Nitrous oxide emissions from excreta deposited onto tropical pasture in Kenya

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Summary

Nitrous oxide emissions from excreta deposited onto tropical pasture in Kenya

Livestock production systems are dominant greenhouse gas sources in sub-Sahara African (SSA), including emissions of nitrous oxide (N_2O) from excreta deposited onto pasture. However, a limited number of studies are available to date. Using the default emission factor (EF) from the Intergovernmental Panel on Climate Change (IPCC) guidelines, which represents a global mean value mainly based on observations in OECD countries, may cause strong uncertainties due to differences in the management practices, climate and soils between OECD countries and countries and livestock systems in SSA. To narrow uncertainties of national greenhouse gas inventories in SSA it is thus needed to develop region-specific N_2O emission factors for excreta dropped on tropical pasture which do consider regional characteristics.

In my thesis the calculation of N_2O emission factors for excreta dropped on tropical pastures was based on a series of field trials in which I varied dung quantity, quality and urine amount. These field trials were carried out during the dry and the rainy season to also investigate how weather conditions and hydric seasons (dry versus rainy) are affecting emissions. In a final experiment I investigated the importance of soil properties on N_2O emissions due to excreta deposition.

Measurements of N_2O fluxes were done using an automated chamber systems as well as manual chambers to measure N_2O. The use of automated chambers allowed to obtain a high resolution dataset and to quantify short-term emission peaks. However, as the number of automated chambers

was limited also manual chambers were used, specifically while investigating the effect of soil types on N_2O emissions from excreta dropped on tropical pastures.

My results show that differences in dung mass, dung quality and urine volume did not cause significant differences in terms of dung or urine N_2O EF. Surprisingly, dung application only marginally stimulated the soil N_2O fluxes regardless of the mass or quality of dung patches, with EFs being a magnitude lower as found for dung applications to temperate grasslands. This significant difference in N_2O EFs was due to the low nitrogen (N) concentration, high carbon (C) to N ratio, as well as the rapid crust formation of dung under tropical weather and soil conditions. In contrast to my observations for dung, application of urine to tropical pastures resulted in rapid increases in N_2O fluxes, though the stimulatory effect was higher in the rainy as compared to the dry seasons. Again, EFs for urine were significantly lower in my experiments as compared to experiments carried out in temperate regions.

The influence of soil type on N_2O emissions after excreta deposited onto pasture was also tested during my study using intact soil cores from five soil types that are wide-spread in Kenya. Through two field trials, I could show that soil properties were significantly affecting soil N_2O emissions as triggered by urine but not dung additions to pastures, with highest N_2O emission responses found for soils with pH values close to neutral and rich in clay.

Based on my findings I calculated a specific cattle excreta N_2O EF (0.13%) as well as disaggregated EFs for dung (0.05%) and urine (0.29%) for Kenya, thereby considering excreta properties, soil properties and climate (rainy and dry seasons). The cattle excreta N_2O EF was calculated assuming N partitioning between urine and dung as 34:66 based on the feed diet in SSA. This N excreta partitioning is different from the IPCC 2019 refinement, which assumes a N

partitioning between urine and dung as 66:34 and causes an overestimation of N_2O emissions from excreta on tropical rangelands in dry climates (IPCC 2019: 0.20%; my study: 0.13% of excreta N deposited onto rangeland).

In the frame of manure decomposition experiments, with dung from cattle, sheep and goat being investigated and exposed at four sites differing in climate conditions, I could show that cattle manure is decomposing faster than manures from sheep or goats. This difference in manure decomposition could be linked to manure cellulose concentration, with higher values supporting higher decomposition rates. Moreover, I found that manure decomposition rates were higher under wetter conditions, but that even after more than one year (378 days) approx. 30% of cattle manure and 50% of goat and sheep manure remained.

My studies show that distinct, so far not sufficiently accounted differences in N_2O EFs and manure decomposition dynamics exist for excreta dropped on tropical pastures if compared to existing studies for temperate regions. These differences are mainly due to lower quality of excreta in terms of N concentrations and composition of the manure with regard to e.g. lignin or cellulose, i.e. factors linked to feed supply and feed quality. In addition, environmental factors such as pronounced dry periods affect N_2O emissions from excreta and the decomposition of manures. These pronounced differences in decomposition dynamics and N_2O EFs might be addressed in further studies in order to narrow uncertainties in N_2O emissions from livestock systems in the tropics and subtropics.

Zusammenfassung

In Sub-Sahara Afrika (SSA) sind Tierproduktionssystems oft die dominierende Quelle des Treibhausgases Lachgas (N_2O). Hauptursache hierbei sind N_2O Emissionen die mit der Deposition von Tierexkreten auf tropischen Weiden verbunden sind. Der bisherige Ansatz solche N_2O Emissionen mit Emissionsfaktoren des IPCC (Intergovernmental Panel on Climate Change) abzuschätzen, das heißt unter Nutzung eines Emissionsfaktors der vor allem auf Basis von Messungen in OECD Ländern und damit unter zumeist temperaten Bedingungen entwickelt wurde, ist aller Voraussicht mit großen Unsicherheiten verbunden. Dies gilt insbesondere da klare Unterschiede in Tierhaltungssystemen und klimatischen und edaphischen Faktoren bestehen, welche bekanntermaßen N_2O Emissionen aus Tierexkretdepositionen auf Weideland beeinflussen können. Meine Dissertation hatte daher das Ziel die bestehenden Unsicherheiten zu N_2O Emissionsfaktoren zu Dung und Urin Exkreten auf tropischem Weideland durch eigene Untersuchungen in Kenia zu reduzieren.

In meiner Doktorarbeit habe ich die Kalkulation von N_2O Emissionsfaktoren von Exkreta-Depositionen auf tropischem Weideland auf einer Serie von Felduntersuchungen basiert, wobei ich Einflussfaktoren wie Exkreta-Menge oder Exkreta-Qualität variiert und Versuche in unterschiedlichen hygrischen Jahreszeiten (Trocken- versus Regenzeit) durchgeführt habe. In einem weiteren Experiment habe ich zudem den Einfluss von Bodeneigenschaften wie pH oder Textur auf die N_2O Emissionen aus auf Weideland deponierten Exkreta untersucht.

Die N_2O Flussmessungen wurden sowohl mit automatischen Kammersystemen wie auch mit manuellen Kammern durchgeführt. Durch die Nutzung eines automatischen Kammersystems war

es mir möglich auch zeitlich sehr kurze Emissionsspitzen zu quantifizieren. Allerdings war die Anzahl von Auto-Kammern beschränkt, so dass ein Teil der Versuche, insbesondere meine Versuche zum Einfluss von Bodeneigenschaften auf Exkreta N_2O Emissionen von Weideland, mit manuellen, statischen Kammern durchgeführt wurden.

Meine Resultate zeigen, dass Unterschiede in der Dungmenge, Dungqualität oder des Urinvolumens keinen signifikanten Einfluss auf den N_2O EF für Exkreta-Depositionen auf tropischem Weideland haben. Überraschenderweise war der stimulierende Effekt von Dungdepositionen auf Weideland auf N_2O Emissionen marginal, wobei die beobachtenden N_2O EF ca. eine Größenordnung niedriger waren als für temperate Systeme beobachtet. Dieser signifikante Unterschied in Exkreta N_2O Emissionsfaktoren für Dung konnte auf signifikant niedrigere Exkreta N Gehalte, bei Zugleich höheren Kohlenstoff (C) zu N Verhältnissen sowie auf eine schnelle Krustenbildung beim Dung, unter den gegebenen tropischen Wetter- und Bodenbedingungen, zurückgeführt werden. Im Gegensatz dazu waren bei Applikation von Urin auf Weideland schnelle und deutliche Zunahmen der N_2O Flüsse zu beobachten, wobei der stimulierende Einfluss von Urin auf N_2O Emissionen in der Regenzeit höher als in der Trockenzeit war. Aber auch hier gilt, dass die gefundenen EF für N_2O für Urindeposition auf tropischem Weideland signifikant niedriger waren als in vergleichbaren Studien für temperate Weidegebiete beschrieben.

In weiteren Versuchen habe ich zudem den Einfluss von Bodenfaktoren auf die mit der Exkreta-Deposition auf Weideland verbundenen N_2O Emissionen untersucht. Hierbei wurden fünf typische, tropische Böden in Kenia in die Untersuchung im Rahmen von zwei Feldversuchen miteinbezogen.

Meine Ergebnisse zeigen, dass Böden mit höheren Tongehalten und mit Boden-pH Werten im neutralen Bereich die höchsten N_2O Emissionen nach Exkreta-Deposition aufweisen.

Basierend auf meinen Ergebnissen habe ich einen für die Deposition von Rinderexkreta auf tropischem Weideland spezifischen N_2O Emissionsfaktor für Kenia berechnet. Dieser beträgt 0.13% des in den Exkreta enthaltenden Stickstoffs und setzt sich zusammen aus einem EF für Dung (0.05%) und Urin (0.29%), wobei Effekte von Exkreta-Eigenschaften, Bodeneigenschaften sowie klimatische Gegebenheiten (Trocken-/ Regenzeit) berücksichtigt sind. Die Kalkulation dieses Rinderexkreta Emissionsfaktors beruht dabei auf einer Splittung der N-Exkretion in Urin und Dung von 34:66, wie dies in verschiedenen Untersuchungen für SSA gezeigt wurde, und damit invers zur Annahme des IPCC (66:34) in seiner revidierten Fassung. Gerade dieses dürfte aber zu einer Überschätzung der N2O Emissionen bei Anwendung des derzeitigen IPCC Ansatzes führen (IPCC 2019 Exkreta N_2O EF: 0.20%; diese Studie: 0.13% des Exkreta N deponiert auf Weideland).

Im Rahmen von Dekompositionsstudien für Rinder-, Schaf – und Ziegendung, habe ich Dung auf tropischem Weideland an vier, klimatisch deutlich unterschiedlichen Untersuchungsstandorten deponiert. Über alle Standorte hinweg zeigen meine Untersuchungen, dass der Dung von Rindern schneller kompostiert als der von Schafen oder Ziegen und das die Dekompositionsgeschwindigkeit von Dung unter feucht Bedingungen deutlich erhöht ist. Zellulose und Lignin-Gehalte des Dungs spielen hierbei eine große Rolle. Allerdings war selbst nach über einem Jahr (378 Tage) ca. 30% des Rinderdungs und 50% des Schaf- und Ziegendungs noch nicht mineralisiert.

Meine Studien zeigen, dass ausgeprägte, bisher nicht ausreichend untersuchte Unterschiede in den N_2O EF und Dekompositionsgeschwindigkeiten von Exkreta Depositionen auf tropischem

Weideland im Vergleich zu Untersuchungen in temperaten Gebieten existieren. Diese Unterschiede beruhen vor allen auf niedrigere N-Gehalte und höhere Lignin-Gehalte (bei Dung) in den Exkreta in meinen Untersuchungen, welche insbesondere mit niedrigerer Futterqualität zusammenhängen dürften. Darüber hinaus spielen aber auch Umweltfaktoren, wie ausgeprägte Trocken- und Regenzeiten oder auch Bodeneigenschaften eine Rolle bei der Erklärung von Differenzen in N_2O Emissionsfaktoren zwischen meiner Studie und Studien in temperaten Regionen. Diese bedeutenden Unterschiede in N_2O EFs und der Dekompositionsdynamik von Dung sollten in zukünftigen Studien weiter untersucht werden, da nur so die bestehenden Unsicherheiten zu N_2O Emissionen aus tropischen Weidegebieten weiter gemindert werden können.

Contents

Summary i

Zusammenfassung iv

1 Introduction 1

1.1 Excreta properties influenced by animal live weight and feed diet 5

1.2 Soil properties 6

1.3 Manure decomposition 6

1.4 Objectives 7

2 Description of farming system and study sites 9

2.1 Smallholder mixed-crop and pastoral livestock systems 9

2.2 Study sites 10

3 Description of N_2O emissions from excreta deposited onto tropical pasture in Kenya 12

3.1 Effect of excreta properties on N_2O emissions from excreta deposited onto tropical pasture in Kenya 12

3.1.1 Feed diet effect on excreta properties 12

3.1.2 Experimental design 14

3.1.3 Results and Discussions 18

3.2 Effect of soil properties on N_2O emissions from excreta deposited onto tropical pasture in Kenya 24

3.2.1 Soil properties 24

3.2.2 Experimental design 25

3.2.3 Results and discussions 28

4 Description of manure composition dynamics influenced by manure type and climatic conditions in tropical pasture in Kenya 34

4.1 Manure properties and weather conditions in the four sites 34

4.2 Experimental designs 35

4.3 Manure decomposition 36

5 Conclusion and outlook 40

Bibliography 42

Publication 55

Acknowledgements 56

Chapter One

1 Introduction

As one of the main challenges facing humanity, climate change has gained growing international attentions. As of 2017, the global mean surface temperature (GMST) was 1.0 °C higher than that of the pre-industrial period (1850-1900) and it is still increasing at the rate of 0.2 ± 0.1 °C per decade (IPCC, 2018). In addition, the incidence of extreme events (droughts, floods, heat waves…) has increased and the precipitation pattern has also changed, with increases in the frequency and intensity of heavy precipitation in more regions than those with decreases (Hoegh - Guldberg O et al., 2018). This is a consequence of the increased concentrations of atmospheric greenhouse gases (GHG). Carbon dioxide (CO_2) is the single most important anthropogenic GHG and its concentration in the atmosphere has reached 407.8 ± 0.1 ppm, which is 1.47 times higher compared with the pre-industrial period (before 1750) (World Meteorological Organization and Global Atmosphere Watch, 2019). It is estimated that CO_2 in the atmosphere contributes to 66% of the radiative forcing, and that anthropogenic emissions are mainly from fossil fuel combustion, industrial processes and agriculture, forestry and other land uses (AFOLU). Atmospheric methane (CH_4) is 2.59 times higher than pre-industrial levels and current concentrations are 1869 ± 2 ppb (World Meteorological Organization and Global Atmosphere Watch, 2019). With a global warming potential 28-times that of CO_2 over a 100-year time horizon on a per mass basis, CH_4 contributes approximately 17% of the radiative forcing. CH_4 sources include natural sources mainly from wetlands and termites, constituting 40% of the total CH_4 emissions, and anthropogenic sources constituting 60% of the total CH_4 emissions mainly associated with agriculture, fossil exploitation and landfills emitted. As the most potent of the three primary

greenhouse gases, nitrous oxide (N_2O) in the atmosphere reached 331.1 ± 0.1 ppb in 2018, which is 1.23 times higher than the pre-industrial level (World Meteorological Organization and Global Atmosphere Watch, 2019). Nitrous oxide has a global warming potential approximately 265 times more powerful than CO_2 and it contributes 6% of radiative forcing (Davidson, 2009). N_2O production is mainly related to microbial production processes in soils, sediments and water bodies (Butterbach-Bahl et al., 2013). Anthropogenic N_2O emissions constitute approximately 40% of the whole emissions and are mainly from agricultural activities, especially fertilizer and manure from livestock production system.

Table 1 Anthropogenic N_2O budget for the year 2006 (Ciais et al., 2013)

Anthropogenic N_2O sources	Mean (Tg N_2O-N yr^{-1})	Range (Tg N_2O-N yr^{-1})
Agriculture	4.1	1.7-4.8
Fossil fuel combustion and industrial processes	0.7	0.2-1.8
Biomass and biofuel burning	0.7	0.2-1.0
Rivers, estuaries, coastal zones	0.6	0.1-2.9
Atmospheric deposition on land	0.4	0.3-0.9
Atmospheric deposition on ocean	0.2	0.1-0.4
Human excreta	0.2	0.1-0.3
Surface sink	-0.01	0– -1
Total anthropogenic sources	6.9	2.7-11.1

The AFOLU sector is an important anthropogenic GHG source contributing approximately 24% of the global anthropogenic GHG emissions (IPCC, 2014). The CO_2 emissions from AFOLU constitute approximately 10-13% of the global anthropogenic GHG emissions and are mainly from land use and land use change and the subsequent depletion of soil organic carbon (C) stocks. Enteric fermentation from ruminants, manure management and rice production are the main CH_4 sources in the agriculture sector (Tubiello et al., 2013). Agricultural activities including the use of organic and inorganic fertilizers and excreta from livestock are dominant N_2O sources (Table 1). Livestock production systems contribute 12% of the global anthropogenic GHG emissions and generate 80% of all agricultural non-CO_2 emissions. As the largest CH_4 sources, enteric fermentation from ruminants comprise 40% agricultural GHG emissions production (Tubiello et al., 2013). Manure-related emissions including manure left on pasture, manure applied to soils and manure management contribute approximately 26% to total GHG emissions from all agricultural sources (Tubiello et al., 2014). Specifically, emissions from excreta (i.e. dung and urine) deposition comprise 15% of the total GHG emissions from the agricultural sector (Tubiello et al., 2014). The N_2O emissions from global livestock production systems comprise 22% of total anthropogenic N_2O emissions (Oenema et al., 2005; Taghizadeh-Toosi et al., 2011). A new modelling study estimated that N_2O emissions from livestock excreta N deposition on pasturelands and rangelands was 1.31 Tg N_2O-N yr^{-1}, which is equivalent to 19% of total anthropogenic N_2O emissions (Dangal et al., 2019).

The significant N_2O emissions from excreta deposited on pasture are mainly due to the large grazing area and numerous grazing animals. Grasslands cover up to 40 % (59 million km^2) of the world's ice-free land area (Hufkens et al., 2016) and they are mostly used for grazing (Salvati and

Carlucci, 2015; Zhou et al., 2018). As the most common fed animals, the population of cattle, goat and sheep is estimated at 1.5 billion and 1.7 billion, respectively, and will continue to increase due to the growing demand for livestock products (FAO, 2009). Up to 75-90 % of nitrogen (N) ingested by grass-fed animals is returned to the soil through dung and urine excretion (Saggar et al., 2013; Bell et al., 2015). The form of N in a cattle dung patch is mainly organic and the amount of N is calculated as equivalent to up to 1130 kg N ha^{-1} (Saarijärvi et al., 2006), while the form of N in a urine patch is mainly urea and the N loading rates are found to be equivalent to 200-2000 kg N ha^{-1}, with an average of 613 kg N ha^{-1} for dairy cattle and 345 kg N ha^{-1} for beef cattle (Selbie et al., 2015). The amount of N in excreta patches would significantly exceed plant N demand if all excreta-N were mineralized. Hydrolysis of urea and mineralization of organic N to ammonium (NH_4^+) provide a pool of available N for the nitrification and denitrification processes, and thus N_2O production (Butterbach-Bahl et al., 2013). Therefore, urine and dung patches on grassland are regarded as GHG hotspots (Cai et al., 2017).

In general, N_2O emissions from cattle urine are higher than cattle dung due to higher N availability in urine and better incorporation as urine infiltrates into the soil (Cai et al., 2017). Besides, the N partitioning between cattle urine and dung is estimated to range from 50:50 to 75:25 depending on the crude protein (CP) in feed diet. Hence, excreta-N partitioning should be considered to better estimate regional N_2O emission inventory. A number of recent studies (Chadwick et al., 2018; Thomas et al., 2017; Krol et al., 2016; Bell et al., 2015;Van der Weerden et al., 2011) recommend developing specific and disaggregated emission factor (EF) for dung and urine to better quantify the sources and more effectively target mitigation strategies. However, the Intergovernmental Panel on Climate Change (IPCC) Tier 1 approach developed a default EF of 2% for excreta based

on the studies conducted in temperate regions, mostly in developed countries (Smith et al., 2007). The N partitioning of cattle urine and dung in developed countries such as western European countries is assumed to be 60:40, which means that urea in urine is the main form of the excreted N (Chadwick et al., 2018). By contrast, the N in cattle dung estimated to be 66% of the total amount excreted N in sub-Saharan Africa (SSA) due to the low CP concentration in the feedstuff (Rufino et al., 2006; Onyango et al., 2019). Due to the difference in the N partitioning, it is straightforward that the N_2O emissions from excreta patches is higher in developed countries compared with those in SSA. Since most studies have been conducted in developed countries, studies in developing countries – especially in SSA are necessary to better reflect the native conditions such as climate, soil and managements practices.

1.1 Excreta properties influenced by animal live weight and feed diet

The excreta properties differ due to differences in the feed intake (Yamulki et al., 1998) and animal live weight (Goopy et al., 2018). As estimated by Haynes and Williams, (1993), the mean weight of a single dung patch ranges from 1.5-2.7 kg and the mean volume of a single urine patch is in the range of 1.6-2.2 L. Larger dung patches of 3 kg fresh weight from 500 kg Friesian milking cows in Brazil (Sordi et al., 2014) and smaller urine patches of 1.2 L from beef cattle have also been reported (Selbie et al., 2015). Besides, the quality of feed diet also influences the N concentration in excreta patches, especially dung patches, as reported by many studies (Korir et al., 2016; Sørensen et al., 2003; van Vliet et al., 2007). Such differences in excreta quantity and quality might result in different EFs and should be considered when developing the disaggregated dung or urine EF.

1.2 Soil properties

Soil N availability is not the only driving factor influencing the magnitude and temporal dynamics of soil N_2O emissions, whereby some key soil properties also play a major role (Ghezzehei et al., 2019; Wang et al., 2018; Neira et al., 2015; Butterbach-Bahl et al., 2013). As the major driver of N_2O emissions (Butterbach-Bahl et al., 2013; Wanyama et al., 2018), soil moisture could be influenced by soil texture and soil organic carbon content (SOC). Through controlling the availability of oxygen (O_2) in soils, soil moisture can influence N_2O production processes (Balaine et al., 2016; Butterbach-Bahl et al., 2013; Davidson et al., 2000). With a low water-filled pore space (WFPS, approximately 40%), nitrification is the predominant process, while high soil WFPS (50-70%) favors denitrification. Meta-analysis studies have illustrated that soil texture, pH and bulk density also influence soil N_2O emissions from livestock excreta applied to pasture soils (Y. Wang et al., 2018; Cai and Akiyama, 2016). However, to our knowledge, no field study to date has investigated the influence of soil properties on N_2O emissions from excreta patches.

1.3 Manure decomposition

Grazing animals consume large amounts of biomass that often reduce litter inputs to soil (Güsewell et al., 2005; Tanentzap and Coomes, 2012). However, most of the nutrients taken up by the grazing animals is excreted back onto the grazed lands in the form of dung and urine. For example, it is estimated that excreta-N comprises approximately 75% of ingested N by grass-fed animals (Oenema et al., 2005). The N concentration in cattle dung and urine patches were calculated to be equivalent of up to 1130 kg N ha^{-1} (Saarijärvi et al., 2006) and 613 kg N ha^{-1} (Selbie et al., 2015), respectively. Due to the high concentrations of easily-decomposable C and N, excreta is regarded as the fast cycle that compares to the litter decomposition (Bakker et al., 2004). In addition, adult

cattle can excrete as much as 23 kg dung and 21 L urine per day. Considering the grazing area and number of livestock, manure may play a major role in C and N cycling in the grassland ecosystems. Soils are poor in nitrogen in extensive areas in SSA, thus, nutrient cycling is critical to maintain the productivity of the land (Powell et al., 1996). However, the dynamics of C and N changes after manure deposition are seldomly studied.

1.4 Objectives

Therefore, my study aimed to meet the following objectives:

1) Assess the effects of quantity and quality of dung and urine on N_2O emissions from excreta applied to rangelands in both dry and rainy seasons;

2) Examine the effect of application of the same amount of dung and urine to five soil types with differing soil properties on soil- and excreta-derived N_2O emissions in both dry and rainy seasons in a tropical highland climate;

3) Develop a regional-specific emission factor for cattle excreta in SSA, based on the outcomes of 1) and 2);

4) Investigate the effect of manure type and climatic condition on manure decomposition in tropical pastures in Kenya.

Accordingly, my hypotheses were:

1) Specific EFs are influenced by the magnitude of the mass of dung patch, volume of urine patch and N concentration of the dung and urine;

2) Soil properties such as pH or texture have minimal effects on N_2O emissions from dung but significant effects on N_2O emissions from urine deposited onto pasture;

3) Effects of both excreta and soil properties amplify during the rainy seasons and reduce during dry seasons;

4) The manure decomposition rates are higher for manure with higher N concentration and lower C/N ratio, as well as being higher under wetter and warmer conditions.

Chapter Two

2 Description of farming system and study sites

2.1 Smallholder mixed-crop and pastoral livestock systems

Smallholder mixed-crop and pastoral livestock systems (Figure 1) are the dominant farming systems across savanna ecosystem which is the main ecosystem type and occupies approximately 65% of the African continent (Brümmer et al., 2008). These farming systems are highly diverse with small-scale farm (1-2.5 hectares), few animals, different animal species and mainly rely on free-range grazing. However, an estimated 123 million cattle were fed in these farming systems (Thornton and Herrero, 2014). The systems are also characterized by feed variability, for instance, animals may not receive sufficient food in the dry seasons (Korir et al., 2016). The fodder materials are typically high in fiber with low digestibility and low protein content compared with temperate feeds. Consequently, this results in a low urine-N: dung-N partition and low N content compared to livestock systems in developed countries (Rufino et al., 2006). As livestock production predominantly relies on free-range grazing during the daytime, with animals being kept in kraals or confined areas close to the homestead only during the night, it is estimated that a minimum of 40% of excreta is deposited on range lands (Rufino et al., 2006).

Figure 1 Typical smallholder farms in western Kenya

2.2 Study sites

The experiment was set up on grassland at the campus of the International Livestock Research Institute (ILRI), Nairobi, Kenya (1°16’13″S; 36’43023″E; altitude 1,809 m above sea level [asl]).

The grassland was dominated by a mixture of Kikuyu grass (*Pennisetum clandestinum* Hochst. ex *Chiov*.) and Rhodes grass (*Chloris gayana Kunth*, Figure 2). No animals were grazing on the grassland, but the grass was manually cut approximately every 2 weeks during the rainy season and no cutting took place during the dry season. According to Climate-Data.org (https://en.climate-data.org/africa/kenya/nairobi/nairobi-541/), the long-term mean annual precipitation is 869 mm, with distinct dry and rainy seasons. On average, July is the driest month with 14 mm precipitation, while most precipitation falls in April, with 191 mm. The mean annual air temperature is 19.0 °C with only 4 °C difference between the warmest (March) and coolest month (July). Soils are classified as well-drained, deep humic nitisols (IUSS Working Group WRB, 2007) and a clay texture with 24% sand and 63% clay in the uppermost 10 cm. The topsoil has a pH of 6.2 ± 0.1

and a bulk density of 0.8 g cm^{-3}. The C and N concentrations are 25.08 ± 0.03 g C $kg^{-1}$and 2.31 ± 0.01 g N kg^{-1}, respectively.

To record the real-time weather information, a meteorological station was installed at the experimental site. The station comprises a tipping rain gauge (ECRN-100 high-resolution, Decagon, Pullman, WA; USA) for precipitation, temperature/relative humidity sensors (ATMOS 14, Decagon, Pullman, WA; USA) for air temperature and humidity every 5 min, and Decagon 5TM sensors for soil moisture and temperature at 0.05-m soil depth every 5 min.

Chapter 3

3 Description of N_2O emissions from excreta deposited onto tropical pasture in Kenya

3.1 Effect of excreta properties on N_2O emissions from excreta deposited onto tropical pasture in Kenya

3.1.1 Feed diet effect on excreta properties

In close collaboration with colleagues in Mazingira, ILRI, a parallel animal feeding trial (Figure 2) was conducted to investigate the effect of feed diet on urine and dung properties and the N partitioning between urine and dung. In this animal feeding trial, 14-month-old boran steers (*Bos indicus* L., 183 kg live weight) were fed at different maintenance energy requirement (MER) levels. At either 40 or 60% MER levels, the steers were fed with only Rhodes grass hay, while the steers were fed with Rhodes grass hay for 80% MER plus cotton seed meal for 10% MER and molasses for 10% MER at 100% MER level. Total tract digestibility was calculated as 55.3%, 59.1%, and 61.5% for 40%, 60%, and 100% MER, respectively. Steers were fed at respective MER levels for 5 weeks with 3 weeks for adaptation and 2 weeks for sample collection. The steers fed at 40% and 60% MER levels lost weight, and marginally gained weight fed at 100% MER in those 5 weeks. Another 2 weeks of refeeding was added with all steers fed ad libitum with Rhodes grass hay plus cottonseed meal and molasses. Pasture-fed steers at ILRI farm with estimated MER levels of 130-140% were also included to represent the good feed condition.

The MER of each steer was calculated as follows: MER (MJ) = 0.0819 × live weight (kg) + 21.625 (National Research Council (U.S.), 1989).

Figure 2 The ongoing animal trial fed with different MER levels and the boran steer

Throughout the animal trials, the number of dung excretions and the total daily amount of dung were recorded within two days. Both the number of dung excretions per day and the total daily dung weight increased with increasing MER level. The weight of a fresh dung patch ranged from 0.6 to 0.9 kg, with the smallest dung patch from steers fed at 40% MER and the largest patch from steers fed at 100% MER (Table 2).

The total amount of urine and dung were also collected on a daily basis during the sampling period and further analyzed for nitrogen content by Wassie et al. (2019) within the same animal trials. The total amount of dung per patch was consistent with our observation. The urination event was not recorded, although the total volume of urine per day increased with increasing feed level. However, the urine-N concentration per liter was similar among the different feed levels, with a value of 2.5 g N/L (Table 3). More N was voided through excreta with increasing feed MER level. However, the N partitioning between urine and dung was decreased with more N excreted in the form of dung following increment of feed MER level (Table 3).

Table 2 Number of dung excretions per day and total daily dung weight as recorded during the two days of observations of the feed quantity/quality trial at the International Livestock Research Institute, Nairobi, Kenya

Treatment	Day 1		Day 2	
	Number of excretions per day	Total daily fresh/dry weight (g)	Number of excretions per day	Total daily fresh/dry weight (g)
40% MER	6	3563/1018	7	4078/1165
60% MER	7	5027/1360	8	5735/1552
100%MER	10	7577/1841	9	8485/2061

Table 3 The average urine-N, dung -N, their ratio, the total volume of urine and total dung dry matter per day from the steers fed at 40%, 60% and 100% MER level (Wassie et al., 2019).

	40%MER	60%MER	100%MER
Urine-N (g/d)	9.4	8.6	11.4
Dung-N (g/d)	9.8	14.3	25
Urine-N: dung-N ratio	0.96	0.60	0.46
Urine volume (L/d)	3.6	3.8	5.0
Dung DM (kg/d)	1.0	1.4	2.0

3.1.2 Experimental design

In experiment I, we investigated the effect of dung mass on N_2O emissions. Three treatments (no addition as control, addition of 0.5 kg fresh dung, and addition of 1.0 kg fresh dung) were tested

with three spatial replicates. The dung application rate corresponded to the average dung weight as observed in one of our animal trials (Table 2). Measurements were conducted from 8 March to 12 April 2016 (Trial 1, dry season) and 13 June to 22 July 2016 (Trial 2, transition period).

Table 4 The trial number, season, observation period and related dung type in Experiment II.

Trial Nr.	Season	Observation period	Dung type
Trial 1	Dry season	16 August to 19 September 2016	60% MER vs 100% MER
Trial 3	Rainy season	11 December 2016 to 8 January 2017	
Trial 2	Rainy season	14 October to 15 November 2016	40% MER vs farm dung
Trial 4	Dry season	8 January to 12 February 2017	

In experiment II, the effect of dung quality on N_2O emissions was explored. Dung from steers fed at 40%, 60% and 100% MER in the parallel animal trials and dung from free-grazing cattle in the ILRI farm were used. Since only nine chambers were available and three spatial replicates within each treatment were needed, we split the experimental set up into two groups. One group had dung from cattle fed at 60% and 100% (trials 1 and 3) and another group had dung from cattle fed at 40% MER and dung from free-grazing cattle (trials 2 and 4, Table 4). Overall, four trails were conducted during both dry and rainy seasons and each trial had control treatment.

In experiment III, three different volumes (0.5 L, 1.25 L and 2.0 L) of urine and two different volumes (0.5 L and 2.0 L) of water were applied, and a control there was no urine or water applied

to test urine volume effect on N_2O emissions. Each treatment had three replicates. Water applications were used to eliminate the re-wet effect by water in urine. Measurements of N_2O fluxes were both conducted in dry season (from 20 May to 04 September 2018) and rainy season (from 20 September to 30 November 2019).

Figure 3 Automated chamber system used in the experiment

Gas sampling and measurements from the experiments were conducted using the automated chamber system. The automated chamber system (Butterbach-Bahl et al., 1997) comprised an automated gas sampling system (IDAWs), a cavity ringdown laser absorption spectrometer (G2308, Picarro Inc., Santa Clara, CA, USA) and nine chambers (Experiments I and II) or eighteen chambers (Experiment III, Figure 3). The stainless steel frames (0.50 m × 0.50 m × 0.05 m) were inserted into the soil, 0.5 m away from each other. Subsequently, the opaque chambers (0.50 m × 0.50 m × 0.15 m in height) with seals on the bottom were put onto the frames and fastened with

clips to keep them airtight. Soil N_2O fluxes were measured semi-continuously in 84/140/277-minutes time resolution depending on the measuring time per chamber and the number of chambers.

The GHG fluxes were calculated from the linear changes in the headspace gas mixing ratios during chamber closure and corrected with the regional atmospheric pressure and temperature using the following formula:

Mvkorr = 0.02241 × ((273.15 + self.temp) / 273.15) × (PRESSURESEALEVEL / 820)

self.fluxLin = (slopeLin × self.vol × MW[self.gas] × 60 × 1000000) / (self.area × Mvkorr × 1000000000)

Where Mv is the molar volume (m^3 mol^{-1}), self.fluxLin is the linearly-calculated flux (µg N_2O-N m^{-2} h^{-1}), slopeLin is the concentration change over time (ppb min^{-1}), MW is the molar weight (28 g N_2O-N mol^{-1}), and self.area is the chamber area (m^2).

Cumulative emissions were calculated using trapezoidal integration. Net cumulative emissions were calculated by subtracting cumulative emissions from control plots – i.e. plots not receiving dung – from cumulative emissions of plots with dung, or plots with water, from cumulative emissions of plots receiving the same volume of urine. The N_2O EF (i.e. the proportion of applied N emitted as N_2O) was calculated by dividing the net cumulative emissions by the amount of added excreta-N according to the following equation:

$$N_2O\ EF\ (\%) = \frac{\text{Net cumulative } N_2O \text{ emission(g } N_2O\text{-N)}}{\text{Nitrogen content applied as excreta (g N)}} \times 100$$

3.1.3 Results and Discussions

In experiment I, N_2O fluxes were only marginal stimulated after different quantity of dung application, with most values below 5 µg N_2O-N m^{-2} h^{-1} during the dry season and below 10 µg N_2O-N m^{-2} h^{-1} during the transition period. Heavy rainfall increased the flux in all treatments after the long-term dry season. Net cumulative N_2O emissions from grassland plots to which 0.5 or 1.0 kg dung was added based on dung dry matter were calculated and no significant differences among them were found (Figure 4; **Zhu et al., 2018 [I]**). The results were contrary to our hypothesis and a global analysis by Shcherbak et al. (2014) that N_2O emissions from soils increased exponentially with increasing N fertilizer application rate. Our hypothesis was based on more easily-accessible N and C substrates and the notion that the more water in the larger dung patches was likely more favorable for N_2O production via closely-coupled nitrification and denitrification. However, the dung dried and quickly formed a crust irrespective of the mass of dung patches due to the relatively low humidity and higher solar radiation in our study site, which limited the interactions between dung patch and soils. The mass of dung did not influence the N_2O emissions, this is in line with the findings by another study in tropical regions of Brazil by Sordi et al. (2014).

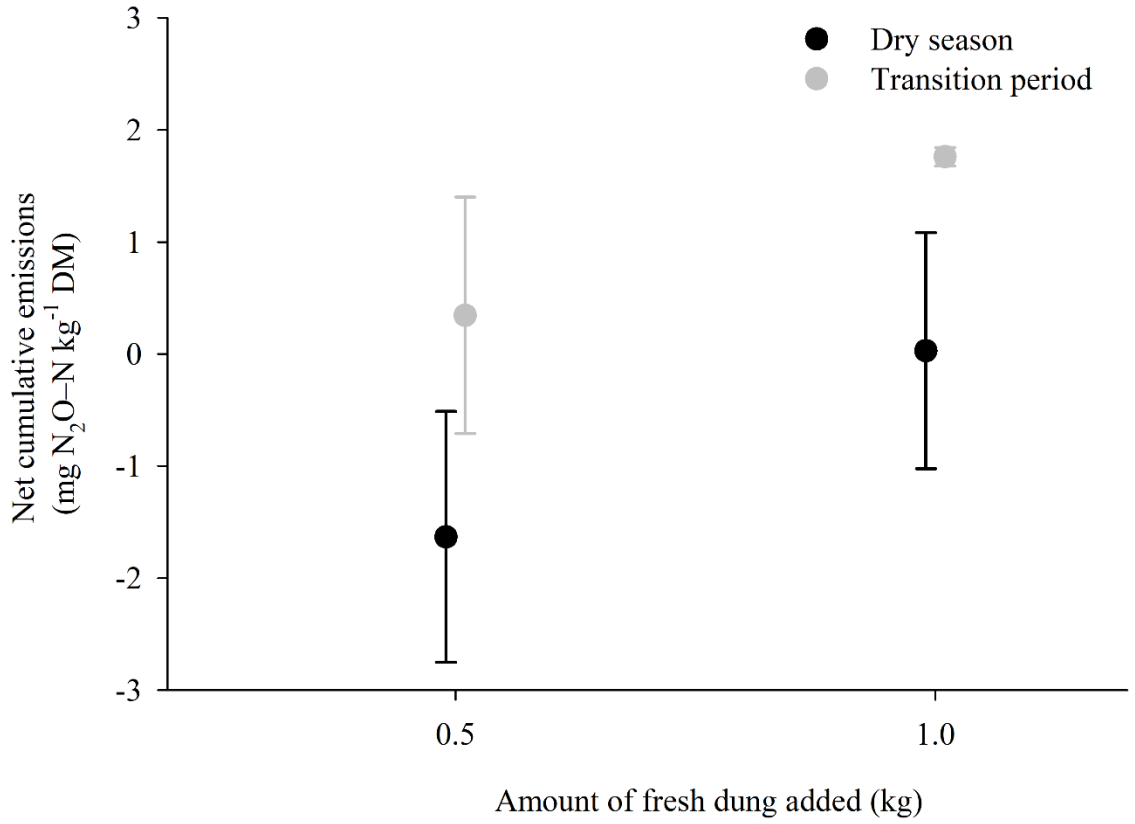

Figure 4 The net cumulative emissions for 0.5 kg and 1.0 kg dung addition to tropical pasture during both dry and transition period.

Similar to the results from experiment I, the N_2O fluxes after dung application were similar to those from the control plots (mostly < 10 µg N_2O-N m^{-2} h^{-1}) and highest fluxes (30 µg N_2O-N m^{-2} h^{-1}) were only observed concomitant with rainfall event in all treatments in experiment II. After observing no effect of period on N_2O emissions for the individual control plots, we calculated and compared the net cumulative emission from treatments with different quality of dung added (Figure 5; **Zhu et al., 2018 [I]**). Contrary to our expectations, no significant effect of dung quality on N_2O emissions was observed. A minor stimulating effect on N_2O emissions after dung

application was also reported by Pelster et al. (2016) and Tully et al. (2017) in Kenyan rangelands. This might be attributed to the low N concentration (9.7-16.5 g N kg^{-1} dry matter) and high C/N ratio (23-41) in the dung used in our experiments compared with developed countries (Chadwick et al., 2018; Krol et al., 2016; Rochette et al., 2014) which might not create adequate conditions for denitrification. As most of the N in dung was in organic form, it can only be used after being mineralized (Pelster et al., 2016). However, the high amount of C and the high C/N ratio might result in rapid soil N immobilization and subsequently less available NO_3^-, and thus reduced N_2O production (Pelster et al., 2012). Another possibility was that a large part of the N lost in the form of NH_3 volatilization due to the low air humidity and intensive radiation in our study site. Laubach et al. (2013) reported that 12% of the N was lost through NH_3 volatilization within 10 days after dung deposition. However, considering the low N concentration in the dung patch in our study, the N loss pathway remains unclear.

The stimulated N_2O emissions after dung deposition mainly occurred in the first two or three weeks after application due to the fast crust formation, as also reported by other studies in tropical areas (Tully et al., 2017; Pelster et al., 2016; Lessa et al., 2014; Mazzetto et al., 2014). After crust formation, the nutrients in the dung was hardly available for the soil microbes, which limited the microbial activity. In addition, even rainfall events hardly enhance nutrient availability after curst formation, which also hampers the N leaching. In our study, the N_2O fluxes from plots where dung was added were similar with those from control plots in the first two weeks, and rainfall did not cause significant differences among treatments. The flux measurement frequency in experiment I and II was > 10 times per day, which enabled us to largely reduce the uncertainties caused by the

high temporal variation and calculate accurate emissions (Barton et al., 2015). Hence, the emission factors calculated based on 25 to 29 days measurements in our study were realistic.

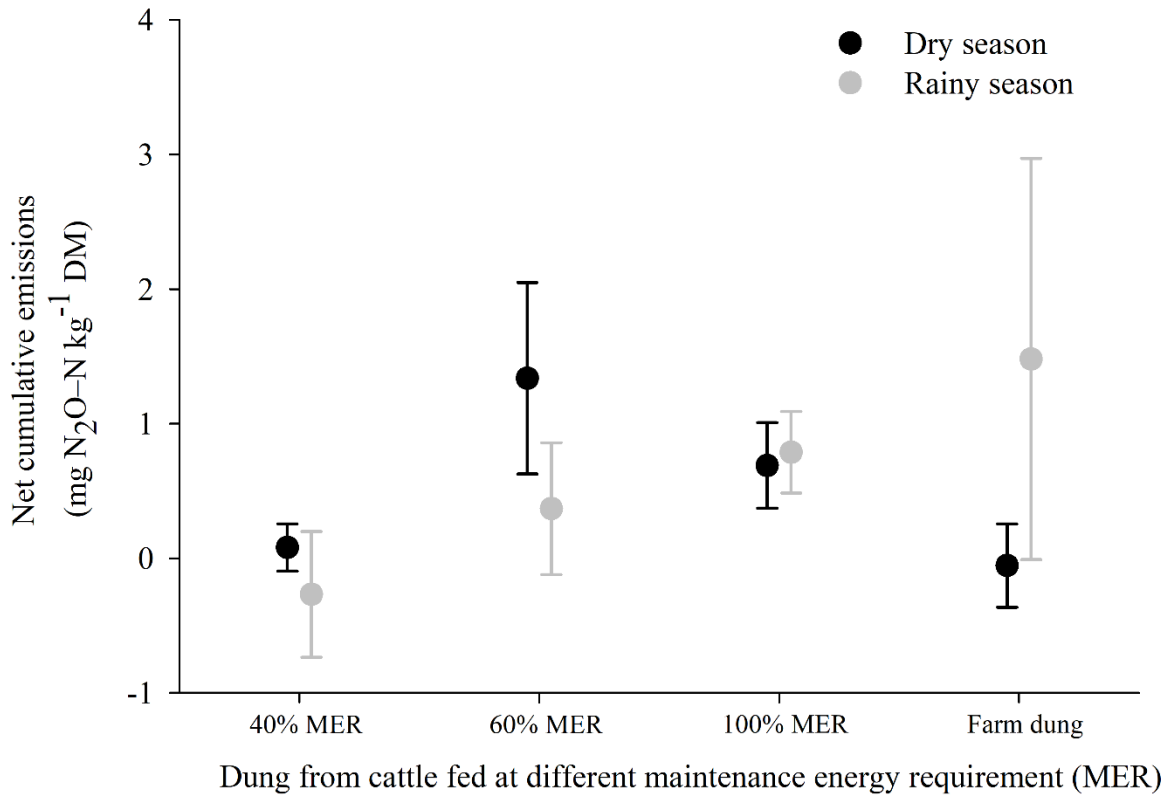

Figure 5 Net cumulative N_2O emissions for dung from steers fed at different MER level added to tropical pasture during both dry and rainy seasons.

The dung N_2O EF ranged from -0.01 to 0.01%, which was negligible (**Zhu et al., 2018 [I]**). This result was consistent with Tully et al. (2017) who reported dung EF of 0.00 and 0.04% in western Kenya during the rainy and dry seasons, respectively. Another study on a rangeland in Nairobi by Pelster et al. (2016) estimated dung N_2O EF in a range of 0.04 and 0.36% and suggested a value of 0.2% for Kenya. A global meta-analysis found a mean cattle dung EF of 0.28% (Cai and

Akiyama, 2016). All of these studies suggest that the IPCC default value of 2% overestimated N_2O emissions for cattle dung (IPCC, 2006). However, our region is in the tropics with precipitation less than 1,000 mm, which is classified as a dry climate and thus our results are within the range of recently refined IPCC 2019 estimates (0.07%) for cattle dung under a dry climate (Kristell et al., 2019).

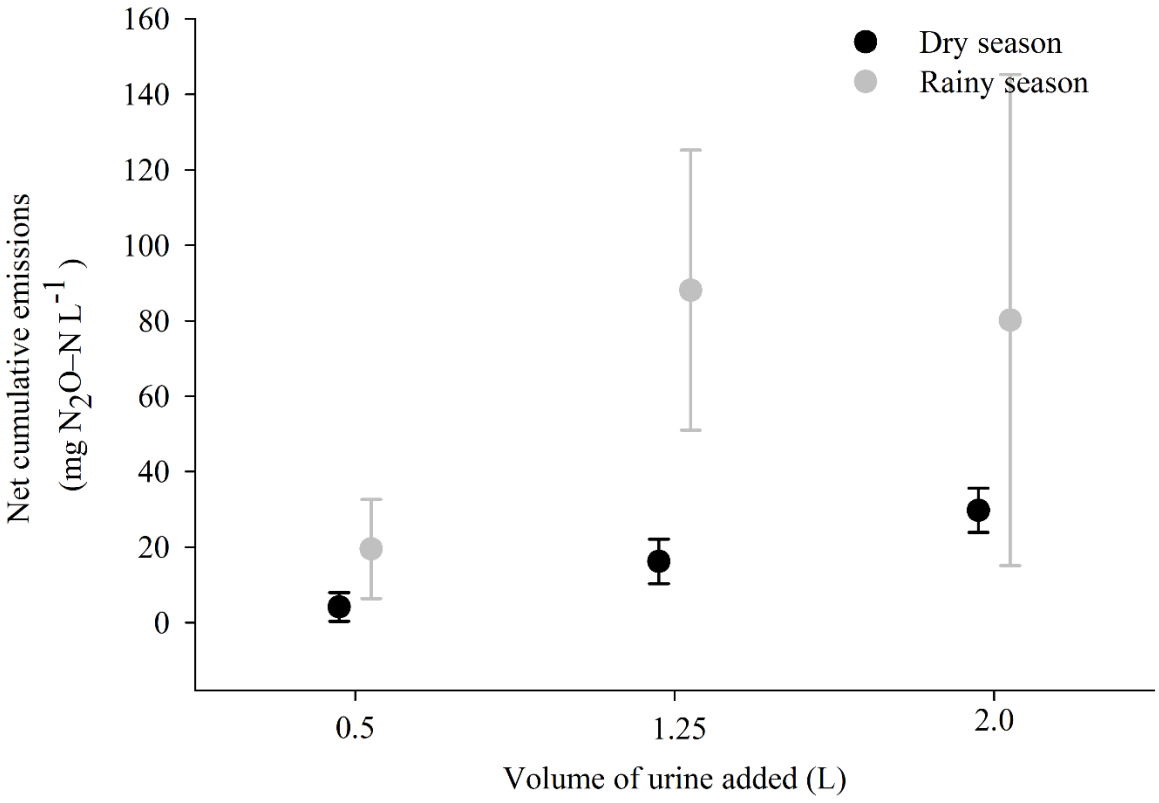

Figure 6 Net cumulative N_2O emissions for different volume of urine addition to tropical pasture during both dry and rainy seasons.

In experiment III, N_2O fluxes were immediately increased after urine application with similar patterns across the 3 volumes. The magnitude of the highest fluxes was differed with 594 µg N_2O-

N m^{-2} h^{-1} from plots receiving 1.25 L urine in rainy season and 108 µg N_2O-N m^{-2} h^{-1} from plots receiving 2.0 L urine in dry season. The net cumulative N_2O emissions were influenced by volume of urine and by seasons with higher value in rainy season (Figure 6). However, no volume effect on urine N_2O EF could be defined in both dry and rainy season.

This does not fit in with our hypothesis, as higher volumes of urine contains more available N which might have a higher EF. Sordi et al. (2014) reported the urine EF decreased with the increment in urine volume and attributed to higher volume urine might percolate deeper into the soil which resulted in less N remained available for N_2O production in top soil. In addition, the N_2O production in deeper profile might be absorbed during upward path and not reach the atmosphere (Neftel et al., 2000).

Previous studies in SSA has shown the main N_2O emissions after urine application occurred within a month (Pelster et al., 2016; Tully et al., 2017). Our observation with automated chamber system for more than two months was robust to capture the pulse N_2O emissions. The urine N_2O EF ranged from 0.04 to 0.40%, with an average of 0.05% in dry season and 0.28% in rainy season. Those values are similar with Tully et al. (2017) who reported that the urine N_2O EF was 0.05% in dry season and 0.18% in rainy season in western Kenya pasture. The aggregated urine N_2O EF for both dry and rainy seasons was 0.17% which was lower compare with 0.32% by the IPCC 2019 refinement for the urine N_2O EF in a dry climate.

3.2 Effect of soil properties on N_2O emissions from excreta deposited onto tropical pasture in Kenya

3.2.1 Soil properties

Based on a soil map for Kenya (Jaetzold et al., 2010), five soil types were selected (Table 5). Intact soil cores (26 cm diameter and 12 cm depth) were taken from fourteen smallholder farms with grazing pasture or rangeland. Meanwhile, soil samples were also taken from the adjacent area for soil property analysis. All of the soil cores and soil samples were immediately transported to Mazingira Centre of the International Livestock Research Institute (ILRI) in Nairobi, Kenya. In addition to the five different soil types, pure sand (washed, dried and sieved under mesh width of 2 mm) was included in this study as a control. Soil cores were put into holes (30 cm diameter, 15 cm deep), where 3 cm sand was placed at the bottom. Sand was also used to fill the gaps between soil core and the hole before inserting the chamber collar.

The soil properties among the five types of soils varied largely. The soil bulk density ranged from 0.94 (Gleysols) to 1.17 g cm^{-3} (Acrisols). The soil texture of different soil types corresponded with the soil map, with the highest clay content of 53% in Ferralsols and the lowest clay content of 29% in Acrisols, while there was opposite trend for sand content with highest sand content of 53 % in Acrisols and lowest sand content of 25% in Ferralsols. All soils were slightly acidic with a narrow range from 5.3 to 6.4 among all soil types. Gleysols had the highest soil organic content (SOC) of 46 g C kg^{-1} dry matter (DM), while the SOC in Cambisols was only 35 g C kg^{-1} DM. The N content was similar, ranging from 2.80 to 3.91 g N kg^{-1} DM across the five soils.

As a control, pure sand had a bulk density of 1.07 g cm^{-3} and a pH of 8.4. The SOC in pure sand was only 0.3 g C kg^{-1} and the N content was negligible.

Table 5 Soil type, farm area, coordinate and soil core collection dates during the two trials (soil classification based on IUSS Working Group WRB, 2014).

Soil type	Area	Longitude	Latitude	Soil cores collection dates	
				Trial 1	Trial 2
Gleysols	Kapnyeberai	35.2°E	0.22°N	19-Feb-18	25-Apr-18
Gleysols	Kapnyeberai	35.2°E	0.22°N	19-Feb-18	25-Apr-18
Gleysols	Songoliet	35.3°E	0.22°N	19-Feb-18	25-Apr-18
Gleysols	Songoliet	35.1°E	0.03°N	19-Feb-18	25-Apr-18
Nitisols	Kapnyeberai	35.2°E	0.22°N	20-Feb-18	26-Apr-18
Nitisols	Kapnyeberai	35.2°E	0.22°N	20-Feb-18	26-Apr-18
Nitisols	Kapnyeberai	35.2°E	0.22°N	20-Feb-18	26-Apr-18
Nitisols	Lelmokwo	35.2°E	0.33°N	20-Feb-18	26-Apr-18
Acrisols	Kiborgok	35.0°E	0.03°N	21-Feb-18	27-Apr-18
Acrisols	Kiborgok	35.0°E	0.02°N	21-Feb-18	27-Apr-18
Cambisols	Tindinyo	35.0°E	0.17°N	21-Feb-18	27-Apr-18
Cambisols	Tindinyo	35.0°E	0.18°N	21-Feb-18	27-Apr-18
Ferralsols	Lelmokwo	35.2°E	0.34°N	22-Feb-18	28-Apr-18
Ferralsols	Lelmokwo	35.2°E	0.35°N	22-Feb-18	28-Apr-18

3.2.2 Experimental design

In experiment IV, intact soil cores of different soil types were collected to investigate the effect of soil properties on N_2O emissions after excreta deposition in both dry and rainy seasons. According

to Krol et al. (2016), the loading rate of urine was 12.5 L m^{-2} and it was 12 kg m^{-2} for dung. Based on our chamber area (0.04 m^2), 0.5 kg dung or 0.5 L urine were used in this study. Besides, our previous studies in Chapter 3.1 and Sordi et al. (2014) reported that N_2O EFs were not influenced by dung mass or urine volume. The same volume of water was also used to eliminate the water effect by urine.

Within each soil type, twelve soil cores were divided evenly into four sets: with no addition, with 0.5 L distilled water addition, 0.5 L urine addition and 0.5 kg fresh dung addition. Trial 1 was conducted from 25 March 2018 to 14 May 2018 and Trial 2 was conducted from 02 July 2018 to 04 October 2018. After a few days of pre-measurements, water, dung and urine were applied at each trial.

Fresh dung was collected from the cattle fenced in the pen with concrete floor each morning of the application day. Fresh urine was collected from three steers with harness connected with tubes into bucket, where 50 ml 20% HCl was placed to acidify the urine (pH below 3) and later stored at -20 °C in a freezer to minimize N loss. Urine was thawed and re-adjusted pH to the original pH with 2 *M* NaOH on the day of application.

Figure 7 Photos of different soil cores with manual static chambers

To measure N_2O gas samples, static chambers with 72 Polyvinyl chloride collars (22.5 cm inner diameter and 13 cm height) inserted 8 cm into the soil and twelve opaque, airtight lid were used (Figure 7). After the lids were clamped to the collars, four 20 ml gas samples were collected by syringe via a sampling port in the center of the lid with 10 minutes interval. The airtight lid was removed after sampling. Gas samples were immediately analyzed for N_2O concentrations using a gas chromatograph (SRI 8610C, SRI Instruments, Torrance, CA, USA) fitted with an electron capture detector (ECD). The N_2O concentration within the samples was calculated by the linear relationship between the peak area and calibration gas concentration. Gas fluxes were estimated by the concentration change over time using linear or non-linear regression while correcting with air pressure and temperature in the study area by idea gas law through the following formula:

$F = b \times M_W \times V_{Ch} \times 60 \times 10^6 / (A_{Ch} \times V_m \times 10^9)$

Where F is the flux rate (μg N_2O–N m^{-2} h^{-1}), b is the slope of concentration change over time (ppb min^{-1}), M_W is the molecular weight (28 g N_2O–N mol^{-1}), V_{Ch} is the chamber volume (m^3), A_{Ch} is the chamber area (m^2), and V_m is the corrected standard gaseous molar volume (m^3 mol^{-1}). After comparing the R^2 value of linear or non-linear regression, N_2O fluxes were finally decided.

3.2.3 Results and discussions

The N_2O fluxes had a small increase after dung application in all soil types. The peaks were higher in rainy season ranging from 78 to 369 μg N_2O-N m^{-2} h^{-1} than 36 to 88 μg N_2O-N m^{-2} h^{-1} in dry season. However, the soil N_2O fluxes responded to the urine addition in a highly variable manner across soils. After urine application, the highest peaks both occurred in Acrisols with 8760 μg N_2O-N m^{-2} h^{-1} during the rainy season and 935 μg N_2O-N m^{-2} h^{-1}in dry season, while the lowest peaks were 542 μg N_2O-N m^{-2} h^{-1} in Sand during rainy season and 33 μg N_2O-N m^{-2} h^{-1} in Ferralsols during dry season, respectively. The elevated N_2O fluxes from urine also lasted for longer compared with dung fluxes (**Zhu et al., 2020 [II])**. Based on a field study conducted in Canada for a whole year, most of the N_2O emissions occur within the first few weeks following excreta deposition (Rochette et al., 2014). The stimulation on soil N_2O fluxes after urine application might be longer. Due to urea hydrolysis after cattle urine application in New Zealand, it was reported that elevated soil NH_4^+-N lasted for 27 days, while soil NO_3^--N concentration returned to the background level after around 40 days (Clough et al., 2009). Similarly, Sordi et al. (2014) showed the N_2O fluxes ascending and descending dynamics lasted for 41±41 days after excreta deposition. As mentioned in the previous study, the elevated N_2O fluxes lasted for around

2 weeks (Pelster et al., 2016; Tully et al., 2017) in SSA. Thus, our results based on 39 and 73 observation days were robust to represent the emissions after excreta application.

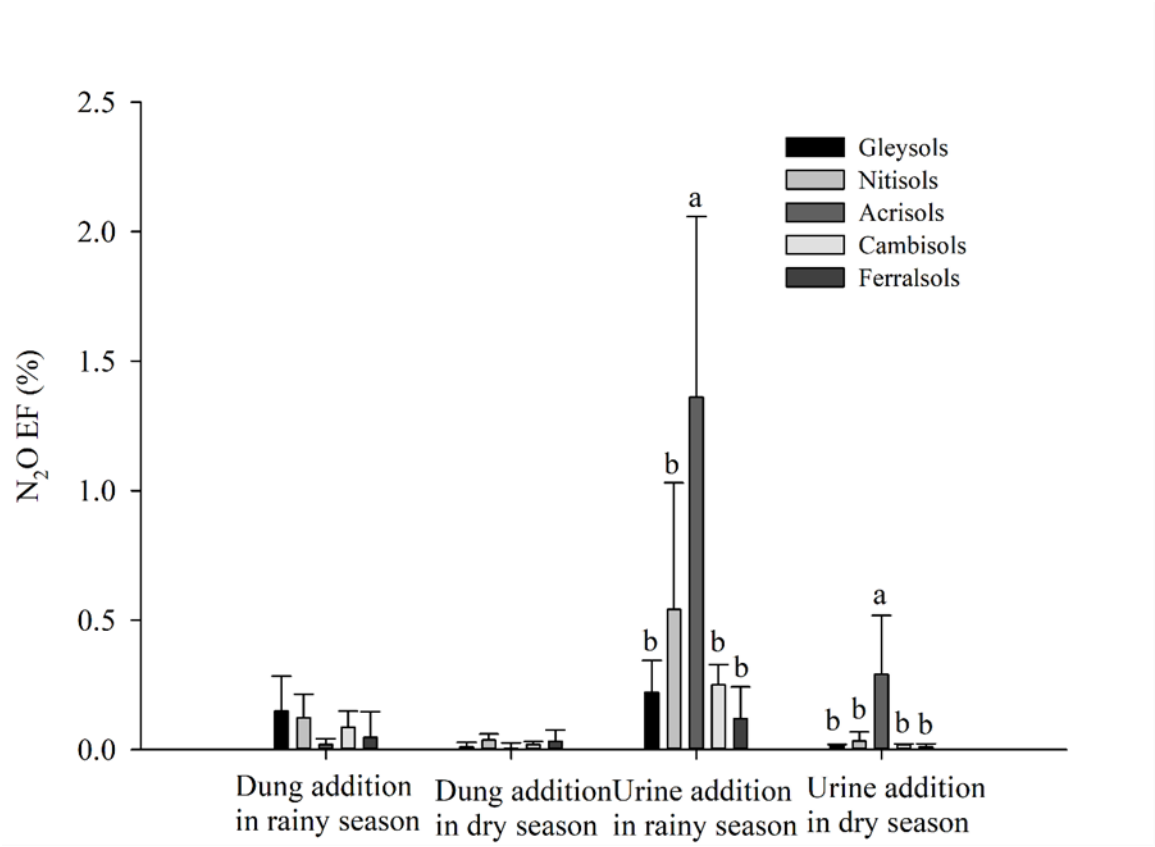

Figure 8 Dung and urine N_2O EF after applied onto different soils during both rainy and dry seasons

The cumulative N_2O emissions from plots with dung addition were similar across all soils and no differences among soil types could be found in both seasons. Therefore, soil type effect on cattle dung EF could not be defined. This is in line with our hypothesis that the N_2O emissions from dung application would not be influenced by the underlying soil properties. This could be partly explained by 1) the low N concentration (12 and 14g N kg^{-1} dry matter), 2) mainly organic N form in dung, which limited N availability and 3) rapid crusting of dung in our study area, which further

hampered incorporation of dung-N into soil (**Zhu et al., 2018 [I])**. The lack of measurable effects on N_2O emissions after dung application was also reported by Van der Weerden et al. (2011) under different soils in three regions in New Zealand. The dung N_2O EFs were higher in the rainy season (0.04-0.15%) relative to what was observed during the dry season (0-0.04%) despite the higher N concentration (14 vs 12g N kg^{-1} dry matter) and lower C/N ratio (29 vs 36) in the dung used in the dry season. More frequent and heavy rainfall in the rainy season hindered the formation of crust (Mazzetto et al., 2014) and the higher soil moisture, which would be more favorable for denitrification and thus higher N_2O production. The average dung EF among the five soils was 0.05% (Figure 8). This value was lower than the documented regional cattle dung EF of 0.28% for Kenya. However, our value was consistent with the value of 0.07% for cattle dung under dry climate from the IPCC 2019 refinement (Kristell et al., 2019).

In contrast to dung, the cumulative N_2O emissions after urine application largely varied across soils in both dry and rainy seasons. The urine N_2O EF ranged from 0.12 to 1.36% in the rainy season and 0.01 to 0.12% in the dry season. The highest N_2O EF was observed for Acrisols and the lowest for Ferralsols in both rainy and dry seasons (Figure 8; **Zhu et al., 2020 [II]**). The higher urine N_2O EF in the rainy season is unsurprising due to the minimal air temperature change across the year and distinct rainfall patterns. Through modulating soil O_2 and nutrient availability (Butterbach-Bahl et al., 2013), higher soil moisture in the rainy season favors coupled nitrification and denitrification, which is the main N_2O production after urine application (Monaghan and Barraclough, 1993). The same results have also been reported in Kenya (Tully et al., 2017) and Brazil (Mazzetto et al., 2014).

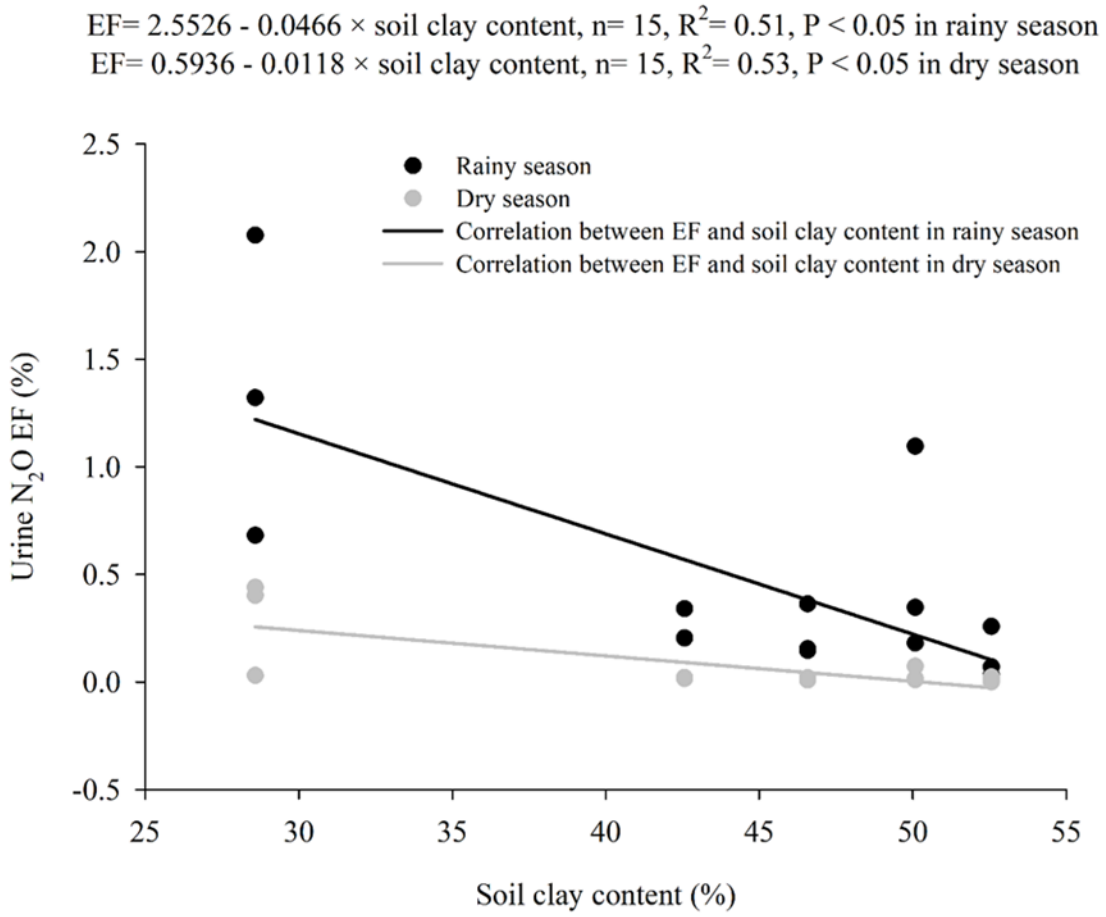

Figure 9 Correlations between urine N_2O EF with soil clay content during both rainy and dry seasons

The different urine N_2O EF could be partly explained by the soil clay content (Figure 9) and soil pH (Figure 10). Higher clay content could retain more water and reduce gas diffusivity, which might promote complete denitrification with the end product of N_2 and lower N_2O emissions (Gu et al., 2013; Weitz et al., 2001). Besides, N_2O emissions from clay-rich soils after urine application are generally low (Jarecki et al., 2008; Zhou et al., 2017) due to the high ammonium sorption capacity, which reduces the NH_4^+ and NO_3^- ion availability (Venterea et al., 2015). Soil pH is normally negatively correlated with N_2O emissions through modulating the N_2O/ N_2 ratio (Wang

et al., 2018). However, the soils used in our study were slightly acidic (pH range: 5.3-6.4) and the positive correlation might due to the stimulated microbial activities due to the pH increase.

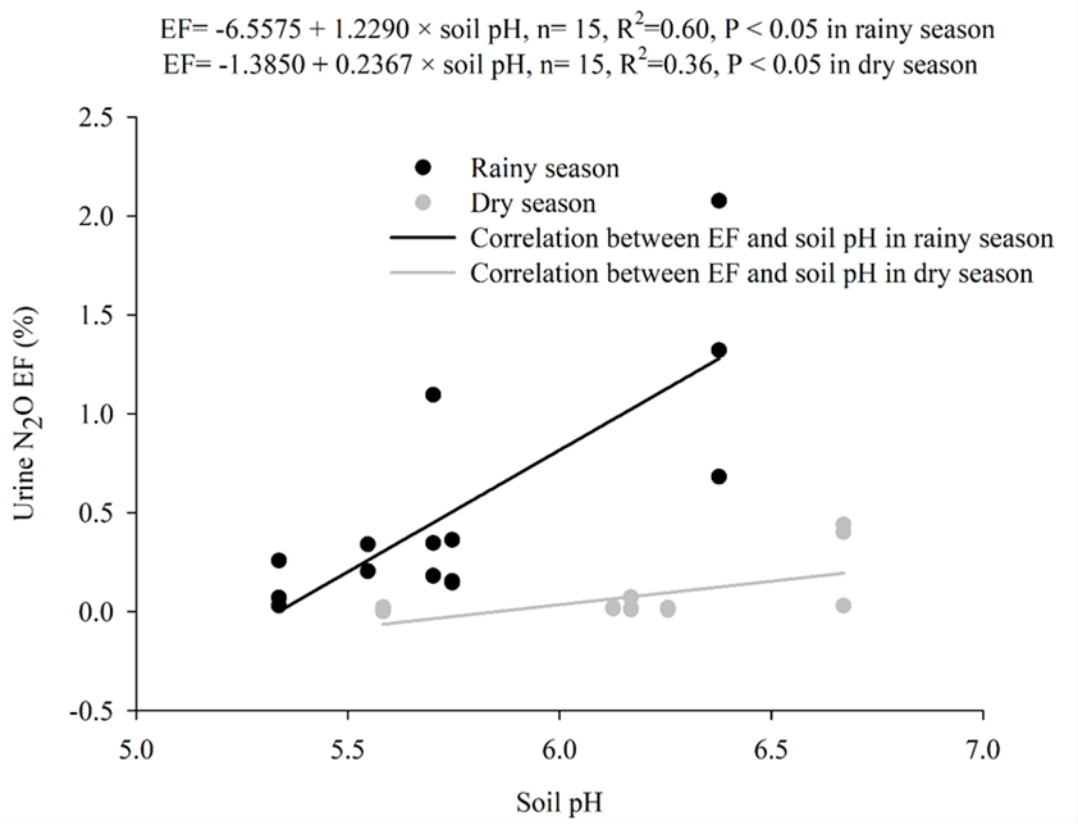

Figure 10 Correlations between urine N_2O EF with soil clay content during both rainy and dry seasons.

The average of cattle urine N_2O EF was 0.29%, which was slightly higher than studies by Barneze et al. (2014) in Brazil with a value of 0.2% and Tully et al. (2017) in Kenya with an average value of 0.13%. Our value was also lower than another study in Kenya that reported the cattle urine N_2O EF of 1.4% (Pelster et al., 2016). However, our value was similar to the 0.32% defined by IPCC 2019 refinement for cattle urine in a dry climate (Kristell et al., 2019).

However, the IPCC 2019 refinement calculated the aggregated default value of 0.2% assuming a urine: dung-N ratio of 0.66:0.34 based on the study in New Zealand (Kelliher et al., 2014). Nonetheless, Wassie et al. (2019) reported that N partitioning between urine and dung varied from 0.33:0.67 to 0.5:0.5 through an animal trial with different MER feed level in SSA. This is in line with a previous study that estimated the average N partitioning between urine and dung as 0.34:0.66 in SSA (Rufino et al., 2006). If we calculate the aggregated value for cattle excreta with the urine-N: dung-N of 0.34:0.66, the aggregated value was 0.13%, which is slightly lower than the value of 0.2% stated by the IPCC 2019 refinement.

Chapter 4

4 Description of manure composition dynamics influenced by manure type and climatic conditions in tropical pasture in Kenya

4.1 Manure properties and weather conditions in the four sites

Manure was collected from the same source but at different dates, which resulted in small differences for the same manure among sampling days. However, manure property was significantly affected by animal species, whereby cattle manure had the highest C concentration (387-431 g C kg^{-1} DM) and the lowest N concentration (13.2-16.4 g N kg^{-1} DM) while sheep manure had the lowest C concentration (355-371 g C kg^{-1} DM) and the highest N concentration (25.2-28.2 g N kg^{-1} DM). Cattle manure also had higher initial hemicellulose and cellulose concentrations and sheep manure generally had higher or similar lignin concentrations among the different sites.

Table 6 Cumulative precipitation, mean air temperature and aridity index during our 378 days observation period at a given site. Aridity index was calculated using the following equation: Aridity index= precipitation/potential evapotranspiration.

Site	Cumulative precipitation (mm)	Mean air temperature (°C)	Aridity index
Taita	328	22.3	0.18
ILRI	1372	18.0	0.92
Kapiti	575	18.6	0.37
Embu	209	22.9	0.12

According to the aridity index (AI), the Taita site (AI = 0.18) and Embu site (AI = 0.12) were defined as arid with higher mean air temperature of 22.3 and 22.9 °C and less precipitation of 328 and 209 mm during our 378-day observation. The Kapiti site had lower mean air temperature of 18.6 °C and more precipitation of 575 mm and defined as semi-arid with AI of 0.37, while the ILRI site had the lowest mean air temperature of 18.0 °C and more precipitation of 1372 mm which was humid with AI of 0.92 (Table 6).

4.2 Experimental designs

In experiment V, three types of manure were used including cattle manure, sheep manure and goat manure. To work on the same dry matter basis (20 g) per litter bag, we used 100 g fresh cattle manure, 70 g fresh sheep manure and 30 g fresh goat manure. Fresh manure was weighed and put into 1-mm nylon mesh bags with the dimensions 12.5 cm×10 cm. These litter bags allow the diffusion of water, nutrient and microbial but do not permit the entry of soil fauna. Manure bags (3 manure type × 3 replicates × 23 sampling dates × 4 farms) were applied on an open and flat area (2.5 m × 6.0 m) with grass on the surface being cut with scissors at the ground level (Figure 11). The experiment was set on 31 March 2018, 06 April 2018, 11 April 2018 and 12 April 2018 at Taita, ILRI, Kapiti and Embu sites, respectively.

After setting up, three bags for each manure type were sampled weekly for the first 7 weeks, then bi-weekly in the rainy seasons and monthly in the dry seasons. In total, 23 sampling dates over a 378-day period were conducted at each of the four sites.

The decomposition rate (k, year^{-1}) of manure was calculated using the following equation:

$$k = \ln \frac{M_t / M_0}{-1/t}$$

where M_t and M_0 are the mass of manure at the time of t and 0, respectively, and t is the time in the year.

Figure 11 Manure decomposition trial at Kapiti Research Station in South Central Kenya

4.3 Manure decomposition

The manure decomposed following the same pattern across manure types and sites, with a rapid decrease in dry matter during the first weeks, after which the decrease rate slowed down over time. Consequently, the total C and N content also rapidly decreased in the first weeks. The concentrations of hemicellulose in manure tended to decrease while concentrations of lignin tended to increase. Compared with N, more C was lost, which resulted in a decrease in C/N ratios over time. Generally, cattle manure had higher decomposition rates (0.288-0.989 k, year^{-1}) than

for sheep manure (0.200-0.457 *k*, year^{-1}) or goat manure (0.234-0.750 *k*, year^{-1}) at all sites. These values are in line with a study by Markewich et al. (2010) which reported rates of decomposition for cattle manures in western Kenya of 0.907 and 1.214 *k*, year^{-1} for medium- and low-quality cattle manure, respectively (**Zhu et al., 2020 [III])**. However, contrary to our hypothesis, cattle manure with a lower initial N concentration and a higher initial C/N ratio – which normally negatively correlated with decomposition (Bradford et al., 2016) – had higher decomposition rates than sheep and goat manure. This might be explained by the higher initial concentrations of cellulose, which were around 25% in cattle manure, but <20% in sheep and goat manures. In addition, lignin compounds are more recalcitrant to decompose than other plant components (Morvan et al., 2006) which could provide physical protection to N-containing compounds (Fioretto et al., 2005; de Bruijn and Butterbach-Bahl, 2010). This might be the reason why lignin concentration increased in cattle manure and slightly increased or remained constant in goat and sheep manure after 378 days decomposition.

The decomposition rates for the same manure were generally high at the ILRI site. This could be partly explained by the higher cumulative precipitation at ILRI (Table 6 and Figure 12). Higher water availability in manure and soil caused by more frequent and heavier rainfall could stimulate microbial biomass and activities, and thus the decomposition process (Jacobson et al., 2015; Gliksman et al., 2017). Besides, more frequent and heavier rainfall might have increased the nutrient leaching from litter bags (Wang et al., 2009).

By contrast, warmer sites receiving less precipitation resulted in the manure losing moisture and forming a crust quickly, which led to slowly decomposing. In addition, the air temperature (18.0-22.9) was similar and not the limited factor, so water availability is the key factors in such

condition. Prieto et al. (2019) also reported that the interactive effect of warming and rainfall reduction was negative on litter decomposition in a semi-arid shrubland and explained to both indirect desiccating effect of warming and direct reduce of water input.

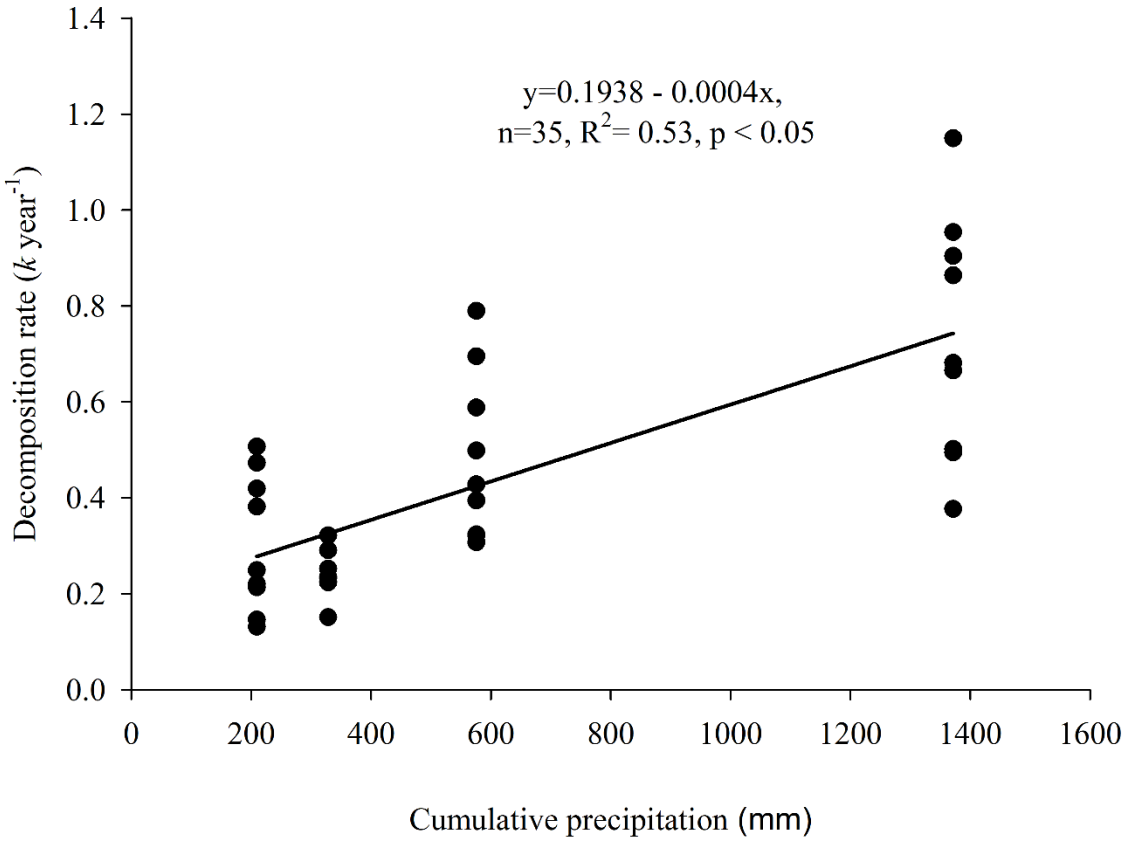

Figure 12 Linear regression between cumulative precipitation and manure decomposition rate

To better reflect the water balance at each site, the aridity index was calculated based on the weather information during our observation period, and it also showed a positive correlation with decomposition rates similar to precipitation (Table 6 and Figure 13).

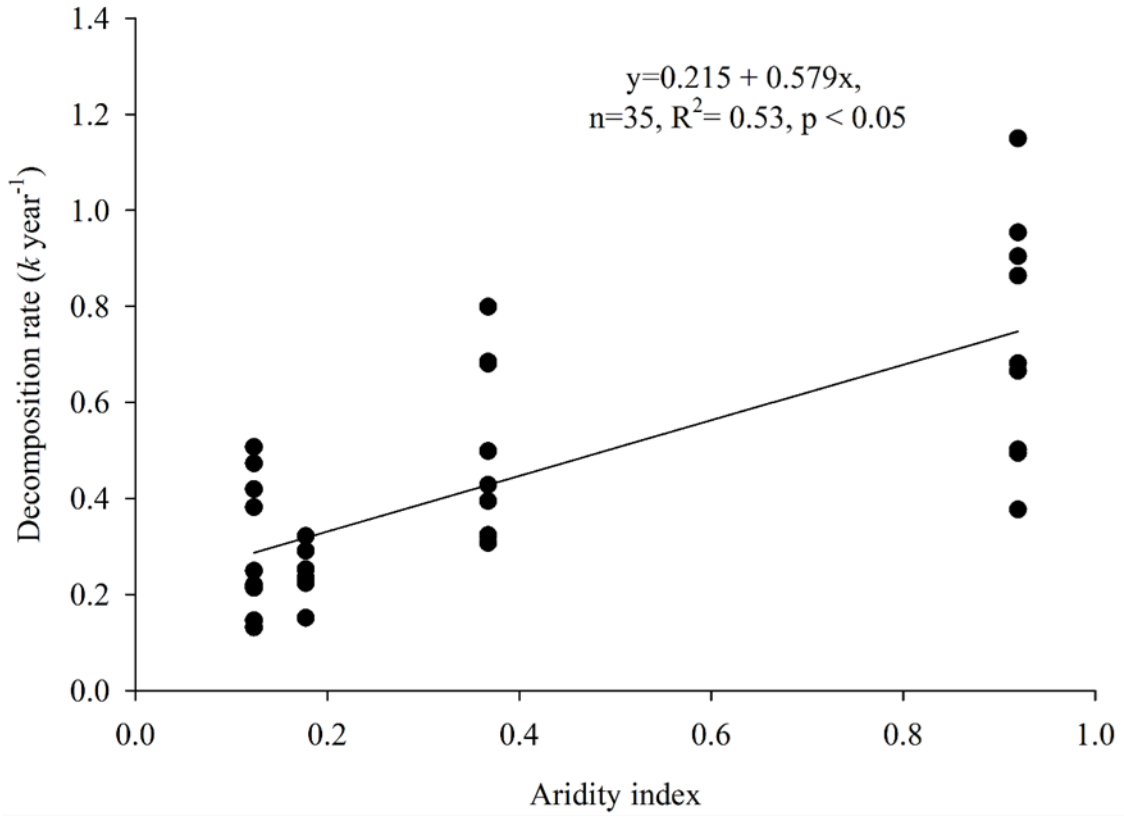

Figure 13 Linear regression between aridity index and manure decomposition rate

Chapter 5

5 Conclusion and outlook

Even though livestock systems are important N_2O sources both globally and especially in SSA, to date the data availability is insufficient to reliably assess livestock-related N_2O emissions in most SSA countries. This leads to large uncertainties in global N_2O emission inventory as emissions from livestock excreta deposition account for 32% of global N_2O emissions from agricultural sources. Using state-of-art instruments such as an automated chamber system, we developed a specific regional N_2O emission factor (EF) for excreta in Kenya through a series of field studies considering the diversity of management practices, soils and climate.

Smallholder farms in SSA are highly diverse in feed diet and animal live weight as different animal species. Based on an animal feeding trial by our colleagues at ILRI, Mazingira, the differences on dung mass, urine volume and the N concentration in dung were observed from the steers fed at different MER level. However, these differences did not lead to any difference in N_2O EF, which means that a single EF for dung or urine can be used for Kenya regardless of the quantity and quality of dung or urine patches.

Furthermore, soil type affected urine N_2O EF, but not dung. The urine N_2O EF largely varied among soil types, ranging from 0.12 to 1.36% in the rainy season and 0.01 to 0.29% in the dry season. These differences were likely associated with the soil clay content and soil pH. Our results clearly demonstrate that soil type has to be considered when estimating the excreta EF after deposition onto grassland. However, the current IPCC methodology does not take soil into account when estimating the N_2O EF for cattle excreta, which might cause large uncertainty.

Considering the effects of both excreta properties and soil properties, the average dung N_2O EF of 0.05% and urine N_2O EF of 0.29% are close to the IPCC 2019 refinement for cattle dung (0.07%) and urine (0.32%) in dry climate. However, the IPCC 2019 refinement calculated the aggregated excreta EF of 0.2% assuming the N partitioning between urine and dung as 66:34 based on the value from New Zealand. This is not the case for SSA due to the low quality of feed diet and a urine-N: dung-N ratio generally below 50:50. Using a urine-N: dung-N ratio of 34:66, the aggregated cattle excreta EF was 0.13%, which was lower than the IPCC value. The N partitioning between urine and dung has to be considered regionally, as this value was estimated as 60:40 in Europe. To reduce uncertainty, specific regional excreta EF and disaggregated EF for urine and dung are still recommended.

More and faster C loss than N loss was observed through a 378-day manure decomposition experiment. Cattle manure decomposed faster than that of sheep or goat manure due to the initial concentration of easily-decomposed cellulose and recalcitrant lignin. More cumulative precipitation increased manure decomposition. As the air temperature was similar in our four sites, the aridity index is more strongly influenced by precipitation and also showed positive correlation with manure decomposition. Understanding the C and N dynamics after manure deposition could better quantify the N_2O sources and develop a more effective mitigation way. However, 30% of cattle manure and 50% sheep or goat manure still remained after the 378-day experiment. To improve the grassland productivity, manure needs to be better utilized, especially in SSA, due to the depletion of N in soils and minimal artificial fertilizer application.

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RESEARCH ARTICLE
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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.,

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).

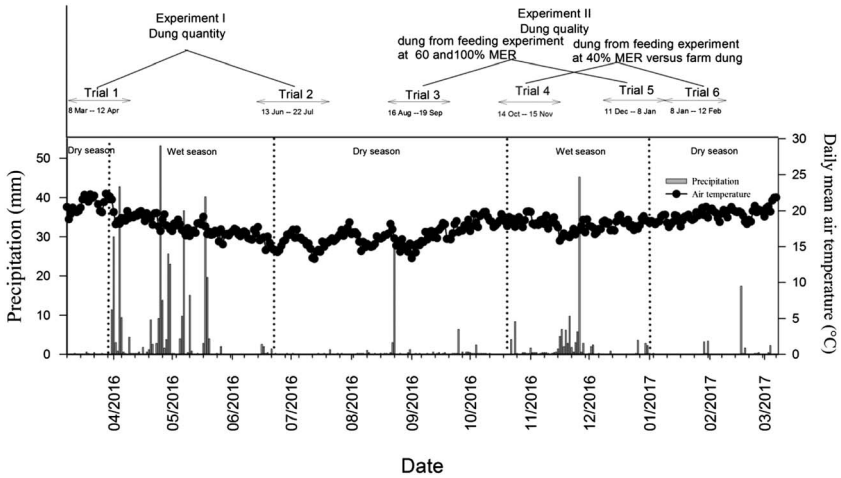

Figure 1. Duration and time of the individual experiments. MER stands for maintenance energy requirement. The lower panel shows air temperature (markers) and precipitation (bars) from 8 March 2016 to 7 March 2017. To note: There were no predung application measurements for the very first part of the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

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

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):

$$\mathrm{CH_4EF}\ (\%) = \frac{\text{Cumulative } CH_4 \text{ emssion } (g\ CH_4\text{-}C) \text{ from dung appplication-Cumulative } CH_4 \text{ emission } (g\ CH_4\text{-}C) \text{ from control}}{\text{Carbon content in applied dung } (g\ C)} \times 100$$

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

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).

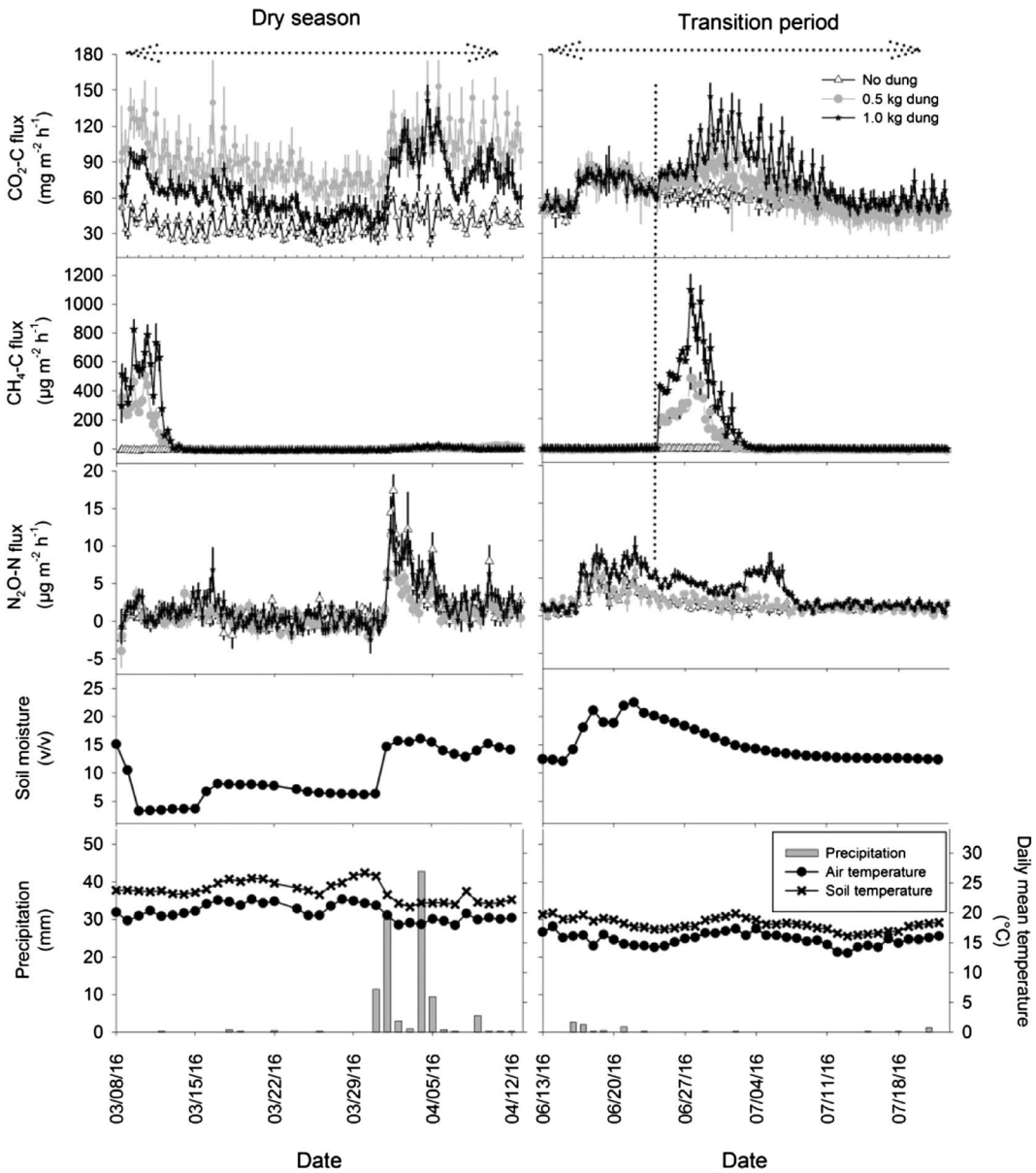

Figure 2. Dynamics of CO_2-C, CH_4-C, and N_2O-N fluxes as affected by additions of different amounts of farmyard dung to grassland. The lower panels show the observed temporal dynamics of mean daily soil moisture (0.05-m depth), soil temperature (0.05-m depth), air temperature, and the daily sum of 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	Net cumulative emissions	Cumulative emissions	Net cumulative emissions	Cumulative emissions	Net cumulative emissions
		Mg CH_4–C m^{-2}	Mg CH_4–C kg^{-1} DM	g CO_2–C m^{-2}	g CO_2–C kg^{-1} DM	Mg N_2O–N m^{-2}	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

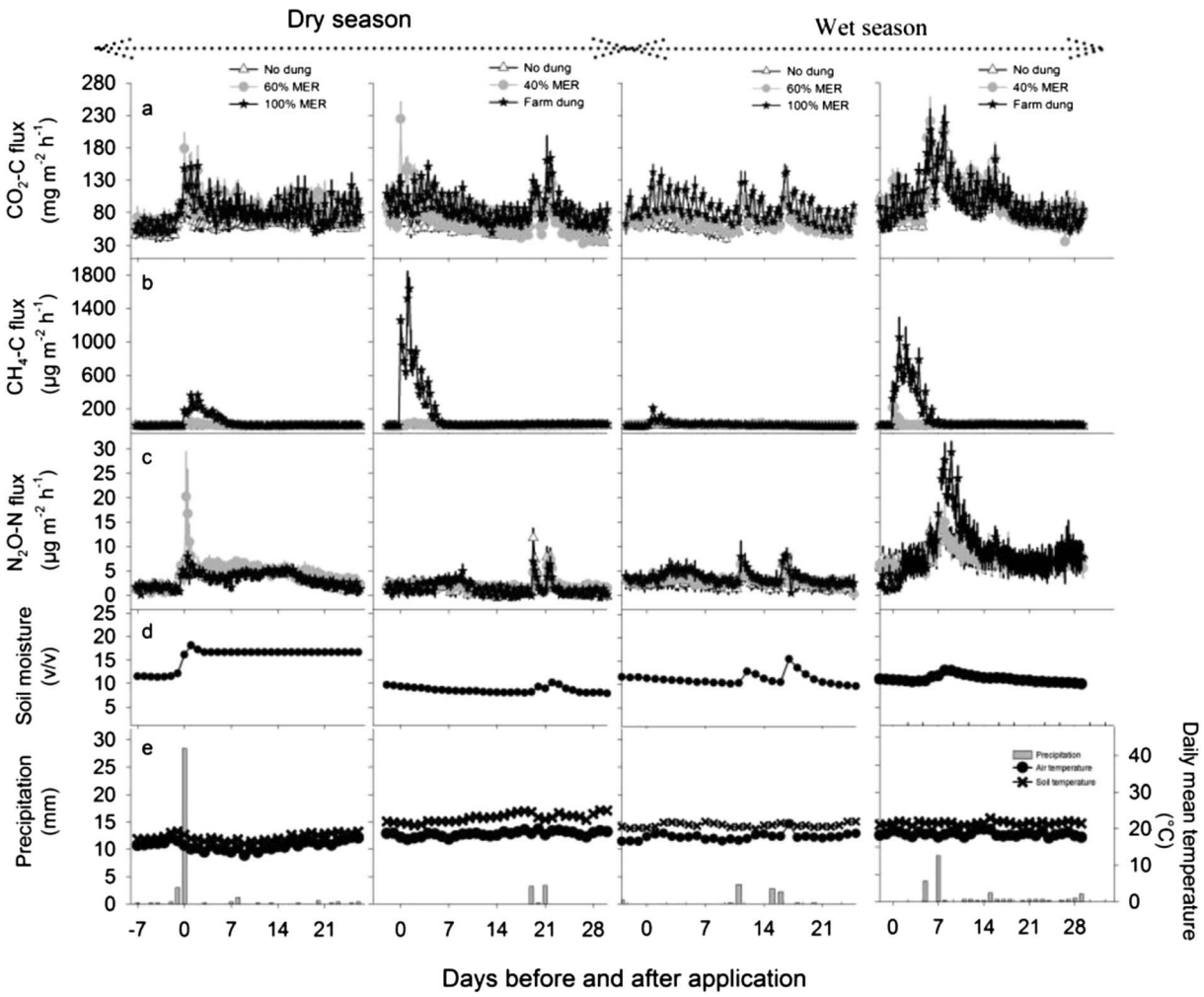

Figure 3. Dynamics of (a) CO_2-C, (b) CH_4-C, and (c) N_2O-N fluxes from grassland soils to which dung of different quality was added (control: no dung; dung from cattle fed at 40, 60, or 100% MER and farm dung). The lower panels show the observed temporal dynamics of (d) mean daily soil moisture (0.05-m 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	Net cumulative emissions	Cumulative emissions	Net cumulative emissions	Cumulative emissions	Net cumulative emissions
		Mg CH_4–C m^{-2}	Mg CH_4–C kg^{-1} DM	g CO_2–C m^{-2}	g CO_2–C kg^{-1} DM	Mg N_2O–N m^{-2}	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

Period	40% MER		60% MER		100% MER		Farm cattle	
	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

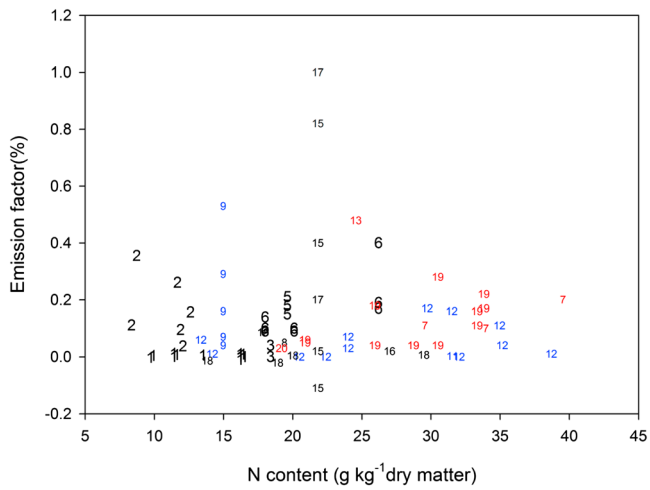

Figure 4. Relationship of the cattle dung N content with the N_2O emission factor in our (1) and previous studies (2–20). The numbers refer to individual studies as listed in Table 6. The numbers in bold and with increased font size refer to studies in tropical regions. The colors refer to the length 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.

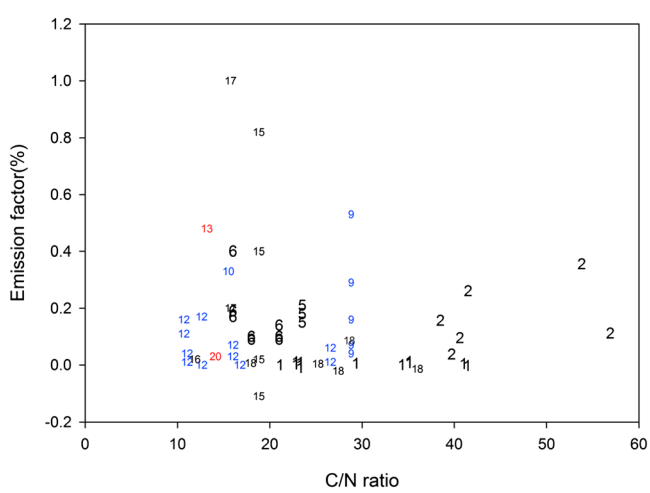

Figure 5. Relationship of the cattle dung C/N ratio with N_2O emission factors reported in our (1) and previous studies (2–20).The numbers refer to individual studies as listed in Table 6. The numbers in bold and with increased font size refer to studies in tropical regions. The colors refer to the length 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

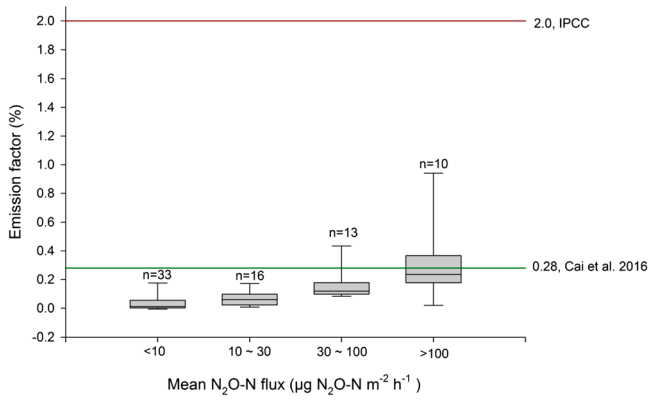

Figure 6. N_2O emission factors for different mean N_2O-N flux classes (0–10, 10–30, etc.) from dung applied to grasslands in previous studies as well as our studies. Note: The horizontal lines indicate the Intergovernmental Panel on Climate Change Tier 1 default value and the estimated emission factor 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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Influence of soil properties on N_2O and CO_2 emissions from excreta deposited on tropical pastures in Kenya

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ABSTRACT

Urine and dung patches deposited by grazing cattle on grassland are an important source of nitrous oxide (N_2O). While a number of studies have investigated the effects of excreta on soil N_2O fluxes in developed economies and in China, observations in sub-Saharan Africa (SSA) are scarce. Moreover, the effects of soil properties (e.g. pH or texture) on N_2O emissions from excreta patches have hardly been studied. In this study we investigated the importance of soil properties on N_2O and carbon dioxide (CO_2) emissions from cattle excreta (dung, urine, and manure [dung + urine]) for five typical tropical soils in Kenya. For this, intact soil cores were translocated from Western Kenya (Nandi county) to Nairobi, where N_2O and CO_2 fluxes were measured over four individual periods (two during dry seasons and two during wet seasons). Fluxes were measured for between 25 and 73 days following surface application of excreta, depending on how quickly emissions returned to baseline. Both dung and manure applications led to increased CO_2 and N_2O fluxes during both dry and wet seasons. On average, the N_2O emission factor (EF) for manure was higher than for dung. The EFs during the wet season were higher for both the dung (0.12%) and urine (0.50%) compared to the dry season EFs (0.01% and 0.07% for dung and urine respectively). Soil type had no measurable effect on N_2O and CO_2 emissions for either dung or manure application. In contrast, soil clay content was negatively ($P < 0.05$) and pH positively ($P < 0.05$) correlated with N_2O emissions after urine application. Assuming an excreta-N ratio of dung to urine of 66:34, as evidenced in earlier studies for SSA, and averaging across all treatments and soils, we calculated a cattle excreta N_2O EF of 0.14%, which is one magnitude lower than the IPCC default N_2O EF of 2%. Our results call for a revision of the IPCC guidelines for calculating N_2O emissions from excreta deposition on tropical rangelands.

1. Introduction

Nitrous oxide (N_2O) is a potent greenhouse gas with a global warming potential 265 times greater than that of carbon dioxide (CO_2) on a per mass basis over 100-years (IPCC, 2014), and it is estimated to account for 6% of total anthropogenic global warming (Davidson, 2009). Current atmospheric concentrations of N_2O are approximately 18% higher than pre-industrial levels and concentrations were projected to further increase by 35–60% between 2007 and 2030 (Smith et al., 2007). More specifically, annual N_2O emissions from excreta (i.e. dung and urine) deposited by grazing livestock were estimated to be 1.5 Tg N_2O–N yr^{-1}, which is equivalent to 41% of all N_2O emissions derived from global livestock production systems and 22% of total anthropogenic N_2O emissions (Oenema et al., 2005; Taghizadeh-Toosi et al., 2011). Smallholder mixed-crop and pastoral livestock systems, which rely on native pasture grazing are widespread across sub-Saharan Africa (SSA), and support an estimated 123 million cattle (Thornton and Herrero, 2014). Due to the free grazing in daytime, it is estimated that in SSA more than 40% of total cattle excreta are deposited on rangelands (Rufino et al., 2006). As a result, direct N_2O emissions from urine and

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dung excreta on SSA pastures were estimated to cause 62% of total annual N_2O emissions in the region (Tully et al., 2017).

Up to 75–90% of nitrogen (N) ingested by grass-fed animals is returned to the soil through dung and urine excretion (Saggar et al., 2013; Bell et al., 2015). As adult cattle can excrete as much as 23 kg dung and 21 L urine over 13 dung and ten urine patches per day (Haynes and Williams, 1993), the amount of N contained in individual urine or dung patches by far exceeds plant N demands in the deposited area (Di and Cameron, 2007; Chadwick et al., 2018). In fact, the amount of N in a dung patch was calculated to be equivalent to up to 1130 kg N ha^{-1} (Saarijärvi et al., 2006), while N loading rates for urine patches were found to be equivalent to 613 kg N ha^{-1} for dairy cattle and 345 kg N ha^{-1} for beef cattle (Selbie et al., 2015). Thus, high environmental N losses due to nitrate (NO_3^-) leaching, ammonia (NH_3) volatilization, or other gaseous emissions (N_2O, nitric oxide [NO] and dinitrogen [N_2]) are often associated with N excretion on rangeland (Chadwick et al., 2018).

Although there are many studies focusing on N_2O emissions from excreta patches, most of these have focused either on timing (Lessa et al., 2014; Rochette et al., 2014; Bell et al., 2015) or application rate (Van Groenigen et al., 2005a; Sordi et al., 2014; Zhu et al., 2018) of excreta, while neglecting the influence of soil properties on N_2O emissions. However, the magnitude and temporal dynamics of soil N_2O emissions are determined not only by soil N availability but also by key soil properties (Butterbach-Bahl et al., 2013; Neira et al., 2015; Samad et al., 2016; Wang et al., 2018a,b; Ghezzehei et al., 2019). For example, soil texture and soil organic carbon content (SOC) affect the water holding capacity and therefore the gas diffusivity of soils. Both factors are crucial parameters controlling the availability of oxygen (O_2) in soils and consequently soil microbial processes, thus N_2O production (Davidson et al., 2000; Butterbach-Bahl et al., 2013; Balaine et al., 2016). Consistent with this, two other previous studies suggested that certain soil properties such as soil texture, pH and bulk density may affect denitrification rates and soil N_2O emissions from livestock excreta applied to pasture soils (Cai and Akiyama, 2016; Wang et al., 2018a,b). However, to our knowledge, no previous study has systematically explored how soil properties may affect N_2O emissions from excreta patches.

In general, N_2O emissions from animal excreta on grasslands have been found to scale linearly with the dung mass (Zhu et al., 2018) or urine volume (Sordi et al., 2014), with higher N_2O emissions from cattle urine compared to cattle dung due to higher N availability in urine and greater interactions of urine with the soil microbial communities as it infiltrates into the soil (Cai et al., 2017).

Depending on diet, N partitioning between cattle dung and urine is thought to range from 50:50 to 25:75 (Valk, 1994; Webb and Misselbrook, 2004; Van der Weerden et al., 2011; Chadwick et al., 2018). The split depends on the crude protein (CP) intake and concentration in feedstuffs. In Western Kenya, CP of feed ranges from 3.2 to 14% (Onyango et al., 2019), which is much less than that in intensive production systems such as in the USA where CP concentrations range from 17 to 23% (Council, 2015; Korir et al., 2016). The low CP concentration of feeds in SSA therefore results in an average N partitioning between cattle dung and urine of 66:34 (Rufino et al., 2006).

Given the differences in N_2O emission factors (EF) between dung and urine (Cai et al., 2017), disregarding excreta-N partitioning might cause large uncertainties when estimating regional N_2O emission inventories. However, the Intergovernmental Panel on Climate Change (IPCC) guidelines do not disaggregate the EF for urine and dung, rather they propose a default N_2O EF of 2% excreta-N (IPCC, 2006). As such, many recent studies have suggested that disentangling N_2O emissions from urine and dung is critical to improve our understanding of N_2O emissions from grazed pastures, better assess this key source for atmospheric N_2O and identify potential mitigation options (Van der Weerden et al., 2011; Krol et al., 2016; Chadwick et al., 2018). Despite differences in climate and soils, most countries in SSA, due to the scarcity of local and regional studies, still use the IPCC default value to estimate their country-level greenhouse gas (GHG) inventories, even though this may not accurately reflect the specific conditions of SSA (Ogle et al., 2014). The objectives of our study were to: a) examine the influence of soil type on the CO_2 and N_2O emissions from single dung, urine or manure patches after deposition on tropical pasture; b) quantify the cumulative CO_2 and N_2O emissions from urine and dung applications to the five soil types; and c) examine whether CO_2 and N_2O emission magnitudes from dung, urine or manure patches would differ between dry and wet seasons. We hypothesized that: a) soil properties such as pH or texture have minimal effects on N_2O emissions from dung but significant effects on N_2O emissions from urine deposited onto pasture; b) soils with a high SOC would have higher N_2O emissions from urine deposited onto pasture than soils with low SOC; c) N_2O emissions would be higher and effects of soil properties would be stronger during the wet than during the dry season; and d) soil CO_2 fluxes would be stimulated by all additions of excreta.

2. Materials and methods

2.1. Soil and site description

For our study, five soil types including (i) poorly-drained Gleysols, (ii) well-drained Nitisols, (iii) well-drained Acrisols, (iv) well-drained Cambisols, and (v) well-drained Ferralsols (IUSS Working Group WRB, 2014) differing in SOC (34–45 g C kg^{-1} dry matter), clay content (29–53%) and pH (5.3–6.4) were selected in Nandi County, Western Kenya (Table 1). Sampling locations were selected based on a soil map for Kenya (Jaetzold et al., 2010) and the willingness of smallholder farmers to participate in the study. In total, 14 farms with grazing pastures or rangelands were chosen. There were four farms each for Nitisols and Gleysols, while two farms each were allocated for the other soil types. Before taking the cores, grass was cut down to 2 cm above the soil surface, after which intact soil cores with a diameter of 26 cm and a depth of 12 cm were collected with spades. In the area immediately adjacent to the soil cores, we took soil samples using 100 cm^3 soil cylinders to measure soil bulk density (BD) and with a 4.5 cm diameter soil auger to 5 cm depth to measure soil carbon (C), N and pH (for a detailed description see Saiz and Albrecht, 2016).

The intact soil cores were carefully wrapped in plastic bags and put into 50 L-buckets. Small holes were made to allow for gas exchange, and the cores were immediately transported to the Mazingira Centre of the International Livestock Research Institute (ILRI) in Nairobi, Kenya (S 1°16′13"; E 36°43′23"; altitude 1809 m asl). At the study site, soil cores were embedded into a flat grassland dominated by a mixture of Kikuyu grass (*Pennisetum clandestinum* Hochst. ex Chiov.) and Rhodes grass (*Chloris gayana* Kunth), i.e. grass species that were also found in the swards of the sampled rangelands. In addition to the five different soil types, dried and sieved (mesh width 2 mm) sand was included in this study as a control. Unfortunately, the sand contained high amount of soil, which still supported microbial activities. The soil cores were placed into holes (30 cm diameter, 15 cm deep) in an existing grassland immediately adjacent to the Mazingira Centre that had been lined with the same sand at the bottom. The sand was also used to fill the gaps on the sides of the cores in order to separate the soil cores from the adjacent soil. This procedure did not likely significantly affect N movement in the soil as the main rooting depth of the sward was generally less than 15 cm and as vertical N transport with the soil water movement was not influenced. After installation, soil cores were left to settle for three weeks to allow for equilibration with environmental conditions at the ILRI site. Thereafter, the grass was cut down to 2 cm again and excreta applied in the following days. Note that before the start of each trial, new soil cores were obtained from the field sites in Nandi County, transferred to ILRI, Nairobi, and placed in newly opened holes to avoid any legacy effects from the previous trial(s). A meteorological station was installed to record precipitation (tipping rain gauge, ECRN-100 high-resolution, Decagon, Pullman, WA; USA) and air temperature

Table 1
Soil pH, bulk density, carbon and nitrogen concentration, C/N ratio, soil sand and clay content of the different soils used in the experiment.

Soil type	pH	Bulk density (g cm^{-3})	C content (g kg^{-1} dry matter)	N content (g kg^{-1} dry matter)	CEC (cmol kg^{-1})	C/N ratio	Sand (%)	Clay (%)
Gleysols	6.0 ± 0.3^{bc}	0.94 ± 0.13^{b}	45.6 ± 9.3^{a}	3.91 ± 0.92^{a}	151 ± 16^{a}	12 ± 0.5^{bc}	27.2 ± 14.4^{bc}	46.6 ± 2.9^{a}
Nitisols	5.9 ± 0.5^{c}	0.95 ± 0.11^{b}	42.9 ± 8.7^{a}	3.63 ± 0.93^{ab}	147 ± 62^{ab}	12 ± 0.9^{b}	25.2 ± 5.0^{c}	50.1 ± 5.3^{a}
Acrisols	6.4 ± 0.6^{b}	1.17 ± 0.14^{a}	35.2 ± 6.1^{b}	3.12 ± 0.64^{bc}	137 ± 37^{ab}	11 ± 0.6^{c}	52.7 ± 1.4^{a}	28.6 ± 21.4^{b}
Cambisols	5.6 ± 0.4^{d}	1.10 ± 0.13^{a}	34.7 ± 6.5^{b}	2.89 ± 0.69^{c}	85 ± 43^{bc}	12 ± 0.8^{b}	37.7 ± 2.8^{b}	42.6 ± 4.2^{a}
Ferralsols	5.3 ± 0.5^{e}	0.98 ± 0.08^{b}	37.0 ± 5.1^{b}	2.80 ± 0.61^{c}	67 ± 7^{c}	14 ± 1.7^{a}	24.5 ± 1.4^{c}	52.6 ± 4.3^{a}
Sand	8.4 ± 0.0^{a}	1.07 ± 0.11^{ab}	0.3 ± 0.1^{c}	0.00 ± 0.00^{d}		–	–	–

Values are mean ± standard deviation (n = 3). Different lowercase letters indicate significant differences within columns ($P < 0.05$).

and humidity (ATMOS 14, Decagon, Pullman, WA; USA) at a 5 min resolution. Soil moisture and temperature at 0.05 m depth were also measured during gas sampling (Decagon 5TM sensors, Decagon, Pullman, WA; USA).

2.2. Experimental design

In this experiment, four separate trials were conducted; two during the wet season and two during the dry season. Trial 1 was conducted from 26 July 2016 to 25 August 2016 (dry season) with eight cores of six different soils, resulting in a total of 48 soil cores. We applied 0.5 kg of fresh dung to the soil surface of four cores of each soil type, while the other four served as controls (i.e. no application, Fig. 1). For the following three trials, 12 new soil cores of the same six soils were installed, giving a total of 72 soil cores for each of the remaining three trials. Trial 2 was conducted from 16 October 2017 to 01 December 2017 (wet season). The 12 soil cores of each soil type were divided evenly into three groups that received either no application, or a surface application of either 0.5 kg fresh dung or 0.5 kg fresh manure (dung + urine) (Fig. 2). Trial 3 was conducted from 25 March 2018 to 14 May 2018 (wet season, Fig. 3), while the fourth trial was conducted from 02 July 2018 to 04 October 2018 (dry season, Fig. 4). In these two trials, 12 soil cores of each soil type were divided evenly into four groups with either no application, or with 0.5 L distilled water, 0.5 L urine or a surface application of 0.5 kg fresh dung. Excreta application was done following a few days of background measurements (i.e. prior to excreta application) at the beginning of each trial (see Figs. 1–4). Excreta application rates were scaled to the chamber area (0.04 m^2) based on previous studies that found that emissions scale linearly with mass/volume of applied excreta (Sordi et al., 2014; Zhu et al., 2018). Due to an initial lack of urine-collection harnesses, urine collection was only possible in 2018. Therefore, urine and soil interaction effects on soil N_2O fluxes could only be studied in trials 3 and 4.

Annual rainfall distribution at Nairobi, Kenya (mean 1982–2012: 869 mm) is bimodal with a long rainy season from the end of March to

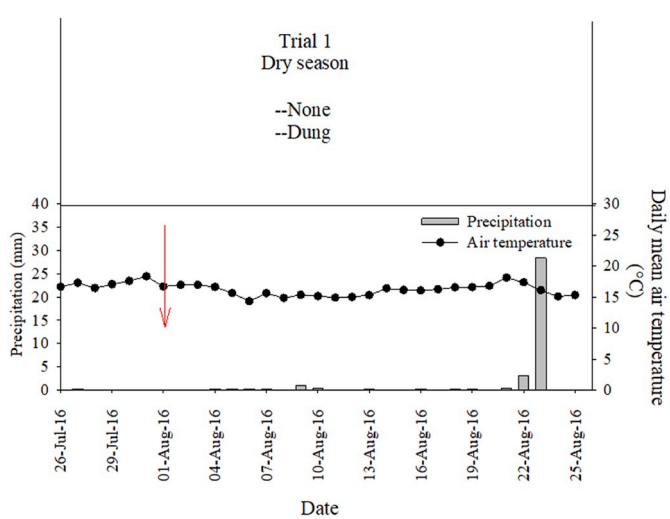

Fig. 1. Observation time and treatments in Trial 1. The lower panel shows air temperature (°C) and precipitation (mm). Red arrows indicate application dates for dung. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

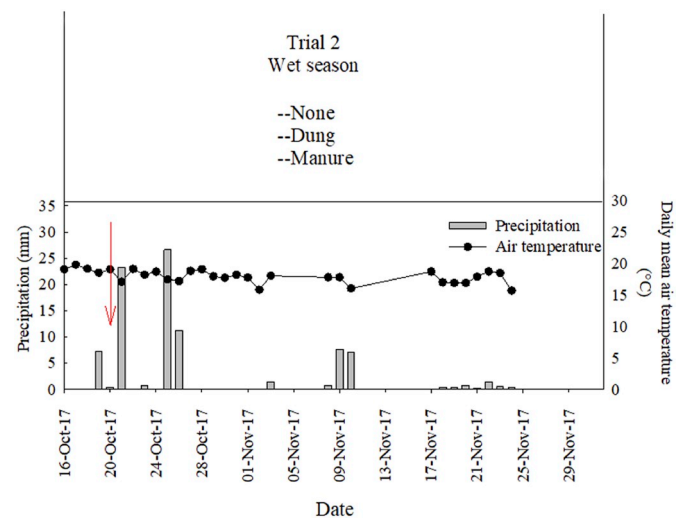

Fig. 2. Observation time and treatments in Trial 2. The lower panel shows air temperature (°C) and precipitation (mm). Red arrows indicate application dates for dung or manure. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

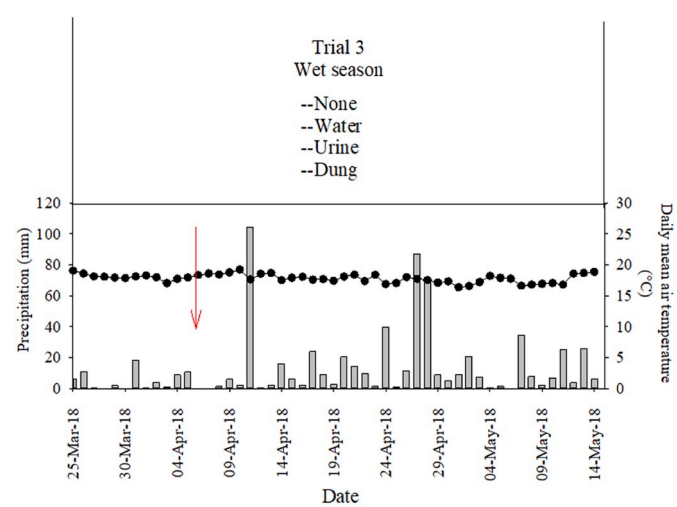

Fig. 3. Observation time and treatments in Trial 3. The lower panel shows air temperature (°C) and precipitation (mm). Red arrows indicate application dates for excreta (or water). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

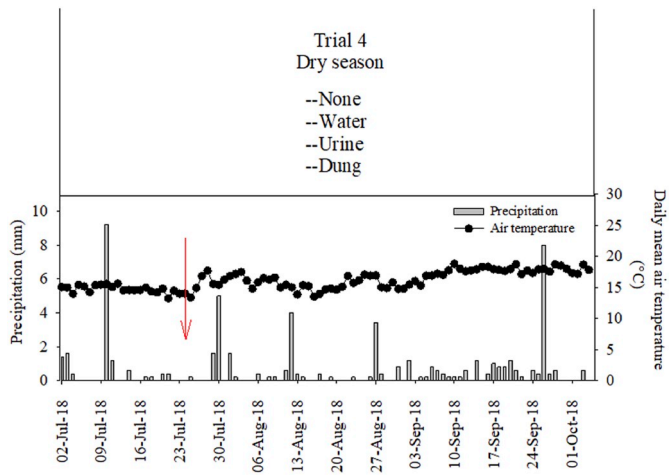

Fig. 4. Observation time and treatments in Trial 4. The lower panel shows air temperature (°C) and precipitation (mm). Red arrows indicate application dates for excreta (or water). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the end of June and a short rainy season from October to December. Trials 1 and 4 were conducted during the dry season, with 34.8 mm precipitation in 25 days (trial 1) and 41.6 mm precipitation in 73 days (trial 4); trial 2 was conducted during the short rainy season with 83.2 mm precipitation in 43 days and trial 3 was conducted during the long rainy season with 596.4 mm precipitation in 39 days. Across our four observation periods, the mean temperatures were 16.0, 17.8, 17.8 and 16.5 °C during trials 1, 2, 3 and 4, respectively (Figs. 1–4).

Fresh dung and manure were collected at the ILRI farm from a cattle herd that was grazed on pasture dominated by a mixture of Kikuyu grass and Rhodes grass during the day and then housed in single-animal pens at night, where they had access to hay from Kikuyu and Rhodes grass. Excreta were collected from the concrete floors of each pen in the mornings, after the animals had been taken outside for grazing. To avoid contamination of dung with urine, only the upper half of dung cakes was collected from the floor. To collect manure (dung + urine), a waterproof plastic sheet was placed on the concrete floor, and the edges were raised upward to capture all the urine and dung from individual animals overnight, which was then homogenized in a bucket. Dung and manure were collected immediately before application to prevent nutrient losses during storage. Fresh urine was collected from three steers, each fitted with collection harnesses. The three urine samples were pooled after collection and the pH was measured immediately. Urine was acidified with 50 ml 20% HCl to a pH ≤ 3 to minimize N loss during collection and storage and then frozen at −20 °C until application. Before application, urine was thawed and the pH re-adjusted to the original pH with 2 *M* NaOH.

2.3. Measurements of CO_2 and N_2O fluxes

For CO_2 and N_2O fluxes measurements, we used closed static chambers (Norman et al., 1997). Polyvinyl chloride collars with 22.5 cm inner diameter and 13 cm height were inserted 8 cm into the soil on the plots with the soil cores. During chamber deployment, the collars were covered with an opaque, airtight lid and four 20 ml gas samples were collected via a sampling port in the centre of the lid at times 0, 10, 20, and 30 min. The airtight lid was removed between samplings. Gas samples were immediately analyzed for CO_2 and N_2O concentrations using a gas chromatograph (SRI 8610C, SRI Instruments, Torrance, CA, USA) equipped with a flame ionization detector (FID) for CO_2 and an electron capture detector (ECD) for N_2O. The oven was operated at 70 °C, and the ECD and FID were heated to 340 and 350 °C. The carrier gas (pure N_2) had a flow rate of 25 mL min^{-1}. CO_2 (5%) in 95% N_2 was used as purge gas (3 ml min^{-1}) for the ECD to avoid cross-interferences of the N_2O signal with CO_2 (Zheng et al., 2004). Greenhouse gas fluxes were calculated on basis of temporal changes of CO_2 and N_2O concentrations in the headspace of closed chambers. Slopes were calculated either using linear or non-linear regression analysis, using R^2 values as decision criteria (Yao et al., 2015). Corrections for air pressure and headspace temperature were applied as described by Wolf et al. (2010). The flux detection limits were 2.3 μg N_2O–N m^{-2} h^{-1} and 1.3 mg CO_2–C m^{-2} h^{-1} (Parkin et al., 2012). The fluxes were calculated by the following formula:

$$F = (b \times M_W \times V_{Ch} \times 60 \times 10^6) / (A_{Ch} \times V_m \times 10^9)$$

Where *F* is the flux rate (mg CO_2–C m^{-2} h^{-1} or μg N_2O–N m^{-2} h^{-1}), *b* is the slope of concentration change (ppm min^{-1} or ppb min^{-1}), M_W is the molecular weight of component (12 g CO_2–C mol^{-1} or 28 g N_2O–N mol^{-1}), V_{Ch} is the chamber volume (m^3), A_{Ch} is the chamber area (m^2), and V_m is the corrected standard gaseous molar volume (m^3 mol^{-1}).

The sampling scheme was as follows: Before excreta application CO_2 and N_2O fluxes were measured for at least three individual days. Following excreta application for the first trial, gas fluxes were measured twice per day for three days, then every two days for the next two weeks, and every three days for the last eight days. For the other three trials, fluxes were measured after excreta application daily for the first week, then every two days for the next two weeks, and three or four days per time for the remaining days. To capture the entire CO_2 and N_2O emissions of the applied excreta, samples were collected for a minimum of one additional week (sometimes longer) after CO_2 and N_2O fluxes had returned to background in all treatments, i.e. to levels as observed for the unamended soil cores. In the majority of cases fluxes returned to background levels within two or three weeks after application of dung or manure. Still, it needs to be noted that during the third trial (wet season) fluxes in the urine treatment only returned to background levels after five weeks. Accordingly, we extended the observation time for the fourth trial to make sure that the entire period with elevated fluxes was captured. In total, CO_2 and N_2O fluxes were measured on 17 occasions/14 days (trial 1), 19 days (trial 2), 18 days (trial 3), and 30 days (trial 4).

Cumulative emissions were calculated using trapezoidal integration. Net cumulative emissions were calculated by subtracting cumulative emissions from control plots, i.e. plots not receiving excreta, from cumulative emissions of plots with excreta. The N_2O EF (i.e. the proportion of applied N emitted as N_2O) was calculated by dividing the net cumulative emissions by the amount of added excreta-N according to the equation:

$$N_2O\ EF\ (\%) = \frac{\text{Cumulative } N_2O \text{ emission}(g\ N_2O - N) \text{ from excreta application} - \text{Cumulative } N_2O \text{ emission } (g\ N_2O - N) \text{ from control}}{\text{Nitrogen content applied as excreta } (g\ N)} \times 100$$

2.4. Soil and excreta analysis

For soil BD determination soil was sampled with a 100 cm^3 ring, the oven-dried at 105 °C for 24 h, and weighed. Soil pH was measured using

a glass electrode in a water:soil suspension of 2.5:1 (weight:weight). Soil CEC was measured by ETH Zurich using $BaCl_2$-TEA method (Hendershot et al., 2007). Water content of dung and manure was determined by drying fresh samples in the oven at 105 °C until constant weight. For total C and N, soil, dung or manure samples were dried at 50 °C for three days, ground and analyzed with an elemental combustion system (Costech International S. p.A., Milano, Italy). Total urine N concentration was analyzed by the Kjeldahl method (Kirk, 1950). Soil texture was estimated based on the soil particle size distribution as analyzed by the hydrometer technique (van Reeuwijk, 2002; Scrimgeour, 2008).

2.5. Data analysis

Two-way ANOVA with Tukey's HSD test was used to test for differences of cumulative emissions from unamended control plots and excreta amended plots and for testing differences between soils during the same trial. The differences of N_2O EF for urine, dung and manure among soils in the same trial were also tested using two-way ANOVA with Tukey's HSD test. All statistical calculations were done in R v3.5.3 (R core team, 2019). The linear regression between urine EF and soil clay content or pH was conducted by Sigmaplot 12.5 (Systat Software, Inc. SigmaPlot for Windows).

3. Results

3.1. Soil properties

The pH of the different soils sampled from individual farms ranged from 5.3 ± 0.5 (Ferralsols) to 8.4 ± 0.0 (Sand) (Table 1). Soil BD varied from 0.94 ± 0.13 g cm^{-3} (Gleysols) to 1.17 ± 0.14 g cm^{-3} (Acrisols). The Gleysols had the highest C and N concentrations (45.6 g ± 9.3 C kg^{-1} dry matter [DM] and 3.91 ± 0.92 g N kg^{-1} DM), while the Sand had the lowest C and N concentrations (0.3 ± 0.1 g C kg^{-1} DM and 0.0 ± 0.0 g N kg^{-1} DM). There were large differences in the proportion of sand and clay in the different soils, with the sand content ranging from 24.5% (Ferralsols) to 52.7% (Acrisols) and clay content ranging from 28.6% (Acrisols) to 52.6% (Ferralsols) (Table 1).

3.2. Excreta properties

Dung properties used in the different trials varied substantially (Table 2) because the quality of the grasses in the pasture differed between dry and wet season. As a result, the dung used in trial 1 (dry season) had the highest N content (18.10 ± 0.12 g N kg^{-1} DM) and lowest C content (397.2 ± 0.2 g C kg^{-1} DM), while the lowest N concentration and highest C concentration was measured in the dung used for the third trial (wet season), which had 11.75 ± 0.09 g N kg^{-1} DM and 422.7 ± 0.3 g C kg^{-1} DM (Table 2). Consequently, the dung C/N ratio also varied significantly from 22 (trial 1, dry season) to 36 (trial 3, wet season). Compared to dung, manure had a higher water content, and lower C and N concentrations. Urine N concentration also varied between the two trials and ranged from 6.42 ± 0.23 g N L^{-1} (trial 3, wet season) to 5.69 ± 0.14 g N L^{-1} (trial 4, dry season) (Table 2).

3.3. Background N_2O and CO_2 emissions from different soils

Cumulative CO_2 emissions from the unamended soils only differed significantly between soil types in trial 3 (wet season, 2018) with the lowest emissions from the Sand (1186 ± 235 kg CO_2–C ha^{-1} 39 $days^{-1}$) and the highest from the Acrisols (3234 ± 968 kg CO_2–C ha^{-1} 39 $days^{-1}$) ($P < 0.05$). Cumulative N_2O emissions from the different unamended soils were similar in all of the trials (Tables 3–6). Soil N_2O fluxes were largely similar or below the detection limit (2.3 μg N_2O–N m^{-2} h^{-1}) during the dry season (trial 1 and 4; Figs. 5 and 8) and the only notable fluxes were following a re-wetting by a 28 mm rainfall at the end of trial 1. As expected, rainfall did not cause elevated N_2O fluxes for the sand plots (Fig. 5). During the wet season, soil CO_2 and N_2O fluxes were higher than during the dry season (see Figs. 5–8).

3.4. Influence of soil type and type of excreta on CO_2 and N_2O emissions

In all trials, dung and manure application increased CO_2 fluxes from all soils, with higher fluxes observed during wet seasons than during dry seasons (Figs. 5–8). The highest CO_2 flux of 890 ± 70 mg CO_2–C m^{-2} h^{-1} was measured in Acrisols amended with manure in trial 2 (Fig. 6). Urine application increased CO_2 fluxes more than either dung or water application within 1–2 days in all soils. However, the stimulative effect of urine additions differed between soils, ranging from 456 ± 35 (Gleysols) to 1464 ± 175 mg CO_2–C m^{-2} h^{-1} (Ferralsols) in trial 3 (Fig. 7), and from 71 ± 3 (Acrisols) to 1026 ± 81 mg CO_2–C m^{-2} h^{-1} (Gleysols) in trial 4 (Fig. 8).

The N_2O fluxes also increased following either dung or manure application. Among dung and manure applications, the highest N_2O flux of 461 ± 136 μg N_2O–N m^{-2} h^{-1} was measured for manure amended Gleysols in trial 2 (Fig. 6). Compared to unamended control soils, cumulative N_2O emissions from dung application were similar across all soils during all trials (Tables 3–6), while during trial 2 the manure application resulted in higher cumulative N_2O emissions compared to the unamended controls only in the Gleysols and Ferralsols (Table 4; $P < 0.05$).

Neither the water nor the dung application significantly increased cumulative N_2O emissions relative to the unamended control across all soil types and all trials. However, the response to urine additions was highly variable across soils (Tables 5 and 6), with fluxes ranging from 33 ± 38 (Ferralsols – trial 4; Fig. 8) to 8760 ± 1322 μg N_2O–N m^{-2} h^{-1} (Acrisols – trial 3; Fig. 7). Soil N_2O fluxes were not only larger but also lasted for longer (several weeks) for urine as compared to dung additions (Figs. 7 and 8). As a result, cumulative N_2O emissions were higher for urine application compared to dung application in the Acrisols (Trials 3 and 4) and Sand (only trial 4) (Tables 5 and 6). The urine EF was negatively correlated with soil clay content in both dry (EF = 0.5936–0.0118 × soil clay content, n = 15, $R^2 = 0.53$, $P < 0.05$) and wet season (EF = 2.5526–0.0466 × soil clay content, n = 15, $R^2 = 0.51$, $P < 0.05$), while positively correlated with soil pH in both dry (EF = −1.3850 + 0.2367 × soil pH, n = 15, $R^2 = 0.36$, $P < 0.05$) and wet season (EF = −6.5575 + 1.2290 × soil pH, n = 15, $R^2 = 0.60$, $P < 0.05$).

Table 2
Water content, carbon and nitrogen concentrations and C/N ratio of dung and nitrogen concentrations of urine applied to soil cores during the four trials.

Period	Season	Excreta type	Water content (%)	C concentration (g C kg^{-1} DM)	N concentration (g N kg^{-1} DM or g N L^{-1})	C/N ratio
26-Jul — 25-Aug-16	Dry season	dung	82.7 ± 0.1^{a}	397.2 ± 0.2^{d}	18.10 ± 0.12^{a}	22 ± 0.1^{e}
16-Oct — 01-Dec-17	Wet season	dung	80.6 ± 0.2^{c}	412.0 ± 0.2^{b}	13.60 ± 0.04^{c}	30 ± 0.1^{c}
		manure	81.8 ± 0.1^{b}	399.9 ± 0.5^{c}	11.93 ± 0.03^{d}	34 ± 0.1^{b}
25-Mar −14-May-18	Wet season	dung	81.7 ± 0.2^{b}	422.7 ± 0.3^{a}	11.75 ± 0.09^{d}	36 ± 0.3^{a}
		urine	–	–	6.42 ± 0.23^{A}	–
02-Jul — 04-Oct-18	Dry season	dung	78.9 ± 0.4^{d}	410.3 ± 1.5^{b}	14.19 ± 0.17^{b}	29 ± 0.4^{d}
		urine	–	–	5.69 ± 0.14^{B}	–

Values are mean ± standard deviation (n = 3). Different lowercase letters indicate significant differences among dung and manure property and the uppercase letters indicate the significant difference of urine N concentration ($P < 0.05$).

Table 3
Cumulative CO_2 and N_2O emission and dung N_2O EF (% applied excreta-N) over 25 days as affected by addition of 0.5 kg cattle dung to different soil cores in Trial 1 (01–25 August 2016, dry season).

Soil type	Cumulative CO_2 emissions (kg CO_2–C ha^{-1})		Cumulative N_2O emissions (g N_2O–N ha^{-1})		N_2O EF (%)
	None	Dung	None	Dung	Dung
Gleysols	472 ± 100^{Aa}	754 ± 148^{Aa}	22.3 ± 11.0^{Aa}	49.7 ± 44.1^{Aab}	0.01 ± 0.01^{a}
Nitisols	353 ± 117^{Ba}	684 ± 75^{Aa}	11.6 ± 9.3^{Aa}	31.3 ± 17.4^{Aab}	0.00 ± 0.00^{a}
Acrisols	518 ± 149^{Aa}	743 ± 113^{Aa}	71.8 ± 71.4^{Aa}	108.5 ± 77.6^{Aa}	0.01 ± 0.02^{a}
Cambisols	441 ± 69^{Ba}	753 ± 125^{Aa}	8.9 ± 5.4^{Aa}	29.3 ± 7.3^{Aab}	0.00 ± 0.00^{a}
Ferralsols	386 ± 82^{Ba}	747 ± 132^{Aa}	12.7 ± 13.3^{Aa}	21.4 ± 15.1^{Ab}	0.00 ± 0.00^{a}
Sand	505 ± 140^{Aa}	748 ± 121^{Aa}	15.5 ± 15.4^{Aa}	20.9 ± 21.2^{Ab}	0.00 ± 0.00^{a}

Values are mean ± standard deviation (n = 4); different lowercase letters indicate significant differences among soil types within the same treatment and different uppercase letters indicate significant differences among treatment within the same soil type ($P < 0.05$).

Table 4
Cumulative CO_2 and N_2O emission and N_2O EF (% applied excreta-N) over 43 days as affected by addition of 0.5 kg cattle dung or 0.5 kg cattle manure to different soil cores in Trial 2 (20 October — 01 December 2017, wet season).

Soil type	Cumulative CO_2 emissions (kg CO_2–C ha^{-1})			Cumulative N_2O emissions (g N_2O–N ha^{-1})			N_2O EF (%)	
	None	Dung	Manure	None	Dung	Manure	Dung	Manure
Gleysols	2028 ± 226^{Aa}	2635 ± 520^{Aa}	2951 ± 573^{Aa}	127 ± 37^{Ba}	579 ± 296^{ABa}	916 ± 86^{Aa}	0.13 ± 0.08^{Aa}	0.28 ± 0.03^{Aa}
Nitisols	1890 ± 455^{Aa}	2555 ± 101^{Aa}	2705 ± 232^{Aa}	150 ± 38^{Aa}	644 ± 448^{Aa}	765 ± 56^{Aa}	0.14 ± 0.13^{Aa}	0.22 ± 0.03^{Aa}
Acrisols	1997 ± 134^{Aa}	3034 ± 661^{Aa}	3024 ± 526^{Aa}	218 ± 47^{Aa}	571 ± 253^{Aa}	709 ± 479^{Aa}	0.10 ± 0.06^{Aa}	0.17 ± 0.18^{Aa}
Cambisols	1945 ± 405^{Aa}	3093 ± 544^{Aa}	3163 ± 1041^{Aa}	110 ± 30^{Aa}	818 ± 616^{Aa}	814 ± 538^{Aa}	0.20 ± 0.17^{Aa}	0.25 ± 0.18^{Aa}
Ferralsols	1540 ± 286^{Ba}	3092 ± 833^{Aa}	2949 ± 711^{ABa}	158 ± 148^{Ba}	620 ± 154^{ABa}	916 ± 403^{Aa}	0.13 ± 0.07^{Aa}	0.27 ± 0.09^{Aa}
Sand	1959 ± 1273^{Aa}	2273 ± 328^{Aa}	2501 ± 729^{Aa}	179 ± 112^{Aa}	335 ± 203^{Aa}	264 ± 96^{Aa}	0.05 ± 0.07^{Aa}	0.03 ± 0.05^{Aa}

Values are mean ± standard deviation (n = 4); there was no significant differences among soil types within the same treatment; different uppercase letters indicate significant differences among treatment within the same soil type ($P < 0.05$).

3.5. N_2O emission factors for dung, manure and urine

The N_2O EF for dung in trial 1 (dry season) was negligible and similar among all soil types, varying from 0.00 to 0.01% (Table 3), while the N_2O EF for dung was in the range of 0.00–0.20% in all other trial periods (Tables 4–6). Despite a lower dung N concentration, the dung EF was higher during the wet seasons (Trials 2 and 3) than during the dry seasons (Trials 1 and 4; $P < 0.05$). However, the N_2O EF for dung did not differ across soils. Manure was only applied in the second trial (wet season), and during this period the manure N_2O EF ranged from 0.03 to 0.28% with no differences between the soils tested (Table 4).

In contrast to dung and manure, the urine N_2O EF varied largely among the different soils during both wet and dry seasons. The highest urine N_2O EF was observed for Acrisols both in the wet season (1.36%) and dry season (0.29%), while the lowest urine N_2O EF was measured in the Ferralsols (wet season: 0.12%; dry season: 0.01%). Furthermore, in both wet and dry seasons, the urine N_2O EF was markedly higher for Acrisols than EFs for all other soils (Tables 5 and 6; $P < 0.05$).

4. Discussion

4.1. Influence of soil properties on CO_2 and N_2O emissions after excreta application

In our study, the CO_2 fluxes after dung or manure applications were similar and no soil type effect was detected indicating that the majority of CO_2 releases was from the fresh dung or manure itself. This was similar to observations by Lin et al. (2009) after yak dung application under laboratory conditions. The short-lived CO_2 fluxes after urine application were likely associated with the rapid hydrolysis of urine urea (Ambus et al., 2007; Cai et al., 2017), during which CO_2 is released, and did not cause any significant difference in cumulative CO_2 emissions among soils in any of the trials.

Consistent with our original hypothesis, the N_2O EF from both manure and dung application to pasture was not affected by the underlying soil properties such as e.g. soil C/N ratio, SOC or pH. This was likely because the low dung-N concentrations limited N availability for N_2O formation in the dung itself, and because the rapid crusting of the dung hampered incorporation of excreta-N into the soil (Zhu et al., 2018). However, the presence of termites in some of the soil cores was noted shortly after application, which might have altered soil aeration and thus influenced N_2O production and emission. Also, the termites may have transferred some of the dung/manure away from the area covered by the chambers. Therefore, it is possible that the presence of the termites may have contributed to our observation that soil texture and pH did not have measurable effects on soil N_2O production after dung application. The lack of measurable soil effects was consistent with earlier findings by Van der Weerden et al. (2011) who also did not find differences in the N_2O EF for dung in three contrasting regions with differing soil types.

In line with a recent global meta-analysis (Wang et al., 2018a,b), the N_2O EF for urine application to soils was negatively correlated with clay content. Clay particles hold water tightly in soil aggregates. Moreover, clay soils mostly have a low gas diffusivity due to small mean pore sizes (Weitz et al., 2001; Gu et al., 2013). Both factors favor anaerobic conditions in soils and, thus, might promote complete denitrification towards the end product N_2, causing lower N_2O emissions from clay soils (Weitz et al., 2001). In addition, high clay content generally correlates with a high cation exchange capacity. As a result, the mineralized NH_4^+ ion can be adsorbed due to the high CEC or even fixed to the clay minerals (Chantigny et al., 2004), which decreases soil NH_4^+ ion availability for nitrification and, thus, NO_3^- ion production. Thus, the soil ammonium sorption capacity has been found to affect N_2O production in soils which otherwise showed largely similar physical and chemical properties (Venterea et al., 2015). Consequently, a number of studies have found limited N_2O production following urine application to clay-rich soils (Jarecki et al., 2008; Zhou et al., 2017). Contradicting our findings, a study in Canada reported higher urine EFs in a clay-rich soil in comparison with a sandy loam soil and argued that the good soil aeration in the sandy loam soil limited the N_2O production due to denitrification (Rochette et al., 2014).

In addition to soil clay content, soil pH is also an important factor controlling the magnitude of N_2O emissions after N additions (Wang et al., 2018a,b). Wang et al., 2018a,b found in a meta-data analysis on

Table 5
Cumulative CO_2 and N_2O emission and N_2O EF over 39 days as affected by addition of 0.5 L cattle urine or 0.5 kg cattle dung to different soil cores in Trial 3 (06 April −14 May 2018, wet season).

Soil type	Cumulative CO_2 emissions (kg CO_2–C ha^{-1})				Cumulative N_2O emissions (g N_2O–N ha^{-1})				N_2O EF (%)	
	None	Water	Urine	Dung	None	Water	Urine	Dung	Urine	Dung
Gleysols	2890 ± 737^{Aa}	2610 ± 136^{Aa}	2974 ± 296^{Aab}	3881 ± 1125^{Aa}	219 ± 69^{Aa}	177 ± 73^{Aa}	2044 ± 963^{Ab}	640 ± 332^{Aa}	0.22 ± 0.12^{Ab}	0.15 ± 0.13^{Aa}
Nitisols	2540 ± 939^{Aab}	2499 ± 526^{Aa}	2815 ± 57^{Aab}	3168 ± 498^{Aa}	186 ± 120^{Aa}	448 ± 440^{Aa}	5022 ± 4552^{Ab}	533 ± 365^{Aa}	0.54 ± 0.49^{Ab}	0.12 ± 0.09^{Aa}
Acrisols	3234 ± 968^{Aa}	2158 ± 327^{Aa}	3167 ± 305^{Aab}	3244 ± 958^{Aa}	186 ± 210^{Ba}	194 ± 187^{Ba}	11684 ± 5920^{Aa}	241 ± 273^{Ba}	1.36 ± 0.70^{Aa}	0.02 ± 0.02^{Ba}
Cambisols	2457 ± 123^{Aab}	2398 ± 370^{Aa}	3591 ± 230^{Aa}	3578 ± 32^{Aa}	133 ± 119^{Aa}	101 ± 93^{Aa}	2212 ± 659^{Ab}	377 ± 153^{Aa}	0.25 ± 0.08^{Ab}	0.09 ± 0.06^{Aa}
Ferralsols	2078 ± 172^{Aab}	2503 ± 671^{Aa}	2519 ± 126^{Aab}	3300 ± 572^{Aa}	87 ± 35^{Aa}	240 ± 67^{Aa}	1254 ± 960^{Ab}	218 ± 267^{Aa}	0.12 ± 0.12^{Ab}	0.05 ± 0.10^{Aa}
Sand	1186 ± 235^{Bb}	1397 ± 140^{Ba}	1862 ± 347^{ABb}	3484 ± 145^{Aa}	52 ± 18^{Aa}	88 ± 39^{Aa}	1359 ± 196^{Ab}	178 ± 110^{Aa}	0.15 ± 0.03^{Ab}	0.04 ± 0.04^{Aa}

Values are mean ± standard deviation (n = 3); different lowercase letters indicate significant differences among soil types within the same treatment and different uppercase letters indicate significant differences among treatment within the same soil type ($P < 0.05$).

Table 6
Cumulative CO_2 and N_2O emission and N_2O EF over 73 days as affected by addition of 0.5 L cattle urine or 0.5 kg cattle dung to different soil cores in Trial 4 (24 July — 04 October 2018, dry season).

Soil type	Cumulative CO_2 emissions (kg CO_2–C ha^{-1})				Cumulative N_2O emissions (g N_2O–N ha^{-1})				N_2O EF (%)	
	None	Water	Urine	Dung	None	Water	Urine	Dung	Urine	Dung
Gleysols	1589 ± 136^{Aa}	1989 ± 564^{Aa}	1892 ± 482^{Aa}	1716 ± 530^{Aa}	18 ± 25^{Aa}	75 ± 63^{Aa}	180 ± 43^{Abc}	60 ± 52^{Aa}	0.01 ± 0.01^{Ab}	0.01 ± 0.02^{Aa}
Nitisols	1937 ± 435^{Aa}	1635 ± 201^{Aa}	1918 ± 280^{Aa}	2286 ± 629^{Aa}	65 ± 60^{Aa}	77 ± 14^{Aa}	325 ± 255^{Aabc}	210 ± 81^{Aa}	0.03 ± 0.03^{Ab}	0.04 ± 0.02^{Aa}
Acrisols	2430 ± 370^{Aa}	2256 ± 307^{Aa}	2215 ± 256^{Aa}	1826 ± 133^{Aa}	84 ± 57^{Ba}	107 ± 202^{Ba}	2285 ± 1729^{Aa}	100 ± 37^{Ba}	0.29 ± 0.23^{Aa}	0.00 ± 0.02^{Ba}
Cambisols	1939 ± 438^{Aa}	2403 ± 119^{Aa}	2163 ± 598^{Aa}	2399 ± 1040^{Aa}	41 ± 42^{Aa}	82 ± 33^{Aa}	216 ± 49^{Abc}	117 ± 64^{Aa}	0.02 ± 0.00^{Ab}	0.02 ± 0.01^{Aa}
Ferralsols	1922 ± 398^{Aa}	1965 ± 285^{Aa}	2460 ± 380^{Aa}	2144 ± 485^{Aa}	36 ± 20^{Aa}	40 ± 0^{Aa}	120 ± 86^{Ac}	162 ± 156^{Aa}	0.01 ± 0.01^{Ab}	0.03 ± 0.04^{Aa}
Sand	1927 ± 568^{Aa}	1887 ± 264^{Aa}	2075 ± 209^{Aa}	2208 ± 476^{Aa}	77 ± 27^{ABa}	98 ± 57^{ABa}	1211 ± 17^{Aab}	31 ± 36^{Ba}	0.15 ± 0.01^{Aab}	-0.01 ± 0.01^{Aa}

Values are mean ± standard deviation (n = 3); different lowercase letters indicate significant differences among soil types within the same treatment and different uppercase letters indicate significant differences among treatment within the same soil type ($P < 0.05$).

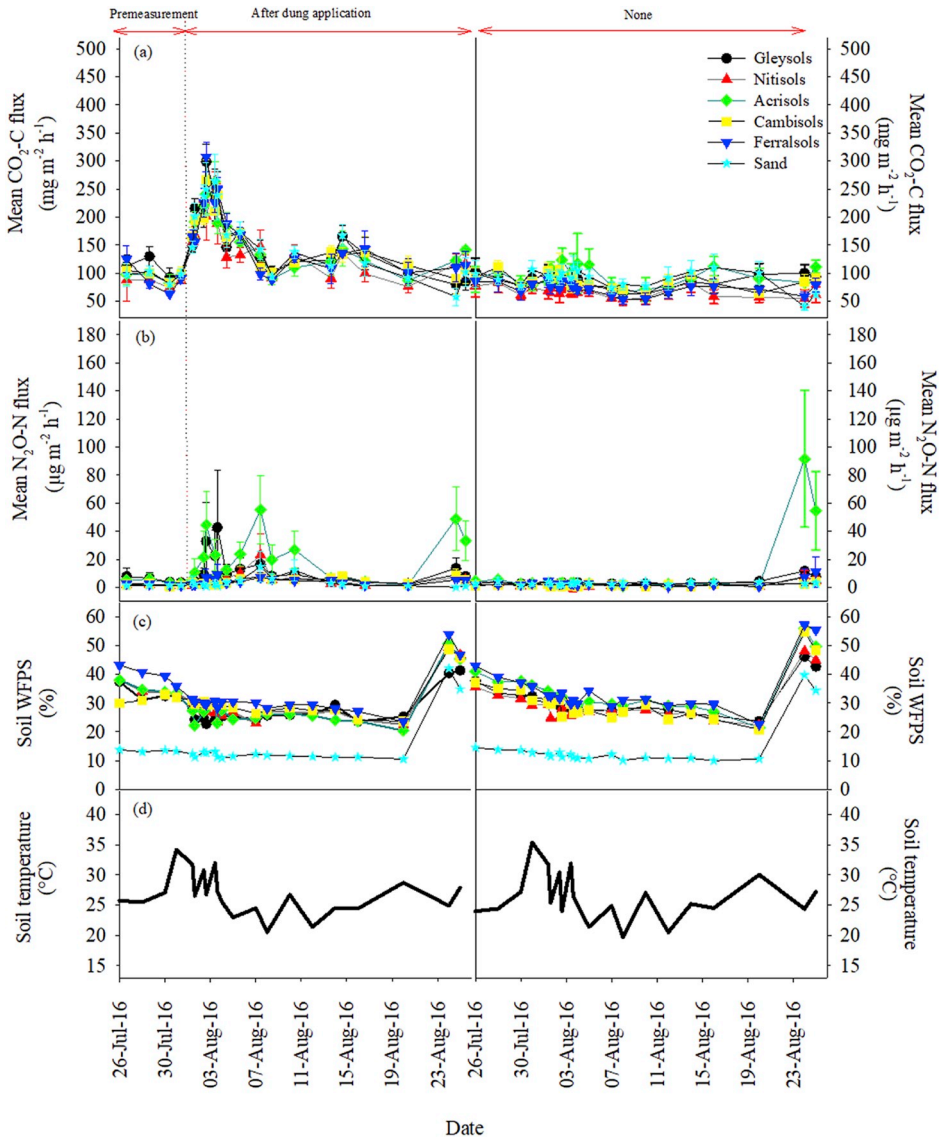

Fig. 5. Dynamics of (a) CO_2 and (b) N_2O daily fluxes from different soil cores to which dung was added (left panel) compared to the same soils without excreta application (right panel) (Trial 1). The lower panels show the observed temporal dynamics of (c) mean daily soil moisture (0.05 m depth), (d) soil temperature (0.05 m depth) measured during gas sampling. Each flux value represents the mean of three chambers (±SE).

N_2O emissions from excreta applied to grasslands, that the magnitude of N_2O emissions is negatively correlated with pH, i.e. that emissions increase at lower soil pHs. However, Khan et al., (2011) reported higher cumulative N_2O fluxes from bovine urine applied onto limed acid soils than on non-limed acid soils. The fact that all the soils except for the sand used in our study (pH 8.4, not included in the analysis) were slightly acidic and fairly similar in pH (pH range: 5.3–6.4) might explain why we found a positive correlation between urine EF and soil pH in both dry and wet season.

Contrary to our expectations, soil C and soil C/N ratio did not have any effect on urine N_2O EFs in our study, contrary to Pelster et al. (2012) who reported a potential C-limitation of soil N_2O emissions after N fertilizer application to temperate soils in Canada. However, because of the high C content in our soils (35 g kg^{-1} compared to 19 g kg^{-1} in the previous study), soil N_2O production was likely N- and not C-limited. This would also explain why urine, but not dung or manure additions significantly stimulated N_2O emissions as urine N is quickly available as urease is ubiquitous in all soils (Van Groenigen et al., 2005b). Moreover, not only N but also significant amounts of water was added with urine applications, which increases soil moisture and soil microbial activity (Marsden et al., 2016). It is noteworthy that in our study water-only additions did not stimulate N_2O emissions while additions of the same amounts of urine did (Figs. 7 and 8), indicating that although significant changes in soil moisture were observed after water application, the effect of the water was minor compared to the urine-N addition effect. As the soil moisture was generally below 60% water filled pore space (WFPS), it is likely that at least in the lighter textured soils nitrification was the dominant process of N_2O production after excreta application (Bell et al., 2015). Generally, high soil moisture promotes N_2O production, as shown in our study when comparing dry and wet season fluxes, but soil moisture can also be related to soil properties causing differences in N_2O fluxes between different soil types. For instance, De Klein et al. (2003) in a study in New Zealand grasslands reported an urine EF of 0.5% for a well-drained stony silt loam soil and 3.7% for a moderately-drained silt loam soil. However, Van der Weerden et al. (2011) reported that N_2O emissions did not always relate to drainage class, when summarizing experimental findings obtained for well- and poor-drained silt loam soils at three sites in New Zealand. Balaine et al. (2013) highlighted that the relative soil gas diffusivity is a key indicator to rank soil N_2O emission potentials.

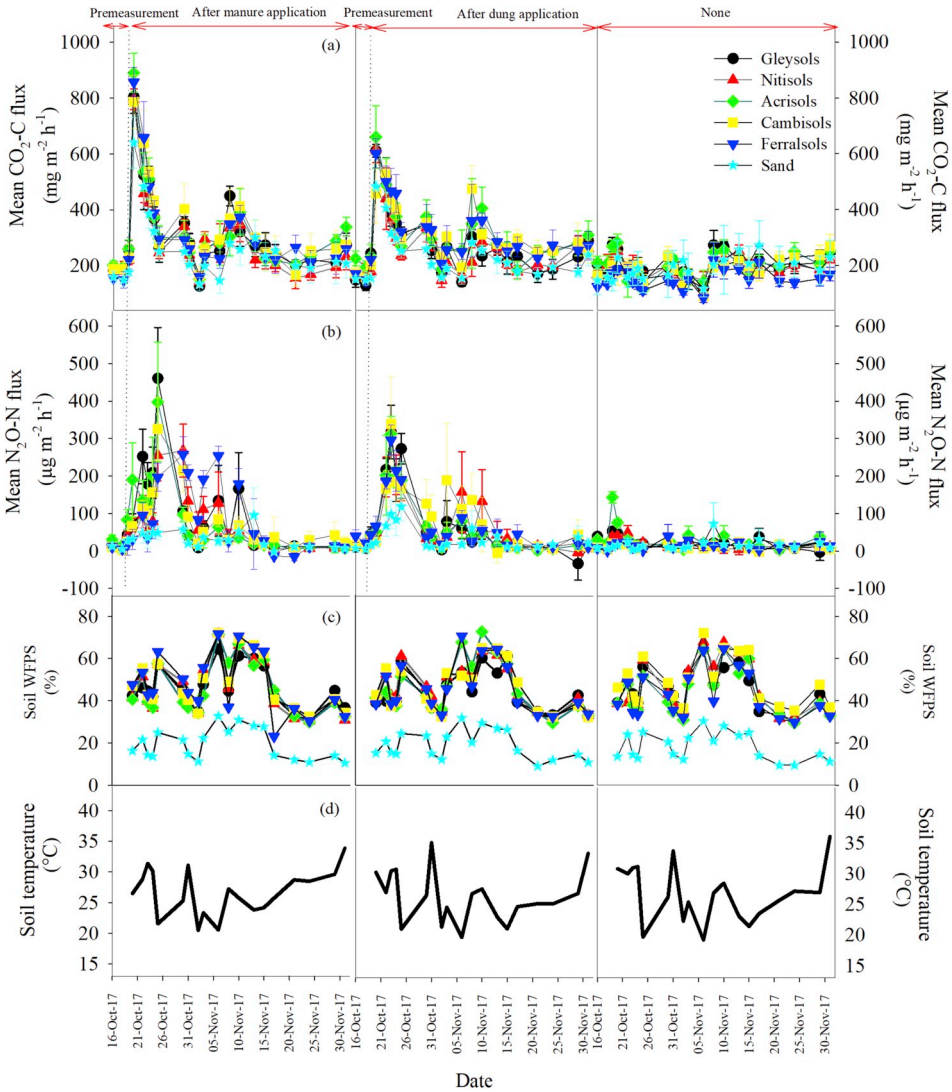

Fig. 6. Dynamics of (a) CO_2–C and (b) N_2O–N daily fluxes from different soil cores to which manure (left panel) or dung (middle panel) was added compared to soils without excreta application (right panel) (Trial 2). The lower panels show temporal dynamics of (c) mean daily soil moisture (0.05 m depth) and (d) soil temperature (0.05 m depth) measured during gas sampling. Each flux value represents the mean of three chambers (±SE).

4.2. Influence of season (wet vs dry) and time elapsed since excreta application on CO_2 and N_2O emissions

The higher soil CO_2 fluxes after excreta application in the wet season compared to the dry season was likely caused by the higher soil moisture, since temperature was similar during all four trials, as both soil moisture and temperature are controlling factors for CO_2 production (Rochette and Gregorich, 1998; Ginting et al., 2003). Similar results had been observed by Zhu et al. (2018) after cattle dung application to an East African grassland.

A one year field observation in Canada showed that most of the emissions occur within a few weeks after excreta application, although smaller peaks may occur later (Rochette et al., 2014). In our study, the flux patterns after dung application were consistent with a previous study for tropical rangeland soils amended with dung by Zhu et al. (2018). They reported that N_2O fluxes stayed elevated for around 14 days after dung application. The rather restricted period of elevated N_2O emissions after dung application can likely be attributed to the relatively low mineral N content of the dung, fast crust formation and the wide dung C/N ratio (22–36, Table 2), which likely promoted N immobilization during C decomposition (Van der Weerden et al., 2011; Zhu et al., 2018).

In the present study, N_2O fluxes returned to background levels within 39–73 observation days after urine application, which is consistent with Sordi et al. (2014) who reported 41 ± 10 days of elevated emissions following cattle urine application to a subtropical pasture of Brazil during three different seasons, after which the N_2O fluxes rapidly diminished and returned to background levels. This was also consistent with another study in New Zealand, which showed that soil NH_4^+-N concentrations returned to background concentrations after 27 days, while elevated soil NO_3^- concentrations were observed for around 40 days after cattle urine application (Clough et al., 2009). Similarly, a study carried out in the UK reported that the majority of N_2O emissions occurred during the first 20 days after sheep urine application onto Eutric Cambisol mesocosms (Marsden et al., 2016). Thus, the frequent gas sampling and the length of the observation period in our study should have been sufficient to capture most of the N_2O emissions caused by excreta application.

The higher N concentration and lower C/N ratio of the dung used during the dry seasons (C/N ratio: 22–29, Table 2) compared to the dung used during the rainy seasons (C/N ratio: 30–36, Table 2) should have translated into more substrate for denitrification and generally greater N_2O production during the dry season; however, the dung N_2O EFs were in fact lower during the dry seasons (range 0.00–0.04%) than during the

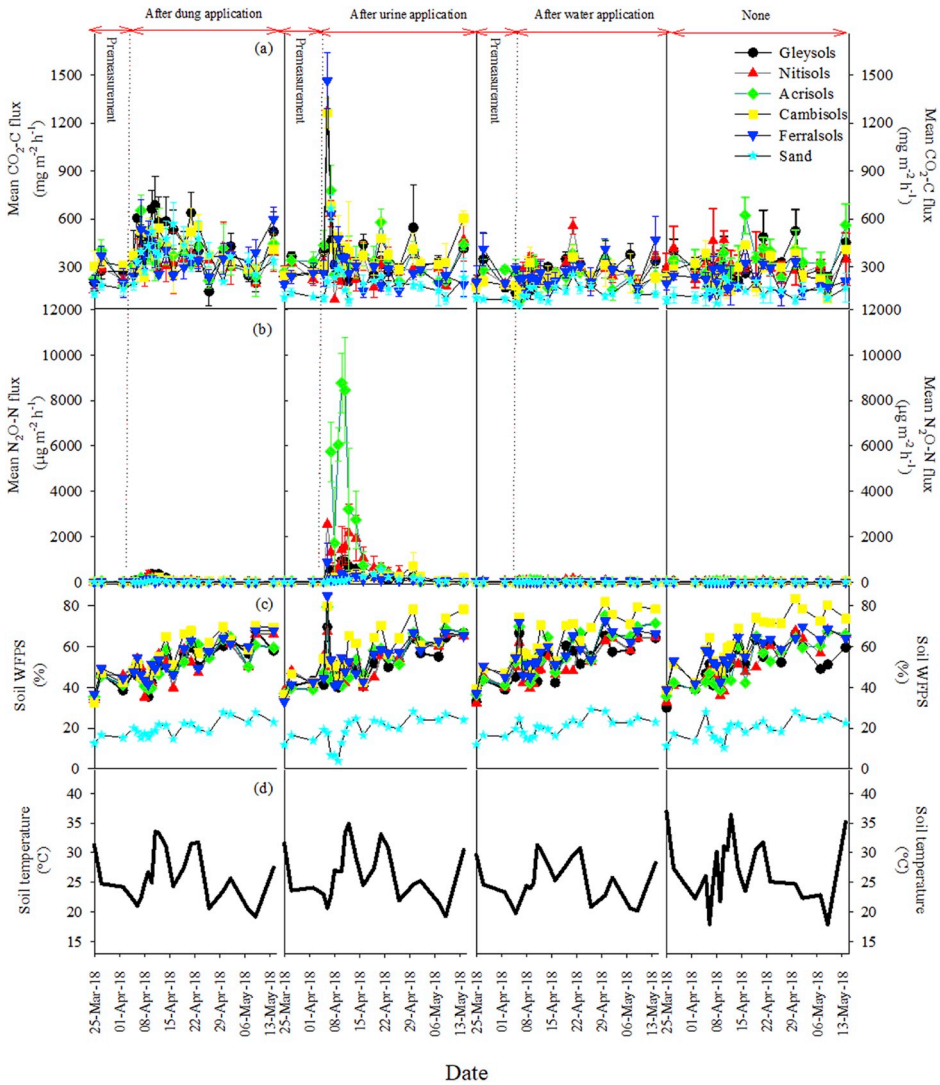

Fig. 7. Dynamics of (a) CO_2–C and (b) N_2O–N daily fluxes from different soil cores to which dung (left panel), urine (second-left panel), water (second-right panel), or nothing (right panel) was added (Trial 3). The lower panels show temporal dynamics of (c) mean daily soil moisture (0.05 m depth) and (d) soil temperature (0.05 m depth) measured during gas sampling. Each flux value represents the mean of three chambers (±SE).

wet seasons (0.02–0.20%). This was likely due to delayed dung crust formation due to rainfall (Mazzetto et al., 2014) and higher dung mineralization activity during the wet seasons with more favorable soil environmental conditions for N_2O production (see soil moisture levels in Figs. 5–8), which is in agreement with our hypothesis. The magnitude of total rainfall may cause differences in the amount of anaerobic microsites in the soil/dung, which could provide suitable conditions for further reduction of the microbially-produced N_2O to N_2 (Sordi et al., 2014; Zhu et al., 2018). This may explain why the dung N_2O EFs during trial 3 (range: 0.02–0.15%), which had 596 mm rain, were lower than for the other wet season period (trial 2; range: 0.10–0.20%), during which only 83 mm of rainfall. Furthermore, this lower EF might also have been related to increased N leaching as a consequence of the heavy rainfalls, although this remains speculation as we did not measure N leaching.

The higher urine N_2O EF in wet season than in dry season was similar to other studies in Brazil and Kenya (Sordi et al., 2014; Tully et al., 2017). Seasonal variations for urine N_2O EFs was also measured under a temperate climate (Van Groenigen et al., 2005b), although the authors attributed this seasonality to the seasonality of background N_2O emissions. Nevertheless, it should be noted that temperate climates show large intra-annual changes in temperature with less seasonality in precipitation rates. In Kenya, air temperature only changes minimally during the year, whereas rainfalls have a distinct seasonality (i.e. a clear separation between dry and wet season). Soil moisture, which exerts significant effects on soil N_2O emissions through modulating soil O_2 concentration and nutrient availability (Butterbach-Bahl et al., 2013), typically mirrors rainfall distribution. Therefore, our observation that excreta effects on soil N_2O emissions were generally stronger in the wet season is not surprising, as soil moisture content was generally higher during the wet season soils favouring coupled nitrification denitrification processes, i.e. the main N_2O production pathway after urine application (Monaghan and Barraclough, 1993). During the dry season, N mineralization and nitrification may be moisture limited, while denitrification would be unlikely as it is limited by the low soil moisture values (Linn and Doran, 1984).

4.3. *Influence of excreta type on CO_2 and N_2O emissions*

We confirmed our fourth hypothesis that both cattle urine and dung applications would stimulate CO_2 fluxes. The cumulative CO_2 emissions after dung application were similar to or slightly higher than after urine application in our study depending on the amount of C applied (Bertora et al., 2008), which is consistent with Lin et al. (2009) who also reported

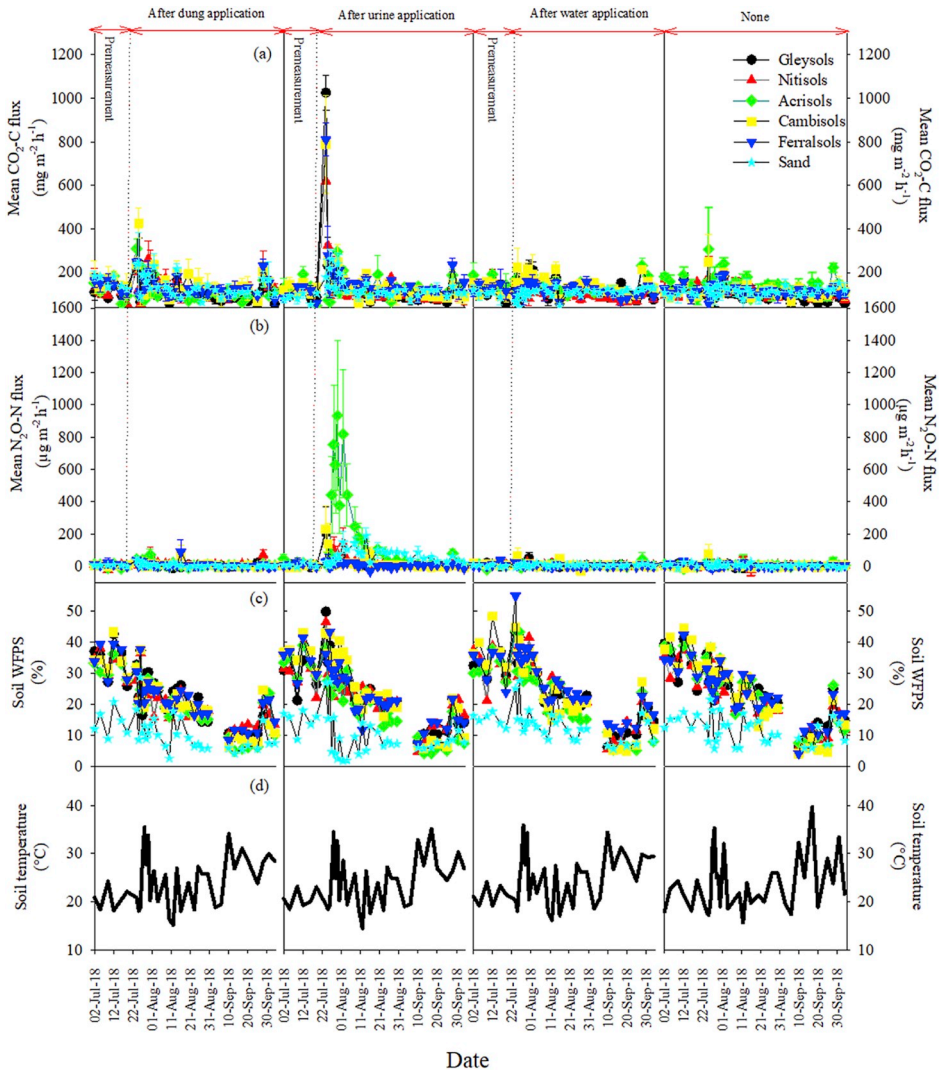

Fig. 8. Dynamics of (a) CO_2–C and (b) N_2O–N daily fluxes from different soil cores to which water, urine or dung was added (Trial 4). The lower panels show the observed temporal dynamics of (c) mean daily soil moisture (0.05 m depth), (d) soil temperature (0.05 m depth) measured during gas sampling. Each flux value represents the mean of three chambers (±SE).

higher cumulative CO_2 emissions from yak dung than from yak urine.

The N_2O EF for dung application (range: 0.00–0.20%, mean = 0.06%) was consistent with another study in Kenya that reported dung N_2O EF ranged from 0.00 to 0.04% (Tully et al., 2017). However, our mean EF was lower than the value of 0.28% synthesized for cattle dung patches through a global meta-analysis by Cai and Akiyama (2016). We suggest that the lower N_2O EF for cattle dung in Kenya in the present study compared to the global mean was caused by low quality feeds (i.e. low protein content) (Wassie et al., 2019) that resulted in lower N excretion and a higher C/N ratio in dung patches. In our study the dung and manure C/N ratios ranged from 22 to 36, much higher than the C/N ratios reported for dung/manure from cattle in temperate regions (<20, see Zhu et al., 2018). Moreover, the soils have low N concentrations (<0.39 g N kg^{-1} DM, Table 1), which could result in rapid immobilization of the applied N causing low NO_3^- availability for denitrification. Furthermore, higher solar radiation and high vapor pressure deficits in the Kenyan highlands leads to fast crust formation on the manure and dung and thus less incorporation of excreta N into the soil (Zhu et al., 2018).

The manure (i.e. a mixture of dung and urine) EF of 0.23% was also similar to a previous study in Kenya that reported a manure EF for the wet season of 0.15% (Tully et al., 2017). The higher manure EF in the present study compared to the dung EF might be explained by synergistic effects of labile C from the dung and inorganic N from the urine in manure, which combined with higher water contents likely promoted N_2O formation in soils and the manure itself (Hyde et al., 2016).

The observed range of urine EFs in our study (0.01–1.36%, with a mean of 0.29%) were within the range of EFs determined by studies carried out in Brazil (0.19–0.33%) (Barneze et al., 2014; Sordi et al., 2014) and Kenya (0.05–0.21%) (Tully et al., 2017). Cattle urine EFs of more than 3% were reported for an incubation experiment conducted at 25 °C and 80% relative humidity in Brazil (Cardoso et al., 2017). However, this previous study might be misleading, as the high temperature and humidity conditions likely favored N_2O production while the incubation system used by Cardoso et al. (2017) did not allow for N leaching; a major N loss pathway of urine N. In their review, Cai and Akiyama (2016) estimated that at least 17.8% of applied urine N is leached under field conditions. Another recent study from Kenya that simulated a 20 mm rainfall event after application, reported a cattle urine EF of 1.2%, which can be ascribed to more favorable soil condition to N_2O production after rainfall (Pelster et al., 2016). As synthesized by Cai and Akiyama (2016) the average N_2O EF for cattle urine was 0.76%, i.e. approx. three times higher than in our study. Although the N application rate of 845 and 749 kg N ha^{-1} applied as urine in our study was in the range of most studies analyzed in the global meta-analysis

(Cai and Akiyama, 2016), the soil properties and climate conditions in our study differed from most of the studies in the Cai and Akiyama review, which were predominantly data from temperate regions.

As the IPCC default value does not disaggregate the EF for urine and dung, we cannot compare the separate EFs for urine and dung in our study with the default value of 2%. However, the N partitioning between dung and urine for cattle excreta in SSA has been reported to be around 2:1 (Rufino et al., 2006), which is consistent with a recent study in Kenya by Wassie et al. (2019) who found that the N partitioning between dung and urine varied from ~1:1 to ~2:1. Those ratios differ significantly from estimates for western European countries (40:60) (Chadwick et al., 2018). If we apply the split as determined by Rufino et al. (2006) (i.e. an excreta-N ratio of dung to urine of 66:34), the overall N_2O EF for cattle excreta on grassland ranges from 0.06 to 0.30% depending on the soil type, with an average of 0.14%. Thus, the EFs found here are approximately one order of magnitude lower than the IPCC default value or the EF determined for livestock excreta in the tropics and subtropics by Albanito et al. (2017) of 1.2%.

5. Conclusion

Here we showed that while soil type had no influence on CO_2 emissions after excreta application it did affect N_2O emissions after urine, but not manure or dung, application. The differences between soils in N_2O emissions following manure application were likely related to soil properties such as texture and soil pH. Environmental conditions (wet and dry season), and the N content of excreta did affect N_2O losses, with a higher N_2O EF (0.29%) from urine application versus application of cattle dung (EF = 0.06%). Based on the excreta N split estimated by existing literature in SSA, the overall mean N_2O EF for cattle excreta deposited onto grassland was 0.14%, less than 10% of the IPCC default value (2%). Our findings suggests that current IPCC methodology provides a substantial overestimation of the N_2O EF for cattle excreta on tropical rangeland. In order to improve estimates and to identify adequate, region specific mitigation options to reduce emissions from livestock excreta, environmental conditions should be taken into account when estimating N_2O losses.

Declaration of competing interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.soilbio.2019.107636.

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Plant and Soil

The effects of climate on manure decomposition for cattle, sheep and goat in Kenyan tropical pastures

--Manuscript Draft--

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Abstract:	Aims Decomposition of manure deposited onto pasture from grazing animals represents an important process for carbon (C) and nitrogen (N) cycles in grassland systems. However, most decomposition studies focus on plant litter while studies investigating manure decomposition are scarce; especially in sub-Saharan Africa (SSA). Methods In this study, we measured decomposition of three types of animal manure (cattle, sheep, goat) over > 1 year using litter bags at four climatically different sites across Kenya. Results Manure dry matter, total C, N and ammonium concentrations decreased exponentially, with the most rapid decrease occurring during the first few weeks following application, followed by slower changes during the following 2-3 months. Rates of N mineralization were lower than those for C mineralization, resulting in decreasing C/N ratios over time. Generally, cattle manure decomposed faster than sheep or goat manure despite having a higher initial C/N ratio and lower N concentration, with decomposition rates for dry matter ranging from 0.200 to 0.989 k year -1 . Cellulose decomposed first, while lignin concentrations increased among all manure types and at all sites. Conclusions Our results show much slower decomposition rates of manures in semi-

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	arid tropical environments of East Africa as compared to the few previous studies in temperate climates.
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Re: Manuscript by Zhu et al. on manure decomposition

Dear editors,

our manuscript "The effects of climate on manure decomposition for cattle, sheep and goat in Kenyan tropical pastures" touches a topic which so far has received astonishing little attention. To our knowledge, globally only four research papers deal with in-situ experiments on the decomposition of manure on pastures. Given the importance of manure return to pastures for grassland C and N cycling this is rather surprising.

In our manuscript we report on decomposition rates for dry matter, C and N of different types of manure, i.e. manures from goats, sheep and cattle. These manures were exposed to different tropical climates and decomposition was studied using the litter bag methodology. Our work shows distinct differences in decomposition rates in dependence of aridity and type of the manure.

We would be delighted to learn that our work is of interest for Plant and Soil

Sincerely yours

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The effects of climate on manure decomposition for cattle, sheep and goat in Kenyan tropical pastures

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Abstract

Aims Decomposition of manure deposited onto pasture from grazing animals represents an important process for carbon (C) and nitrogen (N) cycles in grassland systems. However, most decomposition studies focus on plant litter while studies investigating manure decomposition are scarce; especially in sub-Saharan Africa (SSA).

Methods In this study, we measured decomposition of three types of animal manure (cattle, sheep, goat) over > 1 year using litter bags at four climatically different sites across Kenya.

Results Manure dry matter, total C, N and ammonium concentrations decreased exponentially, with the most rapid decrease occurring during the first few weeks following application, followed by slower changes during the following 2-3 months. Rates of N mineralization were lower than those for C mineralization, resulting in decreasing C/N ratios over time. Generally, cattle manure decomposed faster than sheep or goat manure despite having a higher initial C/N ratio and lower N concentration, with decomposition rates for dry matter ranging from 0.200 to 0.989 k year^{-1}. Cellulose decomposed first, while lignin concentrations increased among all manure types and at all sites.

Conclusions Our results show much slower decomposition rates of manures in semi-arid tropical environments of East Africa as compared to the few previous studies in temperate climates.

Keywords Litter bags · Manure · Mineralization · Climatic conditions · Sub-Saharan Africa (SSA) · Cellulose · Lignin

Introduction

Decomposition is a critical ecosystem function (Wilkinson 2006) that is fundamental to nutrient, carbon (C) and energy cycling within and among ecosystems, and between the biosphere and atmosphere (Schlesinger and Bernhardt 2013; Wieder et al. 2013). More than half of the net primary production (NPP) is returned to the soil through litter decomposition (David et al. 2004), while it also results in CO_2 emissions to the atmosphere of about 60 Pg C yr^{-1} (Houghton 2007). Internal recycling of nitrogen (N) from litter decomposition is also the primary source of N for most ecosystems (Parton et al. 2007; Chapin et al. 2012).

Decomposition largely results from the activities of soil microorganisms and macrofauna, which break down complex non-living organic matter (e.g. protein, cellulose) into smaller oligo- and monomers (e.g. amino acids, sugars) to gain energy and matter to build and maintain their biomass (Handa et al. 2014; Bradford et al. 2016). This process can be divided into two stages: The early decomposition stage, during which up to 40 % mass is lost, is characterized by leaching of soluble compounds and decomposition of solubles and non-lignified cellulose and hemicellulose. The late stage comprises the degradation of lignified tissue, which accounts for 40-100 % of mass loss (Heim and Frey 2004). Litter decomposition rates are driven by multiple biotic and abiotic factors such as litter quality (García-Palacios et al. 2016a; Prieto et al. 2019), decomposer community (Wang et al. 2009; Allison et al. 2013; Matulich and Martiny 2015), and climatic factors, including temperature and moisture (Wang et al. 2010; Veen et al. 2015; Bradford et al. 2016). Litter quality including initial N concentration, C/N ratio and lignin concentration is the predominant factor controlling decomposition through regulating decomposer activities under a given climatic condition (Hishinuma et al. 2017; Pei et al. 2019). Decomposition in natural ecosystems is synchronized with plant growth, while anthropogenic disturbance may retard or accelerate decomposition rates in managed ecosystems (Banegas et al. 2015).

Grasslands are the largest terrestrial ecosystem type, covering up to 40 % (59 million

km^2) of the world's ice-free land area (Hufkens et al. 2016), with most grasslands used for grazing of livestock (Salvati and Carlucci, 2015; Zhou et al. 2018). Grazing animals consume large amounts of biomass that often reduce litter inputs to soil (Güsewell et al. 2005; Tanentzap and Coomes 2012). For example, aboveground biomass and aboveground litter were reduced 63.7 % and 97.1 % under heavily intensified sheep grazing compared with an ungrazed control site in an Inner Mongolia temperate steppe during a two-year study (Bai et al. 2015). Plants ingested by grazing animals are fragmented and digested in their gut before being deposited onto the soil surface. Thereby approximately 75 % of ingested N by grass-fed animals is returned to the soil as excreta (Oenema et al. 2005).

Excreta has high concentrations of easily-decomposable C and N compounds, providing readily-accessible nutrients to soil micro- and macrobiota and plants that may lead to accelerated decomposition rates as compared to plant litter (Knops 2002; Bakker et al. 2004). Adult cattle can excrete up to 25 kg fresh dung and 21 L urine per day (Haynes and Williams 1993), while goats and sheep generally produce much less excreta, but with higher nutrient concentrations that tend to be more widely distributed (Bakker et al. 2004). In general, 5 % and 20 % of pasture surface area are estimated to receive either dung or urine, respectively, every year (Moir et al. 2011; Ward et al. 2018).

Driven by the increasing demand for livestock products it is expected that global cattle populations will increase from 1.5 billion to 2.6 billion and that of goat and sheep from 1.7 billion to 2.7 billion by 2050 (FAO 2009), which will change the volume and likely the type of manure deposited on pasture when herd composition changes. Considering the grazing area and number of livestock, manure plays a major role in C and N cycling in grassland ecosystems. However, quality differences in manure between different animal species may result in differences in decomposition processes and C and N mineralization rates, which in turn might affect C and N cycling in grasslands.

Savanna ecosystems cover about 65 % of the African continent (Brümmer et al. 2008),

with much of this area used by livestock farmers. In much of the arid and semi-arid regions in sub-Saharan Africa (SSA), livestock production generally relies on grazing of native pasture (Thornton and Herrero 2014) with long free grazing time, which leads to more than 40 % of total excreta being deposited on rangelands without further use or management (Rufino et al. 2006). As soils in grazing areas across SSA are often highly weathered and depleted of N, dung and urine play a major role in replenishing C and N pools, which is crucial in maintaining land productivity (Powell et al. 1996). However, studies investigating animal manure decomposition and C and N mineralization in tropical ecosystems and especially in SSA are scarce.

To provide information on how animal manure decomposition affects nutrient cycling in tropical pasture systems, we measured decomposition rates and changes in manure chemistry of three types of manure at four climatically different sites across Kenya for > 1 year (378 days). The objectives were to 1) measure changes in manure C and N concentrations over time after manure deposition; 2) determine if manure type (i.e. animal species) affects manure decomposition rates; and 3) determine how climate affects manure decomposition. We hypothesized that 1) manure decomposition rates would be faster for manure with lower initial C/N ratios and higher initial N concentrations; 2) manure would decompose faster under wetter and warmer climatic conditions; and 3) that C/N ratios in decomposing manure would decrease over time due to slower decomposition of complex N-rich compounds like lignin.

Materials and Methods

Site description

For our study, four sites with different climatic conditions in Kenya were selected. As shown in Table 1, the four sites were: Maktau in the Taita-Taveta county in South Kenya (Taita); the research campus of the International Livestock Research Institute in Nairobi (ILRI); Kapiti Research Station in Machakos county in South Central Kenya (Kapiti), and Machanga in Embu county in Central Kenya (Embu) (Table 1). The long-term mean annual precipitation rates were

lowest at the southernmost site (Taita) and highest at the northernmost site (Embu) (Table 1). Mean annual temperatures ranged from 19.0 to 25.0 °C, with the warmer temperatures associated with the lower elevation sites and cooler temperatures at higher elevation sites.

Experimental design

Approximately 100 g of fresh cattle manure, 70 g of fresh sheep manure and 30 g of fresh goat manure were weighed and put into individual 12.5 cm × 10 cm 1-mm nylon mesh bags. Prior measurement of manure water content showed that this is equivalent to approximately 20 g of manure dry matter (DM), though actual DM contents were finally measured for each sample. The litter bags allow for water, nutrient and microbial passage, while preventing soil macrofauna (e.g. termites) from entering, and have been used widely in decomposition studies (García-Palacios et al. 2016b). Fresh cattle dung was collected at the ILRI Nairobi farm from a cattle herd that grazed on pasture during the day and was housed in single-animal pens at night. Goat and sheep also grazed on the same pasture during the day, but were taken back to a communal barn with a concrete floor overnight, where the manure was collected the following morning. Although the manure was collected from the same source, manure properties differed across different collection days. Consequently, only manure for the Kapiti and Embu sites, which was sampled on the same day, showed the same initial properties, while manure properties for the other two sites differed slightly (Table 2).

Due to logistical reasons, the manure was stored at 4°C for up to two days before being placed in the nylon bags and transported to the various locations, where the litter bags were laid on the soil. For each of the four sites 207 nylon bags were prepared, i.e. 828 bags in total (3 manure types × 3 replicates × 23 sampling dates × 4 sites). Subsamples of the dung were frozen for nutrient analyses. The manure bags were applied to the pasture on 31 March, 06 April, 11 April and 12 April, 2018 for the Taita, ILRI, Kapiti and Embu sites, respectively.

At each site, an open and flat area (2.5 m × 6.0 m) was chosen and the grass on the surface was

cut with scissors at the ground level. After application, 3 bags for each manure type were sampled once per week for the first seven weeks of the experiment. After the initial seven-week period, sampling occurred bi-weekly during the rainy seasons and monthly in the dry seasons. In total, there were 23 sampling dates over a 378-day period for each of the four sites.

Environmental data

Weather stations were installed at each site to collect meteorological data (rainfall and air temperature at all sites, soil temperature and soil moisture at Taita, ILRI and Embu), during the course of the experiment. The climate data were downloaded at least in monthly intervals. PMday.xls was used to calculate daily reference evapotranspiration (ETref) rates using the Penman-Monteith equation as presented by the Environmental Water Resources Institute (EWRI) - American Society of Civil Engineers Committee (ASCE) on Reference Evapotranspiration (ASCE-EWRI 2004). For further details see http://biomet.ucdavis.edu/Evapotranspiration/PMdayXLS/PMday.htm.

Manure analysis

At each sampling date, the surfaces of the collected bags were cleaned carefully with a brush to remove attached vegetation or soil particles before opening the bag followed by measuring the fresh weight of the manure remaining inside the bag. A 5 g sub-sample was oven-dried at 50 °C until constant weight and ground for total C and N analysis with an elemental combustion system (Vario max cube, Elementar analysensysteme GmbH, Germany). Ammonium (NH_4^+) and nitrate (NO_3^-) were extracted from the manure sample by mixing a 1 g subsample with 25 ml 1 *M* KCl in a 50 ml plastic bottle, shaking the slurry on a reciprocal shaker for 1 h, centrifuging at 3000 rpm for 10 minutes and filtering (Whatman No. 42). Extracts were then frozen until colorimetric analyses for NH_4^+ and NO_3^- concentrations (Hood-Nowotny et al. 2010). The rest of the manure was then weighed and oven-dried at 105 °C for 24 h for

determination of water content. Cellulose, hemicellulose, lignin and crude protein (CP) concentrations were measured at time zero, at week 27 and at the end of the sampling period (week 54). The filter bag technique using an ANKOM^{200} Fiber Analyser (ANKOM Technology, Macedon, USA) with sodium sulphate and α-amylase was used to measure neutral detergent fiber (NDF) and acid detergent fiber (ADF) (Van Soest et al. 1991), while acid detergent lignin (ADL) was determined by ashing the remaining fraction in beakers after NDF and ADF determinations (Van Soest 1963). Hemicellulose concentration was calculated as NDF minus ADF and cellulose concentration was calculated as ADF minus ADL, while ADL minus ash was assumed to be equivalent to the lignin concentration (Van Soest et al. 1991). Crude protein concentration was determined by multiplying the N concentrations from the Kjeldahl method by 6.25 (Kirk 1950).

Calculation and data analysis

The decomposition rate (k, year^{-1}) of manure was calculated using the following equation:

$$k = \ln \frac{M_t / M_0}{-1/t}$$

where M_t and M_0 are manure dry matter at time t and 0 (start), respectively; and t is time in years. This equation was also used to calculate rates of C and N mineralization, where M_t and M_0 are total manure C or N content at time t and 0, and t is also time in years.

Two-way ANOVA with Tukey's HSD *posthoc* test was used to test for differences of the various initial manure properties (e.g. C, N concentration and C/N ratio) and for rates of manure decomposition and manure C and N mineralization, using manure types and sites as the two fixed factors. Differences in cellulose, hemicellulose, lignin and CP concentrations among sites and sampling dates within the same manure type were also tested using two-way ANOVA and Tukey's HSD test, respectively. All statistical calculations were done with Aov package in R v3.6.1 (R core team, 2019). The linear regression between decomposition rate and mean temperature, cumulative precipitation and aridity index (AI) was built with Sigmaplot 12.5

(Systat Software, Inc. SigmaPlot for Windows).

Results

Weather condition

Over the 378-day period, the Embu site had the highest mean temperature (22.9 °C) and the lowest cumulative precipitation (209.3 mm), while the lowest mean temperature (18.0 °C) and the highest cumulative precipitation (1371.6 mm) occurred at the ILRI site (Fig. 1b, d). Mean temperatures were within the range of the long-term records for all sites. However, cumulative precipitation at the Taita and Embu sites were 34 % and 50 % less than the long-term means (Ngetich et al. 2014; Pellikka et al. 2018) (Fig. 1a, d), while the ILRI site had 58 % more precipitation than the long-term mean (Pelster et al. 2016) (Fig. 1b). Precipitation at Kapiti during the observation period agreed well with long-term records (Table 1, Fig. 1c). The two warmer sites (Embu and Taita) received less cumulative precipitation during our observation period than the cooler sites (Kapiti and ILRI).

The aridity index was calculated as the ratio of precipitation to potential evapotranspiration based on the actual weather information during the observation period, with lower values indicating drier conditions (Wang et al. 2014). Both the Taita and Embu sites were defined as arid (i.e. aridity index of 0.18 and 0.12 for Taita and Embu sites, respectively). The Kapiti site was considered to be semi-arid (AI = 0.37), while the ILRI site was humid (AI = 0.92), due primarily to more precipitation (Table 1).

Initial manure properties

Due to different manure sampling dates, properties of cattle, sheep and goat manures slightly differed across sites (Table 2). Cattle manure used at the Taita site having the lowest C and highest N concentrations, while the cattle manure used at ILRI site had the highest C and lowest N concentrations. Sheep manure also differed among sites, with sheep manure used at ILRI

having slightly higher C concentrations than that at the other three sites, while sheep manure used at Kapiti and Embu had higher N concentrations. The C and N concentrations of goat manure used at the four sites were similar (Table 2). Sheep manure had lower C and higher N concentrations compared to cattle or goat manure at the same site, while cattle manure showed highest C and lowest N concentrations at all sites except Taita. The C/N ratio ranged from 12.6 to 32.6 with sheep < goats < cattle manure (Table 2). Considering mineral N concentrations in fresh manure, initial NH_4^+ concentrations were highest in sheep manure at Taita, Kapiti and Embu (2898 ± 40, 2871 ± 48 and 2871 ± 48 μg N g^{-1} DM, respectively), while at ILRI NH_4^+ concentrations were highest in goat manure (2764 ± 37 μg N g^{-1} DM). For all manure types, initial NO_3^- concentration was negligible compared to NH_4^+ concentration (Figs. 5, 6, 7).

Initial hemicellulose and cellulose concentrations were higher in cattle and goat manure than in sheep manure at all sites ($P < 0.05$, Figs. 2, 3, 4). Initial cellulose concentrations showed the following order: cattle manure > goat manure > sheep manure, while initial CP concentrations showed the opposite order: sheep manure > goat manure > cattle manure. In contrast, initial lignin concentrations did not differ between the three manure types and the four sites (Figs. 2, 3, 4).

Temporal dynamics of manure component during the experimental period

After 378 days, approximately 70 % of cattle manure and 50 % of goat and sheep manure mass was lost. Temporal patterns of decomposition were similar regardless of manure type and sites. Dry matter decreased rapidly during the first several weeks, with the rate of DM decrease slowing down over time (Figs. 5, 6, 7). Due to the manure DM loss, the total C and N content also decreased rapidly in the first several weeks, though the C loss was faster than N loss. Consequently, C/N ratios decreased over time (Figs. 5, 6, 7). Ammonium concentrations decreased exponentially during the first week after application, and remained low during the rest of the trial (Figs. 5, 6, 7). Crude protein concentrations remained rather constant throughout

the 378 days sampling period, while cellulose concentrations tended to decrease and lignin concentrations increased (Figs. 2, 3, 4). Hemicellulose concentrations in cattle and goat manure tended to decrease, with the exception of the cattle manure at Taita (Figs. 2 and 4). Unlike the other manure types, hemicellulose concentrations for sheep manure generally remained constant with the exception of Kapiti site, where hemicellulose concentrations were lower after 378 days as compared to initial values (Fig. 3).

Decomposition rate and C and N mineralization rates

Decomposition rates varied strongly across sites, but were generally higher for cattle manure (0.288 - 0.989 k year^{-1}) than sheep manure (0.200 - 0.457 k year^{-1}) or goat manure (0.234 - 0.750 k year^{-1}) (Table 3). Cattle and goat manure decomposed more quickly at ILRI site compared with the other sites, while sheep manure decomposition rate at ILRI was higher than that in Taita and Embu sites but similar to the Kapiti site. No significant correlations between decomposition rates and initial ratios of cellulose: N or lignin: N in manures were found.

Between 4.6 ± 0.2 and 42.3 ± 8.5 % of manure N were lost over the entire observation period. The N mineralization rate was higher for cattle and goat manure at the ILRI site compared with that at Taita and Embu sites, whereas there was no difference in N mineralization rates among sites for sheep manure (Table 2). The N mineralization rates (ranging from 0.05 to 0.54 k year^{-1}) were similar among manure types at each site except for the Embu site, where the N mineralization rate from goat manure was lower than that of sheep manure (Table 3). The highest C mineralization rate (1.20 ± 0.15 k year^{-1}) was observed for cattle manure at the ILRI site with 70.8 ± 4.3 % of C being mineralized. In contrast, sheep manure at the Embu site had the lowest C mineralization rate (0.20 ± 0.05 k year^{-1}) with only 18.3 ± 4.5 % of the C being mineralized (Table 3). In general, C mineralization rates were higher at the ILRI site for all manure types, and cattle manure tended to have higher C mineralization rates than the other manure types at all sites.

The dry matter decomposition rate was positively correlated with cumulative precipitation (decomposition rate = 0.1969 + 0.0004 × cumulative precipitation, n = 35, R^2 = 0.52, $P < 0.05$) and negatively with mean air temperature (decomposition rate = 2.081 – 0.080 × mean temperature, n = 35, R^2 = 0.47, $P < 0.05$) regardless of the manure type. The aridity index, which considers both precipitation and air temperature, showed a positive relation with dry matter decomposition rate (decomposition rate = 0.215 + 0.579 × aridity index, n = 35, R^2 = 0.53, $P < 0.05$), indicating faster decomposition at more humid conditions (see Supplementary Material Figs. 1-3).

Discussion

Livestock grazing is the most widespread form of herbivore management and significantly influences the vegetation of grasslands, which cover an area of 28 - 35 million km^2 globally (Tanentzap and Coomes 2012). Grazing animals remove large amounts of plant biomass and translocate nutrients back to soil in the forms of urine and dung (Bakker et al. 2004; Güsewell et al. 2005). Consequently, decomposition of manure is crucial for returning nutrients to soils, and, thus, for maintaining overall ecosystem functioning. However, studies investigating the decomposition processes of manure deposited on pasture are scarce, especially in SSA. To our knowledge, globally only four field studies of manure decomposition *in situ* in grasslands are available, while a rather huge number of studies are available on manure decomposition rates under controlled laboratory conditions. Markewich et al. (2010) investigated the decomposition of manure differing in chemical composition after storage in small-scale Kenyan livestock systems with seven sampling times over 112 days and found that organic N in manure with higher concentrations of N disappeared more quickly than from manure with low initial N. Esse et al. (2001) measured cattle manure and sheep-goat manure decomposition with five sampling times over 17 weeks in south-west Niger and reported that manure decomposed faster on crusted than on sandy soil. Rashid et al. (2013) applied stacked or composted cattle manure to

sandy and peat soils in the Netherlands with three sampling times over 240 days and found that the so-called “home-field advantage”, which had been initially described for plant litter and suggests that litter decomposes faster in its home habitat (Ayres et al. 2009; Strickland et al. 2009), where microbial decomposer communities are adapted to the litter chemistry also had an effect on manure decomposition in grasslands. Somda and Powell (1998) used sheep manure from sheep fed on different diets to study decomposition dynamics in Kenya. On basis of five sampling times over 112 days they showed that decomposition rates were affected by manure quality and seasonality. Given the paucity of data and the importance of excreta for C and N cycling of grassland ecosystems, it is obvious that more frequent samplings over longer time periods are necessary to better understand decomposition dynamics of manures, which are key for rangeland C and N cycling.

Influence of manure type on decomposition rate

Animal species may be an important source of variation in the decomposition rate of livestock manure as manure chemical composition differs due to the digestion capacity of different livestock species, even when fed on the same diet (Schlecht et al. 1997). Furthermore, different livestock species exhibit different feeding practices (i.e. grazers, browsers, intermediate feeders) (Searle and Shipley 2008). As an example, small ruminants actively select plants of higher nutritional value, which results in manure with higher N concentration compared to cattle manure (Shriver et al. 2003; Valdés-Correcher et al. 2019). The higher initial N concentration in sheep and goat manure in our study was consistent with previous studies which also measured higher initial N concentrations in sheep/goat manure, and attributed that to selective feeding on N-rich plants (Esse et al. 2001). However, in contrast to our expectations, and unlike previous studies that found negative correlations between litter decomposition rate and initial C/N ratio (Bradford et al. 2016), manure decomposition rates in our study were higher for cattle manure than for sheep and goat manure despite the lower initial N concentration and higher C/N ratio

in cattle manure. However, our results are consistent with Chen et al. (2019) who reported that chicken manure, with high N concentrations and low C/N ratio, decomposed more slowly than pig manure, highlighting the important role of manure chemical composition for decomposition. Previous studies have found that manure decomposition could be regulated by their main structural components such as cellulose (Morvan et al. 2006; Chen et al. 2019), as cellulose was found to be the primary substrate for glucan depolymerization at a later stage of decomposition after more labile C compounds such as sugars and starch had been depleted (Leitner et al. 2012). This regulation of the decomposition rate by cellulose could explain why cattle manure in the current study decomposed more quickly than the other manures even though it had a higher C/N ratio. Starting concentrations of cellulose were around 25 % in cattle manure, but < 20 % in sheep and goat manures (Figs. 2, 3, 4). Cellulose in manure can decompose very fast because it is already rich in cellulolytic bacteria from the rumen (Liao et al. 2005). In addition, a large fraction of N was tied up in the cellulolignin complex, indicating that the ADF and ADL ratio played a more important role in controlling decomposition rate than the initial C/N ration in our study (see Supplementary Material, Fig. 4). Lignin compounds are known to provide physical protection to N-containing compounds such as cellwall proteins (Fioretto et al. 2005; de Bruijn and Butterbach-Bahl 2010), and as lignin is more recalcitrant to decomposition than other plant components (Morvan et al. 2006), total N concentrations alone might be a poor predictor for decomposition.

Regardless of manure type, manure NH_4^+ concentrations diminished from 551, 2610, 2005 µg N g^{-1} DM for cattle manure, sheep manure and goat manure, respectively to values close to 100 µg N g^{-1} DM within one week after application. Such a rapid change in manure NH_4^+ concentrations has also been reported by Markewich et al. (2010), suggesting that the high mineral NH_4^+ concentration in fresh manure, originating from the rapid decomposition of urea from urine, was either immobilized by the microbial decomposer community, volatilized as NH_3, nitrified and leached or further denitrified and emitted as either NO, N_2O or N_2 (Zhu et

al. 2020). In agreement with our third hypothesis, the C/N ratio became narrower with increasing time of manure decomposition. This was also reported by Glaser et al. (2001) in a 120-day laboratory incubation experiment with cattle manure applied to savanna soils in Northern Tanzania. Comparable observations were also made by Sierra et al. (2013) in composted cattle manure. This shows while C is being mineralized and released as CO_2, mineralized N is retained. Thereby, N immobilization seems to be associated with lignin, as lignin concentrations increased with decomposition time in our experiment (Figs. 2, 3, 4). According to the theory of ecological stoichiometry (Elser et al. 2003; Zechmeister-Boltenstern et al. 2015), microbial decomposers can only build their biomass when they have access to both C and N in a rather narrow ratio (3-10). However, when the substrate that they are growing on has a very wide C/N (> 25), they have to import N from soil into the litter/manure (for example via hyphal networks) leading to net N immobilization into the decomposing substrate (Osono and Takeda 2004; Pei et al. 2019).

Climate effect on decomposition rate

In our study, the decomposition rate was positively correlated with cumulative precipitation, but negatively correlated with mean temperature. Many studies have reported positive effects of warming on litter decomposition (Lu et al. 2013; Yue et al. 2015). This is because warming stimulates microbial activity and accelerates enzyme kinetics (Conant et al. 2011), which are closely linked to decomposition rates (Allison et al. 2013). However, the current study was conducted in the tropics where temperature is not per se limiting. Furthermore, the warmer sites received less precipitation and experienced high vapor pressure deficits that caused rapid water loss and crust formation, likely slowing down microbial decomposition of the manure (Zhu et al. 2018). This is consistent with Prieto et al. (2019) who also found a negative effect of warming and reduced rainfall on litter decomposition in a semi-arid shrubland because of the desiccating effect of warming and decreased water input. In line with this we found a positive

correlation between aridity index and decomposition rate, indicating that decomposition was faster in moister locations. The importance of water availability for litter decomposition has been highlighted by many previous studies (e.g. Wardle et al. 2004; García-Palacios et al. 2016b; Almagro et al. 2017), as rainfall is required to keep manure and soils wet, which in turn stimulates microbial activity and enzyme diffusion, which ultimately drives the decomposition process (Jacobson et al. 2015; Gliksman et al. 2017). Besides, rainfall may also have accelerated nutrient leaching from the litter bags, which likely also contributed to manure mass loss (Wang et al. 2009).

Manure decomposition in other studies

Similar to previous studies (e.g. Chen et al. 2019; Markewich et al. 2010), rates of manure decomposition in our study decreased exponentially with exposure time, regardless of site and manure type. The manure decomposition rates in our study (0.200 to 0.989 k year^{-1}) (Table 3) are comparable to those found by Chen et al. (2019) for pig manure (0.516 k year^{-1}) and chicken manure (0.483 k year^{-1}) applied to cropland. Markewich et al. (2010) also reported decomposition rates for cattle manures differing in quality in Western Kenya. In that study, the decomposition rates for medium and low quality manure were 0.907 and 1.214 k year^{-1}, respectively, i.e. in the range of values found in our study. In addition, in our study 23 - 74 % of the DM had disappeared from the cattle manure bags after 378 days, which is similar to the 39 - 80 % of the DM lost from solid cattle manure bags in a study carried out in the Netherlands (Rashid et al. 2013). However, only 31 - 42 % of N were lost in the current study, which is less than the 56 - 98 % N loss found by Rashid et al. (2013). This difference in N loss rates might be explained by the higher initial manure N concentration and lower C/N ratio in the latter study, as higher initial N concentrations can favor N mineralization (Kuypers et al. 2018). Furthermore, the litterbags used by Rashid et al. (2013) had a larger mesh size (4 mm) than our study (1 mm), through which herbivorous soil fauna can access the manure, which might increase DM as well

as N loss. Similar to our study, Esse et al. (2001) studied the decomposition of sheep-goat and cattle manure in an 18-week field incubation in West Africa and reported that more than half of manure mass remained at the end of the trial, and that over time, N concentrations increased relatively to C concentrations. These authors argued that the observed rather low manure decomposition rates were caused by very high temperature (30 °C) and low precipitation (350 mm), which in their case accelerated crust formation and slowed down the decomposition process.

Soil fauna effect on decomposition

Soil macrofauna and mesofauna (e.g. termites, beetles, nematodes) are able to fragment and shred litter material with their mouthparts, altering litter structure and facilitating microbial access, thus accelerating litter decomposition (Wang et al. 2010). By excluding soil macrofauna in our study, we therefore likely underestimated manure decomposition rates. Indeed, a previous study in Niger, West Africa by Esse et al. (2001) reported that manure in cages with mesofauna decomposed faster than manure in litter bags where meso- and macrofauna were excluded. In addition, a global decomposition experiment by Wall et al. (2008) also demonstrated that soil meso- and macrofauna increased litter decomposition rates in temperate and wet tropical climates. Termites and ants are widespread in the soil in Kenya (Markewich et al. 2010), and previous studies have reported termites appearing rapidly after dung deposition on rangelands in SSA (Pelster et al. 2016; Zhu et al. 2018). Through using the fibrous manure material in their mounds, a significant portion of manure applied may be degraded and/or translocated by the soil fauna (Diamond 1998; Markewich et al. 2010). Besides, application of cattle manure to grassland was shown to increase soil faunal abundance and activity (Forge et al. 2005; Stark et al. 2008), which in turn can increase manure decomposition rates. Therefore, we want to notify the reader that the reported decomposition rates in our study are likely at the lower end of manure decomposition under unrestricted conditions, i.e. when allowing for

interactions with soil fauna.

Conclusion

In this study, we found that manure decomposition rates ranged from 0.200 to 0.989 k year^{-1} and were both influenced by manure chemical composition and climatic conditions. Manure chemical composition (i.e. cellulose and lignin concentration) was a better predictor of manure decomposition rates than manure total N and C/N ratio. Furthermore, as air temperature at our tropical study sites was not limiting, precipitation was the main environmental driving factor for manure decomposition, which was also reflected by the aridity index (i.e. higher AI indicating moister conditions was leading to faster decomposition). Surprisingly, a large proportion of manure (30 % for cattle, 50 % for sheep and goats) mass remained after the 378 days field experiment. Therefore, to enhance our understanding of C and N dynamics in manure decomposition in tropical regions, long-term studies that cover the whole decomposition period are needed. In addition, the role of soil macrofauna on manure decomposition in tropical grassland also needs further investigation as especially termites are known to be keystone species in many savanna ecosystems, but their impact on manure decomposition and ecosystem C and N cycling is still poorly understood.

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Tables and Figures

Table 1 Overview of location of sites (name, coordinates, county), altitude, long-term mean annual precipitation and air temperature and aridity index during our observation period at a given site. Aridity index was calculated using the following equation: Aridity index= precipitation/potential evapotranspiration.

Table 2 Carbon and nitrogen concentrations and C/N ratios of the different manure types at the beginning of the decomposition experiment.

Table 3 Dry matter decomposition rates, and C and N mineralization rates of the different manures at the four sites after 378 days in our experiment

Fig. 1 The daily mean air temperature and precipitation at Taita site (a, from 31-Mar-18 to 13-Apr-19), ILRI site (b, from 06-Apr-18 to 19-Apr-19), Kapiti site (c, from 11-Apr-18 to 24-Apr-19) and Embu site (d, from 12-Apr-18 to 25-Apr-19).

Fig. 2 Cattle manure hemicellulose, cellulose, lignin and crude protein dynamics at Taita, ILRI, Kapiti and Embu sites. Each value represents the mean of two replicates for hemicellulose, cellulose and lignin and three replicates for crude protein (± STDEV); different lowercase letters indicate significant differences among sampling dates within the same site and different capital letters indicate significant differences among sites within the same sampling date ($P < 0.05$).

Fig. 3 Sheep manure hemicellulose, cellulose, lignin and crude protein dynamics in Taita, ILRI, Kapiti and Embu sites. Each value represents the mean of two replicates for hemicellulose, cellulose and lignin and three replicates for crude protein (± STDEV); different lowercase letters indicate significant differences among sampling dates within the same site and different capital letters indicate significant differences among sites within the same sampling date ($P < 0.05$).

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Fig. 5 Dynamics of cattle manure remaining dry mass (a), carbon content (b), nitrogen content (c), C/N ratio (d) and NH_4^+ concentration (e) after litter bags application at Taita, ILRI, Kapiti and Embu sites, respectively. Each value represents the mean of three replicates (± STDEV).

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Fig. 7 Dynamics of goat manure remaining dry mass (a), carbon content (b), nitrogen content (c), C/N ratio (d) and NH_4^+ concentration (e) after litter bags application at Taita, ILRI, Kapiti and Embu sites, respectively. Each value represents the mean of three replicates (± STDEV).

Table 1 Overview of location of sites (name, coordinates, county), altitude, long-term mean annual precipitation and air temperature and aridity index during our observation period at a given site. Aridity index was calculated using the following equation: Aridity index= precipitation/potential evapotranspiration.

Site	Coordinates	County	Altitude (m)	Mean annual precipitation (mm)	Mean annual temperature (°C)	Aridity index	References
Taita	S 3°25' E 38°20'	Taita Taveta	700	400 - 600	25.0	0.18	Pellikka et al. (2018)
ILRI	S 1°16'13" E 36°43'23"	Nairobi	1809	869	19.0	0.92	Pelster et al. (2016)
Kapiti	S 1°37'06" E 37°06'09"	Machakos	1600	550	20.2	0.37	
Embu	S 0°46' E 37°39'	Embu	1100	700 - 900	20.7 - 22.5	0.12	Ngetich et al. (2014)

Table 2 Carbon and nitrogen concentrations and C/N ratios of the different manure types at the beginning of the decomposition experiment.

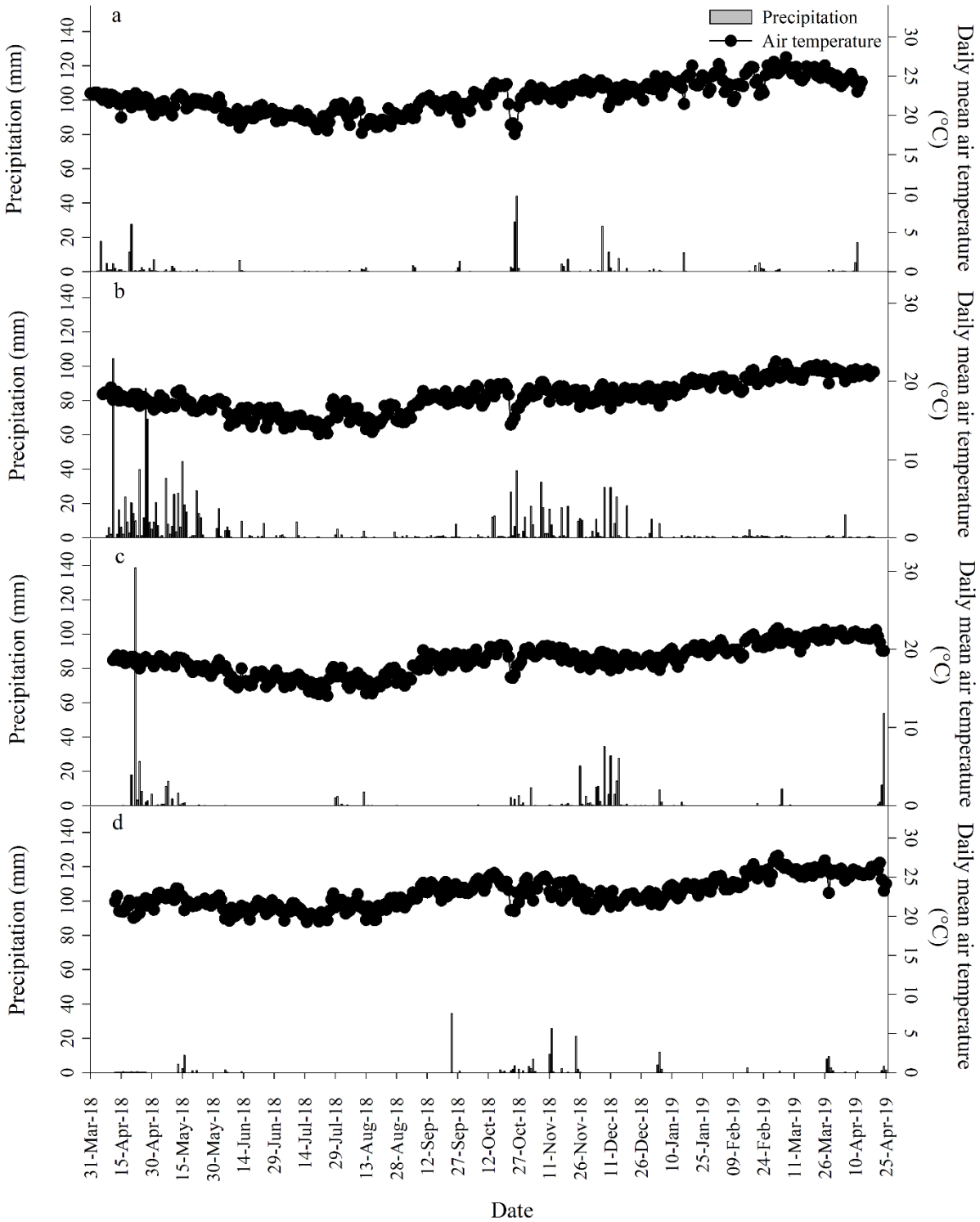
Site	Manure collecting date	C concentration (g C kg^{-1} DM)			N concentration (g N kg^{-1} DM)			C/N ratio		
		Cattle manure	Sheep manure	Goat manure	Cattle manure	Sheep manure	Goat manure	Cattle manure	Sheep manure	Goat manure
Taita	30-Mar-18	387 ± 1^{Bc}	364 ± 0^{Cb}	404 ± 0^{Aab}	16.4 ± 0.3^{Ca}	25.2 ± 0.1^{Ac}	19.1 ± 0.1^{Bb}	23.6 ± 0.4^{Ac}	14.4 ± 0.0^{Ca}	21.2 ± 0.1^{Ba}
ILRI	06-Apr-18	431 ± 0^{Aa}	371 ± 0^{Ca}	404 ± 0^{Ba}	13.2 ± 0.0^{Cc}	26.8 ± 0.1^{Ab}	18.8 ± 0.1^{Bb}	32.6 ± 0.0^{Aa}	13.8 ± 0.1^{Cb}	21.5 ± 0.1^{Ba}
Kapiti	10-Apr-18	421 ± 0^{Ab}	355 ± 2^{Cc}	402 ± 0^{Bb}	16.1 ± 0.1^{Cb}	28.2 ± 0.1^{Aa}	20.3 ± 0.1^{Ba}	26.2 ± 0.2^{Ab}	12.6 ± 0.1^{Cc}	19.8 ± 0.1^{Bb}
Embu	10-Apr-18	421 ± 0^{Ab}	355 ± 2^{Cc}	402 ± 0^{Bb}	16.1 ± 0.1^{Cb}	28.2 ± 0.1^{Aa}	20.3 ± 0.1^{Ba}	26.2 ± 0.2^{Ab}	12.6 ± 0.1^{Cc}	19.8 ± 0.1^{Bb}

Values are mean ± standard deviation (n = 3); different lowercase letters indicate significant differences among sites within the same manure type and different capital letters indicate significant differences among manure types within the same site ($P < 0.05$).

Table 3 Dry matter decomposition rates, and C and N mineralization rates of the different manures at the four sites after 378 days in our experiment.

	Dry matter decomposition rate (k year^{-1})			C mineralization rate (k year^{-1})			N mineralization rate (k year^{-1})		
Site	Cattle manure	Sheep manure	Goat manure	Cattle manure	Sheep manure	Goat manure	Cattle manure	Sheep manure	Goat manure
Taita	0.288 ± 0.035^{Ac}	0.200 ± 0.042^{Ab}	0.234 ± 0.002^{Ab}	0.34 ± 0.03^{Ac}	0.42 ± 0.04^{Ab}	0.42 ± 0.00^{Ab}	0.24 ± 0.03^{Ab}	0.21 ± 0.04^{Aa}	0.05 ± 0.00^{Ab}
ILRI	0.989 ± 0.146^{Aa}	0.457 ± 0.070^{Ba}	0.750 ± 0.134^{Aa}	1.20 ± 0.15^{Aa}	0.67 ± 0.07^{Ba}	1.03 ± 0.13^{Aa}	0.54 ± 0.15^{Aa}	0.36 ± 0.07^{Aa}	0.36 ± 0.13^{Aa}
Kapiti	0.722 ± 0.067^{Ab}	0.318 ± 0.009^{Bab}	0.440 ± 0.053^{Bb}	0.95 ± 0.07^{Aa}	0.49 ± 0.01^{Bab}	0.54 ± 0.05^{Bab}	0.39 ± 0.07^{Aab}	0.27 ± 0.01^{Aa}	0.17 ± 0.05^{Aab}
Embu	0.454 ± 0.065^{Ac}	0.203 ± 0.053^{Ab}	0.257 ± 0.147^{Ab}	0.66 ± 0.06^{Ab}	0.20 ± 0.05^{Bc}	0.41 ± 0.15^{Bb}	0.25 ± 0.06^{ABb}	0.31 ± 0.05^{Aa}	0.06 ± 0.15^{Bb}

Values are mean ± standard deviation (n = 3); different lowercase letters indicate significant differences among sites within the same manure type and different capital letters indicate significant differences among manure types within the same site ($P < 0.05$).



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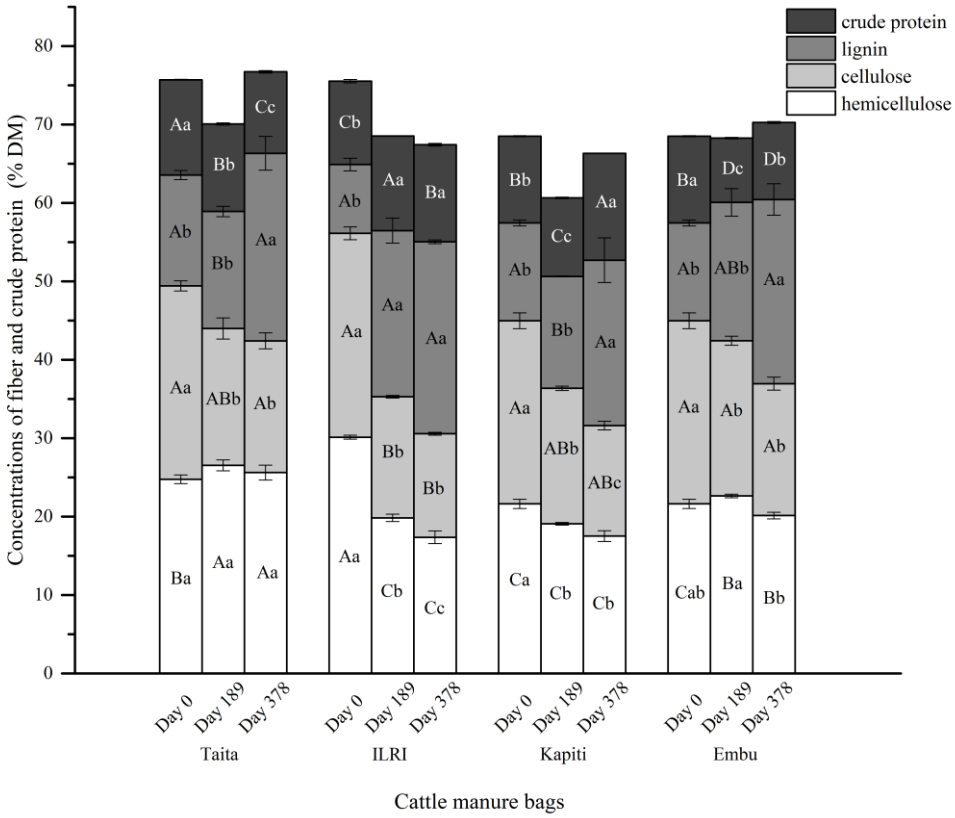

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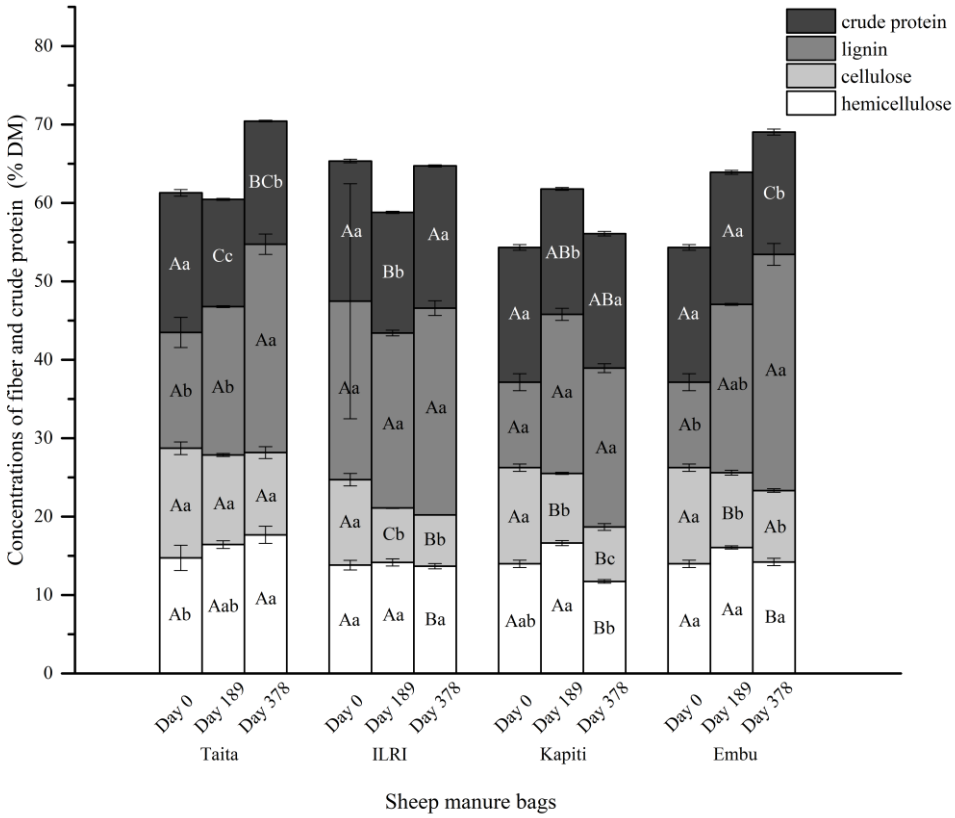

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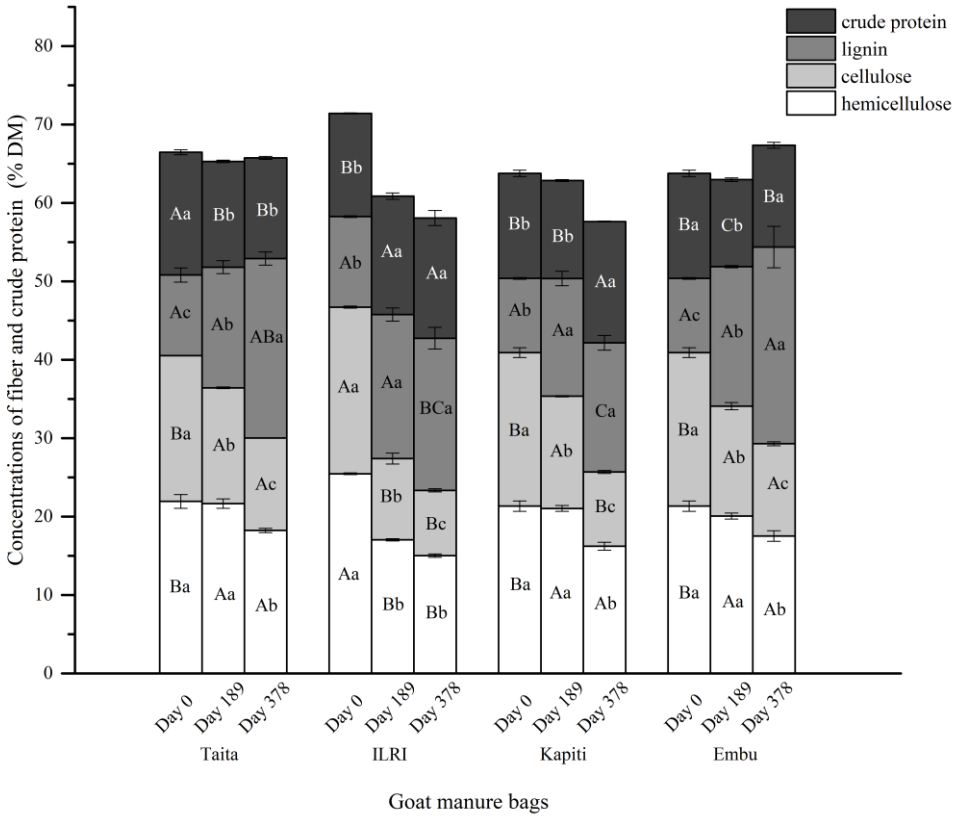

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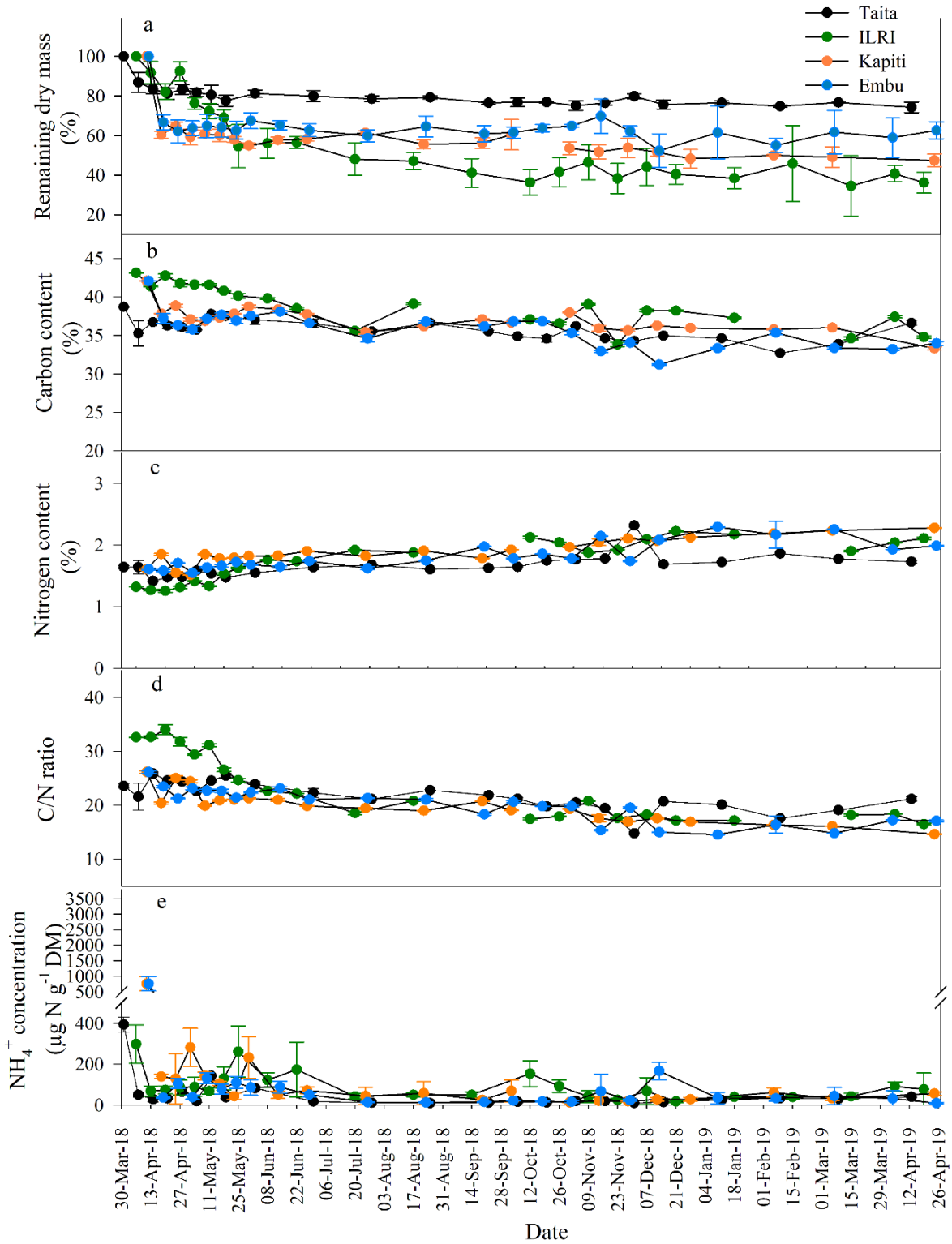

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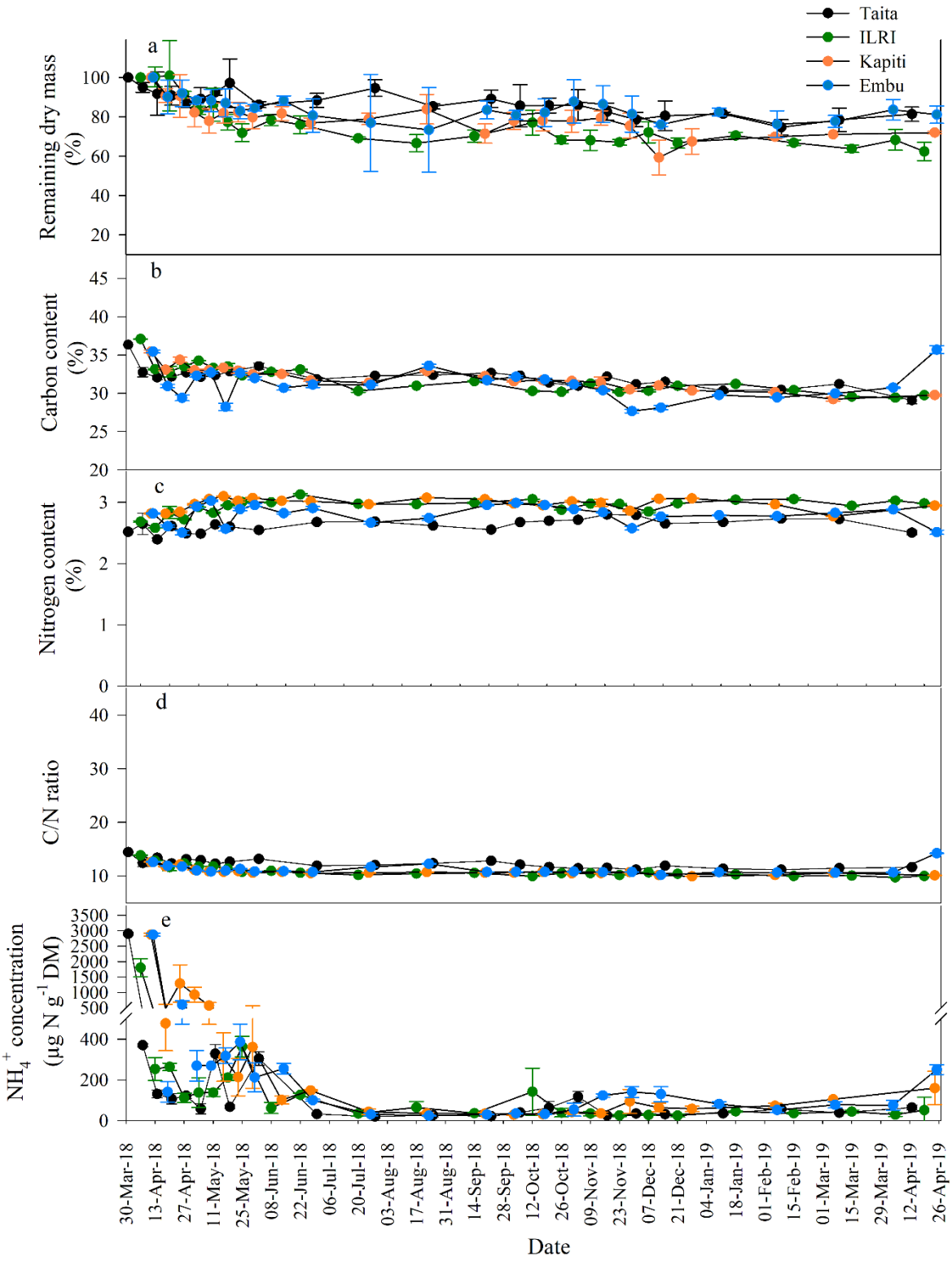

Fig. 6 Dynamics of sheep manure remaining dry mass (a), carbon content (b), nitrogen content (c), C/N ratio (d) and NH_4^+ concentration (e) after litter bags application at Taita, ILRI, Kapiti and Embu sites, respectively. Each value represents the mean of three replicates (± STDEV).

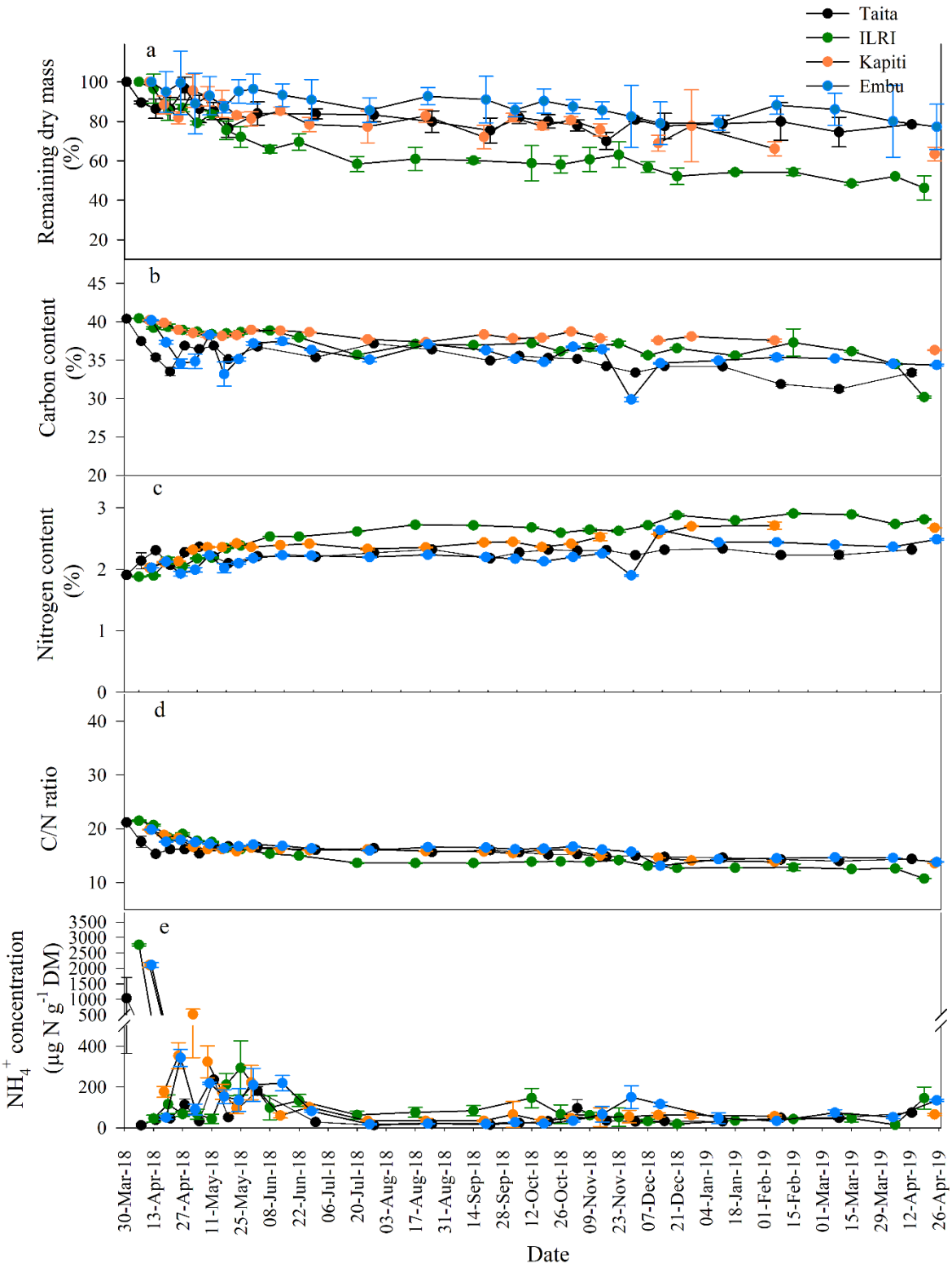

Fig. 7 Dynamics of goat manure remaining dry mass (a), carbon content (b), nitrogen content (c), C/N ratio (d) and NH_4^+ concentration (e) after litter bags application at Taita, ILRI, Kapiti and Embu sites, respectively. Each value represents the mean of three replicates (± STDEV).

Supplementary Information

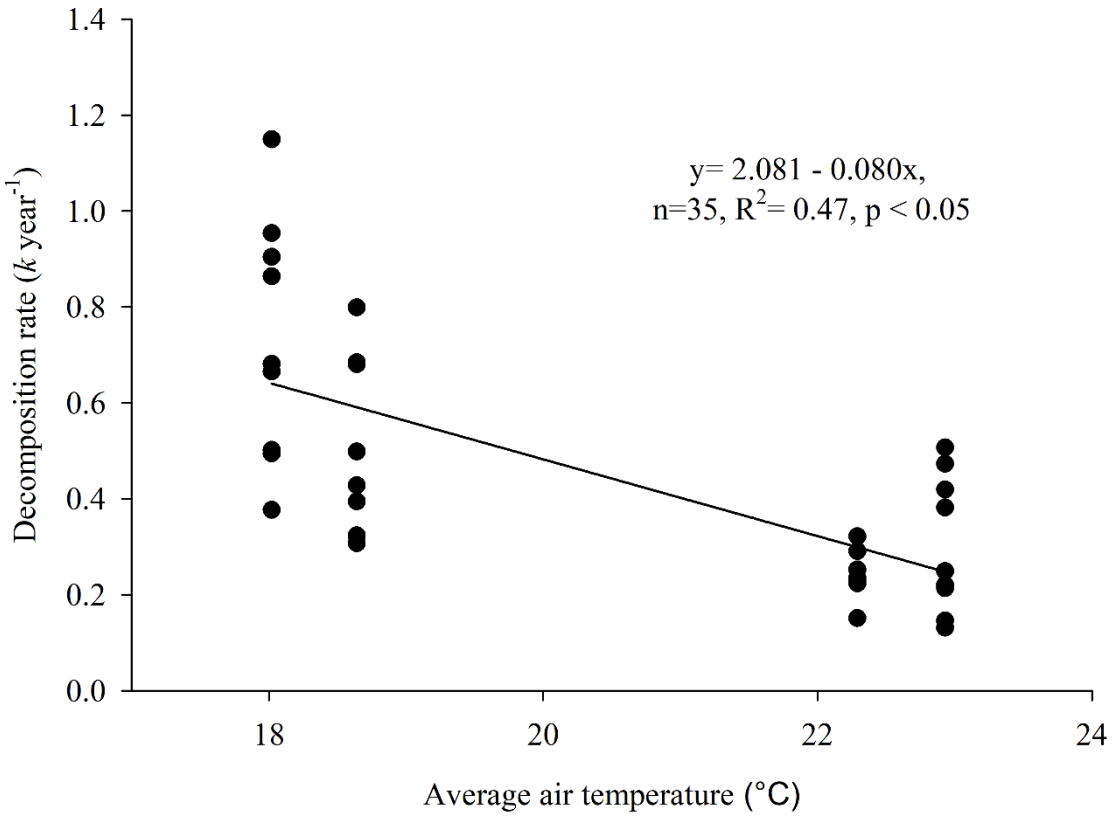

Figure S1 Linear regression between average air temperature and manure dry matter decomposition rate.

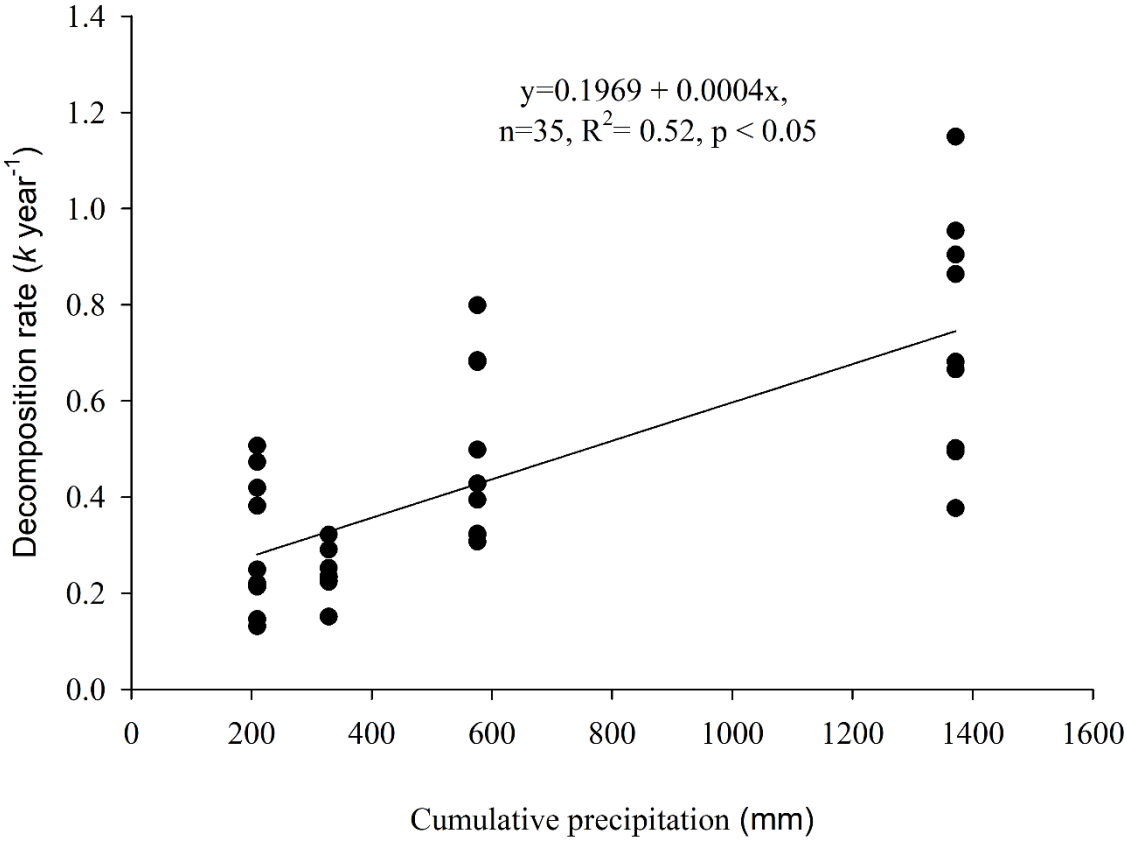

Figure S2 Linear regression between cumulative precipitation and manure dry matter decomposition rate.

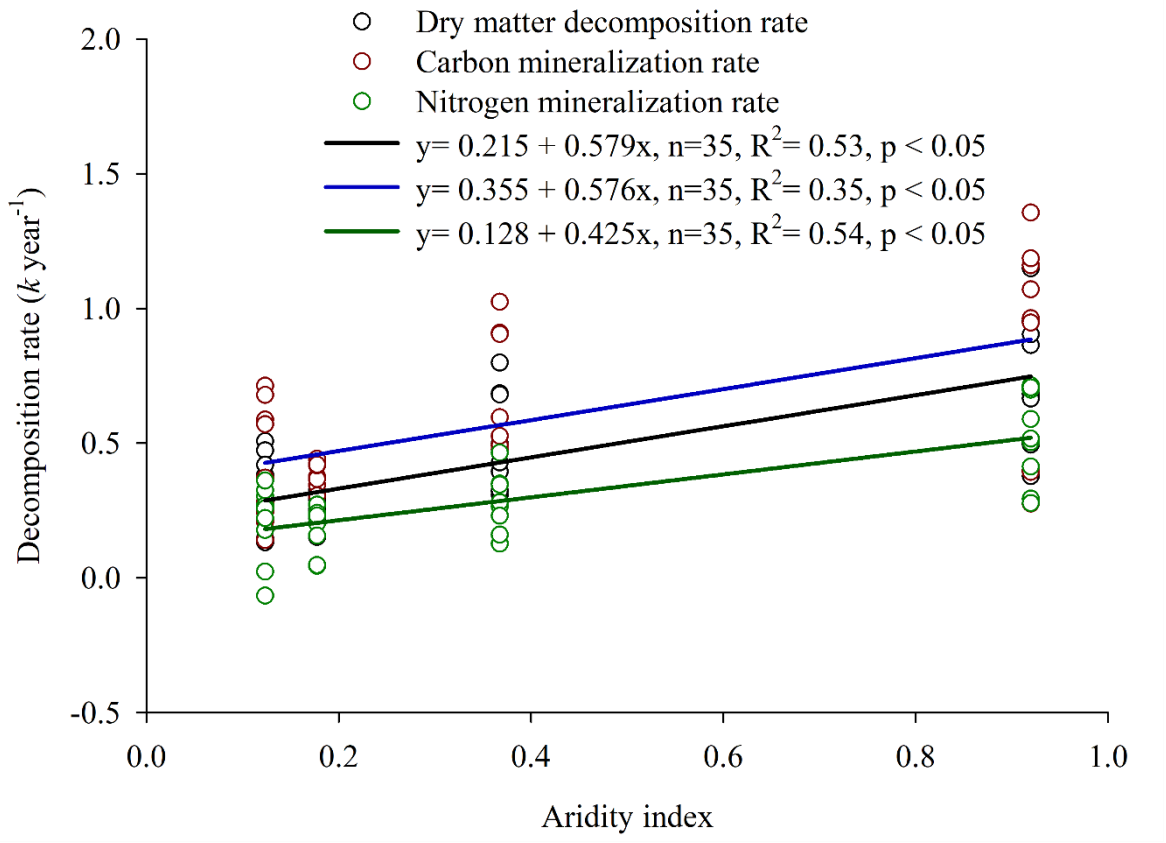

Figure S3 Linear regression between aridity index and rates of decomposition of manure dry matter, manure carbon and manure nitrogen.

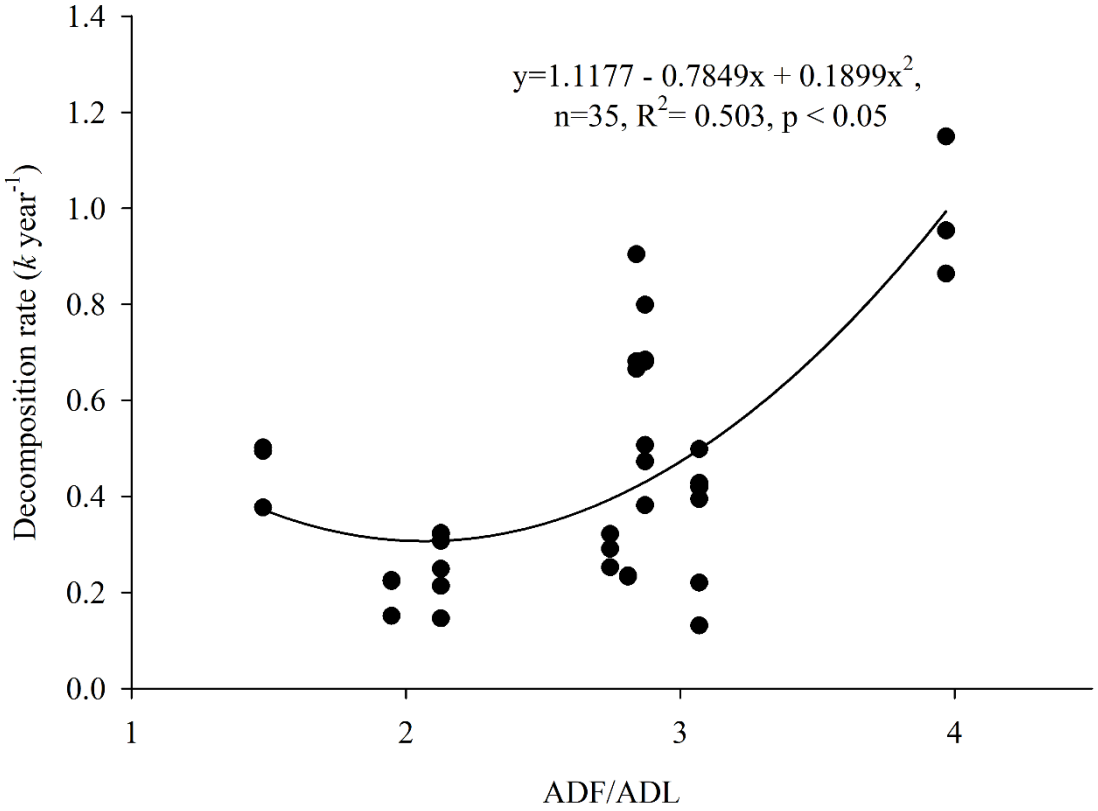

Figure S4 Dependence of the manure dry matter decomposition rate from the ratio of acid detergent fiber (ADF) to acid detergent lignin (ADL).

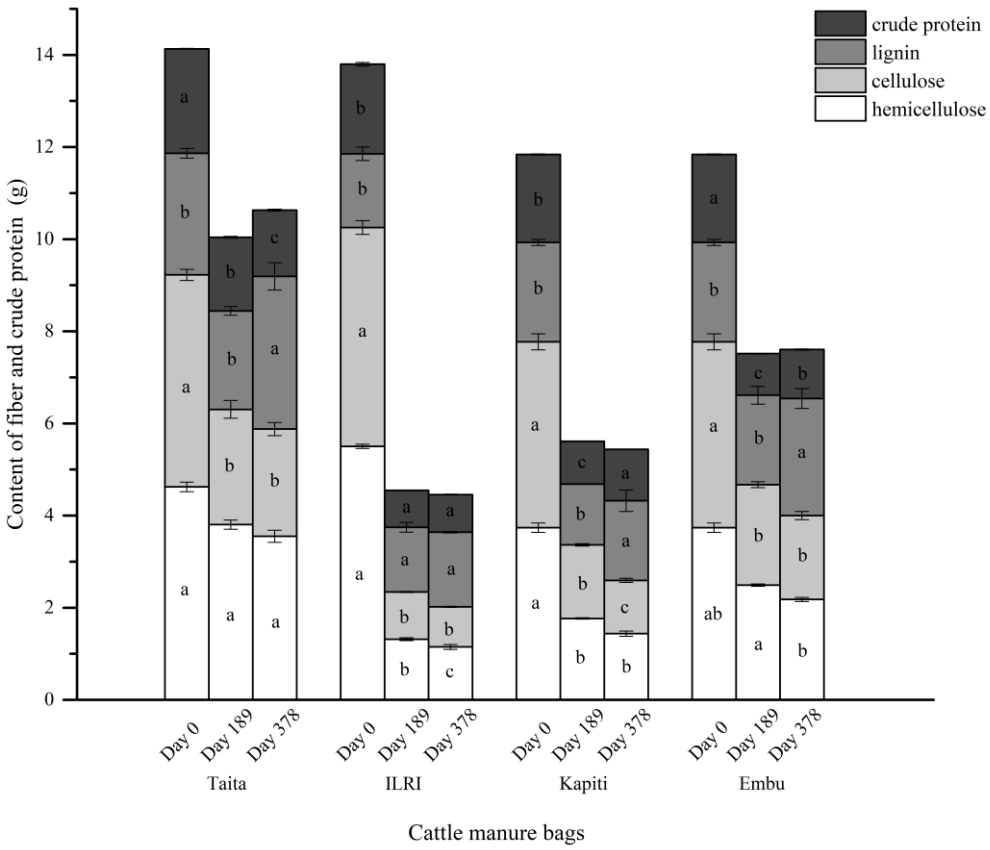

Figure S5 Cattle manure hemicellulose, cellulose, lignin and crude protein dynamics in Taita, ILRI, Kapiti and Embu sites. Each value represents the mean of two replicates for hemicellulose, cellulose and lignin and three replicates for crude protein (± STDEV); different lowercase letters indicate significant differences among sampling dates within the same site ($p < 0.05$).

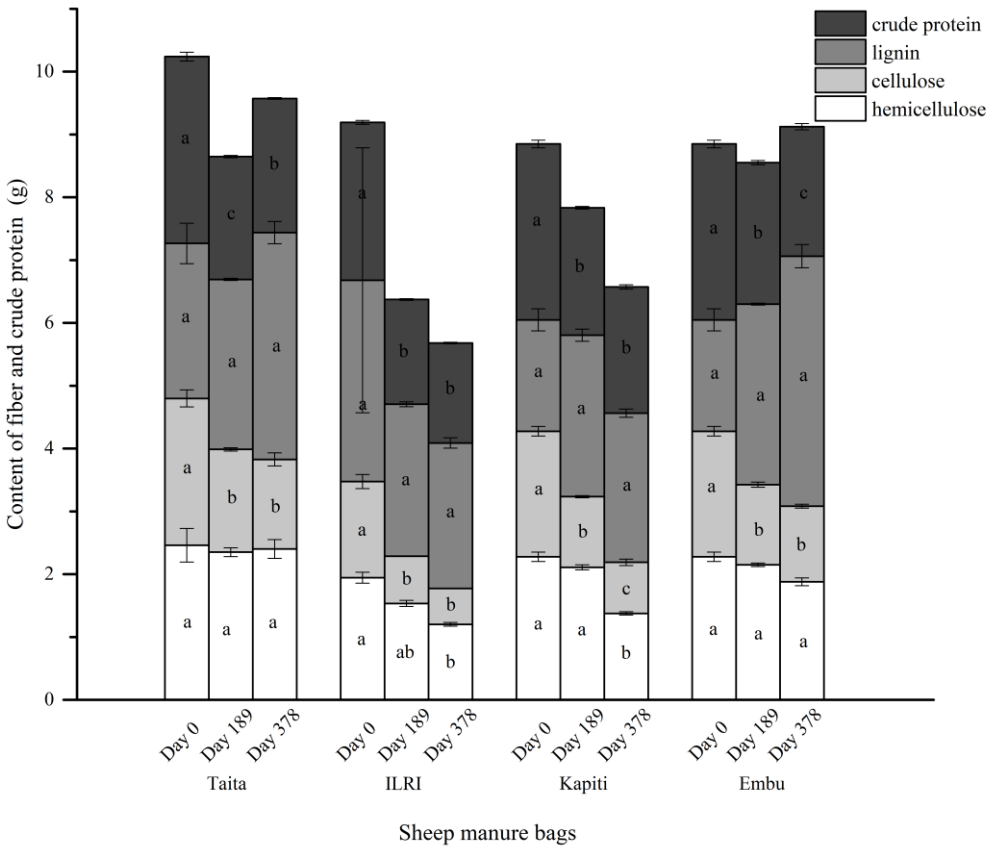

Figure S6 Sheep manure hemicellulose, cellulose, lignin and crude protein dynamics in Taita, ILRI, Kapiti and Embu sites. Each value represents the mean of two replicates for hemicellulose, cellulose and lignin and three replicates for crude protein (± STDEV); different lowercase letters indicate significant differences among sampling dates within the same site ($p < 0.05$).

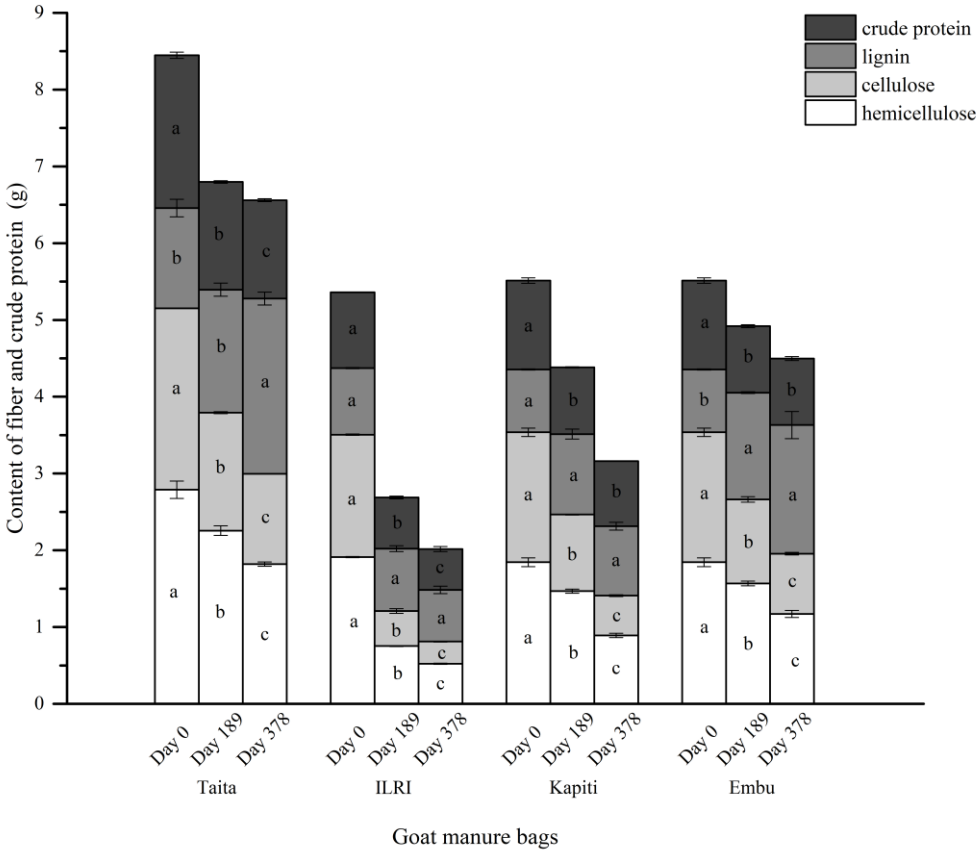

Figure S7 Goat manure hemicellulose, cellulose, lignin and crude protein dynamics in Taita, ILRI, Kapiti and Embu sites. Each value represents the mean of two replicates for hemicellulose, cellulose and lignin and three replicates for crude protein (± STDEV); different lowercase letters indicate significant differences among sampling dates within the same site ($p < 0.05$).