AGU PUBLICATIONS

JGR

Journal of Geophysical Research: Biogeosciences

RESEARCH ARTICLE

10.1002/2016JG003341

# Greenhouse gas fluxes from agricultural soils of Kenya and Tanzania

**Todd S. Rosenstock**[1], **Mathew Mpanda**[2], **David E. Pelster**[3], **Klaus Butterbach-Bahl**[3,4], **Mariana C. Rufino**[5], **Margaret Thiong'o**[1], **Paul Mutuo**[1], **Sheila Abwanda**[6,7], **Janie Rioux**[8], **Anthony A. Kimaro**[2], and **Henry Neufeldt**[1]

[1]World Agroforestry Centre (ICRAF), Nairobi, Kenya, [2]ARI-Mikocheni Campus, World Agroforestry Centre (ICRAF), Dar es Salaam, Tanzania, [3]International Livestock Research Institute, Nairobi, Kenya, [4]Karlsruhe Institute of Technology—Institute of Meteorology and Climate Research, Atmospheric Environmental Research (KIT/IMK-IFU), Garmisch-Partenkirchen, Germany, [5]Centre for International Forestry Research, Nairobi, Kenya, [6]World Agroforestry Centre (ICRAF), Kisumu, Kenya, [7]Department of Applied Statistics, Maseno University, Maseno, Kenya, [8]Food and Agriculture Organization of the United Nations, Rome, Italy

**Key Points:**

- Emissions vary among sites due to fine-scale differences in production conditions
- Heterogeneity and seasonality of fluxes highlight gaps in knowledge of an important source of GHGs
- A targeted effort is needed to understand fluxes in African soils to support sustainable development

**Correspondence to:**
T. S. Rosenstock,
t.rosenstock@cgiar.org

**Citation:**
Rosenstock, T. S., et al. (2016), Greenhouse gas fluxes from agricultural soils of Kenya and Tanzania, *J. Geophys. Res. Biogeosci.*, *121*, 1568–1580, doi:10.1002/2016JG003341.

Received 28 JAN 2016
Accepted 3 MAY 2016

**Abstract** Knowledge of greenhouse gas (GHG) fluxes in soils is a prerequisite to constrain national, continental, and global GHG budgets. However, data characterizing fluxes from agricultural soils of Africa are markedly limited. We measured carbon dioxide ($CO_2$), nitrous oxide ($N_2O$), and methane ($CH_4$) fluxes at 10 farmer-managed sites of six crop types for 1 year in Kenya and Tanzania using static chambers and gas chromatography. Cumulative emissions ranged between 3.5–15.9 Mg $CO_2$-C $ha^{-1}$ $yr^{-1}$, 0.4–3.9 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$, and $-1.2$–10.1 kg $CH_4$-C $ha^{-1}$ $yr^{-1}$, depending on crop type, environmental conditions, and management. Manure inputs increased $CO_2$ ($p = 0.03$), but not $N_2O$ or $CH_4$, emissions. Soil cultivation had no discernable effect on emissions of any of the three gases. Fluxes of $CO_2$ and $N_2O$ were 54–208% greater ($p < 0.05$) during the wet versus the dry seasons for some, but not all, crop types. The heterogeneity and seasonality of fluxes suggest that the available data describing soil fluxes in Africa, based on measurements of limited duration of only a few crop types and agroecological zones, are inadequate to use as a basis for estimating the impact of agricultural soils on GHG budgets. A targeted effort to understand the magnitude and mechanisms underlying African agricultural soil fluxes is necessary to accurately estimate the influence of this source on the global climate system and for determining mitigation strategies.

## 1. Introduction

Agriculture is a significant source of greenhouse gas (GHG) emissions, directly from nitrogen fertilizer use, biomass burning, rice cultivation, livestock manure management, and enteric fermentation from animals and indirectly from land use change [*Davidson*, 2009; *Houghton et al.*, 2012]. Currently, direct GHG emissions from agriculture account for approximately 12% of annual anthropogenic emissions, a reduction from 14% in 2007 [*Intergovernmental Panel on Climate Change* (*IPCC*), 2014]. Despite the decline in relative impact of agriculture versus other GHG sources, the quantity of agricultural emissions in absolute terms has increased and is expected to further increase due to population growth and dietary shifts [*Tilman and Clark*, 2014; *Tubiello et al.*, 2014].

Soils managed for agriculture are both sources and sinks for GHGs [*Kirschke et al.*, 2013; *Smith et al.*, 2008]. The magnitude of exchange of $CO_2$, $N_2O$, and $CH_4$ between the biosphere and the atmosphere caused by biogenic processes including autotrophic and heterotrophic respiration, nitrification and denitrification, and methanogenesis and methanotrophy depends on carbon (C) and nitrogen (N) availability, land management, and environmental conditions (e.g., soil moisture, temperature, and pH) [*Butterbach-Bahl et al.*, 2011, 2013; *Davidson et al.*, 2000b, 2002; *Verchot et al.*, 2000]. The net consequence of these factors is that agricultural soils contribute approximately 39% of global agricultural GHG emissions and 87% of total agricultural $N_2O$ emissions [*FAOSTAT*, 2014a].

Nearly half, 42%, of agricultural soil emissions are produced in tropical countries [*FAOSTAT*, 2014a]. By comparison to temperate regions, however, tropical agricultural soils are underrepresented in global data sets. For example, fewer than 10% of the data in the seminal $N_2O$ data set were collected in tropical developing countries [*Stehfest and Bouwman*, 2006]. The lack of data translates to considerable uncertainty about the impact of tropical agricultural soils on the climate system.

Up to now, few studies have examined GHGs fluxes in managed soils of Africa, despite their areal extent and the likelihood of future intensification. For example, through 2014, there had only been 24 investigations of soil $N_2O$ emissions and the mechanistic controls of $N_2O$ production throughout sub-Saharan Africa (SSA), with only 14 of them being in situ measurements [*Hickman et al.*, 2014]. This small database contrasts starkly with 100 s of studies conducted in temperate regions including Europe and North America [e.g., *Owen and Silver*, 2014; *Rashti et al.*, 2015; *Shcherbak et al.*, 2014; *Snyder et al.*, 2009; *Stehfest and Bouwman*, 2006]. Available data from Africa suggest that the rates of emissions can be higher or lower than expected based on models derived from data collected in other locations [*Dick et al.*, 2006; *Rees et al.*, 2006; *Predotova et al.*, 2010]. Site specific mechanisms, such as rate and quality of substrate inputs (e.g., leaf litter) [*Millar et al.*, 2004; *Baggs et al.*, 2006] and local weather and soil texture [*Mapanda et al.*, 2010] regulate emissions. An equivalent dearth of information exists for soil fluxes of $CO_2$ and $CH_4$ in tropical Africa [*Kim*, 2012; *Kim et al.*, 2015]. It is also unlikely that observed fluxes of these gases will be accurately predicted using existing data given the weathered and often degraded soils, seasonal rainfall patterns, and variable management approaches used in farming systems of Africa.

Current information on fluxes from agricultural soils in SSA is extremely limited in scope. Previous measurements have concentrated on natural ecosystems, e.g., forests and savannahs [*Castaldi et al.*, 2013; *Werner et al.*, 2007; *Zepp and Miller*, 1996]. When agricultural soils are studied, it has been under semiarid conditions [*Mapanda et al.*, 2011] of West or Southern Africa [*Mapanda et al.*, 2010; *Rees et al.*, 2006]. Data available on East African agroecosystems primarily relate to $N_2O$ emissions derived from the integration of nitrogen fixing trees in croplands [*Baggs et al.*, 2006; *Dick et al.*, 2001; *Millar et al.*, 2004] or from tree plantations [*Nsabimana et al.*, 2009], relatively minor cropping systems in the region (see *Pelster et al.* [2015] for exception). Currently available data leave large gaps in the estimation of GHG fluxes of the most common agricultural production systems (e.g., pastures, smallholder, and low-input systems) and environmental conditions (e.g., regions receiving more than 1000 mm $yr^{-1}$ of precipitation).

Agriculture is the primary land use of the countries of the East African Community with both food and feed crops grown for subsistence and commercial production. For example, maize, rice, and cassava are three of the most important subsistence crops covering over 21.6 million ha of harvested land [*FAOSTAT*, 2014b]. Vegetables including indigenous leafy greens, potatoes, and tomatoes as well as animal feeds such as Napier grass (*Pennisetum purpureum* Schumach) or pastures and export crops such as tea are also common depending on agroclimatic zone. Cumulatively, agriculture covers approximately 60% of land in Tanzania, Kenya, Ethiopia, Uganda, Burundi, and Rwanda [*FAOSTAT*, 2014b].

GHG fluxes may vary considerably with soil properties, crop type, and management and climatic conditions. We quantified soil $CO_2$, $N_2O$, and $CH_4$ fluxes at 10 farmer-managed sites in the highlands of Kenya and Tanzania to explore the magnitude, heterogeneity, and annual patterns of baseline fluxes in agricultural soils under various crop types. We expected large variability in fluxes among sites because of differences in nutrient and soil management and microscale environmental conditions. This research provides some of the first GHG flux estimates for a handful of regionally important commercial and subsistence crops providing a critical input to better constrain GHG balances and calibrate agriculture's potential contribution to low emission development in the region.

## 2. Materials and Methods

### 2.1. Site Description

The study was conducted over 12 months (January through December 2013) at eight experimental sites in the highlands of Kenya within 10 km of Kaptumo town and two experimental sites in the Uluguru Mountains of Tanzania in the proximity of the village of Kolero. Kaptumo (35°029′E, 00°007′N) is typical of smallholder farming landscapes in the western Kenyan highlands, elevation 1800 to 2000 m, where dairy and tea are common commercial products and farmers grow a mixture of maize, sorghum, beans, and vegetables for home consumption. Natural vegetation in the area was once tropical forest but was converted to agriculture more than 100 years ago. Precipitation falls in a bimodal pattern with the "long rains" occurring in the 3 months between approximately mid-March and mid-June and the "short rains" taking place mid-October to mid-December, though precipitation is possible at most any time of the year.

**Table 1.** Farm Management at the 10 Farmer-Managed Sites During 2013[a]

| Site | Planting Dates | Soil and Pest Management | Fertilization | Harvest |
|---|---|---|---|---|
| | | *Kaptumo* | | |
| Feed 1 (F1) | 1997 | 15 Dec, herbicide | broadcast following harvest at ~4.1 t fresh manure ha$^{-1}$ application$^{-1}$ | 15–22 Jan; 15 May to 12 June; 2–9 Sept; and 12 Sept |
| Feed 2 (F2) | 2011 | none | Unquantified manure applied from direct deposition during grazing and spreading of slurry from adjacent feeding stalls | 23–30 Jan, grass cut and removed; 27 May to 10 June, grass cut and removed; 13–18 Sept, cattle grazed directly on site; and 6–31 Dec, cattle grazed directly on site |
| Pasture 1 (P1) | native grass pasture | - | Unfertilized | Grazed directly |
| Pasture 2 (P2) | native grass pasture | - | Unfertilized | Grazed directly |
| Tea 1 (T1) | 1969 | 28 Jan, herbicide; 4–11 June, herbicide; and 23 Sept, herbicide | 18 April –57 kg N, 5 kg P, and 9 kg K ha$^{-1}$ as NPK; 15 Oct –57 kg N, 5 kg P, and 9 kg K ha$^{-1}$ as NPK | 1 Jan to 25 March, 3 times a week; 29 July to 12 Oct, 3 times a week; and 30 Sept to 31 Dec, 3 times a week |
| Tea 2 (T2) | 1999 | 24 April, weeded; 8–21 May, weeded; 2 Aug, weeded; and 2 Dec, weeded | 28 May, 56 kg N, 5 kg P, and 9 kg K ha$^{-1}$ as NPK and 22 Oct to 11 Nov, 56 kg N, 5 kg P, and 9 kg K ha$^{-1}$ as NPK | 1 Jan to 20 March, daily and $>$09 Oct , daily |
| Vegetable 1 (V1) | 2012 Nov, indigenous vegetables and pumpkin; 13 May, potatoes; 5–19 Aug, cabbages; and 3 Dec, beans | 7 Jan, hand cultivated; 4–8/4: ploughed; 6 May, herbicide; 11 June, weeded; 30 Sept, weeded; and 18 Dec, ploughed | "handful" manure hole$^{-1}$ at planting; "spoonful" DAP hole$^{-1}$ at 62.5 kg ha$^{-1}$; spoonful DAP hole$^{-1}$ at 250 kg ha$^{-1}$; and spoonful DAP hole$^{-1}$ at 250 kg ha$^{-1}$ | 12 Feb; 22 July; and 16–25 Oct |
| Vegetable 2 (V2) | 2012 Nov, potatoes and 20 May, potatoes | 5 Feb, hand cultivated; 2 April, ploughed; 30 April, ploughed; 3–17 June, weeded; 9 Sept, hand cultivated; and 2 Dec, ploughed | 18 Feb, manure applied as handful hole$^{-1}$ at 2.5 t ha$^{-1}$ and 20 May, manure applied as handful hole$^{-1}$ at 1.25 t ha$^{-1}$ | 18 Feb and 9 Sept |
| | | *Kolero* | | |
| Cassava (C) | 2012 Dec | none | none | 2014 |
| Maize (M) | 2012 Dec and 9 May | None; 5 May, hand cultivated; and 6–15 Nov, hand cultivated | none | 2 May and 2–4 Oct. |

[a] Dates are based on observations of field staff and farmer survey (e.g., fertilizer rates) and are the day and month of 2013 except when noted. Sites in Kaptumo were contained on four farms of decreasing dairy production intensity as follows: F2 and V2 (zero grazing); P2, F1, and T2 (semizero grazing); T1 and V1 (semizero grazing); and P1 (full grazing). P1 was 2.7 ha of native grass pasture grazed by 25 cows. P2 was 0.6 ha of native grass pasture grazed by 17 cows.

Kolero (37°48′E, 07°015′S) sits within the remote Uluguru Mountain Range in Eastern Tanzania at an elevation ranging from 260 to 1250 m. Farmers cultivate maize and cassava on hillslopes, up to more than 30% slope, using slash-and-burn techniques [*Rosenstock et al.*, 2014a]. Virtually all production is for subsistence and there is virtually no use of external nutrient inputs [*Zagst*, 2011]. Annual precipitation trends in the region are similar to that of Kaptumo. Rainfall follows a bimodal pattern with the onset of the rains approximately 2 weeks later than for the Kaptumo site.

Study sites were selected to capture common crop types in the two areas and the region more broadly. In Kaptumo, we measured fluxes at plots of vegetables ($N = 2$, V1 and V2), tea ($N = 2$, T1 and T2), pasture ($N = 2$, P1 and P2), and forages ($N = 2$, Napier grass and Sudan grass, F1 and F2, respectively). Multiple plots of the same farming activities were measured to assess variability in fluxes that may result from land management (e.g., manure additions to vegetables or grazing intensity) or site properties (e.g., soil conditions). In Kolero, fluxes were monitored at two plots, one of maize (M) and one of cassava (C), located on different farms. Farmers managed sites according to their own standard practice (Table 1).

## 2.2. Soil Characterization

Soil chemical and physical properties were analyzed by standard procedures. Soil pH was determined in 1:25 suspension using distilled water. Texture was determined by using Bouyoucos hydrometer after pretreatment with $H_2O_2$ to remove organic matter [*Okalebo et al.*, 2002]. Total N and soil organic C were analyzed by dry combustion using a C/N analyzer (Flash 2000; Thermo Scientific). Bulk density was measured as a composite of five samples taken in each plot from soil depths 0–20 cm in Kaptumo and 0–10 cm in Kolero (Table 2).

**Table 2.** Site and Soil Properties for 10 Land Uses in Kaptumo, Kenya, and Kolero, Tanzania[a]

| Site | Bulk Density (g cm$^{-3}$) | pH | Soil Texture (%) Sand | Silt | Clay | TN (%) | SOC (%) | SOC (Mg C ha) |
|---|---|---|---|---|---|---|---|---|
| | | | *Kaptumo* | | | | | |
| Feed (F1) | 1.02 | 5.9 | 66.3 | 8.6 | 28.5 | 0.33 | 3.73 | 76.1 |
| Feed (F2) | 1.05 | 6.0 | 70.0 | 7.3 | 22.7 | 0.22 | 2.20 | 46.2 |
| Pasture (P1) | 0.85 | 6.2 | 68.3 | 8.6 | 23.0 | 0.32 | 3.51 | 59.7 |
| Pasture (P2) | 1.03 | 6.3 | 60.9 | 6.9 | 32.1 | 0.27 | 3.34 | 68.8 |
| Tea (T1) | 0.92 | 4.1 | 70.0 | 10.9 | 19.0 | 0.33 | 3.42 | 62.9 |
| Tea (T2) | 0.98 | 5.2 | 66.3 | 8.9 | 24.7 | 0.26 | 2.94 | 57.6 |
| Vegetables (V1) | 1.02 | 5.8 | 64.3 | 12.1 | 23.4 | 0.33 | 3.47 | 70.8 |
| Vegetables (V2) | 1.05 | 6.0 | 62.3 | 9.8 | 27.8 | 0.37 | 4.13 | 86.7 |
| | | | *Kolero* | | | | | |
| Cassava (C) | 1.15 | 6.8 | 81.4 | 8.9 | 9.6 | 0.08 | 0.97 | 22.3 |
| Maize (M) | 1.25 | 6.1 | 67.4 | 7.6 | 24.9 | 0.16 | 1.65 | 41.3 |

[a]SOC stocks calculated based on bulk density 0–20 cm in Kaptumo and 0–10 cm in Kolero.

## 2.3. Flux Measurements

$CO_2$, $N_2O$, and $CH_4$ fluxes were measured using vented, static chamber technique with thought taken to minimize sampling artifacts [*Parkin and Venterea*, 2010] and align with the guidelines for GHG measurements in smallholder systems [*Rosenstock et al*., 2013]. Chambers were made of plastic and comprised of two parts: a base (27 × 37.2 × 10 cm) inserted 5–10 cm into the soil and a lid (27 × 37.2 × 12.5 cm). The lids were equipped with 50 cm long (2.5 cm diameter) vents, thermometers to measure internal temperature and gas sampling ports. During measurements, the two pieces were held together with clamps and foam placed between them to form an airtight seal. Five chambers in total were installed at each site. Weeds and other plants, including pasture grass, growing in the chamber were cut to soil height before measurements. Chamber bases were inserted more than 1 week prior to the first measurement and then remained in place throughout the year. When plot cultivation occurred, chamber bases were removed and replaced after soil management. Sites in Kaptumo were sampled approximately weekly and sites in Kolero were sampled approximately twice weekly throughout the course of the year.

During each sampling event, chambers were closed for 30 min with four samples taken at 10 min intervals (0, 10, 20, and 30 min) from each chamber. Gas samples were collected using 60 mL propylene syringes with Luerlocks and immediately transferred into 10 or 20 mL glass vials fitted with crimp seals. The first 30 mL of the sample was used to flush the vial and the remaining 30 mL filled the vial, overpressurizing it to reduce the likelihood of contamination with ambient air. Samples were analyzed as soon after collection as possible based on laboratory capacity, on average 3 weeks after sampling.

Concentrations of $CO_2$, $N_2O$, and $CH_4$ were analyzed using a SRI GHG gas chromatograph (model 8610C; SRI) with a methanizer in conjunction with a flame ionization detector (FID) for $CH_4$ and $CO_2$ and a $^{63}Ni$ electron capture detector (ECD). The gas chromatograph was operated with Hayesep D packed columns (3 m, 1/8″) with oven temperature of 65°C, ECD detector and methanizer temperature of 350°C, and flow rates of 25 mL min$^{-1}$ $N_2$ as carrier gas on both FID and ECD lines. Gas concentrations of samples were calculated based on the peak areas measured by the gas chromatograph relative to the peak areas measured from calibration gases run four times each day. Concentrations were then converted to mass per volume using the Ideal Gas Law and measured chamber volume, internal chamber air temperature, and atmospheric pressure determined by GPS (Garmin Corporation) during sampling. Fluxes were calculated using linear regression of gas concentrations versus chamber closure time. Fluxes were set to zero if the flux was below the minimum detection limit calculated according to *Parkin et al*. [2012].

## 2.4. Soil Nitrate

Soil nitrate ($NO_3$-N) was determined January–March and August–December in Kaptumo and February–April and August–December in Kolero. At each site on each sampling date, we collected a composite of three samples to a depth of 10 cm. Samples were transferred directly to a cooler and kept cool with ice packs in transit to the laboratory where they were refrigerated and extracted within 2 days of sampling. Nitrate was

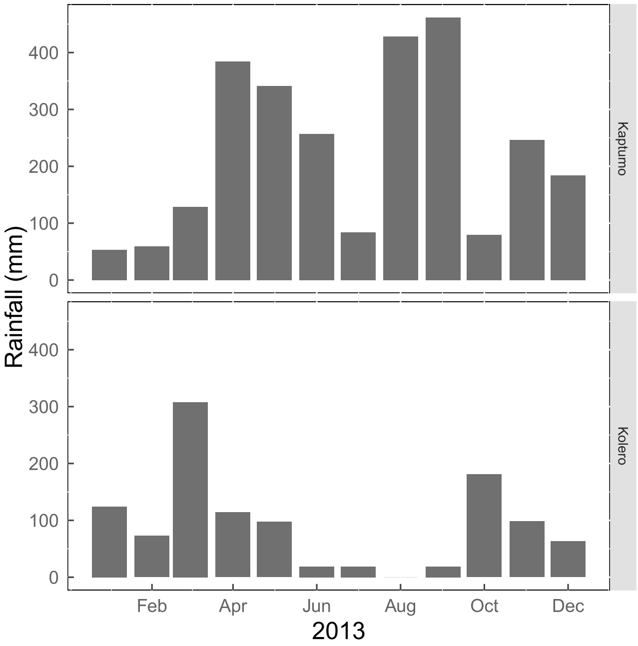

**Figure 1.** Rainfall at the Kaptumo, Kenya, and Kolero, Tanzania, sites in 2013.

determined by extracting 20 g subsamples of field-moist soils with 100 mL of 2 M KCl. The solution was shaken for 30 min on an orbital shaker. Samples were then gravity filtered and allowed to settle overnight. The supernatant was then frozen for later analysis. Analysis was performed by photometric analyzer (Aquakem 200: Thermo Scientific). Concentrations of $NO_3$-N were converted to a soil mass basis based on the water content of the soil measured at time of sampling using the ProCheck GS3 Sensor (Decagon Devices Inc).

### 2.5. Data Analysis

Cumulative estimates of fluxes at each site were estimated for the entire calendar year based on the mean flux of the five chambers at each site and linear interpolation between sampling events using the trapezoidal rule. Seasonal emissions and mean fluxes under various weather conditions were calculated based on wet and dry periods determined by selecting 2 month periods with greatest and least precipitation, respectively (Figure 1). Seasonal periods used were 1 August to 30 September (wet) and 1 January to 28 February (dry) in Kaptumo and 1 March to 30 April (wet) and 1 August to 30 September (dry) in Kolero. Seasonal differences in fluxes were evaluated with Mann-Whitney U for each crop type: forage, pasture, tea, vegetables, and staples. We tested for a crop type by season interaction by two-way analysis of variance (ANOVA). Seasonal emissions were log ($CO_2$ and $CH_4$) and square root ($N_2O$) transformed prior to running the ANOVA. We evaluated the effect of management (i.e., manure use and tillage) on emissions by Mann-Whitney U. Correlations between $CO_2$, $N_2O$, and $CH_4$ flux rates above the minimum detection limit and soil properties including soil temperature, soil moisture, and soil $NO_3^-$ were tested using Spearman's rank correlation.

## 3. Results

During the 12 month period that gas fluxes were measured, precipitation at Kaptumo and Kolero was 2708 mm and 1115 mm, respectively. Annual pattern of precipitation in Kaptumo showed a dampened bimodal distribution (Figure 1). Besides July and parts of February and March, precipitation fell fairly consistently throughout the year, though the months that typically receive more rainfall were considerably wetter (Figure 1). By contrast, precipitation in Kolero showed clear seasonal trends with nearly a 3 to 4 month dry season occurring following the long rain season: from June to September (Figure 1). No long-term records of precipitation were available for the region; however, our technicians and the participating farmers suggested that the amount of precipitation received was consistent with expectations for Kolero but more than normal for Kaptumo.

### 3.1. $CO_2$ Fluxes

Sites included in this study showed a wide range of $CO_2$ emissions. Annual soil emissions ranged from 3.5 to 15.9 t C $ha^{-1}$ (Table 3). Crops not undergoing tillage—tea and cassava—had among the lowest fluxes while cultivated plots such as vegetables higher in absolute terms, yet no measureable effect was found. Where manure was applied such as Sudan grass, fluxes of $CO_2$ were greater than in plots that did not receive manure ($p < 0.01$). The highest emissions were documented in pastures with actively grazing animals where the annual $CO_2$ emissions of soil and vegetation were between 13.4 and 15.9 t C $ha^{-1}$. Lower

**Table 3.** Cumulative Seasonal and Annual Emissions in 2013[a]

| | $CO_2$ (Mg $CO_2$-C $ha^{-1}$) | | | $N_2O$ (kg $N_2O$-N $ha^{-1}$) | | | $CH_4$ (kg $CH_4$-C $ha^{-1}$) | | |
|---|---|---|---|---|---|---|---|---|---|
| Site | Dry | Wet | Annual | Dry | Wet | Annual | Dry | Wet | Annual |
| | | | | *Kaptumo* | | | | | |
| Feed (F1) | 0.9 (0.1) | 1.9 (0.3) | 6.7 (1.2) | 0.1 (0.0) | 0.1 (0.1) | 0.5 (0.2) | −0.0 (0.0) | 2.5 (0.8) | −1.2 (1.8) |
| Feed (F2) | 0.7 (0.1) | 1.6 (0.2) | 10.3 (1.9) | 0.0 (0.0) | 0.1 (0.0) | 0.5 (0.3) | 0.1 (0.3) | −0.6 (3.2) | 7.1 (7.2) |
| Pasture (P1) | 2.1 (0.4) | 2.8 (0.6) | 13.4 (2.6) | 0.1 (0.0) | 0.4 (0.2) | 2.3 (1.3) | −0.2 (0.0) | −0.3 (0.8) | −0.2 (2.4) |
| Pasture (P2) | 1.9 (0.4) | 3.5 (1.0) | 15.9 (3.3) | 0.2 (0.1) | 1.5 (1.0) | 3.9 (2.3) | 1.8 (2.0) | −0.1 (0.5) | 10.1 (10.7) |
| Tea (T1) | 0.7 (0.1) | 0.5 (0.1) | 3.5 (0.6) | 0.1 (0.0) | 0.1 (0.1) | 0.7 (0.5) | 0.0 (0.5) | 0.5 (0.7) | 0.3 (1.6) |
| Tea (T2) | 0.6 (0.1) | 0.9 (0.2) | 5.5 (0.9) | 0.0 (0.0) | 0.0 (0.0) | 0.4 (0.2) | 0.1 (0.1) | 0.5 (0.8) | 0.1 (1.1) |
| Vegetables (V1) | 1.5 (0.3) | 1.1 (0.3) | 7.4 (1.7) | 0.1 (0.1) | 0.1 (0.0) | 0.9 (0.5) | 0.1 (0.7) | −0.1 (0.2) | −0.7 (1.6) |
| Vegetables (V2) | 0.9 (0.1) | 1.1 (0.2) | 6.3 (1.1) | 0.1 (0.0) | 0.0 (0.0) | 0.9 (0.4) | 0.0 (0.1) | −0.2 (0.1) | −0.6 (0.9) |
| | | | | *Kolero* | | | | | |
| Cassava (C) | 0.4 (0.1) | 1.2 (0.2) | 4.4 (0.8) | 0.6 (0.0) | 0.1 (0.0) | 0.4 (0.2) | −0.1 (0.8) | 0.0 (0.1) | 0.8 (5.1) |
| Maize (M) | 0.4 (0.1) | 1.1 (0.2) | 4.2 (0.8) | 0.0 (0.0) | 0.1 (0.0) | 0.9 (0.7) | 0.1 (0.1) | −0.4 (0.5) | 0.8 (5.4) |
| | | $CO_2$ | | | $N_2O$ | | | $CH_4$ | |
| Factor | df | *F* value | *p* value | *F* value | | *p* value | *F* value | | *p* value |
| Crop type | 4 | 18.03 | <0.001 | 0.67 | | 0.631 | 0.18 | | 0.941 |
| Season | 1 | 12.42 | 0.005 | 0.52 | | 0.485 | 0.12 | | 0.740 |
| Crop type * Season | 4 | 5.49 | 0.013 | 1.79 | | 0.206 | 0.51 | | 0.732 |
| Residuals | 10 | | | | | | | | |

[a] Seasonal emissions for Kaptumo, Kenya, and Kolero, Tanzania, based on 2 month periods of driest and wettest weather. Standard errors of emissions are presented in parentheses.

fluxes (less than or equal to 7 t C $ha^{-1}$) tended to be found on lighter textured soils with lower SOC concentrations (C and M) or perennial crop soils with low pH (T1 and T2).

The temporal patterns and magnitude of $CO_2$ fluxes showed large variations across farming sites and farming activities (coefficient of variation (CV) ranged from 37 to 57%) (Figure 2). Some sites such as the Sudan grass (F2), which received intermittent manure slurry, showed steady high rates throughout the second half of 2013. By contrast, fluxes from the Napier grass (F1) were more irregular, fluctuating between very low background emissions during drier periods (June to August) and then showing increases toward the end of the year. Fluxes from pasture and vegetables showed similar degrees of variability between the two sites for each crop group.

Mean $CO_2$ fluxes were significantly different between the wet and dry periods for three of the five crop types examined (Figure 3). Mean dry season fluxes were 47, 65, 90, 108, and 31% of mean wet season fluxes for forages, pasture, tea, vegetables, and staples, respectively. Fluxes were significantly greater in the wet season than the dry season for forages, pastures, and staples ($p < 0.001$). Mean fluxes during wet periods ranged between 50 to greater than 200 mg $CO_2$-C $m^{-2}$ $h^{-1}$, while during the drier season fluxes were mostly between 35 and 150 mg $CO_2$-C $m^{-2}$ $h^{-1}$. Agroecosystems in Kolero showed the greatest differences between wet and dry periods, more than 100% of the mean flux. This pattern—greater emissions during wetter periods—was consistent for all but the vegetable plots (Figure 3). There was a significant effect of crop type, season, and the interaction of these factors on seasonal $CO_2$ emissions (Table 3).

### 3.2. $N_2O$ Fluxes

Cumulative $N_2O$ emissions from the two regions ranged from 0.4 to 3.9 kg N $ha^{-1}$ $yr^{-1}$ (Table 3). Pasture sites had the largest cumulative fluxes, greater than 2 kg $N_2O$-N $ha^{-1}$ $yr^{-1}$, and mean fluxes in pastures were more than double that of any other land cover. However, neither manure nor tillage had a discernable affect on $N_2O$ emissions.

Temporal patterns and peak $N_2O$ fluxes varied by crop type (Figure 4, CVs ranged from 113 to 328%). For example, pulses of $N_2O$ reached nearly 100 μg N $m^{-2}$ $h^{-1}$ following application of fertilizers and rain at both tea sites but were low during the intervening periods. The two vegetable sites showed similar patterns of emissions. The two pastures demonstrated the greatest variability in $N_2O$ fluxes (CVs greater than 325%) both between the two sites and within a given site with variation equivalent to more than 150% of mean fluxes and peak fluxes occurring asynchronously. Furthermore, pastures showed the greatest peak fluxes, above 150 μg $m^{-2}$ $h^{-1}$. Similar to $CO_2$, crops receiving manure fertilizer applications had the greatest cumulative fluxes.

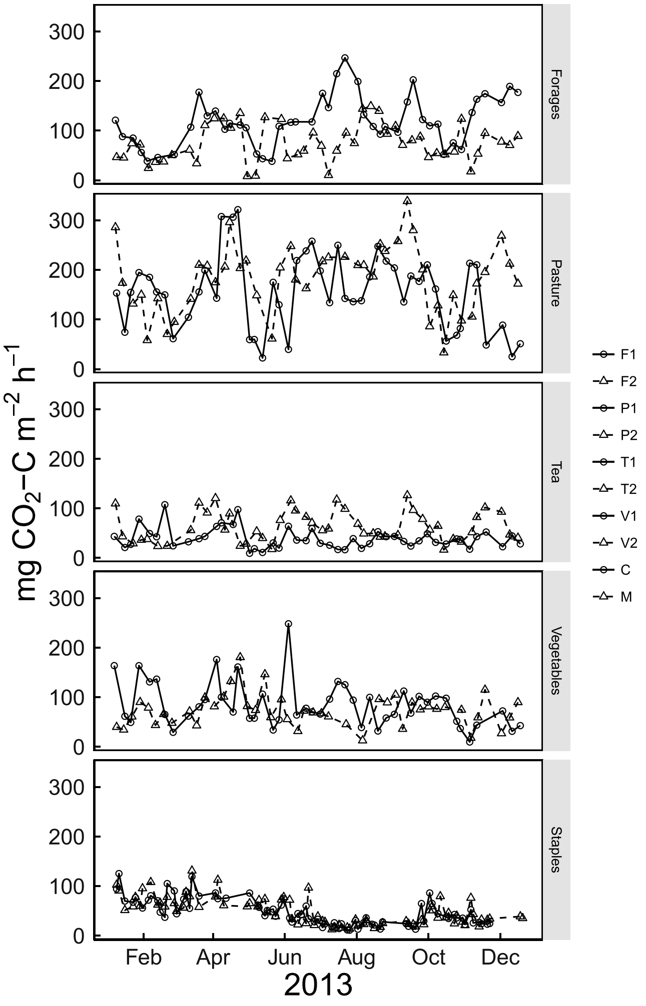

**Figure 2.** Soil emissions of $CO_2$ for ten farmer-managed sites (five chambers site$^{-1}$).

Mean $N_2O$ flux rates ranged from 2 to 62 μg N m$^{-2}$ h$^{-1}$ during the wetter periods and from 1 to 9 μg N m$^{-2}$ h$^{-1}$ during the drier periods. Fluxes for four of the five farming activities were higher during the characteristic wet period than the dry period selected in absolute terms. However, only wet season fluxes of pastures and staples were significantly greater than dry season fluxes (Figure 3). Crop type, season, and the interaction between crop type and season did not have a measurable effect on $N_2O$ emissions (Table 3).

### 3.3. $CH_4$ Fluxes

The studied uplands soils were both sources and sinks for $CH_4$. Annual net $CH_4$ fluxes did not differ between crop types, season, or their interaction and ranged from −1.2 to 10.1 kg $CH_4$-C depending on the site (Table 3). Arable soils tended to consume $CH_4$ while permanent pasture tended to be net sources of $CH_4$. Fluxes of $CH_4$ were often below the minimum detection limit in both the dry and wetter periods for all but one pasture and vegetable site.

### 3.4. Soil Nitrate, Moisture, and Temperature

Across the eight sites in Kaptumo, median $NO_3$-N was less than 12 g kg dry soil$^{-1}$ and typically below 5 kg dry soil$^{-1}$ (Figure 5). However, the inorganic soil N concentrations fluctuated greatly among sampling events evident by the difference by at least a factor of 4 between 25 and 75% quartiles measurements. Plot T1 had the highest median levels of soil $NO_3$-N and widest distribution. Soil inorganic N data were unavailable for the Kolero sites because of logistical challenges of preserving soils at appropriate temperature in transit to the laboratory.

Soil moisture in Kaptumo was typically higher than that found at the site sampled in Kolero (Figure 6). In Kaptumo, water-filled pore space (WFPS) ranged between 20 and nearly 100%. Despite variability, soil moisture in Kaptumo was consistently above 60% toward the end of the year (October–December). In contrast, soil moisture in Kolero was lower, typically below 60% for almost all of the measurements performed.

Three of the nine relationships tested between fluxes and environmental variables showed significant correlations (Table 4). $CO_2$ and $N_2O$ fluxes were positively correlated to soil temperature ($p < 0.001$ and $p < 0.05$, respectively) and soil moisture was positive correlated to $CH_4$ ($p < 0.05$).

## 4. Discussion

Research on soil fluxes of $CO_2$, $N_2O$, and $CH_4$ in sub-Saharan Africa has overlooked many the most common production systems on the continent [e.g., *Hall et al.*, 2006; *Hergoualc'h et al.*, 2007; *Mapanda et al.*, 2010; *Werner et al.*, 2007]. Here, by contrast, GHG fluxes were measured in crop types central to rural livelihoods and landscapes, with widespread distribution across the East African region.

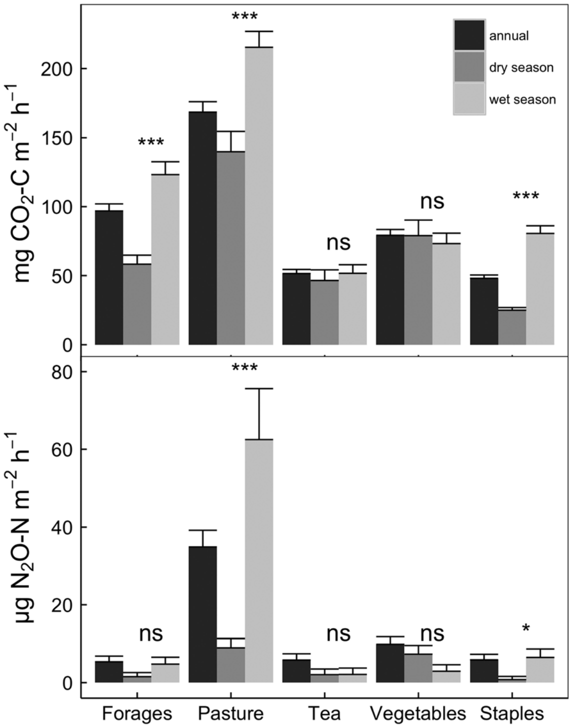

**Figure 3.** Annual and seasonal $CO_2$ and $N_2O$ fluxes in five agricultural land covers in East Africa. "Wet season" refers to 2 months with greatest consistent precipitation in Kaptumo (August–September, 890 mm) and Kolero (March–April, 422 mm), while "dry season" refers to a period with minimal rainfall in Kaptumo (January–February, 113 mm) and Kolero (August–September, 19 mm). $^{***}p < 0.001$, $^{*}p < 0.05$, ns = not significantly different.

$CO_2$ emissions at the 10 sites were generally low by comparison to fluxes in other tropical agricultural soils; for example, emissions ranging from 20 to greater than 38 t $CO_2$-C ha$^{-1}$ yr$^{-1}$ have been observed in agroforestry systems of Brazil, urban gardens in Niger and vegetable production in Uganda [*Koerber et al.*, 2009; *Predotova et al.*, 2009; *Verchot et al.*, 2008]. Seven of the 10 sites studied here emitted less than 8 t $CO_2$-C ha$^{-1}$ yr$^{-1}$ (SOC stocks estimated to be 20 to more than 60 Mg C ha$^{-1}$ in top 20 cm). The low level of emissions found at most of the 10 field sites may perhaps be attributed to the low production intensity of some of these systems (e.g., lack of tillage or low animal stocking rates). Two previous studies explored emissions on low-input agricultural plots of maize and sorghum in sub-Saharan Africa and documented emissions ranging from 2.5 to 4.5 Mg $CO_2$-C ha$^{-1}$ yr$^{-1}$ [*Brümmer et al.*, 2009; *Mapanda et al.*, 2011], which were consistent with emission rates in maize and cassava systems in Kolero. Emissions rates greater than the amount found in Kolero, for example, in vegetables and feed sites of Kaptumo, can likely be attributed to the application of 3 t ha$^{-1}$ of fresh manure applied, tillage in the annual cropping systems, soil disturbance due to grazing in pastures, and higher rainfall at these site by comparison to previous studies. It is important to note that our data for pastures represent cumulative respiration and thus do not differentiate among various sources contributing to gross ecosystem respiration (e.g., microbial, root respiration, or aboveground plant parts). Root respiration is likely to be an important source of $CO_2$ emissions in well-rooted perennial grass and tea systems as aboveground plant parts may contribute largely in pasture systems.

$CO_2$ emissions were greatest (10.4–15.9 t $CO_2$-C ha$^{-1}$ yr$^{-1}$) in Kaptumo when the fields and pastures received manure inputs, either due to farmer application (F1 and F2) or through cattle deposition of feces and urine (P1 and P2). Applications of manures provide easily degradable substrates of C and N catalyzing emissions. Emission rates measured here agree with respiration rates on pastures in the Amazon, 11.0–15.0 t $CO_2$-C ha$^{-1}$ yr$^{-1}$ [*Davidson et al.*, 2000a]. These numbers represent the first measures of GHG fluxes in intensively managed pasture emissions in Africa. With the current extent of pasture-based mixed crop-livestock farming systems in sub-Saharan Africa estimated to be $6.01 \times 10^6$ km [*Thornton and Herrero*, 2010], fluxes from managed pasture represent a potentially large and poorly quantified source of $CO_2$. Emissions, though, need to be considered on balance. Tropical pastures generate herbaceous growth and root biomass C inputs on the order of magnitude from 1 to 12 t $CO_2$-C ha$^{-1}$ yr$^{-1}$, which counterbalances at least part of these losses. Considering the upper end of the range of C inputs is toward the lower end of our range of measured emissions in pastures suggest the possibility that pastures under management and environmental conditions found in Kaptumo would still be net sources of C to the atmosphere.

Comparisons of respiration in this study to previous research, especially that undertaken in sub-Saharan Africa, are confounded by methodological differences. Earlier studies focused on discrete events, such as growing seasons [*Mapanda et al.*, 2011] or sample fluxes at low frequency—monthly [*Koerber et al.*, 2009]. Neither approach is capable of capturing temporal trends well. Our data show large temporal variation over

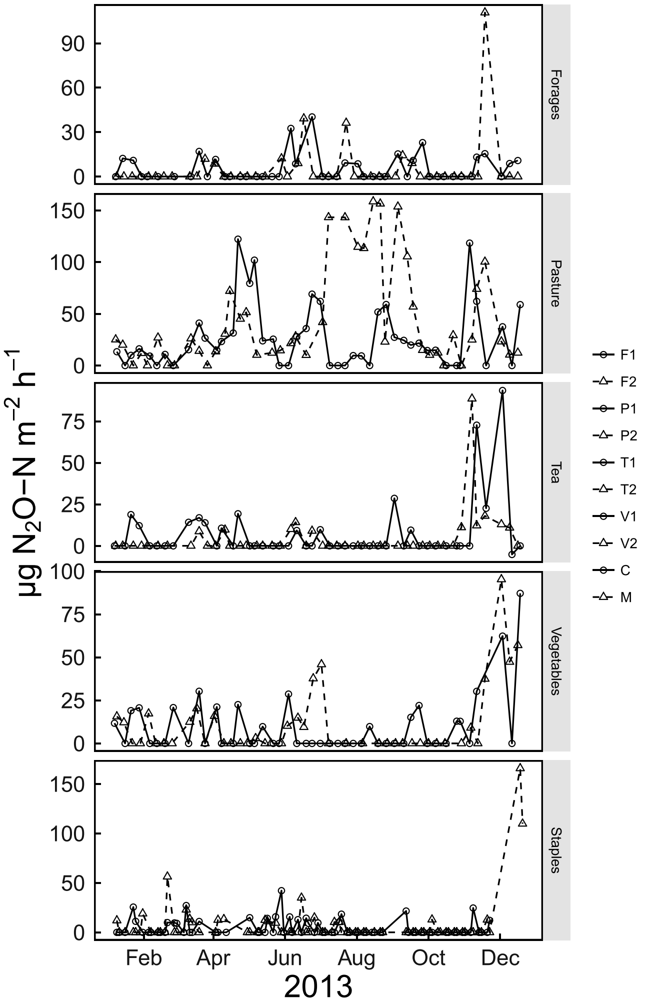

**Figure 4.** $N_2O$ fluxes in 2013 for 10 agroecosystems in 2013 (five chambers $site^{-1}$).

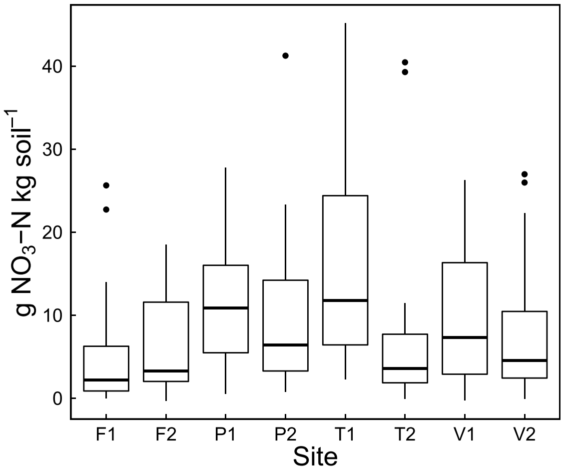

**Figure 5.** Soil nitrate (g $NO_3$-N kg dry $soil^{-1}$) in the eight Kaptumo, Kenya, plots.

the course of the year, evident by differences in mean fluxes between wet and dry periods. Temporal variation of fluxes is common, particularly when precipitation and soil moisture follow seasonal patterns. Consistent with our expectations, wet season $CO_2$ and $N_2O$ were significantly greater than dry season fluxes for many of the farming activities studied. Similarities in the seasonal fluxes in vegetables may be attributed to consistent rainfall throughout the course of the year in Kaptumo. With prevailing trends, infrequent or limited duration measurements compromise the ability to estimate annual emissions or the cumulative impact of agriculture on climate with confidence [*Barton et al.*, 2015]. Subsequent efforts need to ensure appropriate measurement frequency to capture short-term variability ("hot moments") and the annual trends [*Barton et al.*, 2015; *Leon et al.*, 2014].

Measured $N_2O$ flux rates in agricultural managed soils in sub-Saharan Africa have shown a wide range, from $-2$ to more than 300 $\mu g\,N\,m^{-2}\,h^{-1}$ resulting in cumulative emissions of 0.5 to more than 4.1 kg $N_2O$-N $ha^{-1}$ emissions depending on farming system, soil texture, and weather [*Brümmer et al.*, 2008; *Millar et al.*, 2004]. $N_2O$ emissions in this study were consistent with these previous results with average fluxes typically less than 12 $\mu g\,N\,m^{-2}\,h^{-1}$ and cumulative emissions within the range previously found.

Pastures, however, showed considerably higher fluxes, even reaching 159 $\mu g\,N\,m^{-2}\,h^{-1}$. We suspect that the large pulses of $N_2O$ may have resulted from urine and/or feces deposition. Urine provides water, increases pH, and provides labile N to the soil. Even though the N is not in an inorganic form, the urea is quickly broken down to ammonium via hydrolysis. Ammonium can then be converted to $NO_3^-$ (which can produce $N_2O$ as a byproduct), which can then later be denitrified. Though we did not witness animals urinating into the chambers, we did find feces on and near the chambers on multiple occasions and it

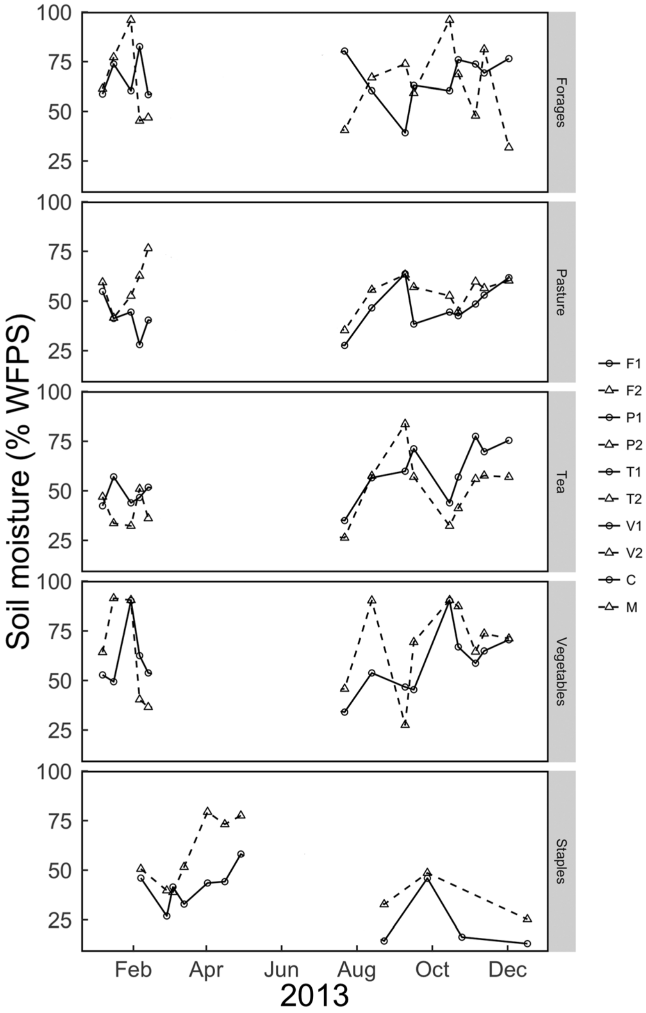

**Figure 6.** Soil moisture in 2013.

is common for cattle to urinate and produce feces simultaneously and in succession. Erratic annual trends in $N_2O$ on the pastures, the high mean flux rates even during dry period ($9\,\mu g\,m^{-2}\,h^{-1}$) and the variation among the chamber evidenced by CV more than 110% lead us to believe this flux is the consequence of urine. It is not possible to correlate pasture $N_2O$ fluxes to soil N from our data here because soil N was measured in the near vicinity of chambers and not within the chambers themselves to avoid soil disturbance. This asymmetry may also explain the lack of correlations between soil N, moisture, and $N_2O$ flux. Controlled experiments with side-by-side applications of urine within and outside the chamber are necessary. Currently, there are no published estimates of urine-derived $N_2O$ fluxes from African pastures (see *van Groenigen et al.* [2005] for data collected in temperate systems). Greater attention is likely warranted to understand this flux given the prevalence of livestock in the region and the climate forcing potential of $N_2O$.

Tea received mineral nitrogenous fertilizers, with farmers reporting application rates of approximately 112 kg N ha$^{-1}$ at both sites. Based on IPCC default emissions factors [*IPCC*, 2006], emissions due to these fertilizer additions were expected to be approximately 1.1 kg $N_2O$-N ha$^{-1}$. $N_2O$ fluxes at both locations—which include fertilizer-induced and background fluxes—were lower than expected, 0.38 and 0.75 kg $N_2O$-N ha$^{-1}$ yr$^{-1}$. That suggests the emission factor from N application in smallholder tea systems would be below 1% of N applied and applying the IPCC default emissions factor for smallholder tea gardens, a regionally important commercial agroecosystem, may overestimate emissions. Caution should, however, be taken when extrapolating our fluxes to annual emissions as it is possible that even at our relatively high temporal sampling frequency for manual chambers in Africa (once to twice a week) we may have missed hot moments and therefore underestimated emissions [*Barton et al.*, 2015]. Further studies that increase replication and sampling frequency across the highland tea zones of East Africa are necessary to substantiate this conclusion.

The mechanism driving the difference in emission rates between the two tea growing sites is not immediately apparent. Clearly, the greater fluxes in T1 are being driven by increased N availability, as median

**Table 4.** Correlation (Rho Values) Between Measured Variables and Gas Flux

| Variable | $CO_2$ | $N_2O$ | $CH_4$ |
|---|---|---|---|
| Soil temperature | 0.26*** | 0.12** | 0.01 |
| Soil moisture (% WFPS) | 0.06 | 0.14 | 0.23* |
| Nitrate ($NO_3^-$) | 0.06 | 0.13 | 0.05 |

*$p < 0.1$.
**$p < 0.05$.
***$p < 0.001$.

$NO_3$-N levels in T1 were more than double that of T2. However, $NO_3$-N was not generally correlated to N fluxes in this study, perhaps because soil moisture was typically below optimal conditions to promote denitrification. Given the similarities in management and the only 0.05% difference in TN pool, perhaps this increased level of inorganic N results from greater nitrification rates in T1. Alternatively, higher fluxes occurred on soils with lower pH and it is well established that lower pH results in higher $N_2O$ losses [*Bakken et al*., 2012; *Kesik et al*., 2006]. Besides N availability and pH, the higher fluxes on T1 might be due to the slightly coarser soil texture facilitating gas diffusion or the SOC levels at the site. T1 had 16% more SOC in 0–20 cm than T2. Higher SOC concentrations can change the $N_2$:$N_2O$ ratio potentially producing more $N_2O$. Contrarily, low SOC concentration constrains denitrification and microbial activity reducing $N_2O$ yield [*Davidson et al*., 2000b]. However, at levels of 2 to 3% SOC, we would not have expected to see C limitation of $N_2O$ production.

Fluxes in the Kolero fields were generally lower than fluxes around Kaptumo. This can plausibly result from the nutrient poor soils that exist in the area and the lack of nutrient additions (mineral or organic) that occurred at either of the sites. However, some $N_2O$ fluxes in Kolero were of similar magnitude to that found in Kaptumo, despite not receiving inputs and having lower C contents. One explanation for this could be related to the response function of $N_2O$ relative to N inputs. Increasing evidence suggests that the rate of $N_2O$ increases nonlinearly after exceeding plant N demand [*Shcherbak et al*., 2014]. Though N was applied in Kaptumo, it was likely below the rate of uptake for productive crop systems in these areas. Alternatively, similarity in fluxes in Kolero and Kaptumo may be related to soil degradation. Though data are scarce, fluxes from degraded soils may be less than healthy counterparts. For example, $CO_2$ emissions in a heavily grazed and abandoned pasture were only 60% of that in active pastures in Brazil [*Davidson et al*., 2000a]. Soil degradation is widespread in sub-Saharan Africa [*Vagen et al*., 2005] due to continuous cropping and highly weathered soils. Soil degradation may help explain why $N_2O$ fluxes measured from unfertilized cropland in sub-Saharan Africa are 33–50% of the global mean of ~1 kg N $ha^{-1}$ $yr^{-1}$ [*Hickman et al*., 2014].

We chose to examine variability within and between crop types to appraise fluxes in key farming activities common in the region, at the expense of creating a sampling design that would allow us to statistically test differences among sites of fewer crop types. Temporal patterns of fluxes show mean and peak fluxes of all gases differed between the Kenya and Tanzania locations, among land uses within a site and between sites of the same land use at the same location. This heterogeneity arose, in some cases, despite what appears to be similar environmental and soil conditions. The two Kaptumo vegetable plots, for example, have roughly comparable pH, bulk density, and soil texture and can be assumed to have near identical weather given they were less than 500 m apart. Yet during the wet period, $CO_2$ and $N_2O$ fluxes were highly variable between sites with peak emissions occurring asynchronously. Heterogeneity in flux rates and patterns among sites of similar land use history and environmental conditions can thus likely be attributed to fine scale differences in crop management, e.g., cultivation techniques, organic matter retention or additions, or cropping patterns. When crops were managed nearly identically but under varying environmental/soil conditions, equivalent heterogeneity in flux rates was found (e.g., tea). Such farmer and environment-induced heterogeneity highlight the challenges of generalizing about and predicting soil GHG fluxes under the diverse management and environmental conditions found in smallholder African farming systems.

## 5. Conclusion

GHG fluxes in agricultural soils of Kenya and Tanzania varied in both patterns and magnitude across crop types and sites. In many cases, the fluxes measured present significant departures from expected results, such as the low emission factor for tea gardens and the high emissions from pastures, likely as the consequence of farm management and microscale and mesoscale weather patterns. The variability and heterogeneity found here point to a large gap in our understanding of GHG fluxes from agricultural soils in Africa. Systematic measurements to better constrain the magnitude of fluxes, full GHG budget including C inputs, and mechanisms driving the heterogeneity in the most common farming activities will be critical to move the conversation beyond quantification and inform policy and low-emission development priorities. Our measurements in remote locations of East Africa were constrained by logistics and available infrastructure, adding challenges to executing the research. Investment is needed to improve scientific infrastructure in sub-Saharan Africa if the global community finds value and demands agricultural and environmental monitoring.

**Acknowledgments**

We thank the UN FAO for their constructive comments. The Government of Finland funded this work through the FAO Mitigation of Climate Change in Agriculture (MICCA) Programme. We thank the CGIAR Research Program on Climate Change, Agriculture, and Food Security and its Standard Assessment of Mitigation Potential and Livelihoods in Smallholder Systems (SAMPLES) Programme for technical and financial support for scientists and laboratories working on this program. All data are archived on ICRAF's Dataverse Web site and available for free upon request [*Rosenstock et al.*, 2014b].

## References

Baggs, E. M., J. Chebii, and J. K. Ndufa (2006), A short-term investigation of trace gas emissions following tillage and no-tillage of agroforestry residues in western Kenya, *Soil Tillage Res.*, *90*, 69–76, doi:10.1016/j.still.

Bakken, L. R., L. Bergaust, L. Binbin, and Å. Frostegård (2012), Regulation of denitrification at the cellular level: A clue to the understanding of N2O emissions from soils, *Phil. Trans. R. Soc. B. 367*, 1226–1234, doi:10.1098/rstb.2011.0321.

Barton, L. B. W., D. Rowlings, C. Scheer, R. Kiese, P. Grace, K. Stefanova, and K. Butterbach-Bahl (2015), Sampling frequency affects estimates of annual nitrous oxide fluxes, *Sci. Rep.*, *5*, 15912, doi:10.1038/srep15912.

Brümmer, C., N. Brüggemann, K. Butterbach-Bahl, U. Falk, J. Szarzynski, K. Vielhauer, R. Wassmann, and H. Papen (2008), Soil-atmosphere exchange of $N_2O$ and NO in near-natural savanna and agricultural land in Burkina Faso (W. Africa), *Ecosystems*, *11*, 582–600.

Brümmer, C., H. Papen, R. Wassmann, and N. Brüggemann (2009), Fluxes of $CH_4$ and $CO_2$ from soil and termite mounds in south Sudanian savanna of Burkina Faso (West Africa), *Global Biogeochem. Cycles*, *23*, 1–13, doi:10.1029/2009GB003616.

Butterbach-Bahl, K., R. Kiese, and C. Liu (2011), Measurements of biosphere-atmosphere exchange of $CH_4$ in terrestrial ecosystems, *Methods Enzymol.*, *495*, 271–287, doi:10.1016/B978-0-12-386905-0.00018-8.

Butterbach-Bahl, K., E. M. Baggs, M. Dannenmann, R. Kiese, and S. Zechmeister-Boltenstern (2013), Nitrous oxide emissions from soils: How well do we understand the processes and their controls?, *Philos. Trans. R. Soc. B Biol. Sci.*, *368*, 20130122, doi:10.1098/rstb.2013.0122.

Castaldi, S., T. Bertolini, A. Valente, T. Chiti, and R. Valentini (2013), Nitrous oxide emissions from soil of an African rain forest in Ghana, *Biogeosciences*, *10*, 4179–4187, doi:10.5194/bg-10-4179-2013.

Davidson, E. A. (2009), The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860, *Nat. Geosci.*, *2*, 659–662, doi:10.1038/ngeo608.

Davidson, E. A., L. Verchot, and J. Cattânio (2000a), Effects of soil water content on soil respiration in forests and cattle pastures of eastern Amazonia, *Biogeochemistry*, *48*, 53–69.

Davidson, E. A., M. Keller, H. E. Erickson, L. V. Verchot, and E. Veldkamp (2000b), Testing a conceptual model of soil emissions of nitrous and nitric oxides, *Bioscience*, *50*, 667–680.

Davidson, E. A., K. Savage, L. V. Verchot, and R. Navarro (2002), Minimizing artifacts and biases in chamber-based measurements of soil respiration, *Agric. For. Meteorol.*, *113*, 21–37, doi:10.1002/2016JG003341/pdf.

Dick, J., U. Skiba, and J. Wilson (2001), The effect of rainfall on NO and $N_2O$ emissions from Ugandan agroforest soils, *Phyton (B. Aires)*, *41*, 73–80, doi:10.1002/2016JG003341/pdf.

Dick, J., U. Skiba, R. Munro, and D. Deans (2006), Effect of N-fixing and non N-fixing trees and crops on NO and $N_2O$ emissions from Senegalese soils, *J. Biogeogr.*, *33*, 416–423, doi:10.1002/2016JG003341/pdf.

FAOSTAT (2014a), FAOSTAT Emissions Database. [Available at http://faostat.fao.org/site/705/default.aspx, Accessed 1 November 2014.]

FAOSTAT (2014b), FAOSTAT. [Available at http://faostat3.fao.org/home/E, Accessed 14 July 2014.]

Hall, N. M., B. Kaya, J. Dick, U. Skiba, A. Niang, and R. Tabo (2006), Effect of improved fallow on crop productivity, soil fertility and climate-forcing gas emissions in semi-arid conditions, *Biol. Fertil. Soils*, *42*, 224–230.

Hergoualc'h, K., U. Skiba, J.-M. Harmand, and R. Oliver (2007), Processes responsible for the nitrous oxide emission from a Costa Rican Andosol under a coffee agroforestry plantation, *Biol. Fertil. Soils*, *43*, 787–795.

Hickman, J. E., R. J. Scholes, T. S. Rosenstock, C. P. Garcia-Pando, and J. Nyamangara (2014), Assessing non-$CO_2$ climate-forcing emissions and mitigation in sub-Saharan Africa, *Curr. Opin. Environ. Sustain.*, *9-10*, 65–72.

Houghton, R. A., J. I. House, J. Pongratz, G. R. van der Werf, R. S. DeFries, M. C. Hansen, C. Le Quéré, and N. Ramankutty (2012), Carbon emissions from land use and land-cover change, *Biogeosciences*, *9*, 5125–5142.

Intergovernmental Panel on Climate Change (IPCC) (2006), Chapter 11: $N_2O$ emissions from managed soils and $CO_2$ emissions from lime and urea application, in *2006 IPCC Guidelines for Natioinal Greenhouse Gas Inventories*, IGES, Japan.

Intergovernmental Panel on Climate Change (IPCC) (2014), *Climate Change 2014: Synthesis Report Contribution of Working Groups I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by Core Writing Team, R. K. Pachauri, and L. A. Meyer, 151 pp., IPCC, Geneva, Switzerland.

Kesik, M., S. Blagodatsky, H. Papen, and K. Butterbach-Bahl (2006), Effect of pH, temperature and substrate on $N_2O$, NO and $CO_2$ production by *Alcaligenes faecalis p.*, *J. App. Microbio.*, *101*, 655–667, doi:10.1111/j.1365-2672.2006.02927.x.

Kim, D.-G. (2012), Estimation of net gain of soil carbon in a nitrogen-fixing tree and intercropping system in sub-Saharan Africa: Results from re-examining a study, *Agrofor. Syst.*, *86*, 175–194, doi:10.1002/2016JG003341/pdf.

Kim, D.-G., A. D. Thomas, D. Pelster, T. S. Rosenstock, and A. Sanz-Cobena (2015), Greenhouse gas emissions in natural and agricultural lands in sub-Saharan Africa: Synthesis of available data and suggestions for further studies, *Biogeosciences Discuss.*, *12*, 16,479–16,526, doi:10.5194/bgd-12-16479-2015.

Kirschke, S., et al. (2013), Three decades of global methane sources and sinks, *Nat. Geosci.*, *6*, 813–823, doi:10.1038/ngeo1955.

Koerber, G. R., G. Edwards-Jones, P. W. Hill, L. M. I. Canals, P. Nyeko, E. H. York, and D. L. Jones (2009), Geographical variation in carbon dioxide fluxes from soils in agro-ecosystems and its implications for life-cycle assessment, *J. Appl. Ecol.*, *46*, 306–314.

Leon, E., R. Vargas, S. Bullock, E. Lopez, A. Rodrigo Panosso, and N. La Scala Jr. (2014), Hot spots, hot moments, and spatio-temporal controls on soil $CO_2$ efflux in a water-limited ecosystem, *Soil Biol. Biochem.*, *77*, 12–21, doi:10.1016/j.soilbio.2014.05.029.

Mapanda, F., J. Mupini, M. Wuta, J. Nyamangara, and R. M. Rees (2010), A cross-ecosystem assessment of the effects of land cover and land use on soil emission of selected greenhouse gases and related soil properties in Zimbabwe, *Eur. J. Soil Sci.*, *61*, 721–733, doi:10.1111/j.1365-2389.2010.01266.

Mapanda, F., M. Wuta, J. Nyamangara, and R. M. Rees (2011), Effects of organic and mineral fertilizer nitrogen on greenhouse gas emissions and plant-captured carbon under maize cropping in Zimbabwe, *Plant Soil*, *343*, 67–81, doi:10.1002/2016JG003341/pdf.

Millar, N., J. K. Ndufa, G. Cadisch, and E. M. Baggs (2004), Nitrous oxide emissions following incorporation of improved-fallow residues in the humid tropics, *Global Biogeochem. Cycles*, *18*, 1–9, doi:10.1029/2003GB002114/pdf.

Nsabimana, D., L. Klemedtson, B. A. Kaplin, and G. Wallin (2009), Soil $CO_2$ flux in six monospecific forest plantations in Southern Rwanda, *Soil Biol. Biochem.*, *41*, 396–402, doi:10.1002/2016JG003341/pdf.

Okalebo, J. R., K. W. Gathua, and P. L. Woomer (2002), *Laboratory Methods of Soil and Plant Analysis: A Working Manual*, Tropical Soil Biology and Fertility, Soil Science Society of East Africa.

Owen, J. J., and W. L. Silver (2014), Greenhouse gas emissions from dairy manure management: A review of field-based studies, *Global Change Biol.*, *21*, 550–565, doi:10.1002/2016JG003341/pdf.

Parkin, T. B., and R. T. Venterea (2010), Sampling protocols chapter 3. Chamber-based trace gas flux measurements, in *Sampling Protocols*, edited by R. F. Follett, U.S. Dep. of Agric.-Agric. Res. Serv. [Available at https://www.ars.usda.gov/SP2UserFiles/Program/212/Chapter%203.%20GRACEnet%20Trace%20Gas%20Sampling%20Protocols.pdf, Accessed 23 May 2016.]

Parkin, T. B., R. T. Venterea, and S. K. Hargreaves (2012), Calculating the detection limits of chamber-based soil greenhouse gas flux measurements, *J. Environ. Qual.*, *41*, 705–715, doi:10.2134/jeq2011.0394.

Pelster, D. E., M. C. Rufino, T. S. Rosenstock, J. Mango, G. Saiz, E. Diaz-Pines, G. Baldi, and K. Butterbach-Bahl (2015), Smallholder African farms in western Kenya have limited greenhouse gas fluxes, *Biogeosciences Discuss.*, *12*, 15,301–15,336, doi:10.1002/2016JG003341/pdf.

Predotova, M., E. Schlecht, and A. Buerkert (2009), Nitrogen and carbon losses from dung storage in urban gardens of Niamey, Niger, *Nutr. Cycl. Agroecosystems*, *87*, 103–114, doi:10.1007/s10705-009-9316-1.

Predotova, M., J. Gebauer, R. V. C. Diogo, E. Schlecht, and A. Buerkert (2010), Emissions of ammonia, nitrous oxide and carbon dioxide from urban gardens in Niamey, Niger, *F. Crop. Res.*, *115*, 1–8, doi:10.1002/2016JG003341/pdf.

Rashti, M. R., W. Wang, P. Moody, C. Chen, and H. Ghadiri (2015), Fertiliser-induced nitrous oxide emissions from vegetable production in the world and the regulating factors: A review, *Atmoph. Environ.*, *112*, 225–233, doi:10.1002/2016JG003341/pdf.

Rees, R. M., M. Wuta, P. A. Furley, and C. Li (2006), Nitrous oxide fluxes from savanna (miombo) woodlands in Zimbabwe, *J. Biogeogr.*, *33*(3), 424–437, doi:10.1111/j.1365-2699.2005.01423.

Rosenstock, T. S., M. C. Rufino, K. Butterbach-Bahl, and E. Wollenberg (2013), Toward a protocol for quantifying the greenhouse gas balance and identifying mitigation options in smallholder farming systems, *Environ. Res. Lett.*, *8*, 021003, doi:10.1002/2016JG003341/pdf.

Rosenstock, T. S., M. Mpanda, J. Rioux, E. Aynekulu, A. A. Kimaro, H. Neufeldt, K. D. Shepherd, and E. Luedeling (2014a), Targeting conservation agriculture in the context of livelihoods and landscapes, *Agric. Ecosyst. Environ.*, *187*, 47–51, doi:10.1002/2016JG003341/pdf.

Rosenstock, T. S., M. Thiong'o, P. Mutuo, and S. Abwanda (2014b), FAO mitigation of climate change in agriculture programme data. [Available at http://thedata.harvard.edu/dvn/dv/icraf/faces/study/StudyPage.xhtml?globalId=doi:10.7910/DVN/24343&studyListingIndex=2_130ea5202e259be6beef3dafe879, Accessed 1 December 2014.]

Shcherbak, I., N. Millar, and G. P. Robertson (2014), Global metaanalysis of the nonlinear response of soil nitrous oxide ($N_2O$) emissions to fertilizer nitrogen, *Proc. Natl. Acad. Sci. U.S.A.*, *111*, 9199–9204, doi:10.1002/2016JG003341/pdf.

Smith, P., et al. (2008), Greenhouse gas mitigation in agriculture, *Philos. Trans. R. Soc. London B Biol. Sci.*, *363*, 789–813, doi:10.1002/2016JG003341/pdf.

Snyder, C. S., T. W. Bruulsema, T. L. Jensen, and P. E. Fixen (2009), Review of greenhouse gas emissions from crop production systems and fertilizer management effects, *Agric. Ecos. Environ.*, *133*, 247–266, doi:10.1002/2016JG003341/pdf.

Stehfest, E., and L. Bouwman (2006), $N_2O$ and NO emission from agricultural fields and soils under natural vegetation: Summarizing available measurement data and modeling of global annual emissions, *Nutr. Cycl. Agroecosystems*, *74*, 207–228, doi:10.1007/s10705-006-9000-7.

Thornton, P. K., and M. Herrero (2010), Potential for reduced methane and carbon dioxide emissions from livestock and pasture management in the tropics, *Proc. Natl. Acad. Sci.*, *107*, 19,667–19,672, doi:10.1002/2016JG003341/pdf.

Tilman, D., and M. Clark (2014), Global diets link environmental sustainability and human health, *Nature*, *515*, 518–522, doi:10.1038/nature13959.

Tubiello, F. N., M. Salvatore, R. Condor, A. Ferrara, S. Rossi, S. Federici, H. Jacobs, and A. Flammini (2014), Agriculture, forestry and other land use emissions by sources and removals by sinks 1990–2011 analysis, *FAO Stat. Div. Working Pap. Ser.*, Rome.

Vagen, T.-G., R. Lal, and B. R. Singh (2005), Soil carbon sequestration in sub-Saharan Africa: A review, *L. Derad. Dev.*, *16*, 53–71, doi:10.1088/1748-9326/8/1/015011.

van Groenigen, J. W., P. J. Kuikman, W. J. de Groot, and G. L. Velthof (2005), Nitrous oxide emission from urine-treated soil as influenced by urine composition and soil physical conditions, *Soil Biol. Biochem.*, *37*, 463–473.

Verchot, L. V., E. A. Davidson, J. H. Cattânio, and I. L. Ackerman (2000), Land-use change and biogeochemical controls of methane fluxes in soils of eastern Amazonia, *Ecosystems*, *3*, 41–56, doi:10.1029/2003GB002210/pdf.

Verchot, L. V., S. Brienzajunior, V. Deoliveira, J. Mutegi, J. H. Cattânio, and E. A. Davidson (2008), Fluxes of $CH_4$, $CO_2$, NO, and $N_2O$ in an improved fallow agroforestry system in eastern Amazonia, *Agric. Ecosyst. Environ.*, *126*, 113–121, doi:10.1002/2016JG003341/pdf.

Werner, C., R. Kiese, and K. Butterbach-Bahl (2007), Soil-atmosphere exchange of $N_2O$, $CH_4$, and $CO_2$ and controlling environmental factors for tropical rain forest sites in western Kenya, *J. Geophys. Res.*, *112*, D03308, doi:10.1029/2006JD007388.

Zagst, L. (2011), Socio-economic Survey CARE-MICCA Pilot Project in the United Republic of Tanzania, Rome.

Zepp, R., and W. Miller (1996), Effects of moisture and burning on soil-atmosphere exchange of trace carbon gases in a southern African savanna, *J. Geophys. Res.*, *101*, 23,699–23,706, doi:10.1029/95JD01371.