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To the Editor
Biogeosciences Journal

Dear Sir

Ref: Minor Revision on manuscript "Soil Greenhouse Gas Emissions under Different Land-Use Types in Savannah Ecosystems of Kenya"

We would like to thank you for the positive feedback we received from you on the above-mentioned manuscript. We also would like to thank our reviewers for their positive feedback and the constructive comments. Based on the comments, we have made all the corrections for areas we agree with the reviewer and given justifications where otherwise in the document attached.

Looking forward to your positive response.

Sincerely

Sheila Wachiye

Minor Revision on manuscript "Soil Greenhouse Gas Emissions under Different Land-Use Types in Savannah Ecosystems of Kenya"

This document is therefore structured with each of the reviewer's comment (indicated by RC) followed by our response (indicated as AC and in italic).

RC: Line 35: could also be microbial/mycorrhizal respiration (especially if respiring root exudates)

AC: We thank the reviewer for pointing this out and we agree. In the abstract, we limited our conclusion to what we observed during the campaigns, while in the discussion we have provided an explanation on the importance of soil microbial/mycorrhizal activity in seasonal variation of soil GHG emissions.

RC: 50(ish): describe how soil can be a sink for N_2O and CO_2 as well; the text as written only describes their source dynamics, which contrasts the first sentence of the introduction.

AC: We have now included this in the first paragraph of the introduction, which reads as below:

Soil is a major source, and in many cases also a sink, of the atmospheric greenhouse gases (GHG) carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) (Oertel et al., 2016). The concentrations of these gases have increased since the onset of the industrial revolution (from about1750), leading to global warming (IPCC, 2013). GHGs trap the long-wave radiation emitted by the Earth's surface, thus increasing surface temperatures (Arrhenius, 1896). Soil CO_2 emissions originate from root & mycorrhiza respiration and heterotrophic decomposition of soil organic matter (Oertel et al., 2016). In addition to being a CO_2 source, by increasing the soil organic carbon (SOC) content soils can also act as a sink for CO_2. N_2O on the other hand can be produced from many pathways in the soil nitrogen (N) cycle, but is considered to result primarily from nitrification and denitrification (Butterbach-Bahl et al., 2013). N_2O uptake into soils is also possible as observed previously (e.g.(Butterbach-Bahl et al., 2002; Rosenkranz 2006; Flechard et al., 2005), and it depends on the complete reduction of N_2O to N_2, the ease of N_2O diffusion within the soil profile, and its dissolution in soil water (Chapuis-Lardy et al. 2007). CH_4 is produced by methanogenesis under anaerobic conditions and consumed by methanotrophic microorganisms under aerobic conditions, with the latter being more important in well-aerated upland soils, which consequently show net CH_4 uptake (i.e. negative flux) (Serrano-silva et al., 2014; Hanson and Hanson 1996).

RC: Line 57: I recommend also adding the critical role of plants to soil GHG fluxes

AC: We have now included the role of vegetation in the second paragraph of our introduction as follows below:

The production and consumption of soil GHGs largely depend on soil physical and chemical properties (Davidson et al., 2006) (e.g. texture, soil organic matter and pH) and are further driven by environmental factors such as soil moisture and soil temperature (Davidson et al., 2006). In addition, vegetation affects both biotic and abiotic factors that drive soil emissions (Raich and Tufekcioglu, 2000; Pinto et al., 2002) and net carbon assimilation (La Scala et al., 2000). Vegetation type directly influences soil physicochemical properties, which in turn modify soil microbial activities (Raich and Tufekcioglu, 2000). It also controls the quantity of plant carbon allocated belowground (Metcalfe et al., 2011) by determining root biomass and litter quality and quantity (Fanin et al., 2011; Rey et al., 2011). Vegetation composition additionally affects root respiration and the associated microbial components. Active roots add directly to soil respiration, while dead roots and root exudates provides carbon as a source of energy and nutrients for soil microbial biomass (Tufekcioglu et al., 2001). Hence, changes in vegetation types and cover due to land-use system and land-use management activities have the potential to modify the soil-to-atmosphere GHG exchange (Raich and Tufekcioglu 2000). Thus, soil GHG emissions and uptake along with their controlling factors differ between biomes based on land use and land-use management.

RC: Line 241: A brief description of the power regression for N_2O would be forthcoming.

AC: We thank the reviewer for this comment. Therefore, we have included both the functions for linear model and the power model as follows.

$$Conc_{CO_2,\ CH_4} = ax + \beta \quad (1)$$

$$Conc_{N_2O} = ax^{\beta} \quad (2)$$

Where $Conc_{CO_2,\ CH_4}$ are the carbon dioxide and methane concentrations in ppm, F_{N_2O} is the nitrous oxide concentration in ppb, a and β are model coefficients, and x is the peak area derived from the GC. Both equations are based on peak area measurements of known standards with our GCs, and while the FID (CO_2 & CH_4 detection) is linear over the entire concentration range, the ECD (N_2O detection) behaves non-linearly and therefore a power function leads to better fits.

RC: Line 250: what if the outlier was at the beginning or end of measurement indicating a potential nonlinearity?

AC: When an outlier was found, it was investigated to establish whether it was genuine or error. Most outliers found were discarded because they were found to be erroneous (i.e. leaky gas vials, etc.). For our study, the majority of the data collected during each campaign fulfilled the conditions for linearity, which is why we fitted linear models.

RC: Line 264: the SI unit for time is seconds, but I am ok with presenting values per hour. (And 'represents'; minor usage errors should be reviewed throughout the manuscript, e.g. a missing period on p. 607 and 'value' on 595).

AC: We thank the reviewer for this observation. We have corrected this throughout the manuscript.

RC: Equation 2: what is the 0.02241?

AC: We apologise for not including the meaning of this value in our equation. The value 0.02241 is the molar volume of an ideal gas in $m^3\ mol^{-1}$ *at standard temperature and pressure following the ideal gas law. We have made this correction in the manuscript.*

SOIL GREENHOUSE GAS EMISSIONS UNDER DIFFERENT LAND-USE TYPES IN SAVANNA ECOSYSTEMS OF KENYA

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Abstract

Field measurement data on greenhouse gas (GHG) emissions are still scarce for many land-use types in Africa, causing a high uncertainty in GHG budgets. To address this gap, we present *in situ* measurements of carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) emissions from the lowlands of southern Kenya. We conducted eight chamber measurement campaigns on gas exchange from four dominant land-use types (LUTs) including (1) cropland, (2) bushland, (3) grazing land, and (4) conservation land between 29 November 2017 to 3 November 2018, accounting for regional seasonality (wet and dry seasons, and transitions periods). Mean CO_2 emissions for the whole observation period were significantly highest (p-value<0.05) in the conservation land (75±6 mg CO_2-C m^{-2} h^{-1}) compared to the three other sites, which ranged from 45±4 mg CO_2-C m^{-2} h^{-1} (bushland) to 50±5 mg CO_2-C m^{-2} h^{-1} (grazing land). Furthermore, CO_2 emissions varied between seasons, with significantly higher emissions in the wet season than the dry season. Mean N_2O emissions were highest in cropland (2.7±0.6 µg N_2O-N m^{-2} h^{-1}) and lowest in bushland (1.2±0.4 µg N_2O-N m^{-2} h^{-1}) but did not vary with season. In fact, N_2O emissions were very low both in the wet and dry seasons, with slightly elevated values during the early days of the wet seasons in all LUTs. On the other hand, CH_4 emissions did not show any significant differences between LUTs and seasons. Most CH_4 fluxes were below the limit of detection (LOD, ±0.03 mg CH_4-C m^{-2} h^{-1}). We attributed the difference in soil CO_2 emissions between the four sites to soil C content, which differed between the sites and was highest in the conservation land. In

addition, CO_2 and N_2O emissions positively correlated with soil moisture, thus an increase in soil moisture led to an increase in emissions. Furthermore, vegetation cover explained the seasonal variation of soil CO_2 emissions as depicted by a strong positive correlation between NDVI and CO_2 emissions, most likely because with more green (active) vegetation cover, higher CO_2 emissions occur due to enhanced root respiration compared to drier periods. Soil temperature did not show a clear correlation with either CO_2 or N_2O emissions, which is likely due to the low variability in soil temperature between seasons and sites. Based on our results, soil C, active vegetation cover and soil moisture are key drivers of soil GHG emissions in all the tested LUTs in South Kenya. Our results are within the range of previous GHG flux measurements from soils from various LUTs in other parts of Kenya and contribute to more accurate baseline GHG emission estimates from Africa, which are key to reduce uncertainties in global GHG budgets as well as for informing policymakers when discussing low-emission development strategies.

Commented [WS1]: RC: 35: could also be microbial/mycorrhizal respiration (especially if respiring root exudates)

Commented [WS2R1]: *AC: We thank the reviewer for pointing this out and we agree. In the abstract we limited our conclusion to what we observed during the campaigns, while in the discussion we have provided an explanation on the importance of soil microbial/mycorrhizal activity in seasonal variation of soil GHG emissions*

KEYWORDS: Carbon Dioxide, Nitrous Oxide, Methane, Bushland, Conservation, Grazing land, Cropland.

1. Introduction

Soil is a major source, and in many cases also a sink, of the atmospheric greenhouse gases (GHG) carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) (Oertel et al., 2016). The concentrations of these gases have increased since the onset of ~~industrialization in 1970,~~the industrial revolution (from about1750), leading to global warming (IPCC, 2013). GHGs trap the long-wave radiation emitted by the Earth's surface, thus increasing surface temperatures (Arrhenius, 1896). Soil CO_2 emissions originate from root & mycorrhiza respiration and heterotrophic decomposition of soil organic matter (Oertel et al., 2016). ~~N_2O~~In addition to being a CO_2 source, by increasing the soil organic carbon (SOC) content soils can also act as a sink for CO_2. N_2O on the other hand can be produced from many pathways in the soil nitrogen (N) cycle, but is considered to result primarily from nitrification and denitrification (Butterbach-Bahl et al., 2013). N_2O uptake into soils is also possible as observed previously (e.g.Butterbach-Bahl et al., 2002; Rosenkranz 2006; Flechard et al., 2005), and it depends on the complete reduction of N_2O to N_2, the ease of N_2O diffusion within the soil profile, and its dissolution in soil water (Chapuis-Lardy et al. 2007). CH_4 is produced by methanogenesis under anaerobic conditions and consumed by methanotrophic microorganisms under aerobic conditions, with the latter being more important in well-aerated upland soils, which consequently show net CH_4 uptake (i.e. negative flux) (Serrano-silva et al., 2014; Hanson and Hanson 1996).

Commented [WS3]: RC: 50(ish): describe how soil can be a sink for N2O and CO2 as well; the text as written only describes their source dynamics which contrasts the first sentence of the introduction.

Commented [WS4R3]: AC: Done

The production and consumption of soil GHGs largely depend on soil physical and chemical properties (Davidson et al., 2006) (e.g. texture, soil organic matter and pH) and are further driven by environmental factors such as soil moisture and soil temperature (Davidson et al., 2006). ~~Thus, soil GHG emissions and uptake along with their controlling factors differ between biomes based on the~~In addition, vegetation affects both biotic and abiotic factors that drive soil emissions (Raich and Tufekcioglu, 2000; Pinto et al., 2002) and net carbon assimilation (La Scala et al., 2000). Vegetation type directly influences soil physicochemical properties, which in turn modify soil microbial activities (Raich and Tufekcioglu, 2000). It also controls the quantity of plant carbon allocated belowground (Metcalfe et al., 2011) by determining root biomass and litter quality and quantity (Fanin et al., 2011; Rey et al., 2011). Vegetation composition additionally affects root respiration and the associated microbial components. Active roots add directly to soil respiration, while dead roots and root exudates provides carbon as a source of energy and nutrients for soil microbial biomass (Tufekcioglu et al., 2001). Hence, changes in vegetation types and cover due to land-use system and land-use management activities have the potential to modify the soil-to-atmosphere GHG exchange (Raich and Tufekcioglu 2000). Thus, soil GHG emissions and uptake along with their controlling factors differ between biomes based on land use and land-use management.

Commented [WS5]: RC: 57: I recommend also adding the critical role of plants to soil GHG fluxes

Commented [WS6R5]: AC: Done

Land-use changes are reportedly the largest source of anthropogenic GHG emissions in Africa (Valentini et al., 2014). However, *in situ* studies on GHG emissions from various ecosystems in remain scarce, particularly from savanna ecosystems (Castaldi et al., 2006). Savanna is an important land cover type in Africa, covering more than 40 % of its total area (Scholes et al., 1997). In Kenya, savanna and grassland ecosystems cover about 80 % of the total area, comprising various land-use types (LUTs) (GoK, 2013). These ecosystems are subject to accelerating land-use change (Grace et al., 2006) due to population growth (Meyer and Turner, 1992) and land-use management activities (Valentini et al., 2014). Conversion of savanna for small- and large-scale livestock production, crop cultivation, and human settlement is common in Africa (Bombelli et al., 2009). As a consequence, vegetation cover, net primary productivity, allocation of carbon and nutrients in plants and soil (Burke et al., 1998) as well as soil GHG emissions are affected (Abdalla et al., 2018; Carbone et al., 2008).

Overgrazing due to overstocking is a major cause of soil and vegetation degradation in large parts of African savannas (Patton et al., 2007; Abdalla et al., 2018). Factors associated with grazing include animal feeding preferences to specific plant species, thus creating higher pressure for those species, which decline in numbers over time, leading to species loss and lower pasture nutritive value (Patton et al., 2007). In addition, soil trampling increases soil bulk density and reduces soil water infiltration

(Patton et al., 2007). Furthermore, high rates of dung and urine deposition, especially around homesteads and waterholes, create high N concentrations that are toxic for many savanna grass species, affecting vegetation cover and composition (e.g. increase of encroaching species such as *Solanum incanum* L., which is toxic for livestock (van Vegten, 1984)). Given that all these factors affect soil properties, soil GHG emissions are most likely similarly affected (Wilsey et al., 2002).

In addition to overgrazing, rapid human population growth leads to more people migrating into savanna ecosystems, which has led to the expansion of cropland (Pellikka et al., 2018; Patton et al., 2007). Brink and Eva (2009) found that the area under cropland increased by 57 % between 1975 and 2000 in Africa. In the Horn of Africa, cropland areas increased by 28 % between 1990 and 2010 (Brink et al., 2014), while wooded vegetation in East Africa decreased by 5 % in forests, 16 % in woodlands, and 19 % in shrublands (Pfeifer et al., 2013). As an additional example, in our study area Taita Taveta County in Southern Kenya, the area under cropland increased from 30 % in 1987 to 43 % in 2011 (Pellikka et al., 2018). However, in the Taita Hills, located in the County, this trend has slowed down in recent years, while the savanna lowlands are still being cleared to make way for new cropland (Pellikka et al., 2013).

Croplands in the Kenyan savannas are mostly managed by smallholder farmers (Waswa and Mburu, 2006). Due to high poverty levels in this region, inputs to improve crop yields, such as the use of fertilizer and herbicide, and mechanized farming are minor (Waswa and Mburu, 2006; CIDP, 2014). Thus, an increase in productivity are mostly via cropland expansion. In spite of this, these smallholder farms are likely to have substantial effects on national GHG emission budgets (Pelster et al., 2017). Until now, only a few studies have investigated soil GHG emissions from such agricultural landscapes (Rosenstock et al., 2016), and these studies were mostly carried out in high-potential farming areas such as the Kenyan highlands, which receive >1000 mm rainfall (FAO, 1996). For example, Rosenstock et al. (2016) showed a large variation of CO_2 and N_2O emissions both within and between four crop types as affected by environmental conditions and land management. However, studies measuring GHG emissions from low-productivity croplands in southern Kenya are to the best of our knowledge still missing. Thus, this study focused on soil GHG emissions from different LUTs relevant for the semi-arid region of Southern Kenya.

Given the vast area covered by savanna, land use and land-cover changes are likely to affect global, regional, and national C and N cycles, and hence the quantification of their role is vital (Lal, 2004; Williams et al., 2007). Studies in Kenya have shown large variations of soil GHG emissions in various savanna ecosystems (Otieno et al., 2010; Oduor et al., 2018), due to land-use (Ondier et al., 2019)

and management activities (K'Otuto et al., 2013). Owing to the high diversity of these savanna ecosystems, such studies may not be entirely representative for every region (Ardö et al., 2008).

The lack of reliable soil GHG flux data from natural savanna and cropland limits our understanding of GHG emissions from African soils (Hickman et al., 2014; Valentini et al., 2014). At the same time, accurate quantification of GHG emissions from multiple LUTs are essential to allow for reliable estimation of Kenya's national GHG inventory (IPCC, 2019). This is particularly important as Kenya currently relies on a Tier-1 approach by using default emission factors (EFs) provided in the Guidelines for Greenhouse Gas Inventories of the UN Intergovernmental Panel on Climate Change (IPCC) to estimate national GHG emission budgets. Following the Paris Climate Agreement (https://unfccc.int/process-and-meetings/the-paris-agreement/d2hhdC1pcy), most countries across the globe, including Kenya, have not only agreed to accurately report their GHG emissions at national scales following a Tier-2 approach (i.e. using localized data) but also to mitigate anthropogenic GHG emissions in the upcoming decades, as is communicated via their Nationally Determined Contributions (NDCs). Both can only be achieved with locally derived data.

To address the lack of localized GHG emission data from different LUTs in Kenya, our study aims at: (1) providing crucial baseline data on soil GHG emissions from four dominant land uses, namely conservation land, grazing land, bushland, and cropland, and (2) investigating abiotic and biotic drivers of GHG emissions during different seasons. We hypothesized that GHG emissions in cropland would be higher compared to grazing land, bushland, and conservation land because of larger nutrient inputs (i.e. fertilization) in managed land. Further, we hypothesized that GHG emissions would differ between seasons; more precisely, we expected higher GHG emissions in the wet season than in the dry season caused by higher soil moisture.

2. Materials and Methods

2.1. Study Area

This study was conducted in the lowlands (800–1000 m a.s.l.) of Taita Taveta County (latitude 3° 25´ S and longitude 38° 20´ E) located in southern Kenya (Fig. 1). Taita Taveta County is one of Kenya's dryland areas, with 89 % of the area characterized as arid and semi-arid area. The county is divided into three major geographical regions, namely the mountainous zone of the Taita Hills (Dawida, Kasigau, Sagalla), Taita lowlands, and the foot slopes of Mt. Kilimanjaro around Taveta. In the lowlands, vegetation types include woodlands, bushlands, grasslands, and riverine forests/swamps.

Tsavo East and Tsavo West National Parks covers ca. 62 % of the county area (CIDP, 2014). The parks are open savanna and bushland that support large herbivores, predators and a wealth of birdlife. There are 28 ranches designated for livestock production and two wildlife sanctuaries (Taita Hills Wildlife Sanctuary and LUMO Community Wildlife Sanctuary). Other important land uses include cropland under small-scale farming (CIDP, 2014), shrublands, and sisal farming (Pellikka et al., 2018). Soil type is characterized by dark red, very deep, acid sandy clay soil (Ferralsols). Our study sites were located in four of these key land uses in the region including cropland, bushland, wildlife conservation land, and grazing land.

The lowland has a bimodal rainfall pattern with two rainy seasons – a long rain season between March and May and a short rain season between October and December (CIDP, 2014). The hot and dry months are January and February while the dry season from June to October is cooler (Pellikka et al., 2018). Mean annual rainfall is 500 mm and the mean annual air temperature is 23 °C, with an average daily minimum and maximum temperature of 16.7 °C and 28.8 °C respectively (CIDP, 2014).

The first site investigated is cropland located in Maktau (1070 m a.s.l., Fig. 1, Fig. 2a) about one and a half hectares, cultivated with maize (Zea mays L.) intercropped with beans. The farm is a typical rain-fed smallholder farm and crop growing closely follows the rainy seasons, with sowing in March, and harvesting in June for beans and August for maize. Animal ploughing is done to prepare land before seeding and weeding is by hand hoeing. Small quantities of fresh and dry manure (roughly 20 kg, accounting for less than 1 kg of N) were used every month to improve soil fertility.

The second site is located in a private bushland in Maktau next to the cropland (1076 m a.s.l., Fig. 1, Fig. 2b). In this region, bushland is found both within the conservation areas and under private ownership. Bushland forms a cover with over 50 % of thorny shrubs and small trees, characterized by *Acacia spp* and *Commiphora spp*. The bushes may vary in height ranging from two to five metres. Herbs and savanna grasses (mostly annual or short-lived perennials) less than one-metre tall form the ground cover. Private bushland similar to our study site is used by the farmers to generate small income from forest products such as timber, poles, and firewood, and charcoal to some extent. Additionally, some grazing occurs primarily by livestock owned by the farmer (CIDP, 2014).

The third site, grazing land (covering approximately 460 km^2) is located in the LUMO Community Wildlife Sanctuary (970 m a.s.l, Fig. 1, Fig. 2c) next to Tsavo West National Park and Taita Hills Wildlife Sanctuary. The sanctuary was formed by merging three ranches, namely Lualenyi and Mramba communal grazing areas and Oza group ranch thus the name "LUMO". This sanctuary is

communally owned (GoK, 2013) designated for community livestock grazing, where wildlife is also present, as conservation areas are not necessarily fenced. However, overgrazing is a major challenge, caused by herders who enter the conservancy illegally especially in the dry season (CIDP, 2014).

The forth site is the conservation land located within the Taita Hills Wildlife Sanctuary (928 m a.s.l., Fig. 1, Fig. 2d) covering an area of ca. 110 km^2. This is a private game sanctuary for wildlife conservation located between LUMO and communal land. The sanctuary is an open savanna grassland dominated by *Schmidtia bulbosa* and *Cenchrus ciliaris* grass species forming an open to closed ground cover, shrublands, and scattered woodlands with *Acacia spp.* as main tree species. However, most trees have been damaged by elephants, leaving the landscape open. The sanctuary is well managed with the application of ecological management tools such as controlled fires. Through these and other conservation efforts, the sanctuary has attracted a higher diversity of large mammals, many of which remain within the unfenced sanctuary throughout the year. Wildlife are the predominant grazers and browsers, although livestock encroachment may be a problem especially during the dry season on the western and eastern borders of the sanctuary (GoK, 2013).

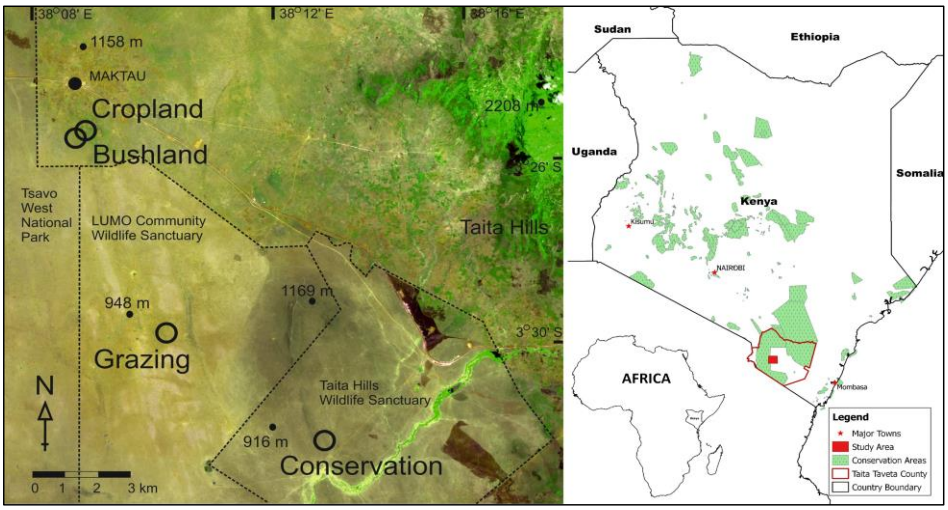

Figure 1. *Location of the study sites cropland, bushland, grazing land, and conservation land in the savanna area in the lowlands of Taita Taveta County in southern Kenya. Image showing the sites is Sentinel-2A acquired from Sentinel's Scientific DataHub (ESA, 2015). The Kenyan and African boundary ©World Resources Institute (retrieved from https://www.wri.org/resources/data-sets/kenya-gis-data)*

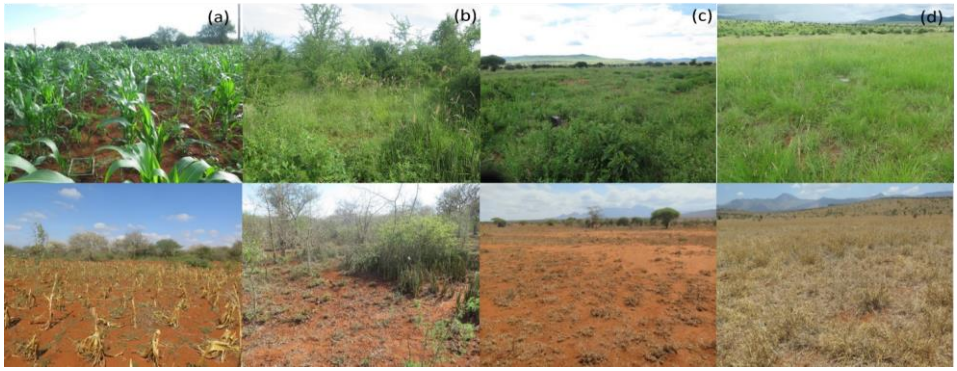

Figure 2: *The four land-use types: (a) cropland, (b) bushland, (c) grazing land, and (d) conservation land. The upper panel shows the land-use types during the wet season, while the lower panel depicts the situation during the dry season. The grey plastic collars visible in upper left photo are frames for the GHG flux chambers.*

2.2. Defining the seasons

We divided our campaigns into dry and wet seasons, based on the agro-climatic concept. The onset of the wet season was the first wet day of a 3 day wet spell receiving at least 20 mm without any 10 day dry spell (< 1 mm) in the next 20 days from 1 March for the long wet season and 1 September for the short wet season (Marteau et al., 2011). Equally, the end of the rainy season was the first of 10 consecutive days with no rain. Thus, for this study, the long wet season (LW) was between 2 March to 4 June 2018, and the short wet season (SW) between 23 October and 26 December 2018. The two wet seasons were separated by two dry seasons, the short dry season (SD) from January to February 2018, and the long dry season (LD) from June to September 2018. We had three campaigns in each of the wet season: the early days of the wet seasons onset (onset-SW, onset-LW), the peak of the seasons (mid-SW, mid-LW), as well as at the end of the seasons (end-SW, end-LW).

2.3. Chamber measurements of greenhouse gas emission

Soil-atmosphere exchange of CO_2, N_2O, and CH_4 were measured in eight one-week campaigns from 29 November 2017 to 3 November 2018 using the static chamber method (Rochette, 2011; Hutchinson et al., 1981). Within each of the four sites (LUTs), three clusters were randomly selected as replicates for soil GHG concentration measurements. In each cluster, three plastic collars (27cm × 37.2 cm × 10 cm) were inserted (5–8 cm) into the soil at least 24 hours before the first sample was taken (see Pelster et al., 2017 for further details). The collars were left in the ground for the entire

measurement period to minimize soil disturbance during measurements (Søe et al., 2004). Any damaged or missing collars (mostly due to livestock or wildlife activity) were replaced, at least 24 hours before the next gas sampling. During each day of a campaign, gas sampling was conducted daily between 7:00 and 11:00 am, which is about the average flux of the diurnal cycle (Parkin and Venterea, 2010).

During each gas-sampling day, grey opaque PVC lids (27 cm × 37.2 cm × 12 cm) covered with reflective tape were placed onto the collars for 30 mins. Lids were fitted with a fan for gas mixing and a vent to avoid pressure differences between the chamber headspace and outside atmosphere (Pelster et al., 2017). A rubber seal was fitted along the edges of the chamber lid and paper clips used to hold the lid and collar in place to ensure airtightness. Four gas samples were then collected every 10 mins (time 0, 10, 20, 30 mins) after lid deployment (Rochette, 2011). The height of each chamber collar was measured on each sampling day to derive the total chamber volume (total chamber height = height of chamber collar sticking out of the soil + height of the chamber lid). A slightly modified version of the gas-pooling method was used to reduce the overall sample size while ensuring a good spatial representation of each LUTs (see Arias-Navarro et al., 2013). Here, 20 ml of headspace air were collected from each of the three chambers at each time interval with polypropylene syringes (60 ml capacity), resulting in a composite gas sample of 60 ml. The first 40 ml were used to flush the vials, and the remaining 20 ml over-pressured into 10 ml glass vials to minimize contamination of the gas with ambient air during transportation (Rochette et al., 2003).

Gas samples were transported to the laboratory (Mazingira Centre, mazingira.ilri.org) and analysed using a gas chromatograph (GC, model SRI 8610C). The GC was fitted with a ^{63}Ni-Electron Capture Detector (ECD) for detecting N_2O concentrations and a Flame Ionization Detector (FID) fitted with a methanizer for CH_4 and CO_2 analysis. The GC was operated with a Hayesep D packed column (3 m, 1/8″) at an oven temperature of 70 °C, while ECD and FID detectors were operated at a temperature of 350 °C. Carrier gas (N_2) flow rate was 25 mL min^{-1} on both FID and ECD lines. In every 40 samples analysed with the GC were eight calibration gases with known CO_2, CH_4, and N_2O concentrations in synthetic air (levels of calibration gases ranged from 400 to 2420 ppm for CO_2, 360 to 2530 ppb for N_2O, and 4.28 to 49.80 ppm for CH_4). Therefore, the gas concentrations of the samples were calculated from peak areas of samples in relation to peak areas of standard gases using

a linear model for CO_2 and CH_4 and a power regression for N_2O using Eq (1) that follows.

$$Conc_{CO_2,\ CH_4} = ax + \beta \quad (1)$$

$$-Conc_{N_2O} = ax^{\beta} \quad (2)$$

Where $Conc_{CO_2,\ CH_4}$ are the carbon dioxide and methane concentrations in ppm, F_{N_2O} is the nitrous oxide concentration in ppb, a and β are model coefficients, and x is the peak area derived from the GC. Both equations are based on peak area measurements of known standards with our GCs, and while the FID (CO_2 & CH_4 detection) is linear over the entire concentration range, the ECD (N_2O detection) behaves non-linearly and therefore a power function leads to better fits.

2.4.Greenhouse gas flux calculations

Soil GHG emissions were determined by the rate of change in gas concentration in the chamber headspace over time by linear fitting. The goodness of fit was used to evaluate the linearity of concentration increases/decreases. The dynamics of the CO_2 concentrations over the 30 min deployment period for each gas concentration was assessed to test for chamber leakage due to the typically more robust and continuous flux of CO_2 (Collier et al., 2014). If the linear model of CO_2 versus deployment time had an $R^2 > 0.95$ using all four-time points (T1, T2, T3, and T4), the measurement was considered valid and four-time points were used for analysing the CO_2, N_2O, and CH_4 emissions. However, if $R^2 < 0.95$ for CO_2 and one data point was a clear outlier, this point was discarded and the three remaining points used for the flux calculation if they showed a strong correlation of CO_2 versus time. Measurements that did not show a clear trend of CO_2 with time were considered faulty, and the entire data point series was discarded. In addition, data points that showed a decrease in CO_2 concentration over time were assumed to indicate leakage and were similarly discarded (chambers were opaque, i.e. photosynthesis was inactive during chamber deployment). However, if no leakage was found, negative CH_4 and N_2O emissions were accepted as the uptake of the respective gas by the soil. Emissions were calculated according to Eq. (1):

$$F_{GHG} = \frac{(\frac{dc}{dt}) \times V_{ch} \times M_w}{A_{ch} \times Mv_{corr}} 60 \times 10^6, \quad (\text{~~1~~}3)$$

Where F_{GHG} = soil GHG flux (CO_2, N_2O, or CH_4), $\partial c/\partial t$ = change in chamber headspace gas concentration over time (i.e. slope of the linear regression), V_{ch} = volume of the chamber headspace (m^3), M_w = molar weight (g mol^{-1}) of C for CO_2 and CH_4 (12) or N for N_2O (2x N = 28), A_{ch} = area

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covered by the chamber (m^2) and Mv_{corr} = pressure- and temperature-corrected molar volume (Brümmer et al., 2008) using Eq. (2). With 60 and 10^6 being, constants used to convert minutes into hours and micrograms respectively. Temperature in Eq. (2) ~~represent~~is the air temperature in the chamber headspace measured during each sampling~~.~~ and 0.02241 the molar volume of a gas at standard temperature and pressure (m^3 mol^{-1}).

$$Mv_{corr} = 0.02241 \frac{273.15+Temp(℃)}{273.15} \times \frac{Atmospheric\ pressure\ at\ measurement\ (Pa)}{Atmospheric\ pressure\ at\ sea\ level\ (Pa)} \quad (2)$$

(4)

The minimum limit of detection (LOD) for each gas was calculated following Parkin et al. (2012) and levels were ± 4.9 mg CO_2-C m^{-2} h^{-1} for CO_2, ± 0.04 µg N_2O-N m^{-2} h^{-1} for N_2O, and ± 0.03 mg CH_4-C m^{-2} h^{-1} for CH_4. However, we included all data in the analysis, including those below LOD in line with Croghan and Egeghy (2003), who noted that including such data provides an insight on the distinct measurements, thus giving clarifying the set of environmental observations.

2.5. Auxiliary measurements

During each gas-sampling day, we measured soil water content (WC) and soil temperature (T) (at a depth of 0–5 cm) adjacent to the collar using a handheld data logger with a GS3 sensor (ProCheck METER Group, Inc. USA). Daily air temperature and precipitation data from November 2017 to November 2018 were obtained from a weather station in Maktau located within the cropland site (Tuure et al., 2019). A soil auger was used to collect soil samples (at a depth of 0–20 cm) during the wet season (22 May 2018) in each site for soil chemical and physical property analysis. For bulk density, we collected a combination of three samples from each cluster close to each chamber collar at depths of 0–10 cm and 10–20 cm using a soil bulk density ring (Eijkelkamp Agrisearch Equipment, Giesbeek, The Netherlands). Samples were stored in airtight polyethylene bags and kept in a cooler box with ice packs before transportation to the laboratory for further analysis. In the laboratory, samples were stored in a refrigerator (4 °C) and analysed within 10 days.

The samples were sieved at < 2 mm before analysis. Soil water content was measured by drying soil at 105 °C for 48 h. Soil pH was determined in a 1:2.5 (soil : distilled water) suspension using an electrode pH meter (3540 pH and conductivity Meter, Bibby Scientific Ltd, UK) and soil texture using the hydrometer technique (Scrimgeour, 2008; Reeuwijk, 2002). Total soil C and N content, a duplicate of 20 g of fresh sample was oven-dried at 40 °C for 48 hours and ground into a fine powder

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using a ball mill (Retsch MM400). Approximately 200 mg of the dry sample were measured by elemental analysis (Vario MAX Cube Analyzer Version 05.03.2013).

2.6. Statistical Analysis

Statistical analyses were carried out using R Statistical Software (R 3.5.2, R Core Team). Spearman correlation coefficients were performed among the variables followed by the Kruskal Wallis test to assess significant differences of GHG emissions between the LUTs and across seasons. A post-hoc analysis involving pairwise comparisons using the Nemenyi test was performed where significant differences existed. Significance level was set at $p < 0.05$. We assessed the correlation between soil GHG emissions with T and WC using several functions based on the coefficient of determination (R^2), root-mean-square error (RMSE) and Akaike's information criterion (AIC). There being no difference in the outputs, we present results from the Gaussian function (O'Connell, 1990) for the correlation between soil emissions and T using Eq. (3), and a quadratic function for correlation with WC using Eq. (4). We also evaluated the combined effect of T and WC on soil GHG emissions by combining Eq. (3) and Eq. (4) into Eq. (5) to assess the effect of these two variables on the emissions.

$$Rs = ae^{(bT+cT^2)} \quad (3\text{~~5~~}5)$$

$$Rs = a + bWC + cWC^2 \quad (4\text{~~6~~}6)$$

$$Rs = e^{(aT+bT^2)} \times (cWC + dWC^2) \quad (5\text{~~7~~}7)$$

Where *Rs* is soil GHG (CO_2 and N_2O) emissions, *T* is soil temperature (°C), and *WC* is soil water content ($m^3\ m^{-3}$), while *a*, *b*, *c*, and *d* ~~represent~~*are* the model coefficients.

After no correlation with T and a weak correlation with WC were observed, we included vegetation cover. For this, we used Normalized Difference Vegetation Index (NDVI) from Moderate Resolution Imaging Spectroradiometer (MODIS) from https://ladsweb.modaps.eosdis.nasa.gov. NDVI quantifies vegetation vigour by measuring the difference between reflectance in near infrared (which green chlorophyll-rich vegetation strongly reflects) and red wavelength areas (which vegetation absorbs) computed using Eq. (6). MOD13Q1 products from MODIS are NDVI data generated from a 16-day interval at a 250 m spatial resolution as a Level 3 product (Didan, K, 2015).

$$NDVI = \frac{NIR+Red}{NIR+Red} \quad (6\text{~~8~~}8)$$

To cover our study period, we selected NDVI data within the campaign dates. If no data fitted within our dates, we used data that were less than five days before or after the campaign dates, assuming that no significant increase or decrease would occur in the vegetation. The pixels containing the study sites were extracted based on the latitude and longitude of each site. Linear functions were applied to the seasonal datasets of Rs with NDVI to assess the contribution of vegetation on soil emissions using Eq. (7) and a combined effect of WC and NDVI on soil CO_2 emissions (Rs) using Eq. (8).

$$Rs = a + bNDVI \quad (7\underline{9})$$

$$Rs = a + bNDVI + (cWC + dWC^2) \quad (8\underline{10})$$

3. Results

3.2. Meteorological data

During the 12-month study period, the long rains lasted from early March to the end of May, while short rains occurred between early September and December (Fig. 3). The total annual rainfall was 550 mm, which is within the average rainfall expected in the area (CIDP, 2014). The mean annual air temperature was 22.7 °C (min=16.7 °C, max=30.5 °C). January was the hottest month (min=17.4 °C, max=31.9 °C), while June and July (min=14.5± 0.2 °C, max=27± 0.1 °C) were the coolest.

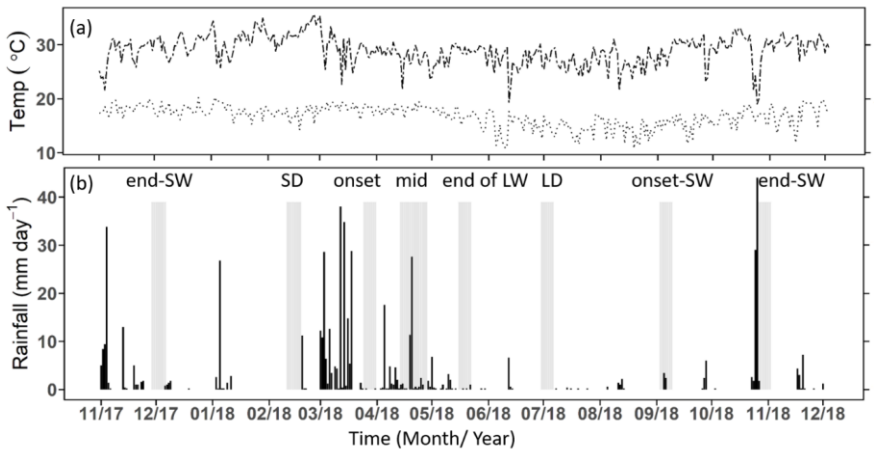


Figure 3: *(a) Daily maximum and minimum air temperature and (b) daily rainfall from lowland in southern Kenya between November 2017 to October 2018 recorded at Maktau weather station. Total annual recorded rainfall was 550 mm. Highlighted grey bars show the days of the sampling*

campaigns (the season above the grey bars denote SW and LW for the short and long wet season with corresponding onset, mid and end of the wet season, and SD for the short dry season and LD for the long dry season).

3.3. Soil characteristics

Sand content was highest in cropland (77±8 %) compared to the conservation land and bushland (ca. 72±1 %) and lowest in grazing land (64.3 ±0.4 %) (See Table 2). Grazing land had the highest clay content (31.7±0.5 %) while cropland (19±2 %) had the lowest. Soil pH ranged between slightly acidic in the grazing land (6.3±0.3), neutral in the bushland (7.2±0.4), and slightly alkaline in the conservation land and cropland (7.5±0.1 and 7.9±0.2 respectively). Carbon content ranged from 0.93 % in the conservation land to 0.60 % in the cropland. Nitrogen content did not vary significantly between sites (mean=0.08±0.01 %).

Table 1: Soil characteristics of the topsoil (a depth of 0–20 cm) from the four land-use types investigated in this study. Values are given as mean ± SE.

					Soil Texture		
Land Use	**% N**	**% C**	**Bulk Density (g cm^{-3})**	**pH**	**% Clay**	**% Sand**	**% Silt**
Bushland	0.08 (0.03)	0.77 (0.5)	1.31 (0.2)	7.2 (0.4)	23.7 (0.7)	71.6 (2.2)	4.7 (2.3)
Conservation land	0.09 (0.02)	0.93 (0.7)	1.27 (0.4)	7.5 (0.1)	26.4 (2.2)	71.6 (0.5)	2.0 (0.0)
Cropland	0.07 (0.04)	0.60 (0.2)	1.26 (0.3)	7.9 (0.2)	19.1 (2.4)	76.9 (8.1)	4.0 (5.1)
Grazing land	0.08 (0.02)	0.83 (0.4)	1.23 (0.5)	6.3 (0.3)	31.7 (0.5)	64.3 (0.4)	4.4 (0.4)

3.4. Soil greenhouse gas emissions

3.4.1. Soil carbon dioxide (CO_2) emissions

Mean annual soil CO_2 emissions were highest in the conservation land (75±6 mg CO_2-C m^{-2} h^{-1}). Concurrently, no significant differences occurred between grazing land (50±5 mg CO_2-C m^{-2} h^{-1}), cropland (47±3 mg CO_2-C m^{-2} h^{-1}), and bushland (45±4 mg CO_2-C m^{-2} h^{-1}). We observed no significant difference in CO_2 emissions between the first three seasons, namely SD, onset-LW and mid-LW. However, towards the end of the wet season (end-LW) in May, CO_2 emissions in the conservation land and grazing land were significantly higher than cropland and bushland ($p<0.05$). Through LD, onset-SW, and mid-SW, CO_2 emissions in the conservation land remained significantly higher, while those from grazing land dropped during LD and were not different from bushland or cropland emissions thereafter.

Generally, CO_2 emissions were higher in the wet seasons than in the dry seasons at all sites (Fig. 4c). At the onset of the rainy season in early March, CO_2 emissions increased at all sites by over 200% from SD to LW and dropped during LD by approximately 70% in grazing land, bushland, and cropland. In the conservation land, the drop from LW to LD was about 20%. In the bushland, the highest seasonal mean fluxes were reached in mid-LW (98±6 mg CO_2-C m^{-2} h^{-1}) while in the conservation land (239±11 mg CO_2-C m^{-2} h^{-1}), grazing land (160±16 mg CO_2-C m^{-2} h^{-1}), and cropland (84±12 mg CO_2-C m^{-2} h^{-1}), the highest were observed during end-LW towards the end May. The lowest seasonal mean CO_2 emissions at all sites were observed during the SD campaign (below 20 mg CO_2-C m^{-2} h^{-1}, Fig 4).

When comparing the two wet seasons (LW and SW), CO_2 emissions were 45 % (bushland), 55 % (conservation land), 56 % (cropland), and 57 % (grazing land) higher in LW than SW (Fig. 5a). For the two dry seasons, CO_2 emissions were significantly higher in LD than SD across all the sites (in SD all sites recorded emission below 20 mg CO_2-C m^{-2} h^{-1}). During the LD, CO_2 emissions were 29 % (bushland), 38 % (cropland), 40 % (grazing land), and 77 % (conservation) higher than during SD. As much as CO_2 emissions in cropland, bushland, and grazing land dropped to less than 30 mg CO_2-C m^{-2} h^{-1} during LD, in the conservation land (118±6 mg CO_2-C m^{-2} h^{-1}) the emissions remained high (Fig 4c).

3.4.2. Soil nitrous oxide (N_2O) emissions

Mean annual N_2O emissions were very low (< 5 µg N_2O-N m^{-2} h^{-1}) at all four sites (Fig. 4d). Cropland (2.7±0.6 µg N_2O-N m^{-2} h^{-1}) recorded the highest mean N_2O emissions than conservation land (1.6±0.4 µg N_2O-N m^{-2} h^{-1}), grazing land (1.5±0.4 µg N_2O-N m^{-2} h^{-1}), and bushland (1.2±0.4 µg N_2O-N m^{-2} h^{-1}). N_2O emissions did not show a clear temporal pattern as observed for CO_2 emissions. Within each season, no significant differences in N_2O emissions were observed among the sites. However, at the onset of the rainy season (onset-LW), there were observable increases in N_2O emissions from all the sites. During this period, mean N_2O emissions at all the sites were ca. 2.6±0.4 µg N_2O-N m^{-2} h^{-1}). By mid-LW and end-LW periods, N_2O emissions had dropped (<1 µg N_2O-N m^{-2} h^{-1}) at all sites. In June during LD, N_2O emissions from the cropland were significantly higher than at the other three sites (2.35±0.03 µg N_2O-N m^{-2} h^{-1}, $p < 0.05$). During this period, the farmer had just harvested his crops.

When comparing the two wet seasons, N_2O emissions did not differ between LW and SW at all sites (Fig. 5b). However, short N_2O emission pulses were observed during both seasons. A notable peak

of about 70 μg N_2O-N m^{-2} h^{-1} was observed in the cropland on 7 April 2018, a week after livestock manure application. It had also rained the night before the sampling day. At the same site, we also recorded a peak of 55.2 μg N_2O-N m^{-2} h^{-1} on 30 September 2018, likely also due to manure application (personal communication from the farmer Mwadime Mjomba). Other notable peaks were 29.9 μg N_2O-N m^{-2} h^{-1} (in the bushland on 3 September 2018) and 26.6 μg N_2O-N m^{-2} h^{-1} (in grazing land on 4 September 2018). These were observed during the SW from chambers with animal droppings. For the dry seasons, N_2O emissions did not differ between SD and LD in the bushland, conservation land, and grazing land, while N_2O emissions in the cropland were significantly higher during LD than SD (Fig. 5b).

3.4.3. Soil methane (CH_4) emissions

Throughout the study period, CH_4 emissions did not vary significantly among sites and seasons (Fig. 4e and Fig. 5c). The studied sites were mostly CH_4 sinks rather than sources, and CH_4 fluxes were very low, ranging from -0.03 to 0.9 mg CH_4-C m^{-2} h^{-1} (Fig. 4e), often below the limit of detection (0.03 mg CH_4-C m^{-2} h^{-1}).

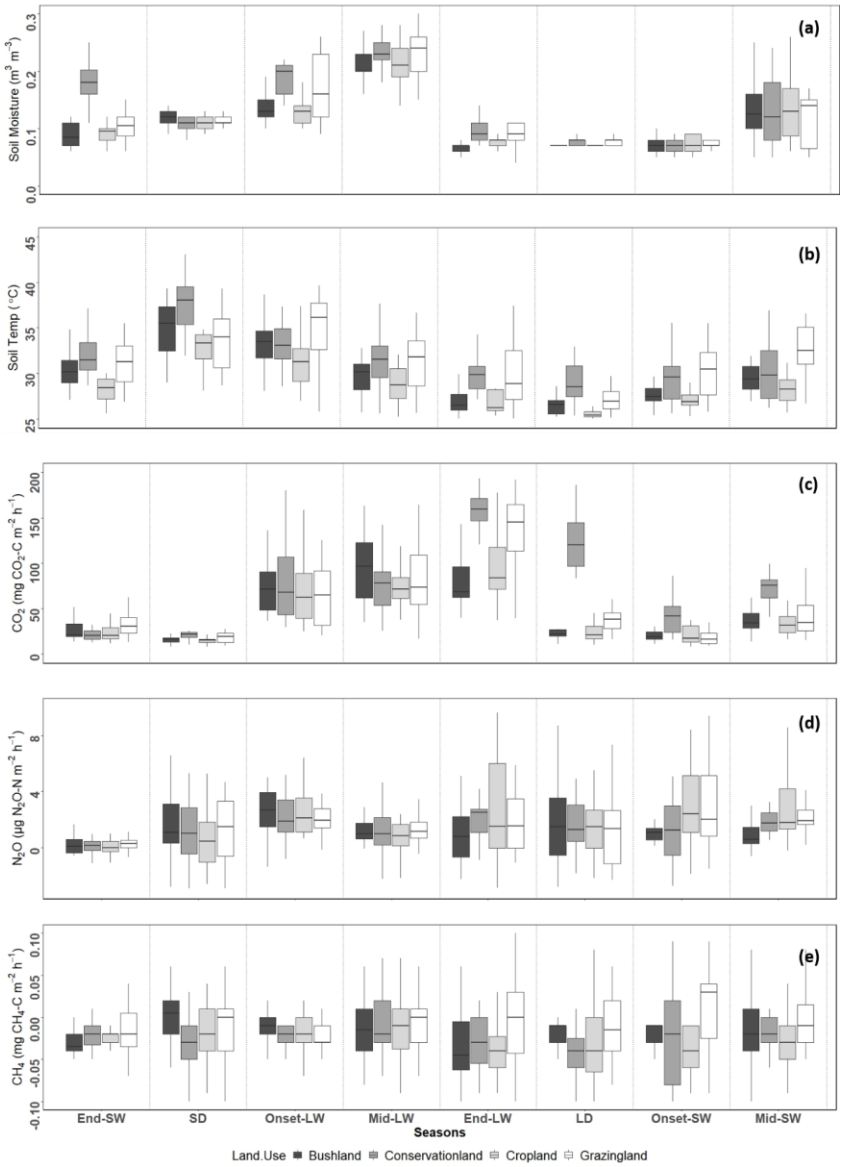


Figure 4: *Box plots showing differences in seasonal means for (a) soil moisture, (b) soil temperature, and soil emissions of (c) CO_2, (d) N_2O, and (e) CH_4 for each site from November 2017 to October 2018. Season abbreviations on the x-axis denote SW for the short wet season and LW for the long wet season with corresponding onset, mid and end of the wet season, along with SD for the short dry season and LD for the long dry season.*

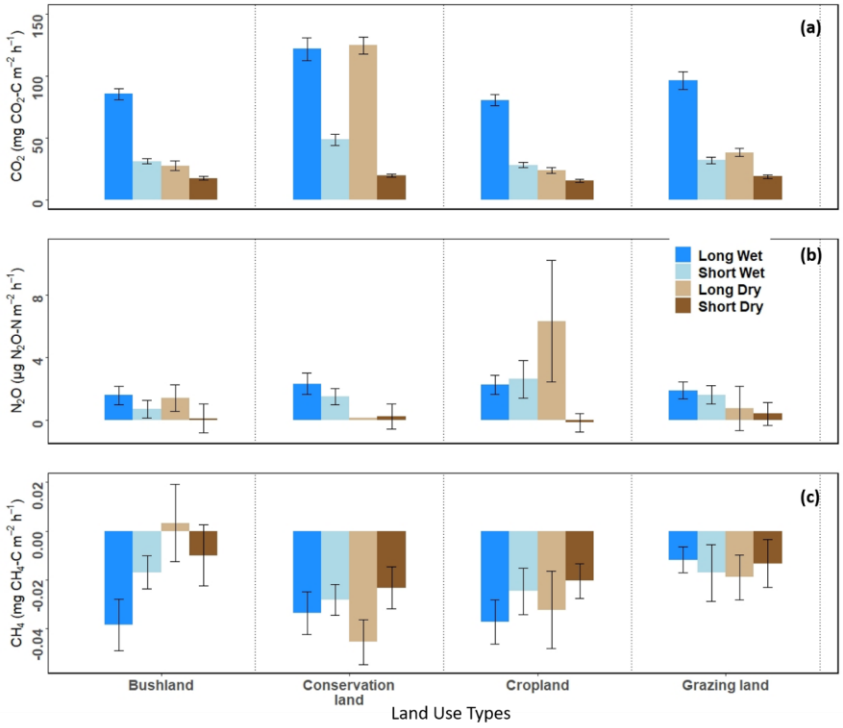


Figure 5: *Seasonal differences in mean (a) CO_2, (b) N_2O, and (c) CH_4 emissions between the long and short wet seasons and the long and short dry season for the four land-use types.*

3.5. Effects of soil temperature, soil water content, and vegetation indices on soil GHG emissions

Soil water content (WC) was highest during the wet season (mean 0.19±0.06 m^3 m^{-3}) and lowest during the dry season (mean. 0.07±0.02 m^3 m^{-3}) at all sites (see Fig. 4a). Soil temperature (T), were highest during the SD (36.7±2.1 ^oC) and lowest (24.5±0.6 ^oC) in LD (Fig.4b). Throughout all the campaigns, mean WC and mean T were highest in the conservation land, followed by grazing land, bushland, and lowest in the cropland. Regression results on soil CO_2 and soil N_2O emissions against T and WC are shown in Table 2. The results showed positive correlations between soil CO_2 emissions and WC ($p < 0.05$). However, the R^2 was very weak at all sites. Conversely, CO_2 emissions showed no correlation with T ($P < 0.05$). We observed no correlation between N_2O and CH_4 emissions with either WC or T ($p < 0.05$). Separating data into the wet and dry season did not improve the correlations.

Table 2: Soil water content (WC) and soil temperature (T) control on carbon dioxide (CO_2) and nitrous oxide (N_2O) denoted by *Rs*, while *a, b,* and *c* ~~represent~~denote the model coefficient

Predictors	Land Use	CO_2-C mg m^{-2} h^{-1}		N_2O-N ug m^{-2} h^{-1}	
Soil water content (WC)		$Rs = a + bWC + cWC^2$			
	Bushland	$6.12WC + 0.92WC^2$	R^2= 0.26***	$19.02WC - 64.11WC^2$	R^2= 0.008
	Conservation land	$135.27WC - 0.57WC^2$	R^2= 0.07**	$11.63WC - 7.736WC^2$	R^2= 0.009
	Cropland	$17.83WC + 0.67WC^2$	R^2= 0.04***	$28.48WC - 66.63WC^2$	R^2= 0.005
	Grazing land	$15.03WC + 0.79WC^2$	R^2= 0.11***	$19.81WC - 53.56WC^2$	R^2= 0.002
Soil Temperature (T)		$R = ae^{(bT+cT^2)}$			
	Bushland	$1.078e^{0.26T-0.004T^2}$	R^2= 0.008	$360.25e^{-0.29T-0.004T^2}$	R^2= 0.008
	Conservation land	$0.001e^{0.81T-0.014T^2}$	R^2= 0.015**	$0.007e^{0.45T-0.008T^2}$	R^2= 0.015
	Cropland	$4.568e^{-0.13T+0.002T^2}$	R^2= 0.008*	$0.007e^{-0.05T+2.42T^2}$	R^2= 0.008*
	Grazing land	$4.136e^{0.18T-0.003T^2}$	R^2= 0.015	$2.366e^{0.05T-0.001T^2}$	R^2= 0.015

***: p<0.0001, **: p<0.001, *: p<0.05

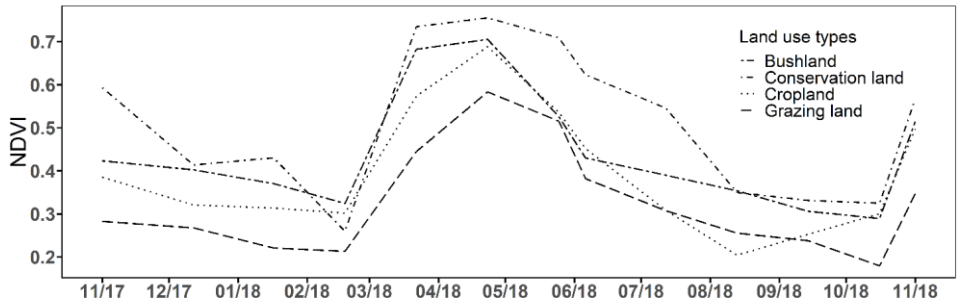
Results from combined WC and T on soil CO_2 and N_2O emissions did not improve the correlation, as shown in Table 3. Thus, we included vegetation indices in our model.

Table 3: Combined effects of soil water content (WC) and soil temperature (T) control on soil CO_2 and N_2O emissions. Soil CO_2 and N_2O emissions denoted by *Rs*, while *a, b, d,* and *e* ~~represent~~signify the model coefficient, (R^2) the coefficient of determination, and AIC the Akaike's information criterion

Functions	Land use	a	b	d	e	R^2	AIC
CO_2-C mg m^{-2} h^{-1}							
$Rs = e^{(aT+bT^2)} \times (dWC + eWC^2)$	Bushland	-0.12	0.001	52.774	-0.527	0.31***	1888
	Conservation land	0.90	-0.016	0.0001	0.000	0.10***	2156
	Cropland	-0.39	0.006	3701.901	-84.001	0.08**	1886
	Grazing land	0.14	-0.003	0.842	-0.008	0.12***	2024
N_2O-N ug m^{-2} h^{-1}	Bushland	-0.50	0.007	2008.345	-58.559	0.009	785
$Rs = e^{(aT+bT^2)} \times (dWC + eWC^2)$	Conservation land	0.56	-0.010	0.000	0.000	0.003	811
	Cropland	0.67	-0.017	0.003	-0.0001	0.089	911
	Grazing land	0.11	-0.003	0.187	-0.005	0.003	770

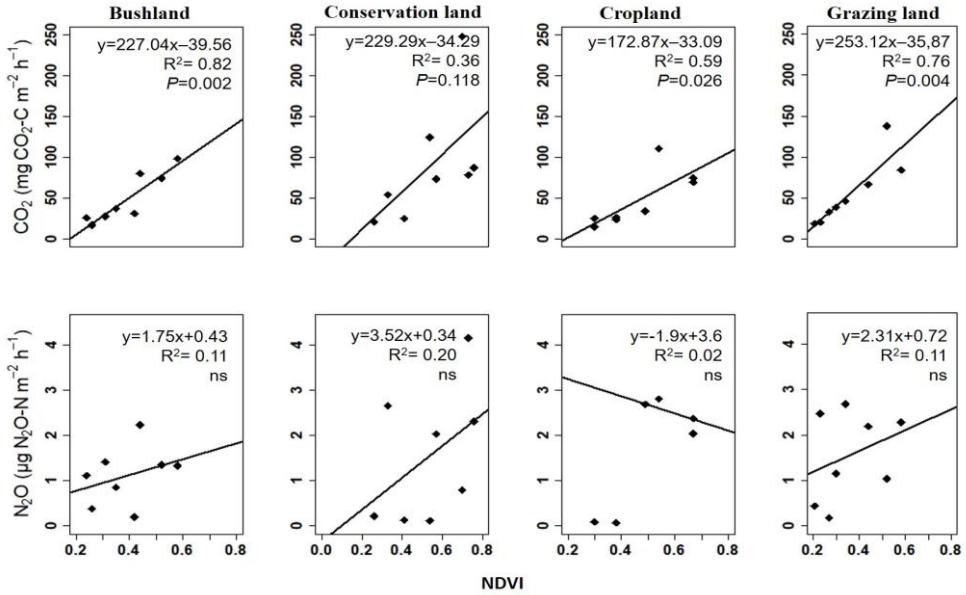
***: p<0.0001, **: p<0.001, *: p<0.05

The annual change in vegetation cover at each site are shown in Fig. (6). The highest NDVI values were observed during the LW in April (ranging from 0.58 to 0.76) and the lowest during the SD (< 0.26). Vegetation greenness increased rapidly from mid-March at all sites coinciding with the onset of the rainy season and remained high (Fig.6). At end of the rainy season, NDVI gradually dropped. Highest NDVI values occurred in the conservation land (0.51±0.05); followed by bushland (0.44±0.05), cropland (0.41±0.05), and the lowest were recorded in the grazing land (0.33±0.05).



475 ***Figure 6:*** *Monthly NDVI time series showing the annual trend in vegetation cover from November 2017 to November 2018 for the four land-use types.*

Regression analysis results shows a positive correlation between NDVI and seasonal CO_2 emissions at all the sites (see Fig. 7). Combined WC and NDVI improved the correlation even further as shown in Table 4. No significant correlation was observed between N_2O emissions and NDVI (Fig. 7).



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ns= not significant

Figure 7: *Linear regression analyses of the measured seasonal means for soil CO_2 and soil N_2O emissions during the campaign from November 2017 to November 2018 plotted against NDVI data acquired during each campaign.*

Table 4: Combined effects of soil water content (WC) and NDVI on soil CO_2 emissions. Soil CO_2 emissions denoted by *Rs*, while *a*, *b*, *c*, and *d* ~~represent~~are the model coefficient and (R^2) the coefficient of determination.

	Land use	a	b	c	d	R^2
CO_2-C mg m^{-2} h^{-1}						
$Rs = a + bNDVI + (cWC + dWC^2)$	Bushland	-29.69	196.47	-83.74	781.29	0.86***
	Conservation land	6.48	382.96	-861.98	-256.66	0.82***
	Cropland	26.94	244.54	-1250.22	3269.46	0.79***
	Grazing land	-97.19	396.41	575.60	-3440.80	0.96***

***: p<0.0001, **: p<0.001, *: p<0.05

4. Discussion

4.1. Soil CO_2 emissions

Soil CO_2 emissions differed significantly between the four LUTs. The highest mean CO_2 emissions were observed in the conservation land followed by grazing land and bushland, and the lowest from cropland. Soil C content, which is the primary source of energy for soil microorganisms that contribute to soil CO_2 emissions (Lal, 2009) also showed the same trend (conservation land > grazing land > bushland > cropland). Therefore, the difference in land use and land-use management activities between our sites played a vital role in modifying both biotic and abiotic factors that drive both soil C content and soil CO_2 emissions (Pinto et al., 2002).

Due to the difference in land use and management, vegetation type and cover differed between our sites. The dense grass network in the conservation land formed an almost closed ground cover, especially in the wet seasons (further confirmed by NDVI values). Being a private sanctuary, only wild mammals (no livestock allowed) grazed and browsed there, and thus we observed less damage on the grass cover throughout all the campaigns as compared to the grazing land (which had large patches of bare soil due to overgrazing) and bushland. This provides a good explanation for the difference in mean CO_2 emissions between these three LUTs, as vegetation is known to affect soil C concentration and root and microbial respiration that directly contribute to soil CO_2 emissions (Fanin et al., 2011; Rey et al., 2011).

With the lowest CO_2 emissions being measured in the cropland, we attribute this observation to the continued tillage and removal of crops and crop residues during land preparation, weeding and harvesting, which affects both root respiration and soil C content (Raich et al., 2000; Nandwa, 2001). In East Africa and especially in smallholder farming systems, most of the crop residues are used as livestock feed and fuel. In addition, manure inputs in cropland are very low (about 20 kg per month

on a 1.5 ha farm) and thus no measurable difference in CO_2 emissions was detected before and after manure input, and with the other LUTs. Several other studies observed the same scenario from low manure input in maize and sorghum plots (Rosenstock et al., 2016; Mapanda et al., 2011, and Pelster et al., 2017).

On average, CO_2 emissions were higher during the wet season than during the dry season. At the start of both rainy seasons (SW, LW), CO_2 emissions increased significantly in all LUTs. Emissions from the conservation land and grazing land are comparable to those in Brümmer et al. (2008), who observed CO_2 emissions ranging between 100 and 250 mg CO_2-C m^{-2} h^{-1} in a natural savanna in Burkina Faso. Several other studies from similar ecosystem have also documented comparable changes in CO_2 emissions with the onset of the rainy seasons (Castaldi et al., 2006; Livesley et al., 2011; Pinto et al., 2002). In the cropland, results in the wet season are similar to those measured by Rosenstock et al. (2016), ranging from 50 to > 200 mg m^{-2} h^{-1}. We attributed the increase in CO_2 emissions in the wet season to the response of soil microbes and vegetation to soil moisture (Livesley et al., 2011; Otieno et al., 2010). Soil moisture connects microorganisms with soluble substrates (Moyano et al., 2013) and increases microbial activity (Davidson et al., 2006; 2009; Grover et al., 2012) and thereby soil CO_2 emissions.

Furthermore, an increase in soil CO_2 emissions during the wet season can also be a result of increased root respiration due to more active plant and root growth (Macdonald et al., 2006). Grasses sprout more rapidly than trees and shrubs with the first rains (Merbold et al., 2009). This provides a possible explanation for the higher CO_2 emissions in the grassy conservation land, grazing land, and bushland compared to cropland during the rainy season. However, grazing land recorded higher CO_2 emissions than bushland (only the farmer's livestock grazed here). The main difference between these two sites – apart from grazing intensity – was that bushland had more trees (*Acacia spp.*) and shrubs (C*ommiphora spp.*) and less herbaceous undergrowth than the grazing land, thus providing shade that might have interfered with growth and regrowth of plants below the canopy. Therefore, grass root production in the open conservation land and grazing land was likely higher than in the bushland (Janssens et al., 2001), although we cannot confirm this because root biomass was not determined in this study. In cropland, all grasses and weeds were cleared during regular weeding and therefore did not play a role in root respiration.

To our surprise, the highest mean seasonal CO_2 emissions in conservation land, grazing land, and cropland were observed at the end rather than at the peak of the wet season. During this time, both soil moisture and soil temperature had dropped in all LUTs. However, our data was only recorded up

to a depth of 5 cm, but roots of perennial grasses, shrubs and trees can tap moisture from greater soil depths (Carbone et al., 2011). According to Carbone et al. (2011), while microbial activity is highest and most variable in the upper soil layers, which are first to wet-up and dry-down, roots can access water reserves in deeper soil layers that take longer to be exhausted, and therefore remain active at the end of the wet and into the dry season.

4.2. Soil N_2O emissions

Our results showed very low N_2O emissions from all LUTs, which we attributed to low soil N content observed in all the sites (see Table 1). Savanna ecosystems are characterized by very tight N cycling, which transcends to low N availability (Pinto et al., 2002 and Grover et al., 2012), and most of this N is rapidly taken up by vegetation, leaving very little for denitrification (Castaldi et al., 2006; Mapanda et al., 2011). The N_2O flux results observed from conservation land, grazing land and bushland are consistent with those observed in a Brazilian savanna by Wilcke et al. (2005), and other studies from similar ecosystems reported comparable N_2O flux magnitudes (Scholes et al., 1997; Castaldi et al., 2016; Mapanda et al., 2010). The higher N_2O emissions observed in June and July from our cropland site after the maize and bean harvests likely occurred due to the disturbance and following absence of live plants, which led to higher soil N availability because of less N uptake by plants and increased root decomposition.

In contrast to the patterns observed for CO_2 emissions, we did not detect any seasonal variations in N_2O emissions. The only exception to the otherwise very low N_2O emissions was after the onset of the rainy season, when N_2O emissions slightly increased at all sites. Such patterns have previously been shown by several similar studies (Scholes et al., 1997; Pinto et al., 2002; Castaldi et al., 2006; Livesley et al., 2011). The increase in N_2O flux at the onset of the rains has been attributed to an increase in microbial activity and therefore faster decomposition of litter and plant residue facilitated by an increase in soil moisture, thus increasing N availability (Rees et al., 2006). Furthermore, according to Davidson et al., (2000) and Butterbach-Bahl et al., (2013), soil moisture affects soil gas diffusion, oxygen (O_2) availability, and the movement of substrate necessary for microbial growth and metabolism.

Negative N_2O emissions were detected during the dry season. Such observations could result from the low N contents observed at all sites coupled with low soil moisture in the dry season, which facilitates diffusion of atmospheric N_2O into the soil. Soil denitrifiers may, therefore, use N_2O as an N substrate in the absence of NO_2^- and NO_3^- (Rosenkranz et al., 2006). Negative N_2O emissions have

575 also been reported in other tropical savanna soils under similarly dry conditions (Castaldi et al., 2006; Livesley et al., 2011).

Manure application in the cropland was very low (< 12 kg of N in 1.5ha for the crop-growing season), and thus N_2O emissions from cropland were low and not different from the other LUTs, which was in contrast to what we had hypothesized. Due to low soil N levels in the cropland, the low amount of
580 manure added was not sufficient to stimulate N_2O emissions, likely because soil N availability was still limiting for plant and microbial growth (Castaldi et al., 2006). Traditional farming systems in smallholder farms in Africa involve repeated cropping with no or very low N inputs that leads to soil N mining over time (Chianu et al., 2012). In line with this, in our cropland site maize and beans are grown during every wet season with no fallow in between years. In addition, the farmer did not use
585 any chemical fertilizer to increase soil N, and the N input from biological N fixation into the soil was likely small because beans were harvested for consumption and bean plant residues were used as livestock feed and not incorporated into the soil. Therefore, the small quantities of manure applied and legume N fixation may have likely been insufficient to compensate for N loss through leaching and crop harvests. According to the Taita Development plan, this is a common scenario in the county,
590 which translates to very low crop yields in this region (CIDP, 2014). Another possible explanation for not detecting the influence of manure on N_2O emissions could be the fact that we did not manage to sample immediately after manure application and therefore might have missed the instant impact of manure application on N_2O emissions. However, similar studies by Pelster et al. (2017) and Rosenstock et al. (2016) also did not see any influence of manure application on soil N_2O emissions
595 and reported N_2O emission values that were generally < 10 μg N_2O-N m^{-2} h^{-1}). Equally, the deposition of dung and urine by animals in the grazing land and bushland did not have any measurable influence on soil N_2O emissions.

4.3.Soil CH_4 emissions

Methane emissions did not vary between the land-use types or with seasons. Most values were below
600 the LOD at all the sites. Soil water content in our study is clearly the limiting factor for methanogenesis, which needs anoxic conditions for a certain period until methanogenic archaea are established (Serrano-silva et al., 2014). Furthermore, soil compaction by animal trampling may have limited CH_4 diffusion into the soil thus limiting CH_4 consumption by oxidation (Ball et al., 1997). In cropland, continuous tillage interferes with soil structure thus affecting the microenvironment that
605 favours methanotrophic (Jacinthe et al., 2014). Additionally, low soil C as observed in all the sites generally leads to low abundance of soil microorganisms and consequently also methane oxidisers

(Serrano-silva et al., 2014). Nevertheless, soils around lakes, waterholes and rivers can be CH_4 sources in semi-arid savanna ecosystems, but those were not investigated during this study.

4.4.Effects of soil moisture, soil temperature, and vegetation indices on GHG emissions

As is common for sub-tropical regions, seasonal variation in soil temperature was small in the study region and therefore soil temperature did not play a big role in modifying soil GHG emissions. Instead, changes in soil moisture were considered to be the main driver of CO_2 emissions in our study, as has previously been highlighted also by other studies (Grover et al. (2012), Brümmer et al. (2009) and Livesley et al. (2011)). However, we did not observe any significant relationship between N_2O emissions with either soil moisture or temperature apart from in the cropland, where we found a positive correlation between N_2O and soil temperature ($p < 0.05$). As much as previous results have sometimes shown a positive relationship between temperature and N_2O emissions (Castaldi et al., 2010), our results are in line with others (Scholes et al., (1997), Brümmer et al. (2008) who were also unable to link soil N_2O emissions to variations in soil temperature. In fact, N_2O emissions were very low during both the wet and dry seasons, which is similar to the findings of Castaldi et al., (2004). The most likely explanation for the lack of seasonality effects on N_2O emissions would be the low soil N levels observed at all the sites, which was probably the most limiting factor for N_2O emissions and thus overruled all other potential controlling factors (Grover et al., 2012).

The vegetation cover as depicted by NDVI ~~represents~~shows the status of the vegetation (~~value~~NDVI values range from ~~-~~+1.0 to -1~~).~~.0). High NDVI values correspond to high vegetation cover, while low NDVI correspond to little or no vegetation (Gamon et al., 1995; Butt et al., 2011). Therefore, the increase in NDVI that we observed at the onset of the rainy season indicates sprouting and regrowth of vegetation at that time, while the drop in NDVI values at the end of the rainy season indicates reduction in vegetation cover due to plant senescence and grazing. In the cropland area, low NDVI coincided with the harvesting of beans and the drying of the maize plants in June and July. Highest mean NDVI values were observed in the conservation land, mainly due to the dense grassy vegetation, while the lowest NDVI values were found in the grazing land, which we had expected because this area has large spots without vegetation due to overgrazing. Results from linear regression analysis showed a strong positive correlation of soil CO_2 emissions with NDVI ($p < 0.05$), explaining between 35 % and 82 % of the variation in soil CO_2 emissions at the four sites. This means that CO_2 emissions were highest when NDVI (i.e. vegetation cover) was high. Thus, the inclusion of both NDVI and soil moisture measurements is essential for reliably predicting soil CO_2 emission from savanna soils, which is consistent with other studies (Reichstein et al., 2003; Anderson et al., 2008; Lees et al.,

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2018). Concurrently, the same relationship between NDVI and N_2O emissions could not be proven in our study.

5. Conclusion

The magnitude and temporal and spatial variability of soil GHG emissions in most developing countries have large uncertainties due to a lack of data, especially in dry areas and ecosystems facing land-use change. In our study, we quantified soil GHG emissions from four dominant LUTs in the dry lowlands of southern Kenya, namely bushland, conservation land, cropland, and grazing land. Our results showed significant variation between seasons and the respective LUTs. CO_2 emissions, in particular, were higher during the wet season, when soil moisture was high, compared to the dry season. Most of the variation in CO_2 emissions could be explained by soil moisture and NDVI, highlighting the importance of including proxies for vegetation cover in soil GHG emissions studies in savannas. N_2O emissions and CH_4 emissions were of minor importance at all sites. However, we acknowledge that we might have missed some episodes of elevated soil N_2O emissions, as these are often episodic and of short duration, for examples after fertilization or precipitation events. Following these results, there is still need for more continuous studies to cover spatial and temporal variation in soil GHG emissions, as well as the inclusion of other LUTs than the ones examined in this study (e.g. wetlands). Nevertheless, we believe that our results are important to reduce uncertainties in GHG emission baselines and to identify reliable and meaningful climate change mitigation interventions by informing the relevant policies.

6. Data availability

The data associated with the manuscript can be obtained from https://figshare.com/articles/Final_data_for_Soil_Greenhouse_Gas_Emissions_under_Different_Land-Use_Types_in_Savanna_Ecosystems_of_Kenya_/11673579

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