342 g CO_2 m^{-2} year^{-1} (net source), with a mean of 255±521 g CO_2 m^{-2} year^{-1} for extensive and 700±717 g CO_2 m^{-2} for intensive grazing (Gilmanov et al., 2010). Mowed areas were also found to act as net sinks (–476±51, Senapati et al. 2014; 313±145 g C m^{-2} year^{-1}, Soussana et al. 2007) or net sources (18±49 g C m^{-2} year^{-1}) (Wohlfahrt et al. 2008) for C in terms of *NEE*. Besides the management regime, climatic factors also affect C balance. Net sink/source activity in dry grasslands highly depends on climatic factors especially on the amount of precipitation (Jaksic et al., 2006; Nagy et al., 2007). However, it is not easy to separate the effects of climate from those of management in grasslands due to their interactions (Reichstein et al., 2013, Senapati et al. 2014). For example high net C sink activity of grasslands can be observed under high precipitation conditions in temperate, dry climate, which can also be due to the interaction with high rates of fertilization (Senapati et al. 2014, Soussana et al., 2010). Climate is expected to change rapidly in Central−Eastern Europe with more frequent heat waves and drought especially during spring and summer periods (Bartholy and Pongrácz, 2008). Droughts were observed to turn grasslands into net C sources at temperate (Nagy et al., 2007, Soussna et al. 2010) rather than in wet, cold climate (Mudge et al., 2011), thus climate change is expected to negatively impact C uptake of grazed grasslands in dry, continental climate.

Besides CO_2 fluxes lateral C and methane-C fluxes affect the total accumulation of carbon for a given system i.e. the net ecosystem carbon balance (*NECB*) (Chapin et al., 2006). Depending on management intensities C is exported from the mown areas and imported to the corral/feeding system and exported from the farm in the form of animal products and manure (Fig. 1). *NECB* of mown areas was found to be lower compared to the grazed treatment (Senapati el al. 2014) but the mown areas were also turned into a net source in terms of *NECB* due to the large amounts of hay removed (Haszpra et al., 2010; Oates and Jackson 2014, Skinner 2008). *NECB* only consists of C fluxes, while it does not express the greenhouse gas balance.

The net greenhouse gas balance (*NGHG*) consists of the total greenhouse gas fluxes (CO_2, CH_4 and N_2O) for a given system (Fig. 1) in CO_2 equivalent, which takes into account the global warming potential (GWP) of the different gases (Soussana et al., 2010). When considering *NGHG* the mowed sites were found to act as net sources (Soussana et al., 2010) but the grazed sites functioned as net carbon sinks (Chang et al., 2015, Soussana et al., 2010), net sources (Levy et al., 2007) or neutral to total net GHG (Schulze et al., 2009). CH_4 and N_2O emissions of the farm depend on livestock management practices and the climate. CH_4 emissions due to enteric fermentation of cattle varies between 27 and 128 kg CH_4 kg head^{-1} year^{-1} (IPCC 2006a) depending on the type of animals, feeding and breeding practices (Smith et al. 2008). CH_4 emissions due to manure decomposition could vary between 1 and 112 kg CH_4 kg head^{-1} depending on the interaction between manure management (storage) and climate (e.g. differences in emissions in wet and warm vs. cool and dry weather conditions) (IPCC 2006a). Soil CH_4 and N_2O fluxes are affected by climatic factors and management regimes through the changes of abiotic (soil temperature, soil water content, pH, aeration of soil) and biotic factors (substrate availability, soil bacteria, impact of grazer animals) (Horváth et al., 2010; Soussana et al., 2010). Wet conditions favour CH_4 and N_2O soil emissions, therefore under drought conditions emissions are expected to be lower. Generally, soils of grasslands' are net sinks for CH_4 but mowing could enhance its uptake (Zhang et al 2012) or have no effect at all (Van den Pol-Van Dasselaar et al. 1999). Grazing managements were observed to weaken CH_4 uptake in semi–arid grasslands (Liu et al. 2007) or to have little impact (Van den Pol-Van Dasselaar et al. 1999). In contrast to CH_4, grasslands are usually net sources for N_2O especially in intensively grazed (e.g. 1.77 kg N_2O-N ha^{-1} year^{-1}, Flechard et al. 2007) and fertilized mown areas but soils can also be a net sink for N_2O via denitrification processes (Chapuis–Lardy et al., 2007).

The system boundaries, i.e. which of the above mentioned fluxes are investigated at plot, field, farm–scale or total thought the value chain (Fig. 1) greatly determine the results of *NECB* and *NHGH* calculations (Oates and Jackson 2014). Different system boundaries are among the reasons (besides uncertainties) why the share of livestock sector in total anthropogenic GHG emission was estimated to vary over a wide range (IPCC, 2007; Schwarzer, 2012; Gerber et al., 2013). Depending on the aims and objectives of the research life cycle assessment (*LCA*) of livestock products (meat, milk) can be performed (Bellarby et al., 2013), which includes farm gate and off-farm emission estimates. However, in contrast to its name, many *LCA*'s are actually lacking the carbon uptake component of grasslands (Nijdam et al., 2012; Opio et al., 2013; Schwarzer, 2012). However, net carbon sink activities of grasslands could partly mitigate GHG emissions from livestock. (Obviously if not included in the balance the sink activity is not represented, although it is the livestock and grassland management which sustains net C sink activity). In this study we used the methodology of Soussana et al. (2010) and followed the terminology of Chapin et al. (2006) for *NECB*. Off–farm emissions related to e.g. fossil fuel emission of transportation, administration, retail etc. were beyond the scope of this study, therefore these emission sources were not assessed. Potentially, grasslands could lose carbon owing to fire, leaching, erosion, and emission of volatile organic compounds (VOC).

Although several techniques were found to decrease GHG emissions from livestock farming (Bellarby et al., 2013; Smith et al., 2008), the greenhouse gas balance of differently managed grasslands (grazed vs. mowed) has not yet been explicitly compared and integrated to calculate farm scale greenhouse gas balance.

Our goal was 1) to test if an extensive cattle livestock farm is a net sink or a net source for GHG in Central–Eastern Europe, 2) to compare the annual GHG balance between the grazed and mowed treatments of the farm and 3) to investigate the role of climate variability in shaping these balances. Based on the farm scale GHG balance we aimed to propose grassland management techniques to mitigate the effects of climate change.

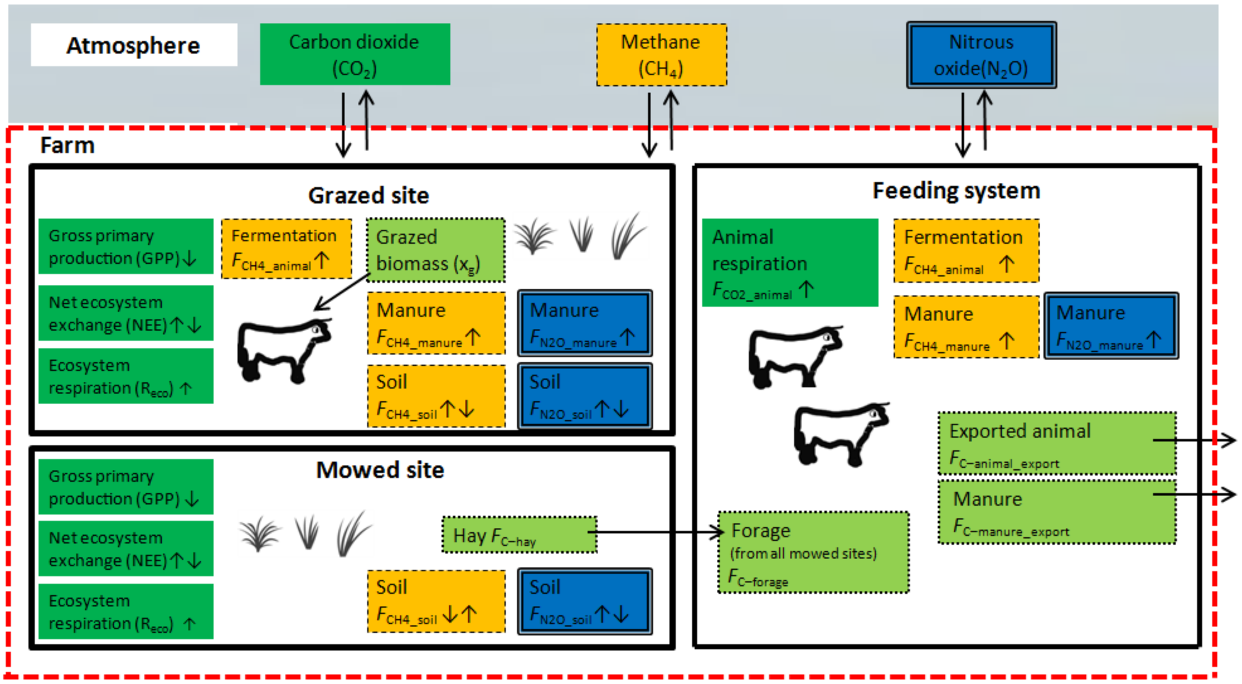

Fig. 1. Illustration of the farm–scale carbon and greenhouse gas fluxes. Arrows pointing up (to the atmosphere) and lateral directions (right) represent net sources, while arrows pointing down represent net sinks to the ecosystem.

2. Methods

2.1. Study area

Our study was conducted on a Hungarian Grey Cattle (*Bos taurus primigenius podolicus*) farm in the Kiskunság National Park between 2011 and 2013 (Bugac, Hungary, 46°41'28"N, 19°36'42"E, 114 m a.s.l.) (Fig. 2a). The farm consists of grazing (~1070 ha) and mowing areas (~847 ha), as well as an open air corral system (~4–10ha), where the animal were fed during winter. Sampling sites for grazed (~3 ha) and mowed (1 ha) treatments were established in the same grassland adjacent to each other (with paired eddy covariance towers 250 m apart). The grazed and mowed treatment was extensive due to the low stocking density (0.64±0.03 number of livestock unit ha^{-1} year^{-1}) and the low frequency of cutting (once per year). Also, treatments lacked fertilization (expect dropping during grazing at grazed areas), irrigation, tillage, or other management techniques (e.g. reseeding). Due to National Park regulations these treatments are not allowed *(Law of Nature Protection, 1996. LIII., 5§ (2) 269/2007. (X. 18.) Parliament Decision)*. The climate is dry continental with a mean annual sum precipitation of 575 mm and a mean annual temperature of 10.4°C (2003–2014). The soil is chernozem type sandy soil with high (above 3%) organic C content (Nagy et al., 2011). The vegetation is characterized by rich sandy grassland (steppe) species (species number above 80 per ha), and its composition was similar in both the grazed and mowed treatments (Koncz et al., 2014). The dominant species were *Poa angustifolia* L., *Carex stenophylla* Wahlenb., *Cynodon dactylon* (L.) Pers. and *Festuca pseudovina* Hack. ex Wiesb.

2.2. Micrometeorology

Precipitation (ARG 100 Tipping Bucket Raingauges, Waterra Ltd.) and air temperature (HMP35AC, Vaisala) were recorded by the meteorological station at the grazed treatments between 2011 and 2013. The treatments were adjacent to each other (250 m) therefore precipitation rates and air temperatures were assumed to be similar. Soil temperature [T_s, °C] and soil water content [*SWC*, %] were measured bi–weekly to monthly in both treatments throughout the whole study period (for more details see Koncz et al., 2015).

2.3. Net greenhouse gas balance (NGHG)

Net greenhouse gas balance *(NGHG)* was calculated based on Soussana et al. (2010) general equation (1.1) in CO_2 equivalent for the grazed (1.2) and mowed (1.3) treatments, and for the feeding system (1.4), as well as for the farm–scale (1.5):

$$NGHG = k_{\mathrm{CO_2}}\left(NECB - F_{\mathrm{CH_4-C}}\right) + GWP_{\mathrm{CH_4}}F_{\mathrm{CH_4}} + GWP_{\mathrm{N_2O}}F_{\mathrm{N_2O}} \tag{1.1}$$

$$NGHG_{grazing} = k_{\mathrm{CO_2}}\left(NECB_{grazing} - F_{\mathrm{CH_4-Cgrazing}}\right) + GWP_{\mathrm{CH_4}}(F_{CH4-animal(p1)} + F_{CH4-manure(p1)} + F_{CH4-soil}) + GWP_{\mathrm{N_2O}}(F_{N2O-soil} + \mathrm{F_{N2O-manure(p1)}}) \tag{1.2}$$

$$NGHG_{mowing} = k_{\mathrm{CO_2}}\left(NECB_{mowing} - F_{\mathrm{CH_4-Cmowing}}\right) + GWP_{\mathrm{CH_4}}(F_{CH4-soil}) + GWP_{\mathrm{N_2O}}(F_{N2O-soil}) \tag{1.3}$$

$$NGHG_{feeding} = k_{\mathrm{CO_2}}\left(NECB_{feeding} - F_{\mathrm{CH_4-Cfeeding}}\right) + GWP_{\mathrm{CH_4}}(F_{CH4-animal(p2)} + F_{CH4-manure(p2)}) + GWP_{\mathrm{N_2O}}(\mathrm{F_{N2O-manure(p2)}}) \tag{1.4}$$

$$NGHG_{farm} = k_{\mathrm{CO_2}}\left(NECB_{farm} - F_{\mathrm{CH_4-Cfarm}}\right) + GWP_{\mathrm{CH_4}}(F_{CH4_{animal}} + F_{CH4_{manure}} + F_{CH4-soil}) + GWP_{\mathrm{N_2O}}(F_{N2O-soil} + \mathrm{F_{N2O-manure}}) \tag{1.5}$$

where k_{CO2} is the multiplier between molar weights of CO_2, carbon (44/12), *NECB* is the net ecosystem carbon balance, $F_{\mathrm{CH4\text{-}C}}$ is the total CH_4 fluxes expressed in carbon, $GWP_{\mathrm{CH4}}F_{\mathrm{CH4}}$ (g CO_2 equiv. m^{-2} $year^{-1}$) is the total CH_4 flux in global warming potential ($\mathrm{GWP_{CH4}}$=34, in 100 year time horizon, IPCC 2013), and $GWP_{\mathrm{N2O}}F_{\mathrm{N2O}}$ (g CO_2 equiv. m^{-2} $year^{-1}$) is the total N_2O flux in global warming potential ($\mathrm{GWP_{N2O}}$=298, in 100 year time horizon, IPCC 2013) for the given system (i.e. grazing, mowing, feeding system, and farm). Calculations of components of *NECB*, F_{CH4} and F_{N2O} fluxes are given in the next chapters ($F_{\mathrm{CH4\_animal}}$ was the fermentation methane emission of the herd, $F_{\mathrm{CH4\_manure}}$ was the manure methane emission, $F_{\mathrm{CH4–soil}}$ was the soil methane flux, $F_{\mathrm{N2O–soil}}$ was the soil nitrous oxide flux, and $F_{\mathrm{N2O–manure}}$ was the manure nitrous oxide flux). The sum of the two periods ($p_1 + p_2$) provides the annual fluxes at farm–scale. In this study fluxes and net balances (*NECB*, *NGHG*) are positive if the given system (grazing, mowing, feeding system, or farm–scale) is a net sink for, while negative if the system is a net source the carbon or *GHG*. Grazing and mowing were parallelly studied throughout the three-year period, thus annual balances under grazing and mowing were compared. Significant differences in the fluxes between the systems were tested with t–test and with ANOVA, followed by Tukey post hoc test in R software (RStudio, Inc, version 0.97.551). Both treatments were applied in the farm (of the total farm area 56% was grazed and 44% was mown), thus the effects of the treatments were area–weighted when calculating the farm scale balance, which also contained the lateral fluxes related to feeding system. The share of each GHG fluxes in total net sink or in total net source activities was given in percentages (%). Due to the fact that the investigation was three years long, the annual farm scale balances could be compared to each other and the averages of the three years could be calculated.

It has to be emphasized that farm scale *NECB* and *NGHG* is a simulation (assumption), which indicates the *NECB* and *NGHG* of the farm if the total grazing and mowing areas are functioning similarly to the investigated (sampled) sites. It is an up–scaling of the measured data to the whole area. See representativeness in the supplementary material (SM 2). Up–scaling is frequently used in point measurements (eddy covariance measurements) based on the vegetation and management types (Schulze et al. 2009).

2.4. Net ecosystem carbon balance (NECB)

Based on CO_2, lateral C and CH_4 fluxes (converted to C based on carbon content), the net ecosystem carbon balance (*NECB)* was calculated for the grazed and mowed treatments, for the feeding system, and for the farm. A negative *NECB*, similarly to *NGHG* represents a net source, while a positive represents a net sink for C in the given system. The *NECB* at the grazed treatment was calculated as follows:

$$NECB_{\mathrm{grazed}} = NEE_{\mathrm{grazed}} + F_{\mathrm{CO2-animal}(p_1)} + F_{\mathrm{CH4-animal}(p_1)} + F_{\mathrm{CH4-manure}(p_1)} + F_{\mathrm{CH4-soil-grazed}} \quad (2),$$

where the NEE_{grazed} is the net ecosytem exchange of the grazed treatment, $F_{CO2\text{-}animal}$ is the animal respiration, $F_{CH4\text{-}animal}$ is the fermentation (rumination) CH_4 flux, $F_{CH4\text{-}manure}$ is the annual CH_4 emissions from manure, $F_{CH4\text{-}soil\text{-}grazed}$ is the soil CH_4 flux for the grazed treatment. Animal respiration (equation 9) and CH_4 fermentation (equation 11) was calculated for the grazing period (p_1), for the period when animals were kept in the feeding system (p_2), and for the whole year (p_1 + p_2 = p). The sum of the two periods (p) provides the annual fluxes at farm scale. Components are discussed throughout the next sections.

The *NECB* at the mowed treatment was calculated as follows:

$$NECB_{\mathrm{mowed}} = NEE_{\mathrm{mowed}} + F_{\mathrm{CH4-soil-mowed}} + F_{\mathrm{C-hay}} \quad (3),$$

where the NEE_{mowed} is the net ecosytem exchange of the mowed treatment, $F_{CH4\text{-}soil\text{-}mowed}$ is the soil CH_4 flux for the mowed treatment, $F_{c\text{-}hay}$ is the hay ($F_{\mathrm{C-hay}}$) exported from the sampled mowed treatment.

The *NECB* related to the feeding system was calculated as follows:

$$\begin{aligned} NECB_{\mathrm{feeding}} = {} & F_{\mathrm{CO2-animal}(p_2)} + F_{\mathrm{C-animal}_{\mathrm{export}}} \\ & + F_{\mathrm{C-forage}} + F_{\mathrm{C-manure}_{\mathrm{export}}} + F_{\mathrm{CH4-manure}(p_2)} + F_{\mathrm{CH4-animal}(p_2)} \quad (4), \end{aligned}$$

where $F_{\mathrm{C-animal\_export}}$ is the export of animal products, $F_{\mathrm{C-forage}}$ is the imported forage from the total mowed sites to the feeding system, $F_{\mathrm{C-manure\_export}}$ is the exported manure from the farm.

Farm–scale *NEE* was calculated based on the proportion of grazed (56%, A_{g}=0.56), and mowed areas (44%, A_{m}=0.44) in the total area (A). This is similar to the proportion of the time spent by the animals in the grazed areas (60%) and in the corral/feeding system (40%), where the forage from the mowed sites was consumed. At farm scale the exported hay from the total mowed areas is equal to the imported forage of the feeding system (note that the sampled mowed treatment refers only to a 1 ha sampling area). *NECB* at farm–scale was calculated as follows:

$$\begin{aligned} NECB_{\mathrm{farm}} = {} & NEE_{\mathrm{grazed}} \times A_{\mathrm{g}} + NEE_{\mathrm{mowed}} \times A_{\mathrm{m}} + F_{\mathrm{CO2-animal}} + F_{\mathrm{CH4-animal}} \\ & + F_{\mathrm{CH4-manure}} + F_{\mathrm{CH4-soil\,grazed}} \times A_{\mathrm{g}} + F_{\mathrm{CH4-soil\,mowed}} \times A_{m} \\ & + F_{\mathrm{C-animal\_export}} + F_{\mathrm{C-manure\_export}} \quad (5). \end{aligned}$$

Loss of carbon through erosion and leaching were assumed to be negligible because the area of the farm was flat and it was covered with closed vegetation (in contrast see Southern European pastures; Van Oost et al., 2007). Also, the farm lacks nearby rivers, and soil tillage was not applied, which could have contributed to erosion and leaching (in contrast see others where leaching affected C balance; Don and Schulze, 2008). Fire did not occur in the grazed and mowed areas (during the study period) and VOC was not assessed in our study.

2.5. Managements

Grazing

The grazing period, at farm scale, usually lasted from May to December. The grazing was rotational within a larger total grazing area (1070 ha) to let the vegetation regenerate between the grazing periods (Fig. 2a). The herd included cows, bulls, and heifers. The average weight of one livestock unit [*LSU*, kg] was calculated as:

$$LSU = \frac{m}{n} \tag{6},$$

where, *m* is the average total mass of the herd [kg], and *n* is the total average number of cattle (2002–2013). The number of animals was expressed in number of livestock units (*NLSU*). Stocking density (*SD*) was calculated based on the ratio of *NLSU* to the total grazing area [ha, *z*] per year [NLSU ha^{-1} year^{-1}]. The amount of biomass (carbon) removed by the grazing animals was estimated according to Vinczeffy (1993):

$$X_{\mathrm{g}} = \frac{DMI_{day} \times 1000 \times NLSU \times y}{z} \times \frac{G_c}{100} \tag{7},$$

where X_g is the dry mass (in carbon) of the estimated grazed biomass [g C m^{-2} year^{-1}], DMI_{day} is the daily dry matter intake [kg day^{-1}] of one livestock unit, 1000 is the mass conversion factor from kg to g, y is the number of grazing days over the year, z is the total grazing area [m^2], and G_c is the percentage carbon content of the plants. DMI_{day} was calculated according to Equation 10.17 (page 10.22) of IPCC (2006b):

$$DMI_{\text{day}} = LSU^{0.75} \times \left(\frac{0.2444 \times NE_{\text{ma}} - 0.0111 \times {NE_{\text{ma}}}^{2} - 0.472}{NE_{\text{ma}}}\right) \quad (8),$$

where NE_{ma} is the estimated dietary net energy concentration of diet (6.5 MJ kg^{-1}, IPCC, 2006b).

Mowing

The sampled mowed treatment (1 ha) was fenced off from the grazed site in 2011 to exclude grazing (Fig 2a). The total mown areas amounted to 847 ha (for representativeness of sampled mown site to total mown areas see SM 2.). Grasslands are often used interchangeably (grazed/mowed) in Central–Eastern Europe, therefore this shift represents a regular management practice. Mowed site was established near the grazed site (250 meters apart) to ensure similarity in soil, vegetation and climatic conditions, allowing for a focus on the obligate effect of management on *NECB* and *NGHG*. The mown grassland was mowed once per year (at 6 cm height) according to the management practice of the National Park. The harvested hay was weighed ($F_{\text{C–hay}}$). Herbage–use efficiency (HUE, %) (Hodgson 1979) for both grazed and mowed treatment was calculated as the proportion of the removed forage (X_g, $F_{\text{C-hay}}$) to the peak biomass.

Feeding system

The fluxes during winter feeding were summarised here (Fig 1.). Fluxes included the animal respiration ($F_{\text{CO2–animal}}$), fermentation methane emission ($F_{\text{CH4\_animal}}$), manure nitrous oxide emission ($F_{\mathrm{N2O-manure}}$), manure methane emission($F_{\mathrm{CH4\_manure}}$) during feeding, and the exported manure ($F_{\mathrm{C-manure\_export}}$), exported animal product ($F_{\mathrm{C-animal\_export}}$), imported forage ($\mathrm{F}_{\mathrm{C-forgae}}$). Fluxes related to winter feeding system were calculated on the farm area bases (1921 ha).

2.6. CO_2 and C fluxes of the farm

NEE (CO_2 flux) was measured by the eddy covariance (EC) technique (for instrumentation and other details see Pintér et al., 2010) in both grazed and mowed treatments. From the raw (10Hz) wind speed (u, v, w) and concentration (CO_2 and H_2O) data half hourly turbulent fluxes were processed by EddyPro® open source software. Gap–filling and flux partitioning was performed according to Reichstein et al (2005). In this study positive *NEE* means a net carbon uptake (net sink) by the ecosystem because all net sinks by the ecosystem are positive in this study. Due to extensive management, animals were scattered around the total grazing area and only occurred for a few weeks in the footprint of the eddy covariance measurements (these data have been removed and gap filled). Therefore, year round animal respiration [$F_{\mathrm{CO2-animal}}$, g CO_2 m^{-2} $year^{-1}$] was estimated separately based on Soussana et al. (2010):

$$F_{\mathrm{CO2-animal}} = \frac{DMI_{\mathrm{day}} \times G_c \times R_{\mathrm{C-animal}} \times NLSU \times p}{A} \times \frac{44}{12} \qquad (9),$$

where $R_{\mathrm{C-animal}}$ is the proportion of the carbon intake by the cattle, which is respired as carbon [62.5%] (Soussana et al., 2010), p is the number of days spent by the animals either in the grazing area (p_1) or in the corral/feeding system (p_2); $p_1 + p_2 = p$ (one year), A is the area of the farm [m^{-2}], 44/12 is the conversion factor from C to CO_2. Leaving out animal respiration would underestimate *NECB* and *NGHG* balances (Jones et al., 2016).

To understand the yearly course of *GPP* and R_{eco} plant biomass and soil respiration (R_s) dynamics were measured. Above and below ground biomass [g C m^{-2}] and R_s was measured bi–

weekly up to monthly in both treatments along transects (for more details see Koncz et al., 2015) (Fig. 2b).

Lateral carbon fluxes [g C m^{-2}] were calculated based on the management data provided by the Kiskunság National Park. These included the animal products ($F_{C\text{–}animal\_export}$) and manure ($F_{C\text{–}manure\_export}$) exported from the farm, the hay ($F_{C\text{–}hay}$) exported from the sampled mowed treatment, and the forage imported from the total mowed sites to the feeding system ($F_{C\text{–}forage}$). After the grazing period (p_1) animals were kept in an open air corral system during winter and early spring (p_2). During this time period animals were fed on forage originated from the mowed areas. The forage was consumed within a year (i.e. the grass, which was cut in June–August was used up until next April–May). By the 31th of December technically they used up 16.6–20% of the forage, which was cut in that year. Forage consumption was calculated on an annual basis. Manure and animal products were exported from the feeding system. Carbon content for animal products (18%) and manure (40%) was based on National Park's data. Carbon content of plant biomass and hay (G_c, %) was measured at the Hungarian Forest Research Institute (Hungarian Standard 1987). Manure was exported from the feeding ($F_{C\text{–}manure\_export}$), where the carbon dioxide release was assumed to be 88%, based on an average 12% of manure C contributing to soil organic carbon (Maillard and Angers, 2014).

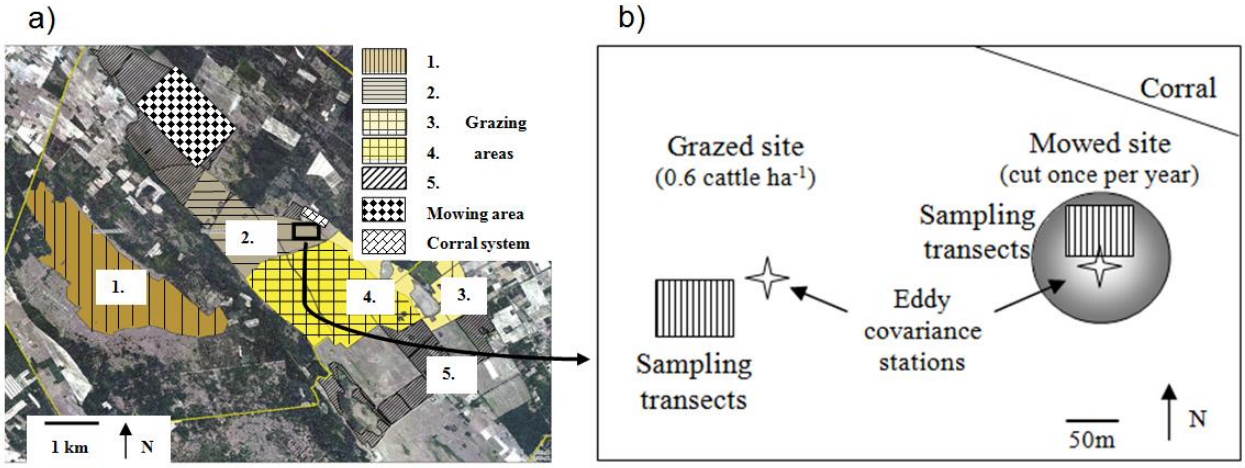

Fig.2. Map of the grazing and mowing areas (a) and the sampling sites (b) on the Grey Cattle farm of the Kiskunság National Park (Bugac, Hungary). Grazing areas were divided into five

sub–areas due to rotational grazing. Grazing (1070 ha), mowing areas (847 ha) and the corral (~4 ha) formed the feedings system and the farm scale (1921 ha).

2.7. CH_4 fluxes of the farm

Soil CH_4 flux [$F_{\mathrm{CH4-soil}}$, g CH_4 m^{-2} year^{-1}] was measured using the static gas flux chamber technique (Horváth et al., 2010). Seven chambers with circular rims were placed permanently along 7 m transects in both the grazed and mowed treatments (Fig. 2b). The rims were pushed 4 cm deep into the soil and were left permanently there to avoid the sudden emission peaks after installation. The rims were covered and closed by the upper part of the chambers only for the duration of sampling. After closure samples were taken at t = 0, 10, 20 min with a syringe. A total of 6 ml of samples were injected into 5.6 ml evacuated tubes. The concentration of CH_4 was measured by a gas chromatograph (HP 5890 II, Waldbronn, Germany) at the Hungarian Meteorological Service (Hungary). Soil CH_4 fluxes were calculated based on the accumulation of CH_4 gas [µg CH_4 m^{-2} h^{-1}] per each chamber (1–7) during the 20 min sampling period based on Horváth et al. (2010):

$$F_{\mathrm{CH4-soil(1)}} = \frac{\Delta C \times M_{\mathrm{CH4}} \times V_{ch} \times 60 \times f}{V_{\mathrm{m}} \times A_{\mathrm{ch}} \times t_{20}} \qquad (10),$$

where ΔC is the difference in mixing ratios [ppb] in chambers at the end and start of samplings, M_{CH4} is the molecular weight of CH_4, V_{ch} is the volume of the chambers [4×10^{-4} m^3], 60 is the time conversion factor for hour [min h^{-1}], f is the factor taking into account the residual pressure in the evacuated tubes (1.233), V_{m} is the molar volume – 24 litres at laboratory temperature [t = 20 °C] during measurements –, A_{ch} is the surface of soil covered by chambers [80 cm^2], t_{20} is the sampling period [20 min]. Measurement campaigns (between 11h to 15h) took place fortnightly during the growing season (April to October) and about every three to four weeks during winter.

Monthly average fluxes were calculated from the average of seven chambers [$F_{\mathrm{CH4–soil(1–7)}}$, µg CH_4 m^{-2} h^{-1}]. From the monthly average the total (sum) of monthly flux was calculated [g CH_4 m^{-2} month^{-1}]. Monthly sum of fluxes (12 month) were added to calculate the total yearly soil CH_4 flux [$F_{\mathrm{CH4–soil}}$, g CH_4 m^{-2} year^{-1}], similarly as described by Horváth et al. (2010).

Fermentation (rumination) CH_4 flux [$F_{\mathrm{CH4–animal}}$, g CH_4 m^{-2} year^{-1}] was estimated based on the IPCC (2006b) methodology:

$$F_{\mathrm{CH4-animal}} = \frac{F_{\mathrm{CH4}} \times NLSU}{A} \times \frac{1}{p} \qquad (11),$$

where F_{CH4} is the average CH_4 emission of one cattle livestock unit in Eastern Europe [58 000 g CH_4 year^{-1} LSU^{-1}] (IPCC, 2006b). The measured weight of one livestock unit in our study (381 kg) was just the same as the default value of the IPCC (2006b) for the region. During grazing period the time frame was p_1, during winter period (feeding) the time frame was p_2, and during the whole year it was p.

Due to extensive management no manure was found in the soil of CH_4 flux measurement chambers (manure did not fall into the chambers), therefore CH_4 emission of the manure [$F_{\mathrm{CH4-manure}}$, g CH_4 m^{-2} year^{-1}] was estimated based on the IPCC (2006b) methodology for the grazing (p_1), and winter period (feeding, p_2) and for the total year (p):

$$F_{\mathrm{CH4-manure}} = \frac{F_{\mathrm{CH4-m(p)}} \times NLSU}{A} \qquad (12),$$

where $F_{\mathrm{CH4–m(p)}}$ is the annual CH_4 emissions from manure for one livestock unit in Eastern Europe [33 000 g CH_4 year^{-1} LSU^{-1}] (Kis–Kovács et al., 2014).

2.8. N_2O fluxes of the farm

Soil N_2O flux [$F_{N2O–soil}$, g N_2O m^{-2} year^{-1}] was measured parallel with the CH_4 fluxes using the same method as described above for CH_4 flux. Soil N_2O flux was calculated similarly to soil CH_4 flux (equation 10) based on 20 min N_2O fluxes [μg N_2O m^{-2} h^{-1}], where M_{CH4} was replaced by M_{N2O} (i.e., the molecular weight of N_2O):

$$F_{\mathrm{N2O-soil(1)}} = \frac{\Delta C \times M_{\mathrm{N2O}} \times V_{\mathrm{ch}} \times 60 \times f}{V_{\mathrm{m}} \times A_{\mathrm{ch}} \times t_{20}} \qquad (13).$$

The concentration of N_2O was measured by a gas chromatograph (HP 5890 II, Waldbronn, Germany) in 2011 at the Hungarian Meteorological Service (Hungary) and similarly at the Department of Chemistry (Szent István University, Hungary) for the samples of 2012–2013. Due to extensive management no manure occurred in the chambers of the soil N_2O flux measurements, therefore the N_2O emission of the manure [$F_{N2O–manure}$, g N_2O m^{-2} year^{-1}] was calculated based on the IPCC (2006b) method for the grazing (p_1), and winter period (feeding, p_2) and for the total year (p):

$$F_{\mathrm{N2O-manure}} = \frac{\frac{N_{\mathrm{ext}}}{1000} \times LSU \times p \times \ MS \times EF \times NLSU}{A} \times \frac{44}{28} \qquad (14),$$

where N_{ext} is the annual average N excretion per one head of cattle (1000 kg) at the region [0.35 kg N head^{-1} day^{-1}] (IPCC, 2006b), MS is the fraction of total annual nitrogen excretion for cattle (93%), EF is the emission factor for direct N_2O emissions from manure management system (0.02 kg N_2O–N/kg N) (IPCC, 2006b).

We performed uncertainty and sensitivity assessments; see methods and results in the supplementary material (SM 1, SM 3, respectively).

3. Results

3.1. Variability of microclimate

Mean annual temperatures during the study period (10.1 °C, 10.8 °C and 10.8 °C in 2011, 2012, and 2013, respectively) were near or above the ten–year average (10.4 °C, 1995–2004). In 2011 and 2012, annual sums of precipitation (436 and 431 mm year^{-1}, respectively) were lower, while in 2013 (590 mm) the sum of precipitation was close to the ten–year average (575 mm). In 2011, we observed that the evapotranspiration (486 mm) was higher when compared to the actual precipitation (436 mm), thus we assumed that water was stored in the soil from the very wet previous year of 2010 (961 mm). Averages (Table 1) and temporal dynamics (Koncz et al., 2015) of T_s and SWC did not differ between the grazed and mowed areas during the study. However, large differences were observed between years as the average SWC decreased by 25% at the grazed and 20% at the mowed treatment from 2011 to 2013 (Table 1).

3.2. Management intensity

The mass of one livestock unit (*LSU*) was 381 kg, which is in agreement with default value of the IPCC (2006b) for the region. The stocking density (*SD*) was 0.64±0.03 NLSU ha^{-1} year^{-1} between 2011 and 2013 (Table 1), which represented an extensive grazing management regime. Daily dry matter intake (DMI_{day}) for one *LSU* was 8.6 kg day^{-1}. Carbon content of plant materials and hay (G_C) was 43%. Based on these data the average estimated grazed biomass during the study period (53.9±6.7 g C m^{-2} year^{-1}, X_g) was lower than the measured harvested hay at the sampled mowed treatment (93.7±31.2 g C m^{-2} year^{-1}, $F_{\mathrm{C–hay}}$) (Table 1). Based on the total biomass and removed forage the HUE was higher for the mowed (63.8±15.1%) compared to the grazed (46.2±1.2%) treatment, which indicated higher usage intensity of the mowed treatment (Table 1).

Table 1
Average soil temperature (T_s), soil water content (*SWC*) (data from April–December) in the grazed (~3ha) and mowed (1ha) treatments, management intensities over the total grazing area (1070 ha) and in the mowed treatment. Grazing period, stocking density and amount of grazed biomass (X_g) were calculated for the total grazing area.

	Grazing			Mowing		
	2011	2012	2013	2011	2012	2013
T_s in sampling treatment [°C]	18.8 (7)^{a}	21.9 (6.8)^{b}	20.1 (7.3)^{a}	18.6 (6.7)^{a}	21.7 (6.4)^{b}	19.7 (7.1)^{a}
SWC in sampling treatment [%]	14.8 (5.5)^{a}	12 (8.5)^{b}	11.1 (7.8)^{b}	13.7 (6.6)^{a}	11.8 (7.5)^{b}	10.9 (6.2)^{b}
Grazing period in the total grazing area [days year^{-1}]	199	229	204	–	–	–
Stocking density at the total grazing area [*NLSU* ha^{-1} year^{-1}]	0.61	0.67	0.63	–	–	–
Harvest days	–	–	–	Aug.–10	Jun.–24	Jul.–01
Grazed biomass (X_g) in the total grazing area and harvested hay ($F_{C\text{–hay}}$) [g C m^{-2} year^{-1}]	48.8	61.4	51.2	119.4	59	102.7
Above ground peak biomass in sampling treatments [g C m^{-2}]	111 (1.4)^{a}	132 (3.9)^{b}	107 (2.9)^{c}	188 (8.1)^{d}	121 (1.8)^{e}	130 (2.1)^{f}
Herbage–use efficiency [%]	44	46.5	47.9	63.5	48.8	79

Different letters ($^{a-f}$) indicates significant differences, i.e. the same letters within management between years indicates no significant differences, and also the same letters between managements within the same year indicates no significant differences, $p<0.05$ (n=14 per year per management, Mann–Whitney test), *LSU* is livestock unit.
Standard deviations are shown in brackets.

3.7. Net greenhouse gas balance (NGHG)

At farm scale the livestock system was a net sink for the GHG in wet (134.7 g CO_2 equiv. m^{-2} $year^{-1}$, 2011), while net source in dry soil moisture condition years (–266.8±213.6 g CO_2 equiv. m^{-2} $year^{-1}$, 2012–2013). On average over the three years the farm was shown to be neutral for GHG (–131.3±282.4 g CO_2 equiv. m^{-2} $year^{-1}$)(Fig. 3, Table 2), as due to the large inter–annual variability of *NGHG*, it was not significantly different from zero (p=0.48, n=3). At farm scale, CH_4 was responsible for 71% of the emissions, while the N_2O for the remaining 29%. CO_2 was responsible for 100 % of net sink activity (10±266 g CO_2 equiv. m^{-2} $year^{-1}$) at farm–scale.

Within farm scale the grazed treatment was a net sink, while the mowed treatment was a net source for GHG (Fig. 3, Table 2). The *NGHG* balance and the total CO_2 and CH_4 fluxes differed significantly between the grazed and mowed treatments, unlike the total N_2O fluxes (ANOVA, $p<0.05$, n=3). In the grazed treatment CH_4 was responsible for 60.9% of the total emissions, while the N_2O for the remaining 39.1%. Within CH_4 fluxes the fermentation contributed by 63.8% to the total CH_4 emission. The net source GHG activity of the mowed treatment was due to the low *NEE* and the high amount of exported hay (Fig. 3, Table 2). In the feeding system, the *NGHG* was found to be positive (net sink) due to the relatively low CH_4 (manure, fermentation) and N_2O emissions (manure) in CO_2 equivalent, which were compensated by the high amounts of imported forage. CH_4 was responsible for 87.9% of the emissions while N_2O for the remaining 12.1%. The results of uncertainty and sensitivity analysis of *NGHG* are shown in the supplementary material.

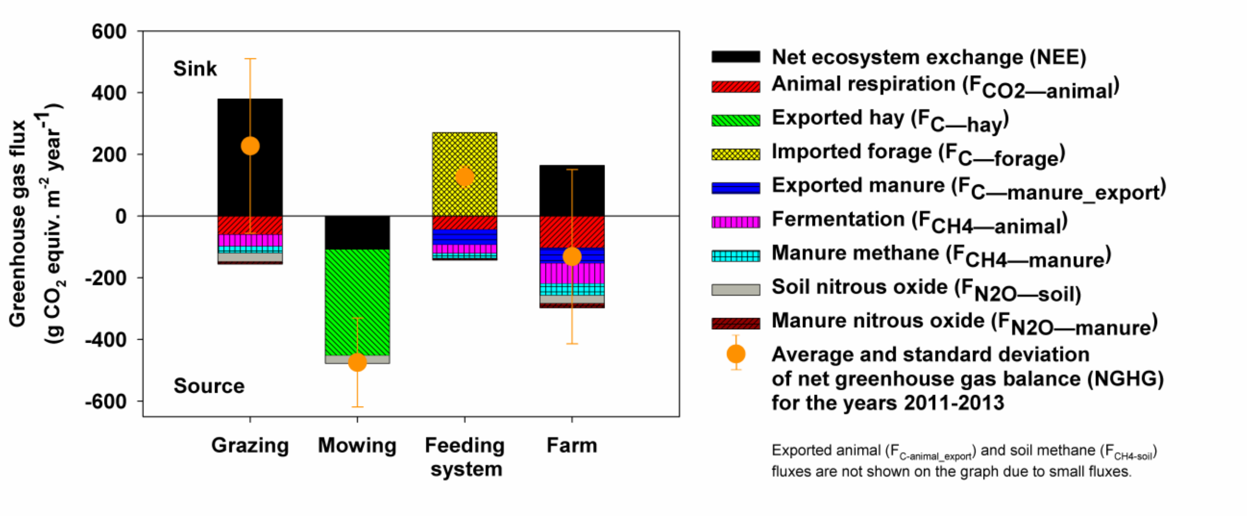

Fig. 3. Net greenhouse gas balance (*NGHG*) and its components in the grazed and mowed treatment, at the feeding system, and at farm–scale. Positive sign represents net sink, while negative sign represents net source for the given system. CO_2: carbon dioxide; CH_4: methane; N_2O: nitrous oxide; equiv: equivalent.

Table 2 Net greenhouse gas balance (*NGHG*) and its components for the grazed and mowed treatments, for the feeding system, and for the farm–scale. Positive sign represents net sink for the given system (highlighted in green and shaded in black and white printed version), while negative sign represents net source of the system. Fluxes of total grazed (1070 ha) and mowed systems (847 ha) are provided here, which were based on measured and up–scaled fluxes in sampled grazed (~3 ha) and mowed (~1ha) sites. The corral has an area of ~4 ha. The area basis for the feeding system and the farm was 1921 ha.

System	*NEE*	$F_{\mathrm{CO2–animal}}$	$F_{\mathrm{C–hay}}$	$F_{\mathrm{C–forage}}$	$F_{\mathrm{C–manure\_export}}$	$F_{\mathrm{C–\ animal\_export}}$	$F_{\mathrm{CH4–animal}}$	$F_{\mathrm{CH4–manure}}$	$F_{\mathrm{CH4–soil}}$	$F_{\mathrm{N2O–soil}}$	$F_{\mathrm{N2O–manure}}$	*NGHG*
						[g CO_2 equiv. m^{-2} $year^{-1}$]						
Grazed	380 (244.6)	–59.9 (7.4)	0	0	0	0	–38.2 (8.67)	–21.7 (2.7)	3.52 (5.44)	–27.9 (1.6)	–8.21 (1.81)	227.6 (283.1)
Mowed	–108.4 (226.9)	0	–343.6 (223.4)	0	0	0	0	0	3.09 (3.72)	–25.6 (8.96)	0	–474.5 (144.3)
Feeding	0	–43.6 (2.7)	0	271 (11.46)	–49.6 (16.3)	–1.51 (0.58)	–27.9 (1.7)	–15.8 (0.97)	0	0	–5.98 (1.71)	126.7 (35.4)
Farm	164.6 (244.6)	–103.5 (4.8)	0	0	–49.6 (16.3)	–1.51 (0.58)	–66 (3.03)	–37.5 (1.7)	3.33 (4.68)	–26.9 (4.87)	–14.2 (1.77)	–131.3 (282.4)
						[%]						
Grazed	99.08	–38.4	0	0	0	0	–24.48	–13.93	0.92	–17.92	–5.27	
Mowed	–22.64		–72	0	0	0	0	0	100	–5.47	0	
Feeding		–28.86	0	100	–37.3	1	–18.4	–10.47	0	0	0	
Farm	98.02	–33.82	0	0	–18.42	–0.49	–21.56	–12.27	1.98	–8.8	4.64	

System	CO_2	CH_4–CO_2 equiv.	N_2O–CO_2 equiv.
		[g CO_2 equiv. m^{-2} $year^{-1}$]	
Grazed	320.1 (271.8)	–56.4 (8.9)	–36.2 (2.5)
Mowed	–452 (131.7)	3.1 (0)	–25.6 (9)
Feeding	176.3 (31)	–43.6 (2.7)	–6 (1.7)
Farm	10 (266.3)	–100.2 (9.4)	–41.1 (6.6)

Legends: *NEE* (net ecosystem exchange), $F_{\mathrm{CO2–animal}}$ (respiration of the herd), $F_{\mathrm{C–hay}}$ (exported hay from the mowed treatment), $F_{\mathrm{C–forage}}$ (imported forage to the feeding system), $F_{\mathrm{C–manure\_export}}$ (exported manure from the feeding system), $F_{\mathrm{C–animal\_export}}$ (exported animal product), $F_{\mathrm{CH4\_animal}}$ (fermentation methane emission of the herd), $F_{\mathrm{CH4\_manure}}$ (manure methane emission), $F_{\mathrm{CH4–soil}}$ (soil methane flux), $F_{\mathrm{N2O–soil}}$ (soil nitrous oxide flux), $F_{\mathrm{N2O–manure}}$ (manure nitrous oxide flux). CO_2: carbon dioxide; CH_4: methane; N_2O: nitrous oxide; equiv: equivalent. The share of each fluxes in total net sink (+) or in total net source (–) activity was given in percentage (%). Standard deviations are shown in brackets. Averages are based on three years (2011–2013).

3.6. Net ecosystem carbon balance (NECB) of the farm

At farm scale the livestock system was shown to be neutral for *NECB* (–1.3±72.6 g C m^{-2} $year^{-1}$) (Table 3). This was due to the high amount of carbon uptake (*NEE*) in the grazed treatment, which compensated for the carbon loss via animal respiration ($F_{CO2\text{–}animal}$), animal fermentation ($F_{CH4\text{–}animal}$) and manure ($F_{CH4\text{–}manure}$) CH_4 emission. Other C fluxes such as the exported number of cattle (0–172 heifer $year^{-1}$) resulted in high meat production (15.9±6.1 t meat $year^{-1}$) but in a low net C export from the farm ($F_{C\text{–}animal\_export}$, 0.41±0.16 g C m^{-2} $year^{-1}$) (Table 3).

Within farm scale the grazed treatment proved to be a net sink, while the mowed treatment was found to be a net source for *NECB* (Table 3). This was due to the significantly higher net carbon sink activity (*NEE*) of the grazed compared to the mowed treatment (paired t–test, p=0.01, n=3) and to the large amount of harvested and exported hay ($F_{C\text{–}hay}$) from the mowed treatment. The harvested hay was higher than the estimated grazed biomass (X_g), which contributed to the higher loss of C in terms of *NECB* in the mowed treatment. Even though there were extra emissions in the grazed treatment (animal respiration, fermentation, and manure CH_4 emissions) compared to the mowed treatment these did not reduce the *NECB* of the grazed treatment below the level of the mowed treatment (Table 3). The feeding system appeared to be a net sink for carbon due to the high amount of imported forage from the total mowed areas ($F_{C\text{–}forage}$) (Table 3).

Table 3 Net ecosystem carbon exchange (*NECB*) and its components for the grazed and mowed treatment, for the feeding system, and for the farm scale. Positive sign represents net sink for the given system (highlighted in green and shaded in black and white printed version), while negative sign represents net source of the system. Fluxes of total grazed (1070 ha) and mowed systems (847 ha) are provided here, which were based on measured and up–scaled fluxes in sampled grazed (~3 ha) and mowed (~1ha) sites. The corral has an area of ~4 ha. The area basis for the feeding system and the farm was 1921 ha.

System	*NEE*	$F_{CH4\text{–}soil}$	$F_{C\text{–}forage}$	$F_{C\text{–}hay}$	$F_{C\text{–}manure\_export}$	$F_{C\text{–}animal\_export}$	$F_{CH4\text{–}animal}$	$F_{CH4\text{–}manure}$	$F_{CO2\text{–}animal}$	*NECB*
					[g C m^{-2} year^{-1}]					
Grazed	103.6 (72.2)	0.08 (0.12)	0	0	0	0	–0.84 (0.19)	–0.48 (0.06)	–16.3 (2.02)	86.1 (74.3)
Mowed	–29.6 (61.9)	0.07 (0.08)	0	–93.7 (31.2)	0	0	0	0	0	–123.2 (35.9)
Feeding	0	0	73.9 (3.1)	0	–15.4 (5.06)	–0.41 (0.16)	–0.61 (0.04)	–0.35 (0.02)	–11.9 (0.73)	45.3 (4.5)
Farm	44.9 (66.7)	0.073 (0.1)	0	0	–15.4 (5.06)	–0.41 (0.16)	–1.46 (0.07)	–0.83 (0.04)	–28.2 (1.3)	–1.32 (72.6)
					[%]					
Grazed	99.93	0.07	0	0	0	0	-4.77	2.71	92.51	
Mowed	–23.9	100	0	–76.1	0	0	0	0	0	
Feeding	0	0	100	0	–53.68	–1.44	–2.14	–1.22	–41.53	
Farm	99.84	0.16	0	0	–33.21	–0.89	–3.14	–1.79	–60.96	

Legends: *NEE* (net ecosystem exchange), $F_{CH4\text{–}soil}$ (soil methane flux), $F_{C\text{–}forage}$ (imported forage to the feeding system), $F_{C\text{–}hay}$ (exported hay from the mowed treatment), $F_{C\text{–}manure\_export}$ (exported manure from the feeding system), $F_{C\text{–}animal\_export}$ (exported animal product), $F_{CH4\text{–}animal}$ (fermentation methane emission of the herd), $F_{CH4\text{–}manure}$ (manure methane emission), $F_{CO2\text{–}animal}$ (respiration of the herd). The share of each fluxes in total net sink (+) or in total net source (–) activity was given in percentage (%). Standard deviations are shown in brackets. Averages are based on three years (2011–2013).

3.3. CO$_2$ fluxes

Yearly fluctuations of cumulative *NEE* were similar between the grazed and mowed sites before the grazing and mowing events (during springs) in 2011 and 2012 (Fig. 4a). Both the grazed and mowed treatments displayed a rapid increase in NEE (net C sink) at the end of spring/beginning of summer, reaching a peak before management events. During early summer of 2011, when the grazing already started but the mowed site was not yet cut, the biomass was higher at the mowed site (Koncz et al., 2015). During this time period the R$_s$ was also higher in the mowed compared to the grazed treatment, due to higher biomass production (see previous study; Koncz et al., 2015). R_s constitutes a major part of R_{eco}, thus higher R_s and biomass at the mowed treatment might have contributed to higher R_{eco} at the grazed treatment during early summer of 2011 (Fig. 4c). Removal of the biomass in 2011 (Koncz et al., 2015) in the mowed treatment caused a slight decline in the course of *GPP* in contrast to grazing (Fig. 4b). By the end of 2011 the mowed treatment had 78% lower cumulative *NEE* compared to the grazed treatment. The decline of *GPP* after the mowing event was also observed in 2012 but it was also pronounced in the grazed treatment under grazing probably due to the relatively dry summer (Koncz et al., 2015). Later on, the autumn of 2012 was relatively wet which contributed to an increase of *GPP* and biomass (Koncz et al., 2015) in both treatments. Although biomass was higher in the mowed treatment during 2012 autumn, it was accompanied by relatively higher R_{eco} (and $R_{s,}$ Koncz et al., 2015) compared to the grazed treatment. The cumulative *NEE* turned out to be a net source in the mowed treatment compared to the grazed treatment by the end of 2012. Biomass remained higher in the mowed treatment in early spring of 2013, which led to slightly higher R_{eco} (and R_s Koncz et al., 2015) in the mowed treatment compared to the grazed treatment prior to grazing and mowing. In the summer of 2013, similarly to previous years, the sudden removal of the biomass in the mowed treatment, in contrast to the prolonged grazing, caused a lack of potential to capture CO_2; hence the sharp increase of *GPP* in the mowed treatment was also greatly weakened in 2013. The average of the R_{eco} over the three years was 4% higher, while the *GPP* was 8% lower in the mowed treatment than in the grazed treatment. This led to net C source activity over the three years in terms of NEE in the mowed treatment compared to the grazed treatment (Table 3). The highest net carbon sink activities (and biomass) among the years were observed in 2011 in both treatments, probably due to the prolonged effects of the very wet year of 2010, which resulted in the highest soil water content in 2011 among the years (Table 1). *NEE* accounted for 98% of the total net sink activity of the farm (Table 3).

Animal respiration ($F_{CO2\text{–}animal}$) varied according to the number of animals and contributed 35% of total emission of the farm (Table 3).

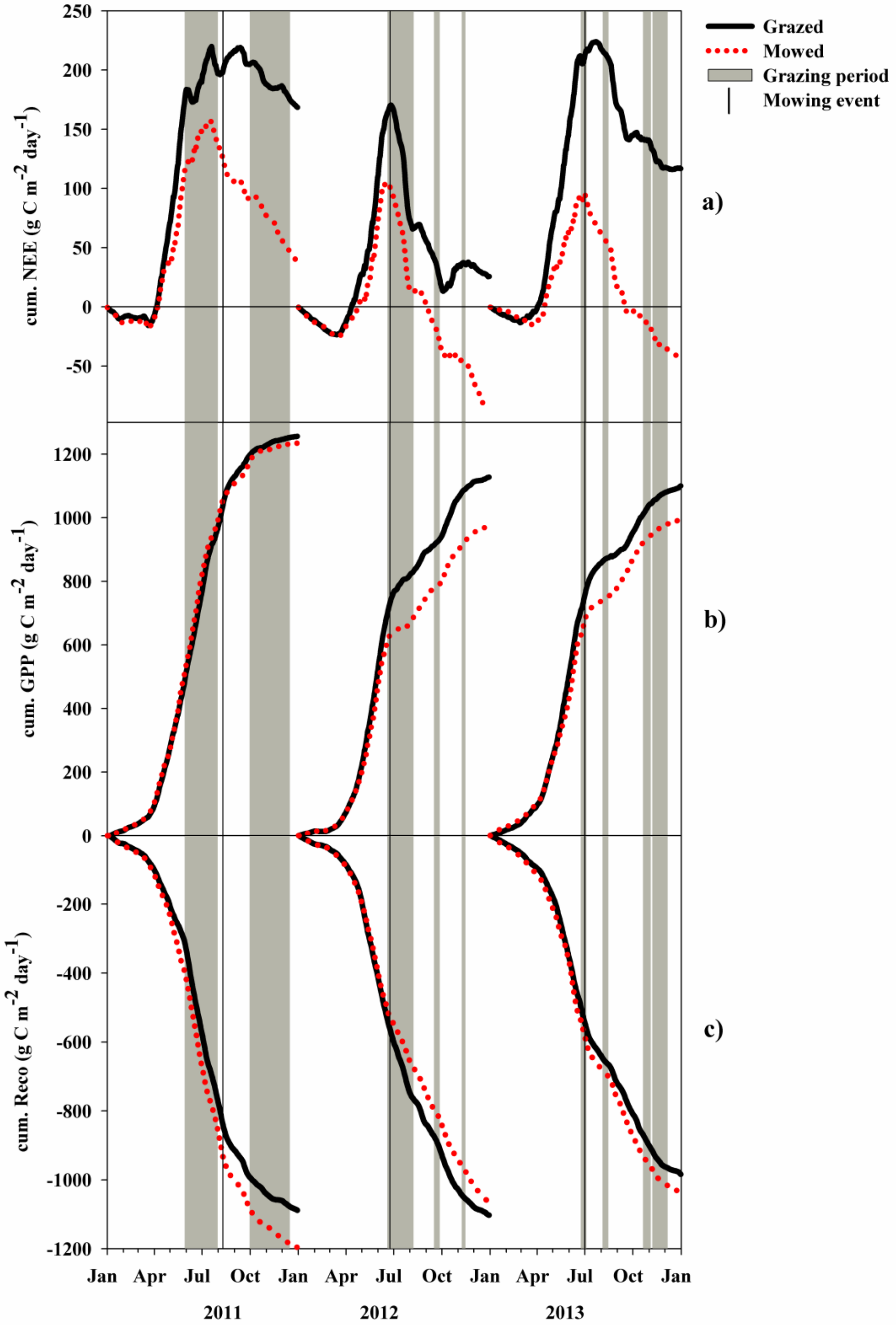

Fig. 4. Yearly course of cumulative net ecosystem exchange (*NEE*) (a), gross primary production (*GPP*) (b), and ecosystem exchange (R_{eco}) (c) in the grazed and mowed treatments in 2011–2013. Note: net sink activity is denoted by positive numbers, net source activity is denoted by negative numbers.

3.4. CH_4 fluxes

Soils were found to be weak net sinks for CH_4 (F_{CH4_soil}) in both the grazed (0.10±0.16 g CH_4 m^{-2} $year^{-1}$) and mowed (0.09±0.11 g CH_4 m^{-2} $year^{-1}$) treatments (2011), but did not vary between treatments (paired t–test, n=19, p=0.79) (Table 2). Soil CH_4 flux was only measured during 2011 due to its low level of contribution to the total greenhouse gas flux of the treatments. Soil net CH_4 sink accounted for 2% of total farm–scale greenhouse gas net sink activity. CH_4 emission due to fermentation ($F_{CH4\text{-}animal}$) accounted for an average of 22% of total farm scale greenhouse gas emissions (Table 3). Manure CH_4 emissions ($F_{CH4–manure}$) (Table 3) were found to be 50% less than the CH_4 emissions from fermentation.

3.5. N_2O fluxes

Soil N_2O emissions accounted for 9% of total farm scale greenhouse gas emission. Soils acted as net sources for N_2O (F_{N2O_soil}) in both the grazed (0.090±0.004 g N_2O m^{-2} $year^{-1}$) and mowed (0.084±0.027 g N_2O m^{-2} $year^{-1}$) treatments and no differences were observed between the treatments and years (2011: n=19, p=0.13; 2012: n=17, p=0.41; 2013: n=19, p=0.78, paired t–test by occasions) (Table 2).

Manure N_2O emission (F_{N2O_manure}) varied according to the number of animals in the grazed treatment and in the feeding system (Table 2). Manure N_2O emissions accounted for 4.7% of total farm–scale greenhouse gas emissions.

4. Discussion

We found that the extensive cattle livestock farm in Central–Eastern Europe (Bugac) was a net sink for the *GHG* in a year of sufficient water supply (2011), while it was a net source in dry years (2012, 2013). Emissions related to fossil fuel use were not estimated in this study, although, due to extensive management, emissions related to fertilization, irrigation, sowing, and land use changes (e.g. conversion from grassland to cropland) were zero in contrast to an intensive farm management regime.

4.1. Net greenhouse gas balance

The livestock system in our study was shown to be a net sink for GHG in a year with high soil moisture conditions (134.7 g CO_2equiv. m^{-2} $year^{-1}$), while it functioned as a net source in years with low soil moisture conditions (–266.8±213.6 g CO_2equiv. m^{-2} $year^{-1}$). On average the farm acted as a net source for GHG (–131.3±282.4 CO_2 g equiv. m^{-2} $year^{-1}$), although it was statistically not significantly different form 0 (neutral) due to large inter–annual climate variability (Table 2). At farm scale the grazed treatment was found to be a net sink while the mowed treatment to be a net source for NGHG. At farm scale 34% of the GHG emission was accounted for by animal respiration, 22% by animal fermentation, 18% by manure export (and related CO_2 emission), 12% by manure CH_4 emissions, 9% by N_2O emissions of the soil, 4.5% by N_2O emissions of the manure, and 0.5% by animal export (total GHG emission=100%). Other studies have also found (although not at paired investigations) that mowed sites were found to be net sources (–141 g CO_2 equiv. m^{-2} $year^{-1}$), while grazed sites to be net sinks (320 g CO_2 equiv. m^{-2} $year^{-1}$) for *NGHG* in European grasslands (Soussana et al., 2010). Based on models and estimations managed European grasslands have been either found to be net sources (DNDC model; Levy et al. 2007), net sinks (19±10 g C–CO_2 equiv. m^{-2} $year^{-1}$, ORCHIDEE−GM model; Chang et al. 2015), or neutrals (–14±10 g C–CO_2 equiv. m^{-2} $year^{-1}$, dual constraint approach; Schulze et al., 2009) for ecosystem−scale *NGHG*, when emissions of N_2O and CH_4 fluxes were included. However, when feeding system (corral, barn) and lateral fluxes were included the farm scale *NGHG* was found to be a net source for greenhouse gases (–50 g C–CO_2 equiv. m^{-2} $year^{-1}$, Chang et al., 2015). Also, full *NGHG* balance for altered grazed and mowed sites was estimated to be net source of GHG (–272 g CO_2 equiv. m^{-2} $year^{-1}$) (Soussna et al 2010). The NGHG balance of abandoned grasslands was rarely investigated. Chang et al. (2015) considered extensively managed grassland as newly abandoned grasslands (with only occasional mowing or rough grazing). Due to the lower number of animals less forage is needed, thus the ORCHIDEE-GM model estimated enhanced sequestration of C in soil (because forage was not exported) (Chang et al. 2015). Consequently, due to the reduction in livestock number the CH_4 emissions from enteric fermentation, and N_2O emissions (related to less nitrogen fertilizer) lowered, the modelled abandoned grassland contributed to net GHG mitigation (net GHG sink) (Chang et al., 2015). However, it should be noted that extensive grazing in our study does not equal to the abandonment of grasslands. Smith et al., (2008), in terms of C balance, summarized that net carbon sink activities on optimally grazed lands were greater than in ungrazed areas. Besides these studies we are not aware of any references with regard to the total net GHG balance of (fully) abandoned grasslands (i.e. lack of grazing for at least of 3 years), although it would be important to investigate it. Different C and N contents of the soil potentially lead to different grass biomass production. Therefore, our assumption that the total grazed and mowed areas have similar C and N contents compared to the sampled sites holds further uncertainties. See uncertainties, representativeness and sensitivity analyses in SM.

4.2. Net ecosystem carbon balance

When integrating all the C fluxes (*NECB*) we found that the mowed treatment lost C, while the grazed treatment was a net sink for C, which is consistent with the only other published study using paired EC towers to investigate *NECB* in grazed and mowed treatments (Senapati el al., 2014). Senapati et al. (2014) also found that the mowed treatment had lower *NECB* (22.7±32.3 g C m^{-2} $year^{-1}$, net sink) due to hay removal compared to the grazed one (140.9±69.9 g C m^{-2}

year^{-1}, net sink). HUE at the mowed treatment was nearly 40 % higher than that of the grazed treatment. The observed high HUE was the dominant factor in turning the grassland into a net source of C. Due to C removal others also found that mown treatments (not in a paired grazed vs. mowed site setup) were net sources of carbon (Haszpra et al., 2010; Skinner 2008). Studies, which included lateral C fluxes found that grazing management regimes usually resulted in higher *NECB* compared to mowing one (Oates and Jackson 2014). For example it has been found that under rotational grazing with cattle on a sub–humid pasture *NECB* was 106±69 g C m^{-2} year^{-1} (net sink), whereas under mowing, *NECB* was a net source; −391±11 g C m^{-2} year^{-1} (Oates and Jackson 2014). On the other hand, in a paired investigation of intensive vs. extensive mowing *NECB* was higher for the intensive (147±130 g C m^{-2} year^{-1}, net sink), compared to the extensive mown grassland (–57+130/–110 g C m^{-2} year^{-1}, net source), which indicated that mown grassland could turn to be a net sink for carbon supposing that fertilization (200 kg N ha^{-1} year^{-1}) is applied (Amman et al. 2007).

4.3. Farm scale carbon and greenhouse gas flux components

Grasslands were proved to be an important net sink for CO_2, in terms of NEE (Gilmanov et al., 2010), although a high variability was observed among grassland sites, which was influenced by climate, management, soil, and vegetation properties (Senapati et al., 2014, Soussana et al., 2010). In our paired study the vegetation composition (Koncz et al., 2014) and the abiotic factors (soil temperature and soil water content) (Koncz et al., 2015) did not differ between the grazed and mowed treatments; eliminating the differentiating effect of these factors on *NEE* between the grazed and mowed treatments. In other studies, differences in vegetation among grassland sites were observed to affect *NEE* (Klumpp et al., 2011; LeCain et al., 2002). In our study *NEE* differed between the treatments due to different management regimes, which influenced the components (*GPP*, R_{eco}) of *NEE*. Similarly to our findings, Soussana et al., (2008) observed a sharp decrease in carbon uptake just after cutting due to the lack of biomass, which led to a reduction of GPP, while the remaining plants parts (roots) had relatively large R_{eco}. In this study both grazing and mowing reduced leaf area index (LAI) (Koncz et al. 2015). After two weeks of grazing LAI only decreased by 24.4±14.3%, while after mowing event (one day) LAI decreased by 66.8±13.9% (Koncz et al. 2015), thereby affecting CO_2 uptake and release by the vegetation. The time course of NEE was markedly affected by mowing and grazing management (Fig 4.a). Mowing caused an abrupt decline in NEE, while the impacts of grazing were more gradual since only part of the available herbage was defoliated. Thus, NEE was more sharply reduced after mowing events then after grazing (Fig. 4). Compared to our study, Soussana et al. (2010) reported 10% higher *NEE* (net sink) in the mown compared to the grazed treatments due to the 10% higher rate of precipitation and the double amount of N fertilizer applied. In a study with adjacent grazed and mowed sites Senapati et al. (2014) also found higher *NEE* (net sink) in the mowed (476±51.8 g C m^{-2} $year^{-1}$) compared to the grazed (231±73.5 g C m^{-2} $year^{-1}$) treatment. On the other hand, in both grazed and mowed treatments the sites were intensively used (frequently mowed and intensively grazed), sown (species number was three) and fertilized (Senapati et al., 2014), which accounts for the high rate of net C sink activities. However, in our study the management intensity was lower and the species number was much higher (species diversity>80, Koncz et al., 2014). During the period under investigation *NEE* was found to be the highest in 2011 (net sink) in our study in both treatments due to favourable soil water conditions, while it was lower in both treatments in 2012 and 2013 when soil water content, which is a limiting factor for biomass accumulation in semi–arid grasslands, was lower compared to the figures recorded in 2011. Drought highly influenced *NEE* and enlarged the differences between management regimes. Droughts are expected to be more frequent occurrences in temperate climate, which will presumably decrease net C sink capacity especially in the mowed treatments.

At farm scale the only net sink activity besides *NEE* was the CH_4 oxidation of the soil ($F_{CH4–soil}$), which was relatively small in both the grazed and mowed treatments (Table 2). On the other hand, the aerated soil is still important at larger scales, as this type of soil is responsible for 10% of the global net CH_4 sink activity (Prather et al., 1996).

Although the largest GHG emitting factor of the farm was animal respiration (Table 2), it should be noted that in an attempt to satisfy the growing meat demands these emissions cannot be reduced. Also, it is important to note that livestock not only meets the growing milk and meat demands (Steinfeld et al., 2006), but it also provides jobs in rural areas (Soussana and Lemaire, 2013). Relevant mitigation would not mean reducing livestock, but improving grazing management (e.g. by optimal grazing intensity) and applying fertilization, irrigation, introducing legume mixtures (Smith et al., 2008 Bellarby et al., 2013), or extending extensive grazing (where appropriate, see 4.4. chapter).

In ours study the rate of CH_4 emissions from fermentation (Table 2) varied as a function of the number of animals based on IPCC (2006b) estimations. CH_4 emission from fermentation could be reduced in several ways such as using antibiotics, vaccination, ionofors, halogens, or probiotics (Smith et al., 2008). However, the Grey Cattle is a natural and cultural heritage, protected by Hungarian national (32/2004. IV. 19. Parliament Decision) and EU law (1300/2011/EU). Therefore, in an attempt to preserve the inherited quality of the Grey Cattle traditional management practices are favoured and the above mentioned treatments are not allowed (Baracskay et al., 2007).

Soil N_2O emission in our study in both the grazed and mowed treatments (Table 2) was 40% lower compared to the average N_2O emission reported for European grasslands (0.14 g N_2O m^{-2} $year^{-1}$) (Flechard et al., 2007). This was probably due to the extensive grazing management regime in the grazed treatment and to the lack of fertilization in the mowed treatment. As a results the N input as a substrate for N_2O production in both treatments was low compared to the investigations of others in which N input and consequently N_2O emission was high (Cowan et al., 2015; Velthof et al. 1996).

4.4. Extensive grazing as a potential mitigation option

Extensive grass fed farming was shown to have lower emission per kg of meat product (19.4–21.6 kg CO_2 equiv. kg^{-1} meat) compared to intensive grain fed management regime (16.4–30.2 kg CO_2 equiv. kg^{-1} meat) (Bellarby et al., 2013). Similarly, in another study, pasture–fed beef had lower environmental effect compared to grain–fed beef due to the lack of irrigation, fertilizers, biocides, and to the low fossil fuel consumption in mechanization (Foley et al., 2011). In contrast, based on Steinfeld et al., (2006) assessment, extensive management had higher *GHG* emissions than intensive management due to the lack of manure treatment (which leads to high CH_4 emissions), and high fermentation rates (due to low digestion efficiency of grasses compared to grains). Certainly there is an obvious need to investigate the GHG balance of intensive vs. extensive farming systems but our study emphasises the fact that this cannot be done without estimating the C uptake (NEE) of grasslands, which is often neglected (Nijdam et al., 2012; Opio et al., 2013; Schwarzer, 2012). We showed that the most important net sink capacity of the farm was indeed the C uptake of the grassland under extensive grazing, while mowing led to a net loss of carbon and *GHG* especially in years with dry soil moisture condition. Grazing optimization through reducing overgrazing and under–grazing was proved to have mitigation potentials (Smith et al., 2008; Herrero et al., 2016), which could contribute to increased sequestration by 130 g CO_2 equiv. m^{-2} $year^{-1}$ (Bellarby et al., 2013). The grazed site was a net sink for CO_2, therefore it is the mowing management which should be improved to increase C uptake. Further research is needed to investigate whether this could be achieved in the region by e.g. allowing organic manuring (where appropriate), by increasing the ratio of native N fixing legumes by mowing after seed dispersal (as N fixing plants increases productivity, Lüscher et al. 2016) or by irrigation. Only a small proportion of grasslands is irrigated (0.1%) and manured (0.8%) in Hungary (Kis–Kovács et al., 2014), therefore there are a number of potential possibilities to increase net C sink capacity with improved grassland management techniques, which might offset the extra emission related to the extra emission sources.

Increasing carbon uptake, however, should not be the one and only goal because grasslands provide a wide range of other ecosystem services. Grasslands prevent soils from erosion (Breshears et al., 2003; Li et al., 2005), provide herbs, and control the spread of invasive species (Haraszthy, 2013). Grasslands and grazing management maintain high biodiversity (Báldi, et al. 2013), sustain traditional management techniques (Scholes et al., 2014) along with related socio–economic values (Henwood, 2010). Animal density (Gibon and Mihina, 2003) and related *GHG* emissions halved during the change of political regime in Hungary during the 1990's (there are now 751 thousands of cattle, KSH, 2015) and also in the Central–Eastern European region (National Reports, 2014). In 2013, European cattle density was almost four times lower in the East (0.12±0.06 cattle ha^{-1}) compared to the West (0.46±0.26 cattle ha^{-1}) (FAOSTAT 2015). Therefore, further reducing the number of animals to mitigate climate change is not an appropriate mitigation option in this region especially as grazing maintains ecosystem services of grasslands and preserves food security and related societal benefits.

In summary, we found that extensively grazed grasslands could act as a net sink for GHGs, in contrast to mowing, therefore it can be considered as a climate friendly management. We suggest, wherever possible, allocating more time to grazing in pastures, rather than to feeding on mown herbage. Possibilities to expand grazing in space beside time (grazing period) should also be investigated. Due to political changes during the 1990's in Hungary, 436 thousand ha of grasslands and croplands remained abandoned and it was estimated that only half of the pastures were used for grazing (Kis–Kovács et al., 2014). Abandonment of Hungarian grasslands led to the decrease in species diversity and to the spread of invasive species (Molnár et al., 2016). Abandonment is also a negative process from the stakeholders' perspective. According to a survey people of the Kiskunság region recognized animal husbandry (including hay, pasture, livestock and agro–biodiversity) as one of the most important ecosytem services besides water regulation (Kelemen et al. 2016). Also, traditional knowledge on grassland management still exists in the region along with the recent conservation/scientific knowledge (Molnár et al., 2016), which could provide a baseline to expand farming. Low profitability of farming could be increased by entering the organic market with meat products. There are a large number of farms in Hungary, which are technically producing organic meat, although not yet officially due to the administrative procedures required (Dezsény and Drexler 2012). Therefore, there is a potential possibility to expand grazing management, although in order to evaluate the feasibility and the full climate change mitigation potential of the expansion of extensive grazing in time and in space a broader ecological, socio–economical and political research is needed.

5. Conclusion

Livestock farming will need to satisfy an ever increasing demand for food while it is threatened by climate change. However, livestock itself contributes to climate change as well, so its emissions should be also reduced. Climate change mitigation options for livestock farming rely on different management regimes. We showed that grasslands under extensive grazing system was a net sink, while mown grassland was a net source for net greenhouse gas fluxes (carbon dioxide, methane and nitrous oxide) in a Grey Cattle livestock system in Hungary (Bugac, 2011–2013). At farm–scale, the investigated farm, which included grazed and mowed grasslands and a feeding system, was found to be a net sink for the greenhouse gases under conditions of good water supply (due to high carbon uptake, i.e. *NEE* of the grasslands), while it was a net source in the two dry years when emissions were not compensated by the low carbon uptake of the grasslands. We propose that mowing management should be improved (e.g. by fertilization, by increasing the ratio of native N fixing legumes, by irrigation) in semi−arid sandy grasslands to avoid potential net carbon loss (in terms of *NEE*) during dry years. We urge that carbon uptake of grasslands should be included in the estimation of livestock farming’s share in total greenhouse gas balances. We suggest that extensive grazing should be supported and extended in space and time (where possible) rather than mowing as grazing could be a solution in combating climate change.

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Supplementary material for the manuscript titled:

SM. 1. Uncertainty assessment

1.1. Methods

Total uncertainty assessment was based on error propagation according to equation 3.2 from the IPCC (2006a). Random error of *NEE* was calculated according to Hollinger and Richardson (2005) and Richardson et al. (2006). Uncertainty for each component of the lateral fluxes was assumed to be 5% (management data were recorded on a daily bases, although error could have still occurred due to human mistakes or forage weight measurement inaccuracies). Uncertainty of fermentation and manure CH_4 fluxes were estimated according to IPCC (2006), while uncertainty of soil CH_4 and soil N_2O fluxes were estimated according to Horváth et al. (2010).

1.2 Results

Total uncertainty of *NEE* on average was 26.3 g C m^{-2} $year^{-1}$, which consisted of random errors (4 g C m^{-2} $year^{-1}$) and errors due to gap filling (26 g C m^{-2} $year^{-1}$). The errors were smaller than the differences between the two treatments in *NEE*. The uncertainty of fermentation CH_4 emission was 20% based on IPCC (2006b), which depends on feed digestibility. The uncertainty of manure CH_4 emission was 10% (IPCC 2006b) and 25% for manure N_2O emission, which depends on the different N excretion rates of animals (Kis–Kovács et al., 2014).

The uncertainty of soil CH_4 and N_2O fluxes due to non–linear gas accumulation rate in the chamber was less than 10% (based on measured data). Also, there is evidence for underestimation of the soil flux using closed chambers, considering the fact that the effective volume of the chamber is larger than its calculated value, since the effective volume also includes the volume of air–filled spaces in the soil below the chamber (Horváth et al., 2010). Based on previous studies at the Bugac research site fluxes could have been underestimated by

25% (Horváth et al., 2010). Error of soil CH_4 and N_2O flux measurements could also arise from non–continuous measurements, which was estimated to be 10% based on Reeves and Wang (2015). The total uncertainty of gas flux measurements with chambers was 28%.

The total uncertainty of *NGHG* at the grazed treatment was 43% and 54% at the mowed treatment. Uncertainties mainly emerged from the large inter–annual variability of *NEE*. The uncertainty values were considerably reduced in our study compared to a single site investigation because of the very similar systematic errors (same methodology) for both grazed and mowed treatments. Nevertheless, it should be emphasized that farm scale *NECB* and *NGHG* are estimations based on the area–weight averages of the fluxes of the sampled mowed and grazed treatments. Uncertainties in our study was similar to others (for *NECB* 20–80%) (Soussana et al., 2010; Mudge et al., 2011).

SM 2. Representativeness of the sampled mowed area

During the selection of the mown EC (eddy–covariance) site it was a selection criteria to be representative of the mown areas in terms of climate, vegetation and soil. The assumption that the vegetation of the mowed sampled area was similar to the total mown areas of the farm was based on field survey and on the vegetation map of Hungary (Bölöni et al., 2011). According to the map the most abundant (20 500 ha) vegetation type in the Kiskunság region is the "closed sand steppe, H5B" which is the same as the sampled site, thus sampled area represents the abundant vegetation type used for grazing and mowing in the region, including the farm area (Bölöni et al. 2011). However, mowed areas also include wet meadows, which have higher productivity, but maybe higher N_2O emissions. Also, according to the Hungarian soil map database (AGROTOPO, http://maps.rissac.hu/agrotopo/) the soil type (chernozem type sandy soil), the soil texture (sand), the soil organic content (around 50 t ha^{-1}), the origin of the soil (alluvial deposit) and the carbonate status (calcerous) of the sampled mowed site was the same as in the total mowed areas. Clearly, measurements on regularly mown sites would be necessary (as sampled site was grazed before 2011), however, grasslands are managed interchangeably at the farm (grazing/mowing), therefore this shift (form grazing to mowing) represents a regular management practice. We acknowledge that up−scaling is an assumption, which holds uncertainty. Last but not least, logistics and security issues were important aspects when we selected mown EC site, as it was not possible to investigate an area which had been mowed for a longer time period.

It has to be noted that the farm scale is an assumption based on up–scaling of point measurements on sampled EC sites. Point measurements can be used for up–scaling based on general management, vegetation, soil and climate characteristics even to regional and European levels (Janssens et al., 2003; Schulze et al., 2009).

SM 3. Sensitivity analysis of the effect of soil organic carbon (SOC) content on net ecosystem exchange (NEE)

The SOC of mowed (3.13±1.18%, 6.03±2.27 kg C m^{-2}) and grazed sampled sites were similar in 2011 at the start of the experiment (3.74±1.00%, 7.02±1.94 kg C m^{-2}, Koncz et al. 2015); although it was statistically significantly different (t-test, p=0.02, n=40 samples per sites). Differences occurred at the beginning of the experiment, therefore it was due to spatial heterogeneity of SOC, which varied highly even within a few meters, rather than to management differences. Sample sites with statistically identical SOC content would have been difficult to select. To test the effect of the different SOC content on NEE, i.e. to assess the sensitivity of NEE to SOC we used simulations with different SOC content with Biome–BGCMuSo model. (Although, the focus of this study is not modeling or simulation.)

Biome-BGC is a widely used and popular biogeochemical model which simulates the storage and flux of water, carbon, and nitrogen between the ecosystem and the atmosphere. Our research group developed Biome-BGC version 4.1.1 to improve essentially the ability of the model to simulate carbon and water cycle in real managed ecosystems. The model version which contains both the former and the new developments is referred to as Biome–BGCMuSo (Biome–BGC with multi-soil layer; Hidy et al., 2016).

The Biome–BGCMuSo was validated using daily eddy covariance data (gross primary production, GPP; ecosytem respiration, Reco; and latent heat flux, LHF) measured in Bugac from 2011 to 2013. The model behavior can be evaluated with the goodness–of–fit of the simulation with the measurement data. This was done using relative error [RE (%); mean difference between the measured and the simulated data relative to the difference of maximum and minimum of the measured data] and square of linear correlation coefficient (R^2) between measured and modeled fluxes. RE was between 10.6–21.2%, R^2 was between 0.63–0.70 regarding to the different reference data using the developed model. The validated model was used to estimate the carbon balance components at Bugac.

The model has two simulation phases: the first is the spin–up simulation, which starts with very low initial level of soil carbon and nitrogen and runs until a steady state is reached with the climate in order to estimate the initial values of the state variables. For the spin–up phase, the 1901–2000 period was used for which the basic meteorological data were available from the CRU TS 1.2 database (Climatic Research Unit, University of East Anglia). The second is the normal simulation phase, which uses the results of the spin–up simulation as initial values for the carbon and nitrogen pools. In this phase, in situ measurements are used. As result of the spin–up phase the modeled soil carbon content was 6.0 kg C m^{-2} in the top soil layer which is fit to the measured average values (6.6±2 kg C m^{-2}). First we run the model from this (6.6±2 kg C m^{-2})

soil carbon condition assuming grazing and mowing. As result of grazing simulation the yearly averaged NEE was 84.4 g C m^{-2} $year^{-1}$ (net sink), while as a result of mowing simulation it was –14.5 g C m^{-2} $year^{-1}$ (net source). As the next step we decreased the soil carbon content with 10% according to the measured soil carbon data from grazed site (5.9 kg C m^{-2}; soil carbon decreased; SCD). As a result of SCD simulation assuming grazing the yearly averaged NEE was 85.1 g C m^{-2} $year^{-1}$, while assuming mowing it was –7.4 g C m^{-2} $year^{-1}$. In the next step we increased the soil carbon content with 9% according to the measured soil carbon data from grazed site (7.2 kg C m^{-2}; soil carbon increased, SCI). As a result of SCI simulation the yearly averaged NEE assuming grazing was 86.5 g C m^{-2} $year^{-1}$, while assuming mowing it was –21.0 g C m^{-2} $year^{-1}$. It can be seen from the simulated NEE data that the effect of the management option (grazing or mowing) is much higher than the effect of the soil carbon content change: the mean of the grazed NEE was 85.33 g C m^{-2} $year^{-1}$, the standard deviation on was 1.07 g C m^{-2} $year^{-1}$, the mean of the mowed NEE was –14.3 g C m^{-2} $year^{-1}$, the standard deviation was 6.8 g C m^{-2} $year^{-1}$, the mean of the NEE difference between grazed vs mowed was 99.63 g C m^{-2} $year^{-1}$, the standard deviation on is 7.53 g C m^{-2} $year^{-1}$.

We concluded that the existing SOC difference between mown and grazed site had minor impact on NEE differences between the two treatments but the management largely affected NEE. The time course of NEE was markedly affected by the cutting and grazing management. Soussana et al. (2008) also observed a sharp decrease in carbon uptake (just after cutting due to the lack of biomass, which led to a reduction in GPP, while the remaining plants (roots) had relatively large R_{eco}.

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