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Nitrous Oxide and Methane Fluxes from Urine and Dung Deposited on Kenyan Pastures

Katherine L. Tully,* Sheila Abwanda, Margaret Thiong'o, Paul M. Mutuo, and Todd S. Rosenstock

Abstract

Livestock keeping is ubiquitous in tropical Africa. Urine and dung from livestock release greenhouse gases (GHGs), such as nitrous oxide (N_2O) and methane (CH_4), to the atmosphere. However, the extent of GHG's impact is uncertain due to the lack of in situ measurements in the region. Here we measured N_2O and CH_4 emissions from cow urine and dung depositions in two Kenyan pastures that received different amounts of rainfall using static chambers across wet and dry seasons. Cumulative N_2O emissions were greater under dung+urine and urine-only patches ($P < 0.0001$), more than three times higher in the wet compared with the dry season ($P < 0.0001$), and higher in the farm receiving higher rainfall overall ($P < 0.0001$). Cumulative CH_4 emissions differed across treatments ($P = 0.012$), driven primarily by soil CH_4 uptake from the urine-only treatment. Cumulative N_2O emissions were positively related to N input rate in excreta. However, the relationship was linear during the dry season ($r^2 = 0.99$; $P = 0.001$) and exponential during the wet season ($r^2 = 0.99$; $P < 0.0001$). Nitrous oxide emission factors were 0.05% (dry season) and 0.18% (wet season) of N in urine and dung+urine, which is less than 10% of the IPCC Default Tier 1 emission factor of 2%. We predict that emissions from cattle urine in Kenya are approximately 1.7 Gg N_2O–N yr^{-1} (FAO estimates 11.9 Gg N_2O–N yr^{-1}). Our findings suggest that current estimates may overestimate the contribution of excreta to national GHG emissions and that emission factors from urine and dung need to account for agroecosystems with distinct wet and dry seasons.

Core Ideas

- Urine and dung deposited on pasture are a significant agricultural source of GHGs.
- N_2O emissions from African cattle excreta were higher in the wet than dry season.
- Urine and the combination of urine and dung had the highest N_2O fluxes across seasons.
- N_2O emissions were positively correlated with excreta N.
- N_2O emission factors from excreta were one-tenth that predicted by the IPCC.

URINE and dung deposited on pasture soils by grazing livestock are significant sources of the greenhouse gases (GHGs) nitrous oxide (N_2O) and methane (CH_4) to the atmosphere (Herrero et al., 2016). Excreta create patches of nutrient-rich soils, stimulating microbial activity and enhancing GHG emissions (Flessa et al., 1996; Moir et al., 2010). Despite the large potential contribution of livestock to climate change outcomes, little is known about GHG emissions from urine, dung, and manure (i.e., dung+urine) in general and specifically under tropical conditions.

Only four studies have quantified GHG emissions from manure under tropical conditions, and these studies report conflicting results. In the tropical region of Brazil, CH_4 emissions from excreta were substantially higher than in temperate conditions (e.g., The Netherlands) (van Groenigen et al., 2005), but N_2O emissions were low [0.2 and 0.7% of excreta N in Barneze et al. (2014) and Lessa et al. (2014), respectively] or undetectable (Mazzetto et al., 2014a). In a subhumid tropical region of Kenya, recent work on a research station reported lower N_2O and CH_4 emissions from livestock fed a legume diet than in temperate conditions (0.15% N in dung and 1.2% N in urine) (Pelster et al., 2016). Available data, therefore, suggest that emissions from manure deposited on tropical pastures may deviate substantially from expectations based on measurements in temperate systems and may poorly inform calculations and actions based on those data.

Environmental conditions are one plausible reason that emissions from manure in the tropics may differ from those in temperate systems. Temperature and rainfall affect the magnitude and rate of GHG emissions from urine and dung patches, with warm and moist conditions creating optimal environments for the production of N_2O and CH_4 (Wang et al., 2013; Maljanen et al., 2007; van der Weerden et al., 2011). Rainfall in the subhumid tropics tends to be seasonal, with pronounced wet and dry seasons. Unlike temperate zones, the subhumid tropics remain warm throughout the year. Warmer and wetter conditions in the tropics can lead to higher N_2O and CH_4 emissions due to greater microbial activity and saturated conditions (González-Avalos, 2001; Rochette et al., 2014). However, recent work in East

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Abbreviations: GC, gas chromatograph; GHG, greenhouse gas.

Africa suggests that N_2O emissions from manure and inorganic fertilizers are lower than average global estimates (Hickman et al., 2015; Pelster et al., 2016), indicating that weather is unlikely to be the only source of differences. Soil properties, pasture species composition, and livestock management all play key roles in determining GHG emissions from tropical pastures (Abalos et al., 2014; Hoeft et al., 2012).

The availability of N (Ball et al., 1997; Castaldi and Smith, 1998) and the availability of organic C (Ineson et al., 1998; Kaiser et al., 1998) control denitrification rates in soils. Therefore, low-nutrient-content feed, and hence the manure from cattle fed this diet, may also contribute to lower N_2O emissions. Cattle in sub-Saharan Africa often feed on native pasture, grasses, and rubbish growing on roadsides and shrubland, which typically has low digestibility and low N content (Rufino et al., 2006; Castellanos-Navarrete et al., 2015). The quantity and forms of N and C in excreta are primarily driven by dietary protein content, sugar content, and feed digestibility (Dijkstra et al., 2011; Merry et al., 2006; Rotz, 2004). Feed composition can affect the C/N ratio in excreta, which in turn affects N_2O and CH_4 emissions (Cardenas et al., 2007). There is also a tradeoff between C and N in terms of nutrient cycling in pastures. Low-quality grasses in African pastures may lead to high C/N in excreta (e.g., >25:1), which leads to immobilization of soil mineral N (Cadisch and Giller, 1997; Giller et al., 1997) and may increase manure C/N in the following season of pasture grasses, effectively amplifying this negative cycle (Haynes and Williams, 1993).

Livestock grazing on native pasture is widespread across tropical Africa. Mixed crop–livestock agriculture covers 6 million km^2 of the continent, roughly an area equivalent to two-thirds the size of Europe or the United States, and supports approximately 123 million cattle (Thornton and Herrero, 2014). These cattle produce significant amounts of manure and hence GHGs. In East Africa, it is estimated that manure deposited on pastures contributes 62% of annual N_2O emissions (FAO, 2016). This study enhances our understanding of N_2O emissions from dung and urine depositions to improve estimates and national GHG inventories (e.g., NAMA [2016]). The aim of this study was to reduce the uncertainty of N_2O and CH_4 fluxes from cattle manure deposited on pastures in tropical Africa. We measured GHG fluxes derived from dung, urine, and dung+urine in controlled experiment plots on two farmer's fields during wet and dry seasons. We hypothesized that the magnitude and pattern of fluxes would differ between treatments, seasons, and sites due to contrasting weather and soil conditions. Our study is the first to report emissions from dung, urine, and dung+urine deposited on farmers' pastures based on samples collected from animals under conventional production conditions in Africa.

Materials and Methods

Site Description

The study was conducted in two farms in Kaptumo, Nandi County, in western Kenya (00°007′ N, 35°029′ E). Sites are 1800 to 2000 m above sea level with a mean annual temperature of 19.6°C and annual total precipitation of 2000 mm, distributed across two rainy seasons. Cumulative precipitation in Kaptumo town was 1795 mm in 2013 and 1541 mm in 2014. Over the study period (October 2013–March 2014), Farm 1 received 954 mm of precipitation, and Farm 2 received 332 mm of precipitation (Fig. 1). The "long rains" occur between mid-March and mid-June, and the "short rains" occur between mid-October and mid-December, although precipitation is possible throughout the year (Rosenstock et al., 2016).

Soils are well-drained, friable clays with a thick humic topsoil derived from biotite gneiss parent material, primarily classified as Humic Nitosols (Dewitte et al., 2013). Topsoils (0–20 cm) are acidic (pH 5.45) and are comprised of 20% clay, 17% silt, and 63% sand (Table 1). The region was previously a tropical forest. Kaptumo is typical of smallholder farming landscapes in western Kenya and is dominated by mixed crop–tree–livestock systems. Nearly every farm in the region produces milk for home consumption and some for sale depending on the season. In the predominant dairy production system, cattle graze in paddocks during the day and return to a shed or small pen in the evening. Farms also tend to produce a mixture of maize, sorghum, beans, and vegetables for household consumption and tea for sale (Zagst, 2011).

In October 2013, we established a randomized complete block design with four replicates on two farms approximately 7 km from each other. At each farm, four 5 m by 5 m blocks were selected, and five pairs of static chambers were installed in a circle, ~2 m apart from one another, within each block (20 pairs of chambers per farm) for measuring gas fluxes. Each pair of chambers was randomly assigned to one of five treatments: cow dung, cow urine, a mixture of cow urine and dung, rain water, and a no-addition control. One plastic chamber was used for gas sampling and one for soil sampling. Because these are working farms, the two focal pastures were grazed the year before the study. However, the sites were protected from livestock access

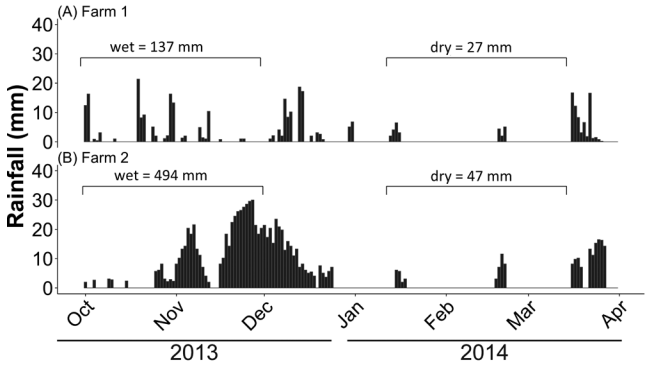

Fig. 1. Trends in rainfall in (A) Farm 1 and (B) Farm 2 across the study period. Brackets represent sampling periods in the wet and dry seasons and the cumulative rainfall over that period.

Table 1. Initial soil characteristics of the two farms in Kaptumo, Kenya.

	Farm 1		Farm 2	
	0–20 cm	20–50 cm	0–20 cm	20–50 cm
pH	5.83	5.74	5.07	5.17
Sand, %	60.00	60.30	66.30	58.90
Silt, %	14.90	6.55	18.50	8.92
Clay, %	21.40	30.70	18.70	34.50
Total N, %	0.46	0.30	0.26	0.34
Total C, %	4.54	2.95	2.66	3.49
Bulk density, g cm^{-3}	0.85	nd†	0.79	nd

† Not determined.

throughout the study. At each site, rainfall was collected via a manual gauge and recorded daily after being measured with a graduated cylinder.

N Analysis in Soil, Urine, and Dung

Before the start of the experiment, we collected soils (0–20 and 20–50 cm) from each block in each pasture. These baseline soils were analyzed for texture using a Bouyoucos hydrometer after pretreatment with H_2O_2 to remove organic matter. We determined pH in a 1:2 (soil/water) suspension. Total N and soil organic C were analyzed by elemental analysis (Flash 2000 series, Thermo Scientific) (Table 1). Soils were collected weekly after the initiation of the experiment (0–5 cm) during the dry season (13 Jan.–7 Mar. 2014) from the soil sampling chamber. Soil cores were composited by treatment at each farm (10 samples per collection date) and kept on ice until transported to the laboratory where they were refrigerated until extraction 1 to 2 d later. Due to logistical issues, soils were not sampled during the wet season. Soils were extracted using 2 M KCl at a 1:10 soil/solution ratio, and extracts were immediately frozen and analyzed within 2 wk. On the day of collection, field-moist soils were weighed, dried for 48 h at 105°C, and reweighed to determine water content. Nitrate-N (NO_3^-–N) in the KCl extracts was analyzed on a photometric analyzer (Aquakem 200, Thermo Scientific). Ammonium-N (NH_4^+–N) is not reported due to high NH_4 reported in blanks, which led us to believe that the reagents were contaminated.

Fresh urine was collected by gently stroking the escutcheon of approximately 25 cows of various ages, including both lactating and nonlactating animals, of Boran, Freisan and cross-breeds. Cows were selected from farms on and surrounding the experimental plots. Diets of animals were not manipulated before urine and dung collection, so the samples represented typical nutrient content of urine and dung in the region at the time of data collection. Samples were thoroughly mixed to form a composite sample across animals. Subsamples (~0.10 g by weight) of composited urine and rain water were digested to quantify total N using a modified Kjeldhal digestion and analyzed on an ultraviolet spectrophotometer (Thermo Scientific, Helios Delta). Subsamples of dry dung were analyzed for total C and N using elemental analysis (Flash 2000, Thermo Scientific). Nitrogen concentrations in urine, dung, and rain water are reported in Table 2.

Application of Urine and Dung

An average cow produces 2 to 3 L of urine over 0.2 m^2 per urination (Haynes and Williams, 1993). Our chambers were 0.0936 m^2; therefore, we used 1.2 L of urine per chamber. Urine was applied the same day as collection and kept in a cooler while collections were ongoing. Fresh dung was collected from cattle pens and grass paddocks, and 500 g was applied inside the chambers. The dung+urine treatments received 500 g dung followed by 1.2 L of urine per chamber. Rain water was collected in buckets from the household reservoir, and then 1.2 L was distributed into each chamber. Treatments were applied at the beginning of the wet season (1 Oct. 2013) and at the beginning of the dry season (14 Jan. 2014). Chambers were not covered between sampling periods, which exposed them to incident rainfall. We did not consider this as a source of N input to the experiment because we estimate that this contributed less than 0.9 g N m^{-2} and because all chambers were exposed to the same rainfall. Thus, N treatments included 0 (control), 0.2 (rain water), 240 (dung), 1066 (urine), and 1306 (dung+urine) kg N ha^{-1} per season (Table 2). Chambers were removed after the last sample collection in the wet season and reinstalled at the beginning of the dry season in a new location in the same block that did not receive a treatment in the wet season.

Flux Measurements

Gas samples were collected daily between 10 am and 12 pm from 1 Oct. to 29 Nov. 2013 (wet season) and from 10 Jan. to 13 Mar. 2014 (dry season). Gas samples were collected using a vented, static chamber technique (Parkin and Venterea, 2010). Each chamber consisted of a lid (27 L × 37.2 W × 12.5 H cm) and a base (27 L × 37.2 W × 10 H cm). The lid formed an airtight seal with foam between the two chambers pieces held together with clamps. Chamber bases were installed at least 7 cm into the soil 24 h before the first measurement. Chambers were equipped with 50-cm vents (2.5 cm diameter), thermometers to measure internal temperature, and gas sampling ports.

During each sampling event, chambers were closed for 30 min, and four samples were taken at 10-min intervals (0, 10, 20, and 30 min) from each chamber. Gas samples were collected by 60-mL propylene syringes with Luer locks and immediately transferred into 10- or 20-mL glass vials fitted with crimp seals. A needle was inserted into each vial, and each vial was flushed with 30 mL of sample. The needle was removed, and the remaining sample (~30 mL) was forced into the vial, causing it to over-pressurize and reduce the potential for contamination with ambient air.

We used a gas chromatograph (GC) (model 8610C, SRI Institute) with a ^{63}Ni electron capture detector to measure N_2O concentrations. The GC was operated with Haysep D packed columns with oven temperature of 65°C and flow rates of 25 mL min^{-1} on both flame ion detector and electron capture detector lines. Carbon dioxide was measured simultaneously by using a methanizer attached to the flame ion detector line (results not reported). Gas concentrations were calculated based on the marginal concentration of peak areas measured by the GC derived from calibration gases run four times each day. We cleaned the GC output data by plotting four CO_2 values per chamber (time 1 to time 4). Carbon dioxide concentrations are typically linear and less variable than N_2O, and, because they were run

Table 2. Urine, dung (dry weight), and rain water N concentrations and calculated mean emission factors by season in two farms in Kaptumo, Kenya.

	N concentration		N input	Emission factor	
				Dry season	Wet season
	%	g L^{-1}	g m^{-2}	%	
Dung	1.86		24	0.04	0.00
Urine		8.81	109	0.05	0.21
Dung+urine			133	0.05	0.15
Rain water		0.0017	0.21	–	–

simultaneously with N_2O, we used them as an initial check on the quality of the GC results. A chamber was excluded from the dataset where the r^2 was <0.8. A chamber with an r^2 of ≥0.8 was excluded if one of the points was five times higher or lower than the other three. We assumed the values were erroneous due to either contamination or leakage.

Nitrous oxide and CH_4–C concentrations were then converted to mass per volume using the ideal gas law and measured chamber volume, internal chamber air temperature, and atmospheric pressure at the site (Butterbach-Bahl et al., 2011). Fluxes were calculated using linear regression of gas concentrations versus time. Fluxes were set to zero if the flux was below the minimum detection limit of the measurement system calculated according to the approach described by Parkin et al. (2012).

Data Analysis

Cumulative gas fluxes from each treatment were quantified from mean fluxes based on wet season (1 Oct.–29 Nov. 2013; 60 d) and dry season (10 Jan.–13 Mar. 2014; 63 d). Cumulative estimates of fluxes were calculated based on the mean flux of four chambers with the same treatment at each plot. Data were interpolated linearly, integrating area under curve between measured fluxes throughout the sampling campaign.

Nitrous oxide emission factors for excreta treatments were calculated by subtracting N_2O–N flux (in mg m^{-2}) in the control plot from the flux in the treatments and then dividing by the total N applied by the dung, urine, or dung+urine.

For comparison to our measured N_2O–N emissions, IPCC-estimated emissions rates are assumed to be 2% of added manure N (IPCC, 2006). We calculated estimated IPCC N_2O emissions over the wet and dry seasons separately by multiplying the amount of N added in each treatment by 0.02 and downscaling to the number days in each season (wet = 60 d; dry = 63 d) to compare fluxes over similar time periods.

We used a linear mixed-effects model (*lme4* package for R) (Bates et al., 2013) to determine significant differences in cumulative fluxes of N_2O–N and CH_4–C among the treatments in the wet and the dry season (separately). The experiments were established as randomized complete block design, and thus blocks were nested within farm. We used Tukey post hoc tests (*multcomp* package for R) (Hothorn et al., 2008) to examine the differences in cumulative fluxes among the treatments. When necessary, we used Box–Cox (Box and Cox, 1964) or log transformations prior to analysis to satisfy the assumptions of the statistical model. To examine the relationship between soil N concentrations and soil N gas emissions, we ran a univariate regression of mean soil N concentrations in the dry season against cumulative N_2O–N emissions. To examine the relationship between the N input rate in the treatments and cumulative N_2O–N emissions, we tested linear and polynomial regressions for best fit.

We conducted Monte Carlo simulations (n = 10,000) to estimate the average and range of cumulative N_2O emissions due to urine excreted from cattle in Kenya. We focus on emissions from urine because we can estimate urine production from cattle population but cannot estimate dung production without cattle body weight, which is highly variable. Emissions were calculated as the product of the following variables: animal population (normally distributed with mean = 18,200,000 ± 531,656 head; FAO [2016]), urine production (uniformly distributed between 5849 and 13,144 L urine yr^{-1}; Haynes and Williams [1993]), N in urine (normally distributed with a mean of 8.81 ± 0.47 g N L^{-1}; our data), and the average emission factor derived from measurements from the urine treatments in the two farms in the wet and dry season in Kenya (normally distributed with a mean of 0.1 ± 0.07% of N applied in urine). Mean and standard deviation for the resulting distribution were calculated after truncating the distribution at zero because soil N_2O uptake (negative flux) has never been measured under urine patches.

Results

Seasonal N_2O and CH_4 Emissions

In the wet season, N_2O emissions increased steadily in the urine and dung+urine treatments for the first week (up to 1076 μg m^{-2} h^{-1}) and then remained high for about 2 wk before decreasing to near background levels (33 μg m^{-2} h^{-1} by 22–26 Oct. 2013) (Fig. 2A, B). In Farm 1, elevated N_2O emissions from urine and dung+urine were coincident with rainfall between 1 and 21 October. In Farm 2, however, the rains hit on 24 October, after N_2O emissions from urine and dung+urine had dropped. Slightly elevated emissions throughout the wet season in Farm 2 may be attributed to the heavy rainfall through late December (Fig. 1B). In the dry season, N_2O emissions increased after the urine addition but returned to near background levels within a week (Fig. 2C, D). Dry season N_2O–N emissions tended to be highest in the urine and dung+urine treatments and increased in mid-January after several days of rain (14 mm in Farm 1 and 17 mm in Farm 2), after which emissions returned to baseline. There was a short-term pulse of CH_4 after excreta additions in both farms and both seasons. The highest pulse of CH_4–C occurred in the dung+urine treatment the same day as the excreta additions, when fluxes increased from 0 to 0.9 mg CH_4–C m^{-2} h^{-1} then decreased to background levels by the next day (Fig. 3C). After these initial pulses, soil CH_4 emissions remained near zero with many of the treatments showed low levels up CH_4 uptake (negative flux) in both the wet and dry seasons (Fig. 3). There did not seem to be a relationship between CH_4 emissions and precipitation during the wet or dry seasons.

Cumulative N_2O and CH_4 Fluxes

Cumulative N_2O–N fluxes were significantly higher in Farm 2 than in Farm 1 (P < 0.0001) in the dry season, and although fluxes tended to be higher in Farm 2 in the wet season as well, the relationship was not statistically significant. Overall, cumulative N_2O–N fluxes were significantly higher in the wet compared with the dry season (P < 0.0001) (Fig. 4A, B). In both seasons, cumulative N_2O fluxes differed significantly among the treatments (P < 0.0001 in both cases) (Fig. 4A, B). In both seasons, dung+urine had the highest fluxes overall (P < 0.0001), and the dung treatments were also higher than the rain water and the control treatments (P < 0.0001). There was no significant effect of season or farm on cumulative or daily CH_4 flux. However, there was a significant effect of treatment overall (P = 0.012), which was driven by the apparent consumption of CH_4 during the wet season in the urine treatments (P = 0.034) (Fig. 4C). This pattern was not significant in the dry season.

Dry season fluxes increased linearly with N input rate, but wet season fluxes increased nonlinearly (Fig. 5). Nitrous oxide emission

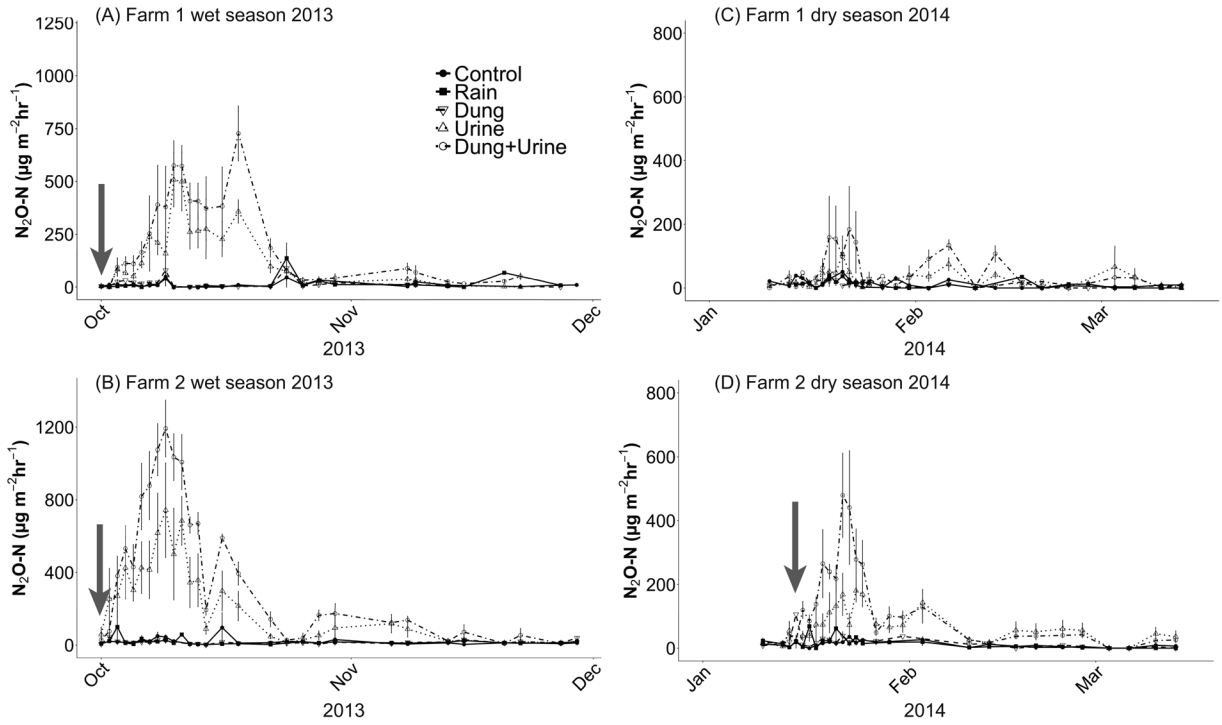

Fig. 2. Mean N_2O–N fluxes ($\mu g\ m^{-2}\ h^{-1}$) from pastures in Farm 1 and Farm 2 in the wet (A and C) and dry (B and D) seasons after amendments of none (solid lines and circles), rain water (solid lines and squares), dung (dashed lines and inverted triangle), urine (dotted lines and triangles), and dung+urine (dashed-dotted lines and open circles). Bars represent SEM.

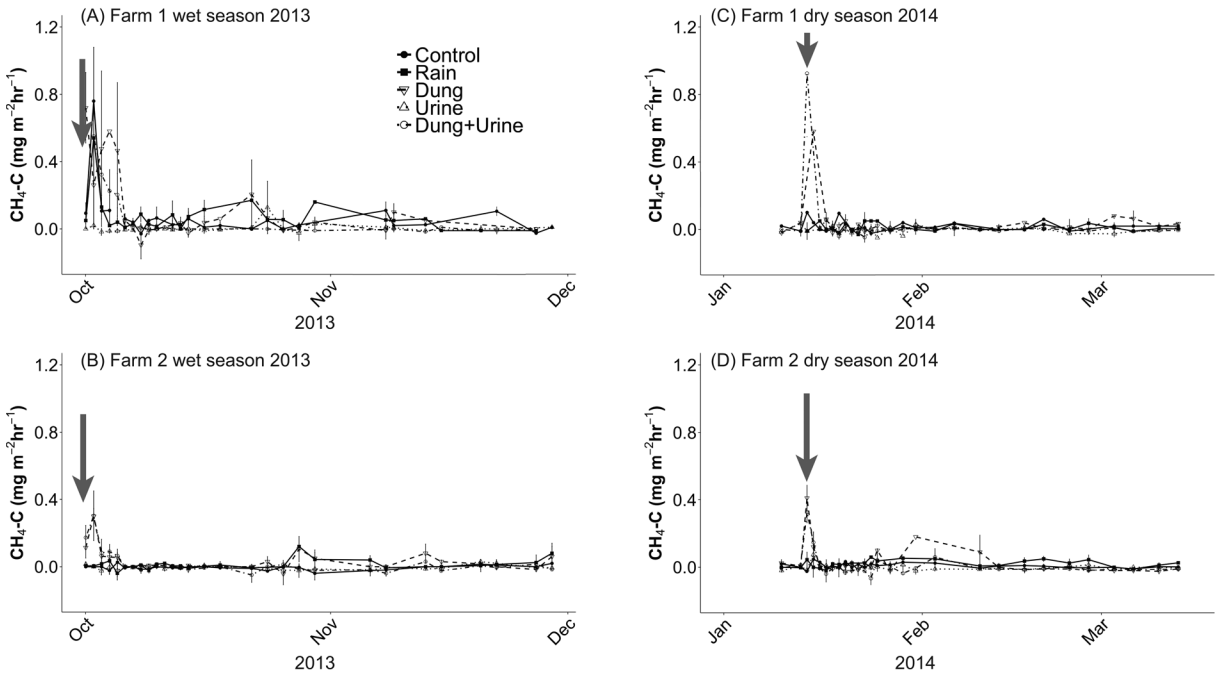

Fig. 3. Mean CH_4–C fluxes ($mg\ m^{-2}\ h^{-1}$) from pastures in Farm 1 and Farm 2 in the wet (A and C) and dry (B and D) seasons after amendments of none (solid lines and circles), rain water (solid lines and squares), dung (dashed lines and inverted triangle), urine (dotted lines and triangles), and dung+urine (dashed-dotted lines and open circles). Bars represent SEM.

factors were higher in the wet season than in the dry season, with the highest emission factors from the treatments that received dung+urine or urine alone (Table 2). Measured cumulative seasonal emissions were on average 65% lower than IPCC-predicted emissions, with predicted dry season emissions 79% higher than our field measurements (Supplemental Table S1).

N_2O Fluxes Correspond to Soil N Concentrations

Similar to gas fluxes, inorganic soil NO_3^-–N was significantly higher in the urine and dung+urine treatments than in the control and rain water treatments ($P = 0.0012$). Although Farm 1 began the study with slightly higher total soil N concentrations (Table 1), over the course of the study, soil NO_3^-–N concentrations remained elevated in Farm 2 (mean, 31.7 $\mu g\ g^{-1}$) compared

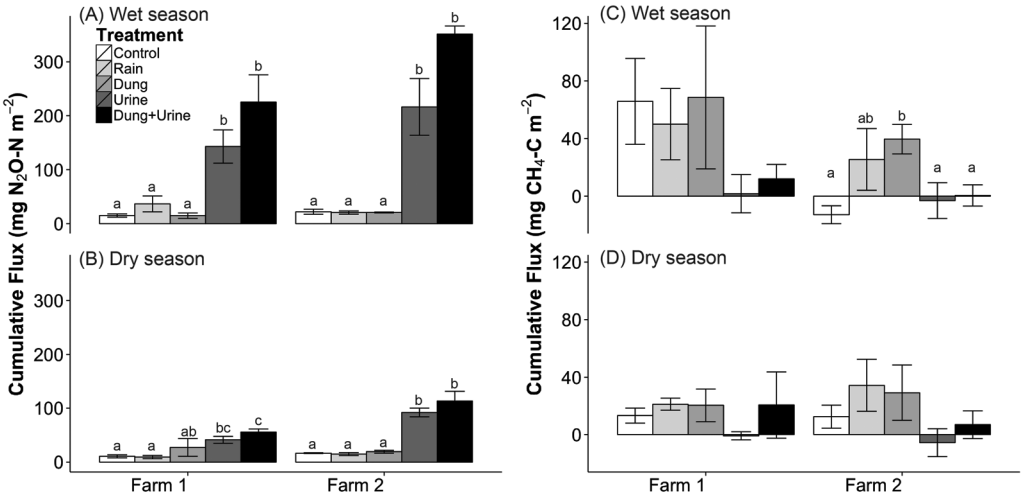

Fig. 4. Cumulative N_2O–N flux (mg m^{-2}) in the (A) wet season and (B) dry season and cumulative CH_4–C flux (mg m^{-2}) in the (C) wet season and (D) dry season in the two farms. Bars represent SEM. Values significantly different at $P < 0.05$ are indicated by different letters above bars.

with Farm 1 (mean, 5.7 μg g^{-1}; $P < 0.0001$) (Supplemental Fig. S1). In the dry season, N_2O–N emissions increased significantly with soil NO_3^-–N concentrations ($r^2 = 0.80$; $P < 0.0001$).

National Level Emissions

All 10,000 simulations of emissions from urine in Kenyan pastures were below the 11.9 Gg N_2O–N estimated by the FAO for 2014 (FAO, 2016). Simulations ranged from 0 to 6 Gg N_2O–N yr^{-1}, and mean emissions were 1.7 ± 1.0 Gg N_2O–N yr^{-1} for the cattle population of Kenya (Supplemental Fig. S2).

Discussion

After the application of excreta, we observed elevated N_2O–N fluxes for several weeks, which returned to baseline levels after a month. Similar patterns have been observed by van Groenigen et al. (2005) in The Netherlands, by Pelster et al. (2016) in Kenya, and by Mazzetto et al. (2014b) and Lessa et al. (2014) in Brazil. Data show several points on both the ascending and descending portions of the flux curve, suggesting that the intervals between gas sampling were sufficient to capture the pulse GHG emissions after excreta emissions in this agroecosystem, and thus the patterns seen are robust against potential sampling error (Fig. 2).

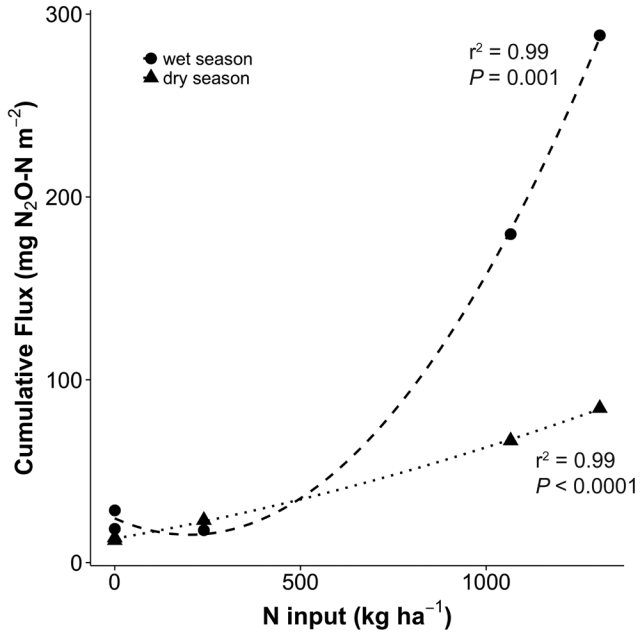

Fig. 5. Cumulative N_2O–N flux in the wet (circles) and dry season (triangles) as a function of N input rate in the treatments. Lines represent best-fit regressions for fluxes in the wet season (dashed lines) and dry season (dotted lines).

The production of N_2O can occur during both denitrification and nitrification in soils (Barnard et al., 2005), but denitrification tends to be the dominant mechanism for N_2O production in soils treated with excreta (Monaghan and Barraclough, 1993). Denitrification is a microbially facilitated process that reduces oxidized forms of N, which means that it is strongly controlled by soil moisture and gaseous diffusion (de Klein et al., 2003). In regions with seasonal rainfall, alternate periods of soil wetting and drying may lead to increased NO_3^- production by nitrification during the dry periods, followed by denitrification in the wet periods. We reported higher N_2O emissions when urine was applied during wet periods compared with dry periods, similar to earlier work by van Groenigen et al. (2005). Similarly, cumulative N_2O fluxes were about 50% higher in Farm 2, where rainfall was nearly three times higher (Fig. 4) and soil NO_3^-–N was five times higher (Supplemental Fig. S1). However, the first flux of N_2O in the wet season occurred before the onset of the rains. This suggests that urea was converted to NH_4^+, oxidized to NO_3^-, and reduced to N_2O, whereas soil moisture was still high after urine treatments. There is also evidence of a second flux after the onset of greater precipitation because remaining N was denitrified after soil moisture increased.

Denitrification is also strongly regulated by the availability of N and tends to increase with increasing N availability (Beauchamp, 1997). Thus, we found the highest cumulative (and daily) fluxes of N_2O in the urine and dung+urine treatments, which added 109 and 133 g m^{-2} per season, respectively (Fig. 2 and 4). Although N_2O emissions increased with inputs, the relationships were not the same between the wet and dry seasons, suggesting that emissions must be measured in each season to capture annual variability in seasonal tropical systems. Emission factors, by definition, are annual estimates and therefore should account for seasonal variation. However, when tropical emissions

are estimated from short-term measurement campaigns, they may miss the large differences between seasons and over- or underestimate the annual average. A nonlinear response of N_2O to soil N inputs has also been observed in an inorganically fertilized maize system in Kenya (Hickman et al., 2015), and there is growing evidence that N_2O losses may respond nonlinearly to N additions in temperate systems (Van Groenigen et al., 2010). Our data agree and suggest that the pattern may hold with organic materials but is unique across seasons (Fig. 5).

Upland soils are typically a sink for CH_4. Methanogenic bacteria thrive in anoxic conditions, which rarely occur in well-drained soils in the highlands of East Africa. However, dung and manure deposited on land can produce localized hotspots of CH_4 production because of the high moisture and C content and the population of methanogens in the material. Cow dung was initially a source of CH_4 at both farms, mainly for the first 2 d after addition, which is similar to earlier studies in temperate (Jarvis et al., 1995; Saggar et al., 2004) and tropical regions (Mazzetto et al., 2014a). The magnitude of CH_4 fluxes after dung addition was similar to Mazzetto et al. (2014a) (i.e., 0–1.0 mg CH_4–C m^{-2} h^{-1}) but substantially lower than Jarvis et al. (1995) and Saggar et al. (2004), who reported fluxes around 13 to 20 mg CH_4–C m^{-2} h^{-1} after initial dung application. Although not significantly different, CH_4 emissions tended to be higher in Farm 1 in absolute terms, the opposite of N_2O emissions. Higher cumulative emissions of CH_4 may also be due to slightly higher soil organic C in Farm 1 than in Farm 2 (Table 1), providing a larger C pool for methanogens after manure deposition. Urine treatments were CH_4 sinks during the wet season (Farm 1), which can occur in some soils during periods of high soil moisture in which gas transport can be limited (Castro et al., 1992).

Methane emissions were significantly lower in the dung+urine than in dung alone, which is easily explained by the redox potential of the different treatments. As oxygen becomes depleted in these localized hotspots, the redox reaction that generates the most energy is denitrification of nitrate to either N_2 or N_2O, with reduction of CO_2 to CH_4 generating substantially less energy. Therefore, the high N treatments (e.g., urine and dung+urine) were less likely to reduce CO_2 to CH_4 than the dung treatment, which provided a high C substrate with a substantially lower N content. Although we observed large pulses of CH_4 emission immediately after excreta application compared with enteric fermentation processes (646 Gg CH_4–C yr^{-1}) (FAO, 2016), CH_4 emissions from manure deposition (18.2 Gg CH_4–C yr^{-1}) (FAO, 2016; IPCC, 2006) comprise a minor component of total CH_4 emissions in Kenya.

Measured N_2O emission factors from excreta ranged from 0 to 0.21% (Table 2), which is far below the 2% IPCC default emission factor (IPCC, 2006) and 1.4% reported for Africa in a recent meta-analysis (Albanito et al., 2017). Our values, which are only 10% of the expected emissions based on IPCC averages, are similar to emissions found in a simulated pasture on the International Livestock Research Institute farm in Nairobi, Kenya (0.1–1.2%) (Pelster et al., 2016), the only other measurements of excreta emissions on pastures in sub-Saharan Africa. Concentrations of N in dung collected from cattle on smallholder farms under production were similar to those found in dung from cattle being fed modified diets by Pelster et al. (2016), which reported concentrations of 1.2% N, compared with 1.9% N in our study. However, urine N concentrations were more than three times higher in our study (8.81 vs. 2.65 g N L^{-1}). The urine N concentrations in our samples and applied in our experiment more closely match previous studies, which show a range of N in urine from 5.9 to 9.2 g N L^{-1} (de Klein et al., 2003; Decau et al., 2004). These findings, therefore, add support to the conclusion by Pelster et al. (2016) that emissions from cattle excreta on pasture may be of lesser consequence for the climate systems than predicted, with this study showing that emissions are minimal across a wider range of N inputs and typical of production conditions characteristic of smallholder dairies in highland Kenya.

A lack of locally relevant emission factors for cattle urine is part of the reason countries in sub-Saharan Africa, including Kenya, use Tier 1 emission factors when reporting emissions from this source in National Communications to the UNFCCC (UNFCCC, 2015). We simulated plausible scenarios of urine-derived emissions given reported and measured levels of uncertainty and found that there is virtually no likelihood that emissions could reach estimated levels. Mean estimates of emissions were less than 2% of those estimated by the FAO using the same animal population size. Even the most extreme emission predictions based on simulations are only about half the current estimates (6 vs. 11.9 Gg N_2O–N yr^{-1}) (FAO, 2016). Although our simulations are rough approximations, they provide clear indications of potential overestimation of urine-derived N_2O–N emission in the tropical African context.

Conclusions

We showed that the largest N_2O emissions occurred in the wet season and in the farm receiving the most rainfall, indicating the importance of climate on N_2O emissions. Soils receiving urine and dung+urine had the largest daily and cumulative N_2O emissions because these treatments provided the largest N inputs. We showed that N_2O emissions from urine and dung under tropical conditions are an order of magnitude lower than those predicted by either the IPCC Default Tier 1 emission factors or a recent meta-analysis for Africa (0.05–0.18% vs. 2 and 1.4%, respectively) (Albanito et al., 2017; IPCC, 2006). The IPCC emission factors are used in the FAO calculations for emissions for manure management from Kenya, which thus overestimate the contribution of manure emissions to the country-wide GHG budget (2 Gg N_2O–N yr^{-1}on average, compared with 11.9 Gg N_2O–N yr^{-1}). We also show that N_2O emissions may respond nonlinearly to N additions in manure in the wet season, which calls into question the use of fixed emission factors for sub-Saharan African agroecosystems. Methane emissions were initially responsive to dung applications but quickly returned to baseline levels, corroborating that manure is not a significant source of CH_4 emissions compared with other agricultural sources, such as enteric fermentation or manure managed in slurry. The deviation from expectation with N_2O fluxes combined with the dearth of available data and the demand for understanding emissions and mitigation potential in light of the Paris Agreement indicate that greater attention is needed to quantify and manage this source of emissions.

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