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Key Points:
- N_2O fluxes from dung patches on rangeland in Kenya are only 1% of current IPCC estimate
- Low emissions are very likely due to low N content of the dung, which results from poor feeds
- Emission factor for dung patches needs to be revised, specifically for livestock systems in developing countries

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Effect of Dung Quantity and Quality on Greenhouse Gas Fluxes From Tropical Pastures in Kenya

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Abstract To improve estimates of agricultural greenhouse gas emissions in sub-Saharan Africa, we measured over six individual periods of 25–29 days fluxes of methane (CH_4), carbon dioxide (CO_2), and nitrous oxide (N_2O) with subdaily time resolution from dung patches of different quality (C/N ratio: 23–41) and quantity (0.5 and 1.0 kg) on a Kenyan rangeland during dry and wet seasons. Methane emissions peaked following dung application, whereas N_2O and CO_2 fluxes from dung patches were similar to fluxes from rangeland soils receiving no N additions. Greenhouse gas emissions scaled linearly with dung quantity during both seasons. Dung with a low (23) C/N ratio produced up to 10 times more CH_4 than dung with a high (41) C/N ratio. Overall, CH_4 emission factors (EFs) ranged from 0.001 to 0.042%, lower than those derived in temperate regions. Cumulative CO_2 and N_2O emissions were similar for all treatments across the different seasons. The N_2O EF ranged from 0 to 0.01%, less than 1% of the Intergovernmental Panel on Climate Change Tier 1 default EF (2%) for N_2O emissions from dung and urine patches, likely because of the low dung N content (9.7–16.5 g N kg^{-1} dry matter). However, these results were consistent with the updated cattle dung EF (0.2%) developed for Kenya in 2016/2017 (EF database ID# 422665). In view of the wide range of climates, soils, and management practices across sub-Saharan Africa, development of robust GHG EFs from dung patches for SSA requires additional studies.

Plain Language Summary With regard to the agricultural sector, livestock production systems are the dominant greenhouse gas (GHG) source. A significant part of emissions due to livestock production is linked to GHG emissions from dung patches on rangelands. While this source is rather well constrained for countries with developed economies, little is known about GHG emissions from dung patches in developing countries, specifically for countries in Sub-Saharan Africa. Based on own measurements and an extensive literature review we show that GHG emissions from dung patches are likely highly overestimated as poor feed quality and differences in environmental conditions strongly limit GHG emissions. Our work calls for a revision of emission estimates from this important GHG source for developing countries.

1. Introduction

Agricultural production systems and particularly livestock systems are major sources of greenhouse gas (GHG) emissions (Lin et al., 2009). While carbon dioxide (CO_2) originating from agricultural sources is mainly linked to land use and land use change and subsequent depletion of soil organic carbon (C) stocks, emissions of nitrous oxide (N_2O), a GHG with a global warming potential approximately 298 times more powerful than CO_2 (IPCC, 2013), are mainly associated with the use of organic and inorganic fertilizers for crop and feed production. Currently, N_2O emissions from agricultural systems have been estimated to contribute approximately 60% to total anthropogenic N_2O emissions (IPCC, 2014). Agricultural sources of CH_4 (global warming potential 25 times higher than that of CO_2, 100-year time horizon on a per mass basis) are dominated by emissions from enteric fermentation, manure management, and rice production (Tubiello et al., 2013).

More specifically, grasslands used for livestock production systems occupy 25% of the Earth's surface and support approximately 1.8 billion livestock units (Krol et al., 2016). These systems generate 80% of all agricultural non-CO_2 emissions (Tubiello et al., 2013), which makes them responsible for about 12% of the global anthropogenic GHG emissions (Havlík et al., 2014). Furthermore, global GHG emissions from livestock systems are projected to increase at a rate of 1% to 1.5% annually (Smith et al., 2016). Global livestock produces approximately 7×10^9 Mg of manure annually, which is consequently a considerable source of N_2O and CH_4 (Thangarajan et al., 2013), with manure left on pasture, manure applied to soils and manure management contributing approximately 26% to total GHG emissions from all agricultural sources (Tubiello et al., 2014), it is noteworthy that N_2O from dung and urine deposited on pasture by grazing livestock represent about one third of all agricultural N_2O emissions (Bogner et al., 2008). Van der Weerden et al. (2011) estimated that for New Zealand dung and urine patches are the largest single source for direct and indirect N_2O emissions, contributing approximately 80% to total national anthropogenic N_2O emissions. For Canada, Rochette et al. (2014) estimated that N_2O emissions from urine and dung patches comprise 11.5% of national agricultural N_2O emissions.

Compared to other continents, agricultural GHG emissions from Africa comprise a higher proportion of total anthropogenic GHG emissions with 14%, 5%, 4%, and 25% of emissions originating from enteric fermentation, manure management, manure applied to soils, and dung and urine left on pasture, respectively (Tubiello et al., 2014). In line with the global trend, GHG emissions from the livestock sector in Africa are also expected to increase due to the projected population increases and subsequent enhanced demand for livestock products (Dangal et al., 2017; Lelieveld et al., 1998).

Dung contains not only a large amount of readily available C, which can stimulate CO_2 and CH_4 emissions from the dung and underlying topsoil (Wang et al., 2013), but also organic and mineral nitrogen (N), as 75–90% of ingested N by grass-fed animals is returned to the soil in form of dung or urine (Oenema et al., 2005). The split between how much N is excreted as dung or urine depends on the dietary protein intake and on its digestibility. Above the required protein intake will increase the proportion of N excreted as urine, while at low concentrations of digestible protein, the proportion of N excreted as dung will increase. The split of cattle in western European countries is assumed to be 40:60; that is, 40% of N is excreted as dung, while 60% is excreted as urine (Chadwick et al., 2018). Reviewing existing literature for tropical livestock systems in Africa, Rufino et al. (2006) found that the total amount of N in dung in relation to total excreted N (dung + urine) ranges from 28 to 99% with a mean value of 66 ± 0.6%. The amount of organic N present in a dung patch can be equivalent to up to 1,130 kg N ha^{-1} (Saarijärvi et al., 2006), far exceeding plant N demand if all dung N would be mineralized. The fate of N in dung patches may differ depending on the environmental situation as N losses can occur along various hydrological and gaseous pathways in the form of NH_3, NO_2, N_2, N_2O, and NO. Furthermore, N can accumulate in the soil. That in turn can stimulate soil microbial activity, leading to anoxic conditions even in the topsoil. As a result NO_3 reduction by denitrification, the main source of N_2O emitted from soils and dung patches is likely to be stimulated as well (Virkajärvi et al., 2010).

There have been a number of studies examining GHG emissions from excreta patches, with most of them being carried out in temperate regions (Hoeft et al., 2012; Kelliher et al., 2014; Ma et al., 2006). In contrast, measurements for the pan-tropics, and particularly sub-Saharan Africa (SSA), are scarce, even though GHG emissions from the agricultural and specifically the livestock sector are the dominant anthropogenic GHG emission source for many SSA countries (Pelster et al., 2016). As livestock production relies predominantly on free grazing during daytime, with animals being kept in kraals or confined areas close to the homestead only during the night, it is estimated that minimum of 40% of excreta are deposited on rangelands (Rufino et al., 2006). As highlighted before, GHG emissions from dung deposited to rangelands in most of SSA are currently estimated using the Intergovernmental Panel on Climate Change (IPCC) Tier 1 approach. The Tier 1 approach uses an emission factor (EF) that is for urine and dung both and was developed in temperate regions; thus, it likely does not reflect the local climate and soil conditions found throughout SSA (Bell et al., 2015). A number of recent studies (Bell et al., 2015; Chadwick et al., 2018; Krol et al., 2016; Thomas et al., 2017; Van der Weerden et al., 2011) suggest that specific EFs should be used for dung and urine as this allows better quantification of the sources and more effective targeting of mitigation strategies.

Smallholder livestock farms dominate the agricultural landscape of SSA and are expected to continue to do so for at least the next 30 years (Assan, 2014). More than 90% of dry matter fed to animals in these systems comes from rangelands, pastures, and annual forages, with only a minor contribution of purchased feeds (Assan, 2014). These fodder materials are typically high in fiber with low digestibility and low protein content compared to temperate feeds. Consequently, this results in low quality and low N content dung (Rufino et al.,

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2006) compared to dung from livestock systems in developed countries. In addition, smallholder livestock production systems in SSA are highly diverse, both spatially (i.e., among regions) and temporally (i.e., between rainy and dry seasons). Accordingly, both the amount and quality of the dung excreted are variable, as is the climate and thus conditions for decomposition. Therefore, IPCC encourages the development of country-specific GHG EFs that better reflect GHG emissions from excreta under existing environmental (climate and soil properties) and livestock management (livestock species, feed supply and quality, and management system) conditions, as these factors are known to alter both nutrient budgets and GHG fluxes (Krol et al., 2016; Pelster et al., 2016). However, the effect of the amount and quality of dung as well as the season (i.e., wet or dry season, which differs markedly with regard to environmental conditions) that the dung is excreted to rangelands on CH_4 and N_2O emissions from dung remains largely unstudied for tropical livestock systems.

To address these questions, this study aimed to (1) quantify GHG emissions from dung deposited on rangelands during both dry and wet seasons, (2) assess the effects of dung quantity and quality on GHG emissions from dung applied to rangelands, and (3) use the outcomes of (1) and (2) to develop regionally appropriate EF for N_2O and CH_4 for dung applied to rangelands.

We hypothesized that (a) dung GHG emissions are higher during the wet season than during the dry season because of the increased soil moisture and rainfall during the wet season, (b) dung GHG emissions increase exponentially with the amount of dung added to the rangeland, (c) both N_2O and CH_4 emissions from dung from cattle fed with a poor quality diet are lower than the emissions from dung from cattle receiving high quality feed, and (d) GHG EFs for SSA are lower than currently used IPCC Tier 1 default EFs.

2. Materials and Methods

2.1. Study Site

The experiment was set up on the campus of the International Livestock Research Institute (ILRI), Nairobi, Kenya (1°16′13″S; 36°43′23″E; altitude 1,809 m above sea level [asl]), with its Mazingira Centre providing the necessary analytical capacity (www.mazingira.ilri.org). The pasture was dominated by a mixture of Kikuyu grass (*Pennisetum clandestinum* Hochst. ex Chiov.) and Rhodes grass (*Chloris gayana* Kunth). The site was not grazed, but grass was manually cut to 2-cm height every 2 to 3 weeks during the wet season. Grass did not need to be cut during the dry season. Soils were well drained, deep humic nitisols (IUSS Working Group WRB, 2007) with a clay-texture (24% sand and 63% clay) in the uppermost 10 cm. The topsoil C content was 25.08 ± 0.03 g C kg^{-1}, while the soil N content was 2.31 ± 0.01 g N kg^{-1}. The pH of the topsoil measured in water (1:2.5) was 6.2 ± 0.1.

A meteorological station was installed directly at the experimental site. Precipitation was recorded with a tipping rain gauge (ECRN-100 high-resolution, Decagon, Pullman, WA; USA). Air temperature and humidity were measured with the temperature/relative humidity sensors (ATMOS 14, Decagon, Pullman, WA; USA) every 5 min, and soil moisture and temperature at 0.05-m soil depth was measured with the Decagon 5TM sensors every 5 min. Precipitation for the period 8 March 2016 to 7 March 2017 (i.e., our observation period) was 607 mm, while the mean air temperature was 17.8 °C (Figure 1), which were slightly lower than the long-term average of 869 mm and 19.0 °C, respectively (Pelster et al., 2016). During the long rains, occurring from end of March to end of June, 395 mm (65% of total precipitation) was observed. In the so-called short rains period, occurring from end of October to end of December, 127 mm of rainfall (21% of total precipitation) was measured. The remaining 14% of precipitation occurred sporadically during the dry seasons.

2.2. Experimental Design

2.2.1. Effect of Dung Quantity on GHG Fluxes

In experiment I, we assessed whether GHG fluxes scale exponentially with the dung quantity added to grassland soils. GHG fluxes from three treatments (a control [no dung addition], addition of 0.5-kg fresh dung per chamber [corresponding to the average dung weight as observed in one of our animal trials, Table 4], and addition of 1.0-kg fresh dung) were measured with three spatial replicates. Experimental periods covered the time from 8 March to 12 April 2016 (Trial 1, dry season) and 13 June to 22 July 2016 (Trial 2, transition period from wet to dry period; Figure 1).

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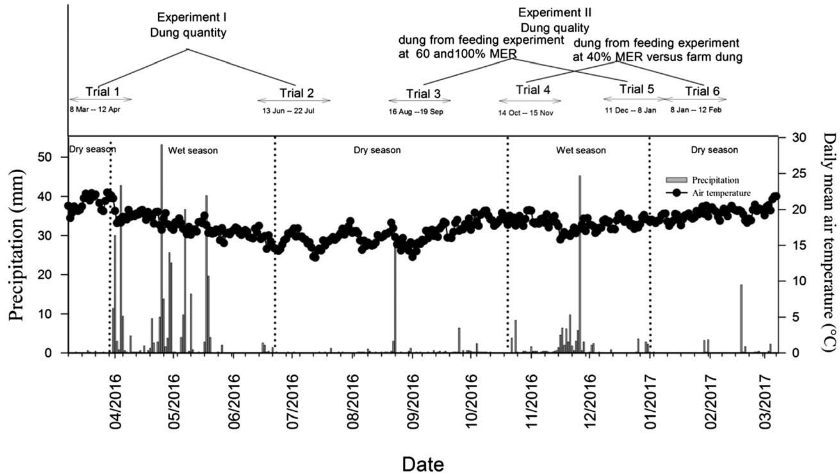

Figure 1. Duration and time of the individual experiments. MER stands for maintenance energy requirement. The lower panel shows air temperature (markers) and precipitation (bars) from 8 March 2016 to 7 March 2017. To note: There were no predung application measurements for the very first part of the experiment.

For each of the trial periods, fresh dung was collected from the ILRI Nairobi farm adjacent to the study site. At the ILRI farm, cattle graze freely during daytime and are taken to an open shed with concrete floor at night. In the early mornings of 8 March and 24 June fresh dung was collected, mixed, weighed, and applied onto the grassland surface within 2 hr following collection such that the dung was in the center of the GHG chambers. GHG emissions during dung mixing were not measured. Subsamples of dung were frozen for later analysis (water content and total C and N content) in the laboratory.

2.2.2. Effect of Dung Quality on GHG Fluxes

In experiment II, we investigated the effect of dung quality on GHG fluxes. Dung of different qualities was obtained from a parallel animal feeding trial where 14-month-old boran steers (*Bos indicus* L.) with an average live weight of 183 kg were fed at different maintenance energy requirement (MER) levels (Daniel Korir, ILRI, personal communication, April 28, 2018). The steers fed at either 40 or 60% MER were provided with only Rhodes grass hay, while animals fed at 100% MER were given Rhodes grass hay (at 80% MER) plus cottonseed meal (10% MER) and molasses (10% MER). Total tract digestibility for the different MER treatments (40%, 60%, and 100%) were 55.3%, 59.1%, and 61.5%, respectively. Each treatment period encompassed 3 weeks for adaptation, 2 weeks for sample collection, and another 2 weeks of refeeding. During adaptation and sample collection period steers were fed at given MER levels, with those being fed at submaintenance levels losing weight, while animals on 100% MER marginally gaining weight. During the refeeding period all steers were fed ad libitum with Rhodes grass hay plus cottonseed meal and molasses. The MER of each steer was calculated as follows:

$$\text{MER (MJ)} = 0.0819 \times \text{live weight (kg)} + 21.625$$ (National Research Council (U.S.), 1989).

Dung from the steers fed at three different MER levels (40, 60, and 100%) was collected early in the morning from individual pens with concrete floors. In addition, we also included dung from pasture-fed steers at ILRI farm (MER levels 130–140%, Daniel Korir, ILRI, personal communication). The dung from these animals was collected by housing the animals in a communal barn with concrete floor overnight and collecting the dung early the following morning. Fresh dung was applied to the rangeland plots as a patch of approximately 3-cm height covering an area of 16 cm × 20 cm (1 kg) in the middle of each chamber within 2 hr following its collection. Subsamples of dung were also frozen for further nutrient analyses. The trials from experiment II were split into two periods for each season because only nine GHG chambers were available. Nevertheless, we measured GHG fluxes for each type of dung during one dry and one wet season (Figure 1), while the control, no-dung amendment treatment was measured during all periods. The first experimental period of experiment II (Trial 3, 16 August to 19 September 2016) consisted of the control, dung from cattle fed at 60% MER, and dung from cattle fed at 100% MER. These treatments were repeated during the wet season (Trial 5, 11 December 2016 to 8 January 2017). The other two experimental periods included the control, dung from cattle fed at 40% MER, and dung from free ranging cattle (Trial 4, 14 October to 15 November 2016, wet season and Trial 6, 8 January to 12 February 2017, dry season).

2.2.3. GHG Flux Measurements

Soil GHG fluxes were measured semicontinuously in 84/140-min time resolution with an automated chamber system (Butterbach-Bahl et al., 1997), consisting of nine chambers, an automated gas sampling system and a cavity ringdown laser absorption spectrometer (G2308, Picarro Inc., Santa Clara, CA, USA) for measurements of N_2O, CO_2, and CH_4 concentrations in the chamber headspace. Nine stainless steel frames (0.50 m × 0.50 m × 0.05 m) were inserted into the soil to the depth of 0.05 m. Opaque chambers (0.50 m × 0.50 m × 0.15 m in height) were fastened to the frames with clips to ensure they were airtight (Butterbach-Bahl et al., 1997). The chambers were divided into three blocks of three chambers, with chambers approximately 0.5 m away from each other. For the first trial, each block of three chambers was closed and sampled for 24 min, before chambers were reopened and the next block was closed and sampled. Following gas sampling of the three blocks a 12-min period followed, which was used for the injection of standard gas for calibrating the GC systems. This resulted in a total measuring cycle of 90 min (3 × 24 + 12 = 84 min) for all nine chambers. Because soil N_2O fluxes tended to be low, we extended the deployment time for the following trials to 42 min, so that each cycle across all three blocks lasted 140 min (3 × 42 + 14 = 140 min). Changes in gas mixing ratios of the headspace of the closed chambers were monitored sequentially in 1-min intervals for each chamber during the deployment. To avoid differences in soil moisture between blocks, chambers were programmed to open automatically during precipitation events.

The GHG fluxes were calculated from the linear change in headspace gas mixing ratios during chamber closure and corrected for atmospheric pressure and chamber air temperature (Butterbach-Bahl et al., 1997). As all the chambers were dark chambers covered with a reflective surface, only respiratory CO_2 fluxes were measured. After chamber installation, but before dung addition, the grass in the chambers was cut to 2-cm height. Except for the first trial, gas flux measurements started a few days prior to dung application in order to assess the spatial variability of background soil GHG fluxes across the individual chambers. Each trial ended when the GHG fluxes had reached background as found prior to dung application. This normally took around 2 weeks, though we continued to measure for another 2 weeks. After each trial chambers were moved to unaffected grassland in order to avoid possible memory effects on GHG fluxes.

2.3. Calculation of Cumulative GHG Emissions and Emission Factors

Cumulative emissions were calculated by linear interpolation between individual GHG flux observations for a period of 29 days in the dung quantity experiment and over 25 days in the dung quality experiment, respectively. Net cumulative emissions on a dry matter basis were calculated by subtracting the emissions from the control (no dung) plots from the total emissions from plots with dung. EFs for CH_4 and N_2O were calculated according to the IPCC methodology (Eggleston et al., 2006):

$$\mathrm{CH_4EF}\ (\%) = \frac{\text{Cumulative CH}_4\text{ emssion (g CH}_4\text{-C) from dung appplication-Cumulative CH}_4\text{ emission (g CH}_4\text{-C) from control}}{\text{Carbon content in applied dung (g C)}} \times 100$$

$$\mathrm{N_2O\ EF}\ (\%) = \frac{\text{Cumulative N}_2\text{O emssion (g N}_2\text{O-N) from dung appplication-Cumulative N}_2\text{O emission (g N}_2\text{O-N) from control}}{\text{Nitogen content in applied dung (g N)}} \times 100$$

2.4. Dung Analysis

Three replicates of fresh dung samples were weighed and then dried in an oven at 105 °C until constant weight to derive total dung water content. Another three dung samples were dried at 50 °C, then ground and weighed for subsequent total C and N content determination with an elemental combustion system (Elemental combustion system, Costech International S.p.A., Milan, Italy).

2.5. Data Analysis

For the dung quantity study, dung properties (water quality, C and N content, and C/N ratio) were compared between seasons using a *t* test, while for the dung quality study, the properties were compared using one-way ANOVA using the dung type as a fixed factor. For the dung quantity study, GHG gross cumulative emissions and net cumulative emissions were analyzed for each period using a one-way ANOVA. For the dung quality study, we first used a one-way ANOVA to determine if there was a period effect for the individual

Table 1
Water Content, Carbon, and Nitrogen Concentrations and C/N Ratio of Dung Applied to Grasslands During Two Different Seasons in Two Experiments

			Dung properties			
Experiment	Season	Dung type	Water content (%)	C_{conc} (g kg^{-1} dry matter)	N_{conc} (g kg^{-1} dry matter)	C/N ratio
Dung quantity	Dry season	Farm dung	84.8 ± 0.1a	377.3 ± 0.7a	16.2 ± 0.3a	23.3 ± 0.4a
	Transition period	Farm dung	84.1 ± 0.2a	368.8 ± 2.5a	16.2 ± 0.1a	22.8 ± 0.3a
Dung quality	Dry season	40% MER	71.9 ± 0.1a	390.8 ± 0.9b	11.4 ± 0.2a	34.3 ± 0.4c
		60% MER	72.5 ± 0.5a	398.8 ± 0.7c	11.4 ± 0.1a	35.1 ± 0.5c
		100% MER	75.8 ± 1.0b	396.2 ± 0.2c	13.5 ± 0.3b	29.3 ± 0.5b
		Farm dung	81.8 ± 0.2c	349.4 ± 3.7a	16.5 ± 0.2c	21.1 ± 0.1a
	Wet season	40% MER	71.0 ± 0.1a	403.3 ± 1.0b	9.7 ± 0.1a	41.4 ± 0.3c
		60% MER	73.6 ± 0.3b	405.1 ± 2.9b	9.9 ± 0.5a	41.0 ± 1.9c
		100% MER	75.6 ± 0.8c	405.8 ± 0.2b	11.6 ± 0.2b	35.1 ± 0.5b
		Farm dung	81.1 ± 0.9d	381.7 ± 0.8a	16.4 ± 0.4c	23.3 ± 0.6a

Note. Values are mean ± standard deviation (n = 3). Farm dung was obtained from pasture fed cattle (MER 130–140%); MER: maintenance energy requirements. Different lowercase letters indicate significant differences within columns for each season ($P < 0.05$).

control plots. Following the results that no period effect was found for the control plots, we decided to analyze the gross and net cumulative fluxes using a two-way ANOVA with dung type as a fixed factor and season (wet or dry) as a random factor. Residuals were tested for normality using Levene's test and where appropriate, the flux data were either square-root or log-transformed to satisfy model assumptions. Where the ANOVA was significant ($P < 0.05$), differences among treatments were determined using Tukey's HSD test. The t tests and one-way ANOVA calculations were done in SPSS 8.0 (SPSS Inc. Chicago, IL, USA), while the two-way ANOVA and multiple comparisons were done using R v3.4.3 (R core team, 2017).

3. Results

3.1. Chemical and Physical Properties of Dung

Water, C, and N contents of the farm dung used in the dung quantity experiments were similar for the dry and wet season, with a C/N ratio in a range of 22.8–23.3 (Table 1). In contrast, dung properties used in the dung quality experiment varied substantially (Table 1). Water content increased with increasing feed supply ranging from 71.9 to 81.8% during the dry season and from 71.0 to 81.1% during the wet season (Table 1). The C/N ratio was widest (34 to 41) for dung obtained from cattle fed at 40 or 60% of MER, and narrowest (21 to 23) for dung obtained from cattle allowed to freely range on the ILRI farm. Dung quality also depended on season, with C/N ratios of dung being less variable during the dry season compared to the wet season (Table 1). In 50% of all cases dung added to rangeland plots disappeared or was fully mixed in soil due to the activity of termites. For the other cases dry matter and C and N concentration of the dung did not change significantly over the 4- to 5-week observation period.

3.2. Effect of Dung Quantity on GHG Fluxes

Although measurement periods were defined as wet and dry seasons based on long-term climate observations for Nairobi, it should be noted that occasional rains also occurred during August 2016 (i.e., the dry season). Conversely, rains were less frequent and less intense during December 2016 (i.e., the wet season) compared to the long-term mean (Figure 1).

During the 2016 dry season cumulative CO_2 emissions from the 0.5-kg dung treatment were approximately equal to those from the 1.0-kg dung treatment. Respiratory fluxes showed a strong temporal variability depending on air temperature (e.g., diurnal variations) and increased following rainfall events toward the end of March and beginning of April 2016 (Figure 2). Application of fresh dung on grassland resulted in a pulse of CH_4 emissions that lasted a few days before decreasing to background values within 6 to 10 days after application (Figure 2). Soil N_2O fluxes from grasslands were in the range of −3 to 17 μg N_2O-N m^{-2} h^{-1} and were only marginally stimulated by the addition of dung. The highest soil N_2O fluxes (17 μg N_2O-N m^{-2} h^{-1}) were observed following rainfall events (Figure 2). Cumulative N_2O emissions from grasslands plots to which 0.5 or 1.0 kg of dung was added were similar to the cumulative N_2O fluxes of the control plots (Table 2).

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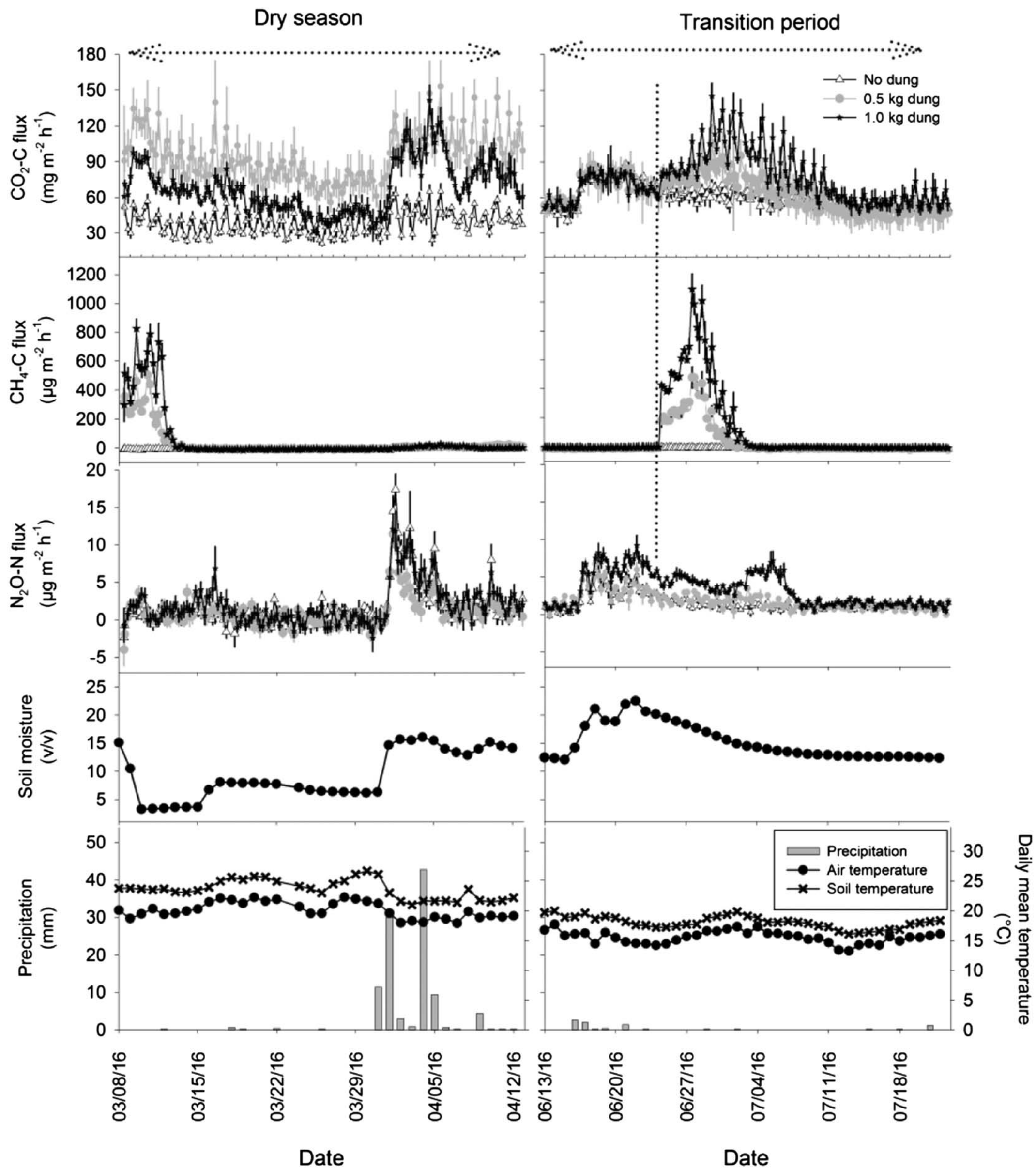

Figure 2. Dynamics of CO_2-C, CH_4-C, and N_2O-N fluxes as affected by additions of different amounts of farmyard dung to grassland. The lower panels show the observed temporal dynamics of mean daily soil moisture (0.05-m depth), soil temperature (0.05-m depth), air temperature, and the daily sum of precipitation as observed at a climate station immediately adjacent to the study site. Each flux value represents the mean of three chambers (±SE), with fluxes being observed in 6-hr time intervals. The dotted lines indicate the timing of dung applications. To note: During the dry season experiment, no premeasurements are available.

During the second measurement period (i.e., the transition between dry and wet season) CO_2 fluxes did not differ between the three treatment prior to dung application; however, mean CH_4 uptake prior to application was slightly higher ($P < 0.001$) in the plots that were receiving 0.5-kg farm dung (fluxes were -7.1 ± 0.4, -11.7 ± 0.4, and -6.4 ± 0.2 μg CH_4-C $m^{-2}\ h^{-1}$ for plots used as control and 0.5- and 1.0-kg farm dung, respectively). Also, mean N_2O fluxes prior to dung application were slightly higher ($P < 0.001$) in the chambers that would receive 1.0-kg farm dung (mean flux rates of 2.73 ± 0.27, 2.91 ± 0.24, and 4.73 ± 0.40 μg N_2O-N $m^{-2}\ h^{-1}$ for the control and 0.5- and 1.0-kg farm dung, respectively).

Across both periods, control plots that did not receive any dung additions continued to act as moderate sinks for atmospheric CH_4 (range: -1.8 to -15.3 μg CH_4-C $m^{-2}\ h^{-1}$). Peak CH_4 fluxes from the plots that received

Table 2
Cumulative GHG Emission and Net Cumulative GHG Emissions Over 29 Days as Affected by Addition of Different Amounts of Cattle Dung to Grassland

Period	Treatment	Cumulative emissions Mg CH_4–C m^{-2}	Net cumulative emissions Mg CH_4–C kg^{-1} DM	Cumulative emissions g CO_2–C m^{-2}	Net cumulative emissions g CO_2–C kg^{-1} DM	Cumulative emissions Mg N_2O–N m^{-2}	Net cumulative emissions Mg N_2O–N kg^{-1} DM
Dry season	Control, no dung	−4.4 ± 1.6a	--	25.6 ± 6.3	--	1.25 ± 0.04a	--
	0.5-kg dung	24.2 ± 10.7b	93.9 ± 34.1	59.6 ± 34.9	111 ± 121	0.76 ± 0.58a	−1.63 ± 1.94
	1.0-kg dung	46.8 ± 20.2b	84.0 ± 34.9	45.4 ± 9.4	32 ± 19	1.27 ± 1.09a	0.03 ± 1.83
Transition period	Control, no dung	−6.22 ± 4.18a	--	39.2 ± 6.5	--	0.82 ± 0.21a	--
	0.5-kg dung	32.6 ± 16.1b	126.0 ± 65.5	42.7 ± 21.5	11.1 ± 49.2	0.93 ± 0.37a	0.35 ± 1.83
	1.0-kg dung	91.6 ± 25.8c	154.1 ± 36.4	53.5 ± 12.7	22.5 ± 15.4	1.94 ± 0.24b	1.76 ± 0.14

Note. Values are mean ± standard deviation (n = 3); different lowercase letters indicate significant differences between treatments during the same period ($P < 0.05$).

1.0 kg of dung were roughly twice as high as the fluxes from plots that received 0.5 kg of dung (826 and 1,089 μg CH_4-C m^{-2} h^{-1} for the 1.0-kg plots versus 504 and 477 μg CH_4-C m^{-2} h^{-1} for the 0.5-kg dung plots during the dry and transition periods, respectively). Net cumulative emissions on a dry matter basis over the 29-day period were similar during the dry season (P = 0.745) for the two quantities of dung added (93.9 and 84.0 mg CH_4-C kg^{-1} dry matter for the 0.5- and 1.0-kg dung additions, respectively) and similar during the transition period (P = 0.551; 126.0 and 154.1 mg CH_4-C kg^{-1} dry matter for the 0.5- and 1.0-kg dung addition, respectively; Table 2).

3.3. Effect of Dung Quality on GHG Fluxes

Similar to the observations made during the dung quantity experiment, CO_2 fluxes measured during the dung quality experiment increased by 9 to 132% for the dung amendments when compared to the control plots. However, due to high temporal and spatial variability (Figure 3), these differences were not significant.

Grassland soils without dung amendment acted as net sinks of atmospheric CH_4 (range: −19.0 to 2.5 μg CH_4-C m^{-2} h^{-1}). Addition of fresh dung resulted in a short (2 to 7 days) pulse of CH_4 emissions, with the highest peak occurring immediately following application of farm dung in both seasons (Figure 3). Cumulative CH_4 emissions from dung taken from the adjacent animal feeding trials (40, 60, and 100% MER) were significantly lower ($P < 0.001$) than emissions from the farm dung during both seasons (Table 3). Although CH_4 fluxes appeared to vary across the seasons (Figure 3), there was no detectable difference between seasons (P = 0.483). The largest emission peak (>1,600 μg CH_4-C m^{-2} h^{-1}) was observed after application of the farm dung to the grassland plot, while the lowest peak (31 μg CH_4-C m^{-2} h^{-1}) was observed from a plot that received dung from cattle fed at 40% MER (Figure 3). In summary, CH_4 emissions from the dung taken from the feed trial (40, 60, and 100% MER) did not differ during the wet season (14 October to 15 November 2016 and 11 December 2016 to 8 January 2017). However, during the dry season, CH_4 emissions from the dung taken from animals fed at 100% MER were higher than emissions from dung taken from cattle fed at 40% and 60% MER.

Dung application to grassland soils did not affect N_2O fluxes, as the fluxes appeared to be related predominantly to rainfall events (Figure 3). Even following rainfall events no significant differences in soil N_2O emissions between control and dung amended plots were found.

4. Discussion

Urine and dung droppings on pastures are regarded as "hot spots" of GHG emissions (Cai et al., 2014). Current estimates assume that emissions due to manure management, which include CH_4 emissions from dung and urine patches on rangelands, represent approximately 10% of total non-CO_2 emissions form livestock production system and 33% of N_2O emissions from agricultural activities globally (Bogner et al., 2008; Herrero et al., 2013; Kelly et al., 2016). Although GHG emissions from the livestock sector are the dominant anthropogenic GHG emission source for many countries in SSA (Tubiello et al., 2013), hardly any measurements on GHG emissions from dung patches are available for this region. Most livestock production in SSA is by

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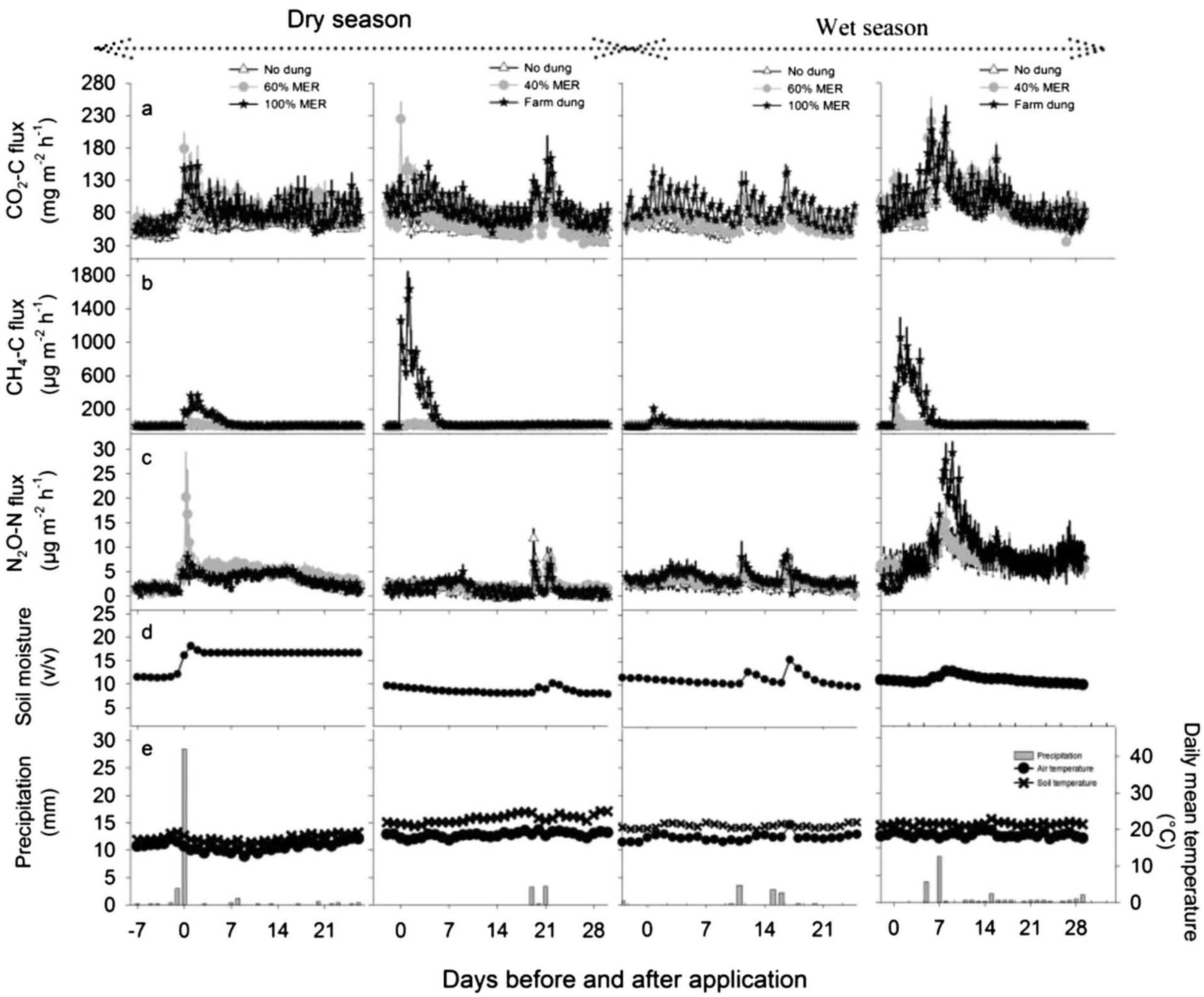

Figure 3. Dynamics of (a) CO_2-C, (b) CH_4-C, and (c) N_2O-N fluxes from grassland soils to which dung of different quality was added (control: no dung; dung from cattle fed at 40, 60, or 100% MER and farm dung). The lower panels show the observed temporal dynamics of (d) mean daily soil moisture (0.05-m depth), (e) soil temperature (0.05-m depth), air temperature, and the daily sum of precipitation as observed at a directly adjacent climate station. Each flux value represents the mean of three chambers (±SE) over a 6-hr period.

smallholder farmers, and in response to varying ecological and socio-economic conditions, livestock production is highly diverse (Herrero et al., 2013). Further, livestock production in SSA is largely dependent on locally produced feed (i.e., rangelands and pastures and crop residues) that is often insufficiently available in quantity and quality, due to the seasonality (Assan, 2014). As a consequence the quantity and quality of the excreted dung also varies, which has subsequent effects on GHG emissions from dung patches.

4.1. Effect of Dung Quantity on GHG Emissions

For most of SSA, the IPCC guidelines suggest using a constant EF where 2% of N applied as excreta to grazing land is lost as N_2O regardless of the excreta type, that is, dung or urine, and the excreta property such as mass or quality of the dung. However, in a global meta-analysis on the response of soil N_2O emissions following N fertilizer amendments to soil, Shcherbak et al. (2014) found that N_2O emissions from soils increase exponentially with increasing rates of fertilization. Larger dung patches would likely provide more easily accessible N and C substrates to the topsoil, thereby stimulating microbial activity in the topsoil (Sordi et al., 2014). Also, larger dung patches might retain more water and remain anaerobic for a longer time, thus promoting greater production of CH_4 and N_2O by methanogenesis and closely coupled nitrification and denitrification. Therefore, N_2O and potentially CH_4 EF from dung patches could also increase with increasing dung quantities dropped on rangelands.

Table 3
Cumulative GHG Emission and Net Cumulative Emission From Grassland Plots Receiving Additions of Dung From Cattle Exposed to Different Feeding Regimes Over an Observation Period of 25 Days

Season	Treatment	Cumulative emissions Mg CH_4–C m^{-2}	Net cumulative emissions Mg CH_4–C kg^{-1} DM	Cumulative emissions g CO_2–C m^{-2}	Net cumulative emissions g CO_2–C kg^{-1} DM	Cumulative emissions Mg N_2O–N m^{-2}	Net cumulative emissions Mg N_2O–N kg^{-1} DM
Dry season	Control, no dung	−5.5 ± 3.7a	--	40.6 ± 11.0	--	1.34 ± 1.00ab	--
	40% MER	−1.2 ± 3.4a	3.7 ± 3.3a	46.5 ± 8.6	4.4 ± 20.4	1.18 ± 0.99a	0.08 ± 0.30
	60% MER	−0.4 ± 3.9a	4.8 ± 8.9a	52.9 ± 22.1	11.9 ± 12.7	3.07 ± 0.76b	1.34 ± 1.23
	100% MER	25.1 ± 1.7b	31.7 ± 6.8b	51.6 ± 21.2	12.3 ± 14.0	2.26 ± 1.05ab	0.69 ± 0.55
	Farm dung	82.1 ± 15.2c	120.2 ± 21.3c	52.6 ± 9.9	15.2 ± 31.3	1.05 ± 1.10a	−0.06 ± 0.54
Wet season	Control, no dung	−5.6 ± 5.1a	--	45.8 ± 15.6	--	3.10 ± 3.21	--
	40% MER	5.0 ± 4.8a	11.0 ± 10.3a	60.9 ± 16.2	4.2 ± 6.8	4.42 ± 3.63	−0.27 ± 0.81
	60% MER	2.8 ± 3.8a	5.9 ± 2.4a	40.2 ± 6.5	5.2 ± 14.6	1.85 ± 1.21	0.37 ± 0.85
	100% MER	4.9 ± 9.5a	8.6 ± 8.5a	50.9 ± 2.2	15.5 ± 11.0	2.23 ± 1.64	0.79 ± 0.52
	Farm dung	73.5 ± 27.0b	107.7 ± 39.5b	61.4 ± 19.8	7.3 ± 20.1	5.85 ± 3.87	1.48 ± 2.58

Note. Values are mean ± standard deviation; different lowercase letters indicate significant differences between the treatments in the same period ($P < 0.05$). Farm dung was obtained from pasture fed cattle (MER 130–140%); MER: maintenance energy requirements.
Important to note is that no period effect was found for the different fluxes measured in control plots during different periods ($P < 0.05$).

As estimates of dung patch mass have been found to vary from 1 to over 3 kg for cattle with a live weight of 450–600 kg (Flessa et al., 1996; Mazzetto et al., 2014; Sordi et al., 2014), it is possible that different EFs need to be determined for different mass patches. The steers used in this study (183-kg live weight) were found to defecate 7 to 10 times per day, with an average defecation depositing between 0.6 and 0.9-kg dung (fresh weight; Table 4). Compared to the above-mentioned studies in Europe, dung patch mass and weights from our study were much smaller, which can be attributed to the lower quality diet, the reduced feed supply and intake, and generally lower livestock live weights in the tropics (Goopy et al., 2018).

Our results, however, indicate CH_4 emissions from dung patches scaled linearly with the quantity of dung applied to the rangeland for both the dry and wet seasons, contrary to a study in Brazil (tropic, 22°46′S, 43°41′W, 33 m asl) that found that the length of the CH_4 emission pulse of freshly dropped dung was positively correlated with weight (Cardoso et al., 2016). This difference from the previous study might be related to the high altitude of Nairobi (~1,850 m asl), which results in relatively low humidity and higher solar radiation causing quick drying of dung irrespective of the weight of the dung. The N_2O EF was similarly not affected by dung weight, consistent with a study undertaken in Brazil by Sordi et al. (2014).

As opaque chambers were used in our study, only respiratory CO_2 fluxes (i.e., the sum of heterotrophic respiration from soils and dung and plant autotrophic respiration) were measured. The observed slight increment in respiratory CO_2 fluxes following dung application is most likely largely derived from the decomposition of easily degradable C compound in the dung as was also described by Ma et al. (2006) on short-term effects of sheep feces droppings on ecosystem respiratory CO_2 fluxes in a typical grassland of Inner Mongolia. The rather minor response of CO_2 fluxes to dung application in our study might also be a result of the formation of a crust within hours of application due to environmental conditions (low humidity and high solar radiation) and/or due to the poor quality of the dung.

Table 4
Number of Dung Excretions Per Day and Total Daily Dung Weight as Recorded During the Two Days of Observations for A Feed Quantity/Quality Trial at the International Livestock Research Institute, Nairobi, Kenya

	40% MER		60% MER		100% MER		Farm cattle	
Period	Number of excretions per day	Total daily fresh/dry weight (g)	Number of excretions per day	Total daily fresh/dry weight (g)	Number of excretions per day	Total daily fresh/dry weight (g)	Number of excretions per day	Total daily fresh/dry weight (g)
Day 1	6	3,563/1,018	7	5,027/1,360	10	7,577/1,841	n.a.	n.a.
Day 2	7	4,078/1,165	8	5,735/1,552	9	8,485/2,061	n.a.	n.a.

n.a.: not available.

In our study, the season (dry versus wet season) had no measurable effect on CH_4 emissions. A previous study found that 80% of total CH_4 emissions occur during the first week after dung application (Nichols et al., 2016), which is consistent with the current study. Even rainfall events following crust formation were not able to revive CH_4 emission. These same observations were also found in studies carried out in Europe and elsewhere (Holter, 1997; Mazzetto et al., 2014).

4.2. Effect of Dung Quality on GHG Emissions From Dung Patches

4.2.1. Feed Quality and N Content of Dung

The amount of N excreted by grazing cattle depends on the protein content of the diet (Lessa et al., 2014). Luo et al. (2014) found that dung from sheep fed either forage rape or ryegrass had N concentrations of 24% versus 8%, respectively. The effect of feed quality on dung N concentrations has also been observed in other studies (Korir et al., 2016; Sørensen et al., 2003; van Vliet et al., 2007). In our study the N content (% dry matter) of dung from cattle fed at different MER levels ranged from 0.97 to 1.65% (Table 1). Thus, the N content of the dung in our study was approximately half of the N concentrations found for cattle dung in the UK (1.6–2.9%) from cattle grazing unfertilized grass, fertilized grass or clover, and cows fed a mix of silage and concentrates (Jarvis et al., 1995). Comparable dung N concentrations to those found in our study were found by Rufino et al. (2006), who concluded that the N content in livestock dung in tropical Africa might be as low as one third of that found for temperate regions, mainly caused by the poor-quality diets.

4.2.2. Dung Quality and GHG Emissions

In agreement with our hypothesis we found that dung from cattle-fed diets below 100% MER emitted less CH_4 than dung excreted by pasture grazed cattle ($P < 0.001$). As cattle in SSA are regularly fed below their MER, caused by low quantity and quality feeds especially during dry periods or droughts, our findings are important for calculating GHG emission inventories throughout much of the arid and semiarid regions in SSA.

The short-term pulses of CH_4 immediately after dung deposition were partly due to the release of CH_4 of enteric origin embedded in the dung along with CH_4 production in the fresh dung, as fresh dung would still host a vital methanogenic population in an anaerobic environment supplied with highly labile organic C (Maljanen et al., 2012; Nichols et al., 2016; Saggar et al., 2004). In our study, the freshly collected dung from the animals that were allowed to graze freely emitted much more CH_4 than the dung obtained from the cattle fed at 100, 60, and 40% MER. However, even the largest CH_4 peak after dung application in our study was only 1.6 mg CH_4-C m^{-2} h^{-1}, much lower than the peaks observed in studies carried out in, for example, Japan (from 3.3 to 13.7 mg CH_4-C m^{-2} h^{-1}) or Germany (30 mg CH_4-C m^{-2} h^{-1}; Flessa et al., 1996; Mori & Hojito, 2015). Studies carried out in the UK (Jarvis et al., 1995), Japan (Mori & Hojito, 2015), and Denmark (Holter, 1997) as well as this study observed a strong trend to increasing CH_4 emission with decreasing C/N ratios. Including such a relationship in international GHG reporting on dung CH_4 emissions might be useful to better account for the observed, comparable low in magnitude, pulse of CH_4 emissions from freshly excreted dung in SSA countries.

Besides the lower N concentrations in the dung from the 40 and 60% MER feeding, we also noted differences in the water content, which was lowest for the dung from the animals fed at 40% MER (Table 1). Higher water content in dung reduces gas diffusion and supports maintenance of anaerobic condition for longer time periods prolonging methanogenesis (Jones et al., 2005). In our study, rainfall during the first week following dung amendment likely delayed crust formation and prolonged anaerobic conditions in the dung patches resulting in CH_4 production (Mazzetto et al., 2014; Yamulki et al., 1999). Due to distinct seasonal variability in rainfalls in our study region (dry and wet season) and the rather minor changes in air temperatures across the year, rainfall would likely be of greater importance for emissions compared to temperate regions.

Contrary to our expectations, there was no effect of diet and associated dung quality on N_2O or CO_2 emissions from dung patches. However, rainfall clearly stimulated N_2O emissions in all plots, including the control plots, which may have masked any dung effect. As a major driver of N_2O emissions, soil moisture is known to regulate soil oxygen concentrations and nutrient availability (Butterbach-Bahl et al., 2013). This is particularly the case as rainfall reduces soil air diffusion, thus promoting the establishment of soil anaerobic conditions. On the other hand, rainfall also promotes the mobility of NO_3 in the soil matrix. Both effects are essential for denitrification and for the production of N_2O during denitrification (Butterbach-Bahl et al., 2013). Still, this does not fully explain why we did not see an effect of dung additions on N_2O emissions even though

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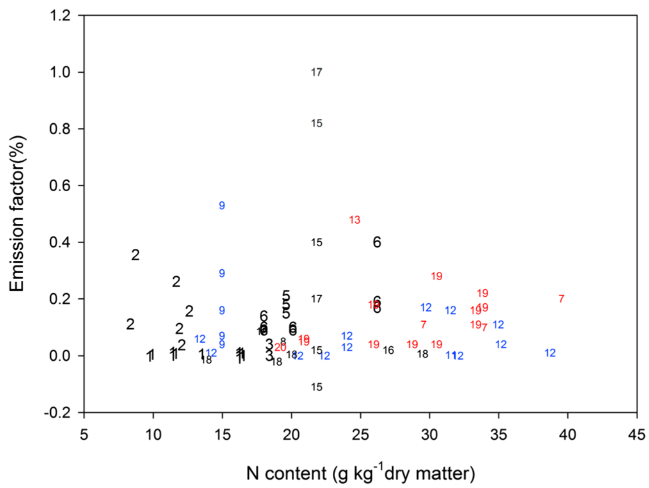

Figure 4. Relationship of the cattle dung N content with the N_2O emission factor in our (1) and previous studies (2–20). The numbers refer to individual studies as listed in Table 6. The numbers in bold and with increased font size refer to studies in tropical regions. The colors refer to the length of the measuring period: black (<90 days); blue (91–180 days); red (>180 days).

additional N, though mostly in organic form, was added to the pasture. We can only speculate that the rather high C/N ratio (21–41) and the low N concentration of the dung, and thus, the low quality of the dung used in our experiments compared to experiments done in Europe and North America (Bell et al., 2015; Rochette et al., 2014; Table 6 and Figures 4 and 5) did not create adequate conditions for denitrification. Our results are in line with the results presented by Pelster et al. (2016), who investigated N_2O EF from feces that were dropped on rangelands and found only a minor stimulating effect on N_2O emissions following dung deposition in Kenyan rangelands. The authors argued that fecal N needed to be mineralized before denitrification could occur (Pelster et al., 2016). In addition, the high amounts of C in the feces and the high C/N ratio likely caused rapid N immobilization, resulting in less available substrate (i.e., NO_3^-) for denitrification and subsequently reduced N_2O production (Pelster et al., 2012). For cool temperate climate conditions in New Zealand, Laubach et al. (2013) observed that approx. 12% of the deposited dung cattle N was volatilized in form of NH_3 within the first 10 days. Given the low air humidity levels and the intensive radiation at Nairobi, even higher NH_3 losses might occur, which might also explain why N_2O emissions were lower than expected. On the other hand, dung from livestock systems in New Zealand typically have a high total ammoniacal N content (Laubach et al., 2013), which does not hold true for the investigated dung in this study, so that the importance of NH_3 losses as a factor reducing N_2O emissions from dung patches in our study remains speculative.

4.3. CH_4 and N_2O Emission Factors

In our study, the EFs were calculated based on 25- or 29-day measurements and it can be argued that after such a short period dung is not yet fully mineralized. However, other studies conducted in tropical areas, such as the studies by Mazzetto et al. (2014) and Lessa et al. (2014) in Brazil or by Tully et al. (2017) and Pelster et al. (2016) in Kenya, show that the stimulating effect of dung deposition on rangelands for soil N_2O (and CH_4) emissions last only 2 to 3 weeks before diminishing and disappearing. This might be due to fast crust formation, losses of dung N along hydrological, and gaseous pathways and immobilization of N in organomineral complexes. In our study the fluxes reached background level approximately 10 days after dung deposition. Even following rainfall events, which generally stimulate soil N_2O emissions, no significant difference between control plots and plots receiving dung was observed after 10–14 days, and our measurements were still conducted for additional 2 weeks. This provides confidence that our calculated EFs are realistic, although due to the short measuring period, these EFs may be subject to a slight underestimation.

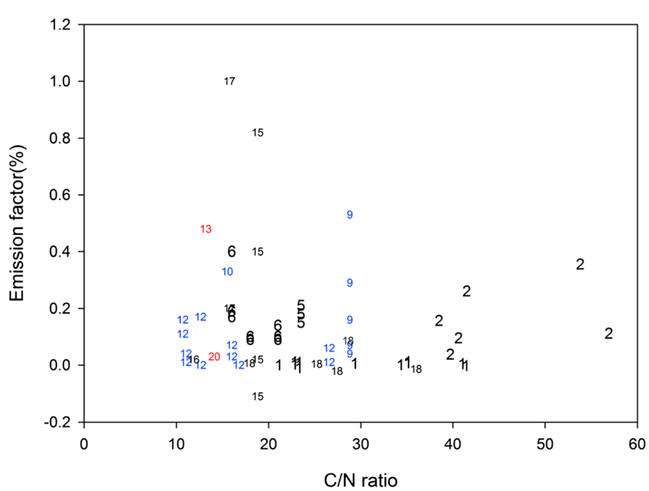

Figure 5. Relationship of the cattle dung C/N ratio with N_2O emission factors reported in our (1) and previous studies (2–20).The numbers refer to individual studies as listed in Table 6. The numbers in bold and with increased font size refer to studies in tropical regions. The colors refer to the length of the measuring period: black (<90 days); blue (91–180 days); red (>180 days).

In our study the EF for CH_4 emissions from fresh dung on rangelands ranged from 0.001 to 0.042% (Table 5), which was lower than a study in Japan (mean: 0.052%, range: 0.010–0.126%; Mori & Hojito, 2015), but in agreement with the EFs for dung deposits on Kenyan rangeland by Boran and Friesian cattle (mean: 0.04%, range: 0.01–0.08%; Pelster et al., 2016). These differences might be explained by the high C/N ratio of the dung in our study, which was confirmed by the strong negative linear relation between the C/N ratio and the CH_4 EF ($CH_4EF = -0.0018$ C/N ratio $+ 0.0705$, $n = 36$, $R^2 = 0.67$, $P < 0.05$). The importance of the dung C/N ratio for CH_4 emissions from dung patches was also highlighted by Pelster et al. (2016). However, based on our data, total CH_4 emissions from dung patches would amount to <100 g CH_4 head^{-1} yr^{-1}, which is small compared to annual CH_4 emissions from enteric fermentation in 1- to 2-year old steers of 30 kg CH_4 head^{-1} yr^{-1} in a study in Kenya (Goopy et al., 2018).

Table 5
CH_4 and N_2O Emission Factors From Dung Deposition to Rangeland in This Study (Based on 25- to 29-Day Observation Period)

Experiment	Period	Treatment	EF	
			CH_4 EF (%)	N_2O EF (%)
Dung quantity	Dry season	0.5-kg farm dung	0.025 ± 0.009	−0.0101 ± 0.0120
		1.0-kg farm dung	0.022 ± 0.009	0.0002 ± 0.0113
	Transition period	0.5-kg farm dung	0.033 ± 0.017	0.0021 ± 0.0113
		1.0-kg farm dung	0.042 ± 0.010	0.0109 ± 0.0009
Dung quality	Dry season	40% MER	0.001 ± 0.001	0.0007 ± 0.0027
		60% MER	0.001 ± 0.002	0.0118 ± 0.0109
		100% MER	0.008 ± 0.002	0.0051 ± 0.0041
		Farm dung	0.034 ± 0.006	−0.0003 ± 0.0032
	Wet season	40% MER	0.003 ± 0.003	−0.0028 ± 0.0083
		60% MER	0.001 ± 0.001	0.0037 ± 0.0086
		100% MER	0.002 ± 0.002	0.0068 ± 0.0045
		Farm dung	0.028 ± 0.010	0.0090 ± 0.0158

Note. Values are mean ± standard deviation. Farm dung was obtained from pasture fed cattle (MER 130–140%); MER: maintenance energy requirements.

The dung N_2O EF in our study ranged from −0.01% to +0.01% (Table 5); that is, the dung essentially did not stimulate N_2O fluxes at all. EFs calculated here were even lower than the earlier study by Pelster et al. (2016), who estimated N_2O EF between 0.04 and 0.36% of applied N for dung deposited on a rangeland in Nairobi, Kenya. However, in the Pelster et al. (2016) study, calculations were based on manual static chamber measurements, with fluxes being determined daily or 2 to 3 times sampling per week, whereas here we measured gas fluxes >10 times per day. This is particularly important as it has been shown that automated soil GHG measurements are needed for calculating accurate emissions over several weeks (Barton et al., 2015). Even measurements frequencies of 2–3 times per week might finally result in an uncertainty of 50% due to high temporal variation of soil GHG fluxes.

Other studies have also found N_2O EFs from dung that were not different from "zero." For instance, studies in Japan (0.004%), China (0.02%), and Ireland (0.003%) showed very low EFs. In contrast, other studies did

Table 6
No., Reference, Year Published, Location, Climate Zone, Observation Period, N Content, C/N Ratio, and N_2O Emission Factors of Available Studies That Investigated N_2O Emissions From Cattle Dung Applied to Rangeland

No.	Reference	Year	Location	Climate zone	Observation period (days)	Number of flux measurements	N content (g kg^{-1} DW)	C/N ratio	N_2O emission factor (%)
1	Our study	2016	Kenya	Tropics	25–29	>10 times per day	9.7–16.5	21.1–41.4	−0.01-0.01
2	Pelster et al. (2016)	2016	Kenya	Tropics	28	13–15	8.4–12.6	38.5–56.9	0.04–0.36
3	Tully et al. (2017)	2017	Kenya	Tropics	60–63	23–26	18.4	--	0.0–0.04
4	Mazzetto et al. (2014)	2014	Brazil	Tropics	30	17	16.3	19.2	Negative
5	Cardoso et al. (2016)	2016	Brazil	Tropics	14–16	14–16	19.6	23.5	0.15–0.21
6	Sordi et al. (2014)	2014	Brazil	Tropics	90	11–13	18.0–26.2	16.0–21.0	0.10–0.45
7	Bell et al. (2015)	2015	UK	Temperate	365	28	29.6–39.5	--	0.10–0.20
8	Hoeft et al. (2012)	2012	Germany	Temperate	77	15	19.4	--	0.05
9	Yamulki et al. (1998)	1998	UK	Temperate	100	16–19	14.97	28.8	0.04–0.53
10	Wachendorf et al. (2008)	2008	Germany	Temperate	171	19	--	15.5	0.33
11	Hyde et al. (2016)	2016	Ireland	Temperate	180	31	31.5	--	0.003
12	Van der Weerden et al. (2011)	2011	New Zealand	Temperate	125–173	24–30	13.4–38.7	10.7–26.5	0.00–0.17
13	Li et al. (2016)	2016	New Zealand	Temperate	271	23	24.6	13.2	0.48
14	Kelly et al. (2016)	2016	Australia	Temperate	86–111	9–13	22–28	--	0.01–0.12
15	Cai et al. (2013)	2013	China	Temperate	15	6	--	18.8	−0.10-0.82
16	Cai et al. (2014)	2014	China	Temperate	61	19	27	11.9	0.02
17	Lin et al. (2009)	2009	China	Temperate	38, 48	15, 21	21.8	15.7	0.20–1.00
18	Mori and Hojito (2015)	2015	Japan	Temperate	78–85	21	13.8–29.5	17.9–36.0	−0.021-0.086
19	Rochette et al. (2014)	2014	Canada	Temperate	365	16–22	20.9–33.8	--	0.04–0.28
20	Thomas et al. (2017)	2017	Canada	Temperate	365	37	19.2	14.1	0.03

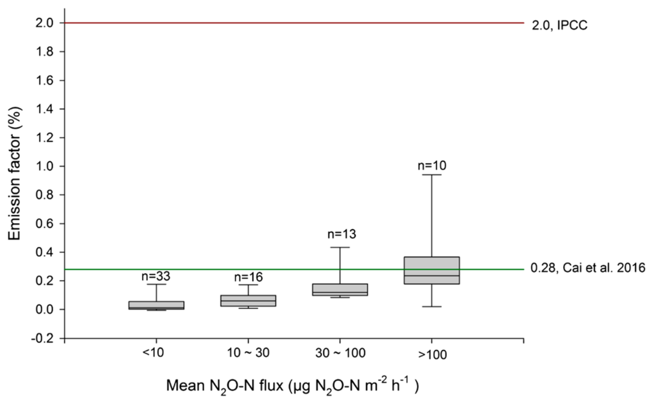

Figure 6. N_2O emission factors for different mean N_2O-N flux classes (0–10, 10–30, etc.) from dung applied to grasslands in previous studies as well as our studies. Note: The horizontal lines indicate the Intergovernmental Panel on Climate Change Tier 1 default value and the estimated emission factor for dung patches on grassland from a recent meta-analysis.

measure EFs up to 1.0% (e.g., Japan 0.86%, China 1.0%, and UK 0.53%, Table 6). All these studies suggest that the IPCC Tier 1 N_2O EF overestimates N_2O emissions from dung patches (Figure 6), which is consistent with the mean EF (0.28%) for cattle dung patches in the meta-analysis by Cai and Akiyama (2016).

Negative net cumulative N_2O emissions as in our study, that is, rangeland plots with dung emitting less N_2O than adjacent control plots, have also been observed in other studies that were carried out in temperate (Mori & Hojito, 2015) or tropical grassland (Mazzetto et al., 2014). This observation might be surprising but was mostly detected in studies where dung with low N contents and high C/N ratios was applied to grasslands (Figures 4 and 5). However, other factors such as rainfall events or extended dry periods during the observation period might also affect the magnitude of N_2O emission from dung patches. Therefore, there is still more research required to fully understand the underlying mechanism leading to N_2O emissions from dung patches being lower than these from adjacent grassland. One possible reason is that the organic matter from dung, with its high C/N ratio, leaches into the soil, subsequently provoking a net N immobilization in the underlying soil. This would reduce the amount of NO_3^- available for denitrification and N_2O production (Xia et al., 2017). Another explanation might be that the wide C/N ratio of the dung and likely of the leachates favors complete denitrification, that is, that N_2 is the sole end product of the denitrification process (Butterbach-Bahl et al., 2013). Nevertheless, from our work as well as from other work undertaken globally (Table 6), it becomes obvious that the default EF for N_2O emissions from cattle dung patches of 2% is too high and even the EF of 0.2% documented by Pelster et al. (2016) for Kenya may still be too high for many SSA countries, so large biases in national GHG inventories can be expected.

5. Conclusion

The N_2O and CH_4 EFs for dung patches from cattle applied to rangelands did not change with the mass of the dung patch indicating that a single EF for dung patches can be used regardless of the size. However, dung quality, which is related to diet quality, did largely influence CH_4 emissions, which could partly be attributed to the original dung water content, but could also be related to the differences in dung N content. Although diet did influence N concentrations in the dung, this did not cause any differences in N_2O fluxes, possibly because N concentrations of the dung were overall substantially lower than in other regions. The N_2O EF of cattle dung patches ranged from −0.01% to 0.01%, much lower than Tier 1 default of 2% (Eggleston et al., 2006) and lower even than a previous study in the same location (Pelster et al., 2016) confirming that regions with poor quality livestock feeds such as SSA should use country and livestock system specific N_2O EFs.

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