Smallholder farms in east African tropical highlands have low soil greenhouse gas fluxes

Pelster, D.E.[1,*], M.C. Rufino[2], T. Rosenstock[3], J. Mango[3], G. Saiz[4], E. Diaz-Pines[4], G. Baldi[5] and K. Butterbach-Bahl[1,4]

[1] International Livestock Research Institute (ILRI), PO Box 30709, Nairobi, Kenya

[2] Centre for International Forestry Research (CIFOR), PO Box 30677-00100, UN Avenue, Nairobi Kenya

[3] World Agroforestry Centre (ICRAF), PO Box 30677-00100, UN Avenue, Nairobi, Kenya

[4] Karlsruhe Institute of Technology – Institute of Meteorology and Climate Research, Atmospheric Environmental Research (KIT/IMK-IFU) Kreuzeckbahnstr. 19, 82467 Garmisch-Partenkirchen, Germany

[5] Grupo de Estudios Ambientales - IMASL, Universidad Nacional de San Luis & CONICET. Ejército de los Andes 950, D5700HHW. San Luis, Argentina

[*] corresponding author: email – d.pelster@cgiar.org

Abstract:

Few field studies examine greenhouse gas (GHG) emissions from African agricultural systems resulting in high uncertainty for national inventories. This lack of data is particularly noticeable in smallholder farms in sub-Saharan Africa, where low inputs are often correlated with low yields, often resulting in food insecurity as well. We provide here the most comprehensive study in Africa to date, examining annual soil CO_2, CH_4 and N_2O emissions from 59 smallholder plots, across different vegetation types, field types and land classes in western Kenya. The study area consists of a lowland area (approximately 1 200 m asl) rising approximately 600 m to a highland plateau. Cumulative annual fluxes ranged from 2.8 to 15.0 Mg CO_2-C ha^{-1}, -6.0 to 2.4 kg CH_4-C ha^{-1} and -0.1 to 1.8 kg N_2O-N ha^{-1}. Management intensity of the plots did not result in differences in annual fluxes for the GHGs measured (P = 0.46, 0.67 and 0.14 for CO_2, N_2O and CH_4 respectively). The similar emissions were likely related to low fertilizer input rates (≤ 20 kg N ha^{-1}). Grazing plots had the highest CO_2 fluxes (P = 0.005); treed plots (plantations) were a larger CH_4 sink than grazing plots (P = 0.05); while N_2O emissions were similar across vegetation types (P = 0.59). This case study is likely representative for low fertilizer input, smallholder systems across sub-Saharan Africa, providing critical data for estimating regional or continental GHG inventories. Low crop yields, likely due to low inputs, resulted in high (up to 67 g N_2O-N kg^{-1} aboveground N uptake) yield-scaled emissions. Improving crop production through intensification of agricultural production (i.e. water and nutrient management) may be an important tool to mitigate the impact of African agriculture on climate change.

1 Introduction:

Increased atmospheric concentrations of greenhouse gases (GHG: CO_2, N_2O and CH_4) over the last century have been correlated to increasing mean global temperature (IPCC, 2013), while N_2O is also the primary ozone-depleting anthropogenically emitted gas (Ravishankara et al., 2009). Globally, agriculture is directly responsible for approximately 14% of anthropogenic GHG emissions while indirect emissions due to conversion of natural landscapes to agricultural systems may contribute an additional 17% (Vermeulen et al., 2012). In less developed countries however, agriculture can account for up to 66% of a country or region's total GHG emission (Tubiello et al., 2014), with African GHG emissions from agriculture and other land uses estimated to be 61% of total continental GHG emissions (Valentini et al., 2014).

In parts of the developing world, such as Sub-Saharan Africa (SSA), smallholder farms (farm size < 10 ha) comprise almost 80% of farmland and up to 90% of the farms (Altieri and Koohafkan, 2008). Thus it is likely that smallholder farms have a large effect on the GHG inventories of many Sub-Saharan countries. Unfortunately, there is a dearth of knowledge on agricultural soil GHG emissions from smallholder systems as only a handful of empirical studies (see Table 1) have measured these (e.g. (Baggs et al., 2006; Brümmer et al., 2008; Dick et al., 2006; Predotova et al., 2010). Previous studies in Africa were also limited in scope; measuring emissions from a low number of sites (generally less than 10) for a short time period (i.e. less than one year), often with low temporal resolution. This shortage of baseline data makes it impossible for many developing countries to accurately assess emissions from soils used for agriculture or to use Tier II methodology, which requires the development and documentation of country or regionally specific emission factors, to calculate GHG inventories (IPCC, 2006). Also, Tier 1 methodology assumes a linear response to fertilizer, which may not accurately reflect emissions in low input systems (Shcherbak et al., 2014). Finally, because most of the research behind the development of the Tier I methodology has been completed in temperate zones, the differences in climate, soils, farm management and nutrient balances (Vitousek et al., 2009) seem to result in consistent overestimates of fluxes (Hickman et al., 2014; Rosenstock et al., 2013b) likely translate to inflated national agricultural GHG inventories in Africa that may result in incorrect targeting and inefficient mitigation measures.

Soil greenhouse gas emission rates have been related to soil properties such as pH, organic carbon (SOC) content or texture (Khan et al., 2011, Chantigny et al., 2010, Rochette et al., 2008, Stehfest and Bouwman, 2006), but also to vegetation (crop) type (Stehfest and Bouwman, 2006) and management operations e.g. tillage, fertilizer type or crop rotation, (Baggs et al., 2006; Drury et al., 2006; Grageda-Cabrera et al., 2004; Halvorson et al., 2008; Yamulki and Jarvis, 2002). In contrast to agricultural systems in most OECD (Organisation for Economic Co-operation and Development) states, smallholder farmers differentially allocate resources based on distance from homestead and perceived soil fertility, specifically manure and fertilizer applications, to their fields resulting in strong gradients in soil fertility (Tittonell et al., 2013). The differences in soil fertility can be predicted using remote sensing tools like “Normalized Difference Vegetation Index” (NDVI) to determine the magnitude and temporal variability of primary productivity (Paruelo et al., 2001). Differences in fertility can also be predicted using farmer questionnaires to determine how farmers allocate resources to the fields and then using this typology of farming activities (hereafter “field typology”) to estimate where soil GHG fluxes would be high. If strong correlations can be demonstrated such fertility gradients may then be upscaled based on either the NDVI or farmer interviews that could allow for effective landscape level predictions based on the field-scale measurements.

The lack of good information on GHG fluxes related to agricultural activities in Africa and specifically on smallholder farming systems is a large data gap that needs to be addressed. The objectives of this study were to gather greenhouse gas flux data from smallholder farms of the western Kenyan Highlands that represent both the diversity in farming practices and landscape heterogeneity typically found for many highland regions in East Africa. We hypothesized that a) in view of low rates of fertilizer applications by smallholders the GHG fluxes are generally at the low end of published fluxes from agricultural land, b) the seasonality of hygric seasons is mirrored by fluxes and c) differences in land productivity as reflected by NDVI and field typology, as well as differences in vegetation can be used to explain spatial variability in field-scale soil greenhouse gas fluxes.

2 Materials and Methods

The study site was a 10 km x 10 km landscape in Kisumu county of Western Kenya (centered at 35.023E, 0.315S); just north of the town of Sondu (Fig. 1), and ranges from a lowland area at approximately 1200 m asl to a highland plateau at approximately 1800 m asl. The site is one of the sentinel sites for the CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS) and is described in much more detail in Sijmons et al. (2013). This site was selected as it was found to be broadly similar in terms of demographics (population density, income, etc) and agro-ecological characteristics (e.g. elevation, temperature, precipitation etc) of other East African tropical highlands (Braun et al., 1997) allowing us to scale up the results to other countries in the region (Sijmons et al., 2013). Mean annual temperature is approximately 23°C and an average annual rainfall is 1150 mm (Köppen classification of a tropical savanna climate [AW]). Temperatures tend to be slightly cooler and precipitation slightly higher in the highlands compared to the lower regions of the study site. Precipitation patterns are bimodal with the "long rains" occurring from April to June (42% of annual precipitation) and the "short rains" occurring from October through December (26% of annual precipitation). The site is primarily composed of smallholder farms typically growing maize (*Zea mays*) and sorghum (*Sorghum bicolor*) during the long rains and beans during the short rains. Based on farmer interviews, approximately 27% of them applied fertilizers (i.e. manure or synthetic fertilizers) to their plots; with application rates being very low. For manure, application rates were approximately 200 kg manure ha^{-1}, which would correspond to approximately 95 kg of C and 5 kg N given typical N contents for cattle in this region (Pelster et al., 2016), while application rates for synthetic fertilizer (two farmers applied diammonium phosphate and one applied urea) were < 50 kg fertilizer ha^{-1} (< 25 kg N ha^{-1}). These fertilizer rates are much lower than rates typical for industrial production where application rates often exceed 150 kg N ha^{-1} for maize production.

Soil types in the study area are highly heterogeneous, ranging from well drained, acidic, nitisols in the upper part of the landscape, to eutric and dystric cambisols in mid-altitude areas and poorly drained planosols in the lower parts (IUSS Working Group WRB, 2015). Selected topsoil characteristics for the different land classes identified in the study region are provided in Table 2.

2.1 Landscape stratification

Differences in management intensity and vegetation were expected to affect GHG fluxes, and so the landscape was stratified to account for the expected variability. The stratification was based on a mixed method landuse classification combining remote sensing and household surveys. For the land classification we followed an approach based on vegetation functioning in terms of the magnitude and the temporal variability of primary productivity (Paruelo et al., 2001). Vegetation primary productivity was assessed through the proxy variable "Normalized Difference Vegetation Index" (NDVI), which allows approximate but widespread characterizations of productivity across space and time and across different ecosystems (Lloyd, 1990; Xiao et al., 2004). We acquired 2001-2012 NDVI data from MODIS (Moderate Resolution Imaging Spectroradiometer). After obtaining the data we selected only those values indicating good to excellent quality conditions (i.e. pixels not covered by clouds, and with a low to intermediate aerosol contamination). Then, we used the program TIMESAT v.3.1. to reconstruct temporal series (Jönsson and Eklundh, 2002).

From the reconstructed temporal series we assessed six functional metrics depicting the magnitude, seasonality and inter-annual variability of productivity. The metrics used were as follows: 1) the mean annual NDVI; 2) the minimum NDVI; 3) the browning rate (rate of NDVI decrease); 4) the peakness of the NDVI; 5) the intra-annual coefficient of variation (CV) of the NDVI; and 6) the inter-annual CV. These metrics allow us to differentiate between land cover types (e.g. cultivated vs. uncultivated) and between different cultivation management approaches (e.g. agroindustrial vs. subsistence) (Baldi et al., 2015). The different elevation bands and soil types resulted in different magnitudes, seasonality and inter-annual variability of productivity with the highlands generally having higher productivity due to the higher rainfall and more fertile soils. We then ran an ISODATA unsupervised classification algorithm (Jensen, 1996), and the resulting spectral classes were aggregated to create patches. After combining minor or sparsely-distributed patches, we ended up with 5 classes, characterized by the following features: 1) lowland subsistence farms with degradation signs (N = 7); 2) lower slopes, moderate sized mixed farms (N = 8); 3) mid slopes, moderate sized, primarily grazing / shrubland (N = 10); 4) upper slopes / highland plateau, mixed farms (N = 22); and 5) mid slopes, moderate sized mixed farms (N = 12).

We also stratified the plots by field typology using the following variables to define a field type score: 1) crop: this score is the sum of the crops each household is cultivating in one plot; 2) fertilizer use: this score distinguishes organic and inorganic fertilizers; 3) number of subplots: which allows us to capture the spatial and temporal allocation of land to crops, crop mixtures, and combination of annual and perennial crops in intercropping, permanent and seasonal grazing land; 4) location of field: the assumption being that fields close to the homestead receive preferential land management (fertilization, addition of organic amendments, weeding etc) when compared to fields that are far away (Tittonell et al., 2013); and 5) Signs of erosion: fields differing in visible sign of erosion obtained a different score depending on the severity. Plots were scored based on the preceding information and those with a higher score were considered field type 1 (N = 17), those with a low score were considered field type 3 (N = 19) and those intermediate plots were assigned a field type 2 (N = 23). It was assumed that field type 1 was the most highly managed (i.e. more fertilizer /manure additions resulting in higher soil C, etc) and field type 3 the least managed (i.e. none to very low fertilizer additions, degraded, low soil C, etc). For a more detailed description of the stratification process see Rufino et al (2015).

Finally, the plots were also stratified by vegetation (cover) type: treed/bush (generally plantations of either *Grevillia spp* or *Eucalyptus spp*) (N = 7), perennial grasses/grazing (N = 15) and annual cropping (N = 37). Initially, the total number of sample plots was 60 with the number per category based partly on the area covered by each specific land classification/field type/vegetation type combination and partly on logistical constraints (i.e. access). One plot however, was converted into a construction site in late 2013, resulting in only 59 plots being measured for the full year.

2.2 Soil core incubation

A soil core incubation study was conducted to compare the effects of the different land-classes, field types and cover types on potential soil GHG fluxes; and to test if potentials of soil GHG fluxes under standardized conditions in the laboratory mirror differences in annual GHG fluxes at observation sites. Five soil cores were collected from 36 out of 59 plots using a 5 cm long PVC pipe (5.14 cm ID). The cores were left intact and taken back to the lab where they were air-dried (2 d at 30°C). One core from each plot was soaked overnight in water and then freely drained for 2-3 hours and then oven-dried (24h at 105°C) to determine maximum water-holding capacity (WHC). Three replicates of the air dried cores for each plot were then placed into a self-sealing 0.50 L glass jar fitted with a septum at 20°C. Air samples (10 mL) from each jar were collected at 0, 15, 30 and 45 min. The air samples were analyzed immediately for CO_2, CH_4 and N_2O by gas chromatography on an SRI 8610C gas chromatograph (9' Hayesep D column) fitted with a ^{63}Ni-electron capture detector for N_2O and a flame ionization detector for CH_4 and CO_2 (after passing the CO_2 through a methanizer). Flow rate for the carrier gas (pure N_2) was 20 mL min^{-1}. Every fifth sample analyzed on the gas chromatograph was a calibration gas (gases with known CO_2, CH_4 and N_2O concentrations in synthetic air) and the relation between the peak area from the calibration gas and its concentration was used to determine the CO_2, CH_4 and N_2O concentrations of the headspace samples. The soil cores were then brought to 25% WHC, left for one hour and then placed in the same jar and the headspace was again sampled and analyzed as above. This was sequentially repeated for the same cores at 35, 55 and 75% WHC. Soil re-wetting is known to result in a flush of nutrients (Birch, 1960) that tends to diminish with subsequent re-wettings. Therefore, for the subsequent re-wettings we also added a dilute KNO_3 solution (equivalent to adding 10 mg N kg^{-1} soil) to replace the N lost.

2.3 Field soil GHG flux survey

At the 59 identified field sites (see above and Fig 1) soil CO_2, N_2O and CH_4 fluxes were measured weekly starting the week of 12 August 2013 through to 12 August 2014 (one full year including two growing seasons) using non-flowthrough, non-steady state chambers (Rochette, 2011; Sapkota et al., 2014). Given the large number of plots and the difficult access, this required four 2-person crews sampling 4 days per week. Briefly, rectangular (0.35 m x 0.25 m) hard plastic frames were inserted 0.10 m into the ground. Fields planted with annual crops were ploughed, either using an oxen-pulled plough or by hand, twice during this period, which meant that the bases needed to be removed and then re-installed, however where possible the chamber bases were left undisturbed for the entire period. For fields planted with annual crops, the bases were installed between the rows and were weeded the same week the farmers weeded the rest of the field. The chambers in the grazing and treed plots would have included some vegetation (primarily grasses), but these were kept short (<5 cm long) by the continual grazing by livestock. On each sampling date, an opaque, vented and insulated lid (0.125 m height) covered with reflective tape was tightly fitted to the base (Rochette, 2011). The lid was also fitted with a small fan to ensure proper mixing of the headspace, and air samples (15 mL) were collected from the headspace at 0, 15, 30 and 45 min after deployment, using a syringe through a rubber septum. The air temperature inside the chambers increased during deployment, which may increase soil microbial activity that could cause an overestimate of the flux. Any increase in temperature inside the chambers would also cause some bias in the calculation of mixing ratios, which given the average change in temperature, we estimated this bias to be about 3%.

To increase the number of sites measured while still accounting for the representativeness of flux measurements in view of expected high spatial variability of fluxes at field scale samples were pooled from four replicate chambers (Arias-Navarro et al., 2013) to form a composite air sample of 60 mL. This method has been found to provide flux estimates within 8% and 4% (for CO_2 and N_2O respectively) of the estimates calculated by separate sampling, although it is unclear which is the more accurate depiction of the true mean. Also, as noted by Arias-Navarro et al. (2013), this precludes the ability to examine on-site variability, however we believed that given the limitations in our sampling and analytic capacity that the trade-off between on-site variability and sampling a broader range of sites was worthwhile given our aims of characterizing emissions in a way that captured both the diversity in farming practices and landscape heterogeneity typically found for many highland regions in East Africa. The first 40 mL of the sample was used to flush 10 mL sealed glass vials through a rubber septum, while the final 20 mL was transferred into the vial to achieve an over-pressure to minimize the risk of contamination by ambient air. The gas samples were analyzed within 10 d of sample collection as described for the soil cores above.

2.4 Calculation of soil GHG fluxes

Soil fluxes were calculated by the rate of change in concentration over time in the chamber headspace (corrected for mean chamber temperature and air pressure) for both the soil core incubation and the field survey, as shown in Equation 1.

Equation 1. $$F_{GHG} = (\partial c/ \partial t)*(M/V_m)*(V/A)$$

Where, F_{GHG} is the flux of the GHG in question, $\partial c/\partial t$ is the change in concentration over time, M is the molar mass of the element in question (N for N_2O and C for CO_2 and CH_4), Vm is the molar volume of gas at the sampling temperature and atmospheric pressure, V is the volume of the chamber and A is the area covered by the chamber.

We validated the data for each chamber/incubation jar measurement by examining the CO_2 concentrations over the 45 minutes. Chambers that experienced a decrease in CO_2 greater than 10% between any of the measurement times were assumed to have a leak and all GHG fluxes were discarded unless the decrease occurred in the last measurement; in this case, the flux rate was calculated with the first three measurement points. In cases where the change in concentration was lower than the precision of the instrument, we assumed zero flux. The minimum flux detection limits (as per Parkin et al. 2012) were 3.61 and 12.46 μg N_2O-N m^{-2} hr^{-1} for the linear and non-linear models respectively and 0.015 and 0.051 mg CH_4-C m^{-2} hr^{-1} for the linear and non-linear models respectively. Also, negative fluxes for CO_2, while negative CH_4 and N_2O fluxes were accepted, as uptake of either in upland soils is feasible. In general, non-linear models are less biased than linear models however they also tend to be very sensitive to outliers (Rochette, 2011). Therefore, when there was a strong correlation for the non-linear model ($R^2 > 0.95$) we used a second-order polynomial; otherwise, we used a linear model. See Rochette and Bertrand (2008) for details on these models. If however the $R^2 < 0.95$ for the non-linear model and <0.64 for the linear model, we assumed there was no valid flux measurement and the data point was thrown out. To minimize measurement error and uncertainty, we used methods that were ranked as either "good" or "very good" for 15 of the 16 criteria selected by Rochette and Eriksen-Hamel (2008), with only the deployment time exceeding the recommended time by about 10%. Cumulative annual fluxes were estimated for the field plots using trapezoidal integration between sampling dates.

2.5 Soil analysis

At the beginning of the experiment and for each sampled site, five replicate soil samples were taken both at 0-5 cm and 5-20 cm depths with the aid of a stainless steel corer (40 mm inner diameter). Samples were individually placed in labelled zip-lock bags. All soil material was oven-dried at 40°C for a week with large clumps being progressively broken by hand. Carbon and nitrogen concentrations were determined on micro-milled powdered samples using an elemental combustion system (Costech International S.p.A., Milano, Italy) fitted with a zero-blank auto-sampler. Soil pH was measured in a 2:1 water:soil solution. Soil texture was determined gravimetrically as described by (van Reeuwijk, 2002).

In addition soil samples were collected periodically (every 2 months) for determination of inorganic N concentrations. Briefly, the topsoil (0-10 cm depth) was collected using a soil auger. Three samples from each plot were collected and placed into a plastic self-locking bag to form one composite sample. These were taken back to the lab and stored (4° C) for less than one week before extraction (1:5

soil:solution w:v ratio) with 2*M* KCl. Extracts were kept frozen until analyzed. Analysis for NO_3-N was done via reduction with vanadium, development of colour (540 nm) using sulfanilic acid and naphtylethylendiamin and measurement of adsorption of light on an Epoch microplate spectrophotometer (BioTek, Winooski, VT, USA). The NH_4-N concentrations were measured using the green indophenol method (660 nm) using the same spectrophotomer (Bolleter et al., 1961).

2.6 Environmental data

Environmental data were collected at two sites, one in the uplands (S 0.35156°, E 35.05590°, 1676 m asl) and the other in the lowlands (S 0.30847°, E 34.98769°, 1226 m asl). Each of the two weather stations was installed at a farm where we also measured gas emissions. Air temperature was measured using a Decagon ECT air temperature sensor (measurement every 5 minutes), while precipitation data were collected with a Decagon ECRN-100 high resolution, double-spoon tipping bucket rain gauge. Soil moisture and temperature were measured using a Decagon MPS-2 Water potential and temperature sensor (Decagon Devices, Pullman, WA, USA). Data were logged on a Decagon Em50 data collection system and downloaded periodically (typically monthly). Also, air temperature, soil temperature and soil moisture (5 cm depth) were measured at each site, at the time of gas sampling using a ProCheck handheld datalogger outfitted with a GS3 sensor (Decagon Devices, Pullman, WA, USA).

2.7 Plant production

To estimate crop yields and crop N content of annual crops in the region, we randomly selected 9 of the annual cropping plots (4 plots with maize, 4 with sorghum and 1 with green grams [*Vigna radiata*]) where we measured gas fluxes. In June 2013, all the plants within a 2.5 m x 2.5 m square near the center of the field (i.e. to avoid edge effects) were harvested and the grains were removed from the plant; both the stover and grains were dried for 48 hours at 60°C and then weighed. A sub-sample of the grains was then ground and analyzed for C and N content on the same elemental combustion system (Costech International S.p.A., Milano, Italy) described above for soil analysis. Yield-scaled GHG emissions (g N_2O-N kg^{-1} above ground N uptake) were calculated for each site by dividing the cumulative emissions for the growing season by the grain yields. The growing season lasted from mid-March to August, which corresponds to the period between preparation of fields for the long rains through harvest and up to the preparation of the fields for the second growing season. No estimate of crop yields (or yield-scaled emissions) was done for the second growing season.

2.8 Statistical analysis

For the soil core incubation study, the flux rates for CH_4, CO_2 and N_2O were compared using ANOVA (AOV in RStudio v. 0.98.953), using the WHC as blocks and cover type, land class, and field type as fixed factors. Because of the imbalanced design, we could not analyze interactions as several combinations had an insufficient number of samples so each of the factors was analyzed independently of the others. When $P < 0.1$, differences between treatments were analyzed using Tukey's HSD. Correlations between maximum flux rates for the intact soil core incubations and total cumulative fluxes for the field measurements were tested using Spearman Rank Correlation, while correlations between GHG fluxes and soil properties were tested using Pearson Correlation. The cumulative field fluxes for a 4-week period during the dry season were compared to cumulative fluxes for a 4-week period during the rainy season using ANOVA, with the season, management practices (ploughed versus not ploughed for CO_2 and fertilized versus not fertilized for N_2O) as fixed factors along with the two-way interaction terms. Cumulative field annual fluxes were compared with ANOVA using an un-balanced design and cover type, land class and field type as fixed factors. In all cases, the distributions of flux measurements were tested for normality using Shapiro-Wilks. Only cumulative N_2O fluxes were not normally distributed and were transformed using the natural log.

3 Results

3.1 Soil core incubation

For the laboratory incubations, there was very little CO_2 efflux (maximum of 7.5 mg CO_2-C m^{-2} h^{-1}) when the soils were air-dried, with increased soil respiration only at higher water contents (Fig. 2). For the five investigated soil moisture levels (air dried, 25, 35, 55 and 75% WHC) soil respiration tended to be highest at 55% WHC (Figs. 2, 3 and 4) and was positively correlated with the soil C and N content (r=0.33, P = 0.005 and r=0.35, P =0.003 respectively). The N_2O fluxes were very low when the water content was less than or equal to 35% WHC and increased exponentially when the water content was increased to 55 and 75% (Fig. 2) and were also positively correlated with total C and N (r = 0.24, P = 0.043 and r = 0.31, P = 0.010 respectively). The soil CH_4 fluxes (mostly uptake) were generally low, ranging from -20 to 20 µg CH_4-C m^{-2} h^{-1} and unlike the previous two GHGs, there were similar flux rates between the three moderate water contents, while there were much lower fluxes at the lowest and highest water contents (Fig 2). Unlike N_2O and CO_2 fluxes, CH_4 fluxes were not correlated with soil C and N contents.

Both the CO_2 and the N_2O fluxes differed by land class (P = 0.001 and 0.061 respectively) with land class 1 (lowland farms with degraded soils) having lower CO_2 fluxes than classes 4 (mid-slope farms and shrub land) and 5 (lowland pasture), while landclass 4 had higher N_2O fluxes than either class 1 or 2 (highland farms) (Fig. 2). As shown in Table 2, land class 1 and 2 also had the lowest soil C and N contents. Grass and grazing plots emitted more CO_2 than annual plots (P = 0.069), while there were no detectable differences in N_2O or CH_4 fluxes between vegetation types (P = 0.603 and 0.457 respectively). Field type had no detectable difference on CO_2, N_2O or CH_4 fluxes (P = 0.179, 0.109, and 0.198 respectively).

3.2 Field meteorological and site observations

For the *in situ* experiments, the soils were slightly acidic to circum-neutral, ranging in pH from 4.4 to 7.5 (mean = 6.0), with C and N contents ranging from 0.7 to 4.0% (mean = 2.2) and 0.07 to 0.33% (mean = 0.17) respectively (Table 2). The C/N ratio ranged from 7.7 to 18.1 (mean = 12.6) while the C and N contents in the top 20 cm of soil were highly correlated with each other (R = 0.976; P < 0.0001). Annual precipitation (15 August 2013 through 14 August 2014) in the lowlands was 1127 mm while there was 1417 mm of precipitation in the highlands, a 25% increase across the 450 m elevation difference between the two stations. The average minimum and maximum daily temperatures in the lowlands were 15.6 and 30.5°C while temperatures were slightly cooler in the highlands, with an average minimum of 12.6 and an average maximum of 26.9°C. Comparing the precipitation at the sites to a long-term 40-year (1960 to 2000) precipitation data set for the two nearby towns of Kisumu and Kericho (data available at africaopendata.org), we see that annual precipitation was within 10% of the long term average. The monthly rainfalls as well were generally similar to long-term trends as well, with the exception of the rainfall in December, which was 26% of the long-term average, and the rainfall in March, which was 2.4 x the long-term mean

3.3 Field scale soil GHG fluxes and ancillary information

Soil CO_2 fluxes during August 2013 ranged from 50 to 200 mg CO_2-C m^{-2} h^{-1}, slowly decreased through to November and remained low (< 100 mg CO_2-C m^{-2} h^{-1}) until the onset of the long rains during March/April 2014 (Fig. 3). The onset of the long rains increased the soil water content from an average of 0.09 m^3 m^{-3} for the week of 3 March 2014 to an average of 0.31 m^3 m^{-3} by 17 March 2014. Within two weeks of this increase in soil moisture, the CO_2 fluxes began to increase, reaching a maximum on 14 April 2014 (mean = 189 mg CO_2-C m^{-2} h^{-1}; Fig. 3).

In general, soil CH_4 fluxes were negative indicating net uptake. Uptake rates tended to stay between 0 and 100 µg CH_4-C m^{-2} h^{-1} from August 2013 until April 2014, after which the variability decreased varying between 0 and 50 µg CH_4-C m^{-2} h^{-1} (Fig. 3). Soil N_2O fluxes were low (generally < 10 µg N_2O-N m^{-2} h^{-1}) for most of the year; with fluxes increasing from a mean of 1.6 µg N_2O-N m^{-2} h^{-1} for the period from October 2013 to March 2014 to a mean of 10.5 µg N_2O-N m^{-2} h^{-1} for the 6-week period just after soil re-wetting in March/April 2014. The inorganic N concentrations in the top 10 cm of soil (approximately 85% N-NO_3 and 15% N-NH_4) generally remained below 20 mg N kg^{-1} soil, although concentrations did increase to around 30 mg N kg^{-1} soil in late December 2013 / early January 2014, shortly after the annual crops planted during the short rains were harvested but before the onset of the long rains in late March / early April 2014.

A comparison of the cumulative fluxes from four weeks in February (end of the dry season) to four weeks in April (immediately following the start of the rainy season) shows greater cumulative CO_2 and N_2O fluxes during the wet season, but no difference in CH_4 fluxes (Table 3). This increase in CO_2 and N_2O fluxes during the onset of the long rains coincided with farmers ploughing their fields and planting and fertilizing their annual crops. However, even though the increase in CO_2 and N_2O fluxes was slightly larger in the managed plots (ploughed for CO_2 and fertilized for N_2O comparisons), neither of these management interventions significantly altered emission rates (Table 3).

Cumulative annual fluxes ranged from 2.8 to 15.0 Mg CO_2-C ha^{-1}, -6.0 to 2.4 kg CH_4-C ha^{-1} and -0.1 to 1.8 kg N_2O-N ha^{-1}. There was no detectable effect on cumulative CO_2 fluxes by field type or land class (P = 0.46 and 0.19 respectively; Fig. 4), although grazed plots emitted more CO_2 than either annual cropland or treed plots (P = 0.005). Cumulative annual N_2O fluxes also did not differ by either field type or vegetation type (P = 0.67 and 0.59 respectively; Fig. 4), however land class did significantly affect N_2O fluxes (P = 0.09; Fig 4) with the flux from land class 3 (mid-slopes, grazing) higher than the flux from land class 4 (upper slopes, mixed farms). Cumulative annual CH_4 fluxes were predominately negative, indicating CH_4 uptake. Cumulative CH_4 uptake rates, unlike N_2O and CO_2, varied by land class (P = 0.01) and land cover type (P = 0.01), but not by field type (P = 0.16; Fig. 4). Uptake of atmospheric CH_4 by soils was greatest in land class 2 (lower slopes, degraded), greater than classes 1 (lowland farms with degraded soils) or 3 (mid-slopes grazing land; Fig. 4). Uptake was also almost 3x greater in treed plots versus those plots with grasses and or those used for grazing (Fig. 4). The difference seems to be primarily due to one grazing plot that was a CH_4 source for 14 of 24 sampling dates (sink for only 4 of 24 sampling dates) between 5 August 2013 and 10 February 2014. This same plot also had the second highest cumulative N_2O fluxes (1.5 kg N_2O-N ha^{-1} yr^{-1}), however the CO_2 fluxes were average (7.2 Mg CO_2-C ha^{-1} yr^{-1}) and the soil organic C and N contents were relatively low (1.2 and 0.10% for C and N respectively) compared to the rest of the plots (Table 2).

Both the soil C and N content were correlated with cumulative CO_2 fluxes (r = 0.411; $P = 0.002$ and r = 0.435; $P < 0.001$, for C and N content respectively). However, the C and N content were not correlated with either the cumulative N_2O fluxes ($P = 0.321$ and 0.365 for C and N respectively) or the cumulative CH_4 fluxes ($P = 0.188$ and 0.312 for C and N respectively). The cumulative CO_2 and N_2O fluxes were also not correlated ($P = 0.188$)

Many of the farmers within the study site complained that the annual crops planted in March 2013 failed due to the poor timing of the rains. Within the 9 fields that we measured, the crop yields ranged from 100 to 300 kg ha^{-1} for maize (n = 4), from 140 to 740 kg ha^{-1} for sorghum (n = 4) and were approximately 20 kg ha^{-1} for green grams (n = 1) during the long rain season (March through June). The low yields resulted in yield-scaled soil N_2O fluxes of up to 67 g N_2O-N kg^{-1} aboveground N uptake.

The maximum N_2O fluxes as observed within our soil core study were correlated with the cumulative annual fluxes as observed at the field sites ($\rho = 0.399$, $P = 0.040$), while CO_2 fluxes followed a similar trend ($\rho = 0.349$, $P = 0.075$), however the CH_4 fluxes from the soil cores were not correlated with measured flux at the field sites ($\rho = -0.145$, $P = 0.471$).

4 Discussion

The CO_2 fluxes were seasonal, and it was thought that management events, such as ploughing fields or fertilizer applications, would affect the flux rates throughout the year. However, during the commencement of the rainy season in March 2014, which coincided with tilling, the ploughed fields did not show significant increases in soil respiration rates beyond the enhancement in soil CO_2 flux due to re-wetting that was also measured in untilled fields. Increased soil respiration due to ploughing however are short-term, usually lasting less than 24 hours (Ellert and Janzen, 1999; Reicosky et al., 2005), so because the chambers needed to be removed before ploughing and were not re-installed until sites were re-visited a week later, the ploughing-induced increase in soil respiration was probably not fully captured. Also, root respiration, which at seeding accounts for 0% of soil CO_2 fluxes but can increase to around 45% of fluxes (Rochette et al., 1999), may also result in greater CO_2 fluxes during the growing season for the annual cropping systems. However, the increase in soil CO_2 fluxes from dry to growing season in annual crops was similar to the increase experienced in the other vegetation types (Table 3; P = 0.39). It is therefore likely that the low yields for the annual crops corresponded with poor root growth and low root respiration rates.

Soil CO_2 fluxes showed cumulative fluxes, (2.7 to 14.0 Mg CO_2-C ha^{-1} yr^{-1}), well within the range of other African studies (Table 1) and were not related to land class or field type, although the higher soil respiration rates from grazing land was inconsistent with a previous study that found similar rates between perennial tropical grasslands, croplands and tree plantations (Mapanda et al., 2010). However, because we did not differentiate between root and microbial respiration it could be that the continual vegetation cover in the grazing plots contributed more root respiration over the year than was found in the annual crops and treed plots.

Methane was generally taken up by these upland soils, however these rates also varied through the year (Fig. 5b). During August 2013, the soils were sinks for CH_4, however as the soils dried, the emission / uptake rates became more erratic until the long rains started again in late March 2014. The CH_4 flux at the soil surface is the result of the balance between production and consumption (Le Mer and Roger, 2001), so the low rates of atmospheric CH_4 uptake during the long rains may be caused by greater soil CH_4 production due to higher soil moisture and anaerobic conditions at depth (e.g. (Butterbach-Bahl and Papen, 2002) overriding the existing methanotropic activity or because the higher water content limited CH_4 diffusion from the atmosphere to the soils.

The CH_4 uptake from these sites were consistent with previous studies in upland agricultural soils and indicate that soils of smallholder farms are sinks for atmospheric CH_4 (Le Mer and Roger, 2001). There were no differences between field types, but regarding cover types, grazing plots took up less CH_4 than treed plots and land class 1 took up less than land class 2 (Fig. 4). The difference between cover types is consistent with previous studies that found that forest soils were greater CH_4 sinks than agricultural soils (MacDonald et al., 1996; Priemé and Christensen, 1999) and high degrees of degradation in land class 1 was likely responsible for reduced CH_4 oxidation rates

The N_2O flux rates remained below 20 μg m^{-2} h^{-1} with the exception of the onset of the rainy season in March 2014 (Fig. 4). According to Linn and Doran (1984) maximum aerobic activity occurs at approximately 60% water filled pore space (approximately 40% WHC for our study,), above which anaerobic processes such as denitrification can occur. The soils in the study area were typically drier than this threshold suggesting that N_2O fluxes were limited by a lack of anaerobic conditions and that the increase in soil water content was responsible for the increases in N_2O fluxes during March 2014. However, soil moisture was greater than 35% WHC during September/October 2013 and March 2014, but it was only in the latter period large increases in N_2O fluxes were observed. The high amounts of soil moisture in March coincided with an increase in inorganic N likely caused by drying and rewetting (Birch, 1960), which can also stimulate N_2O fluxes (Butterbach-Bahl et al., 2004; Davidson, 1992; Ruser et al., 2006). Commencement of the rainy season was also when farmers fertilized, although application rates were low (1-25 kg N ha^{-1}) and did not have a detectable effect on soil inorganic N concentrations, or N_2O emissions (Table 3).

The inability to discern between fertilized and unfertilized plots suggests that the differences in soil fertility and primary productivity were too low to have a noticeable effect on GHG emissions. Alternatively, it is possible that the sensitivity of the monitoring approach was not enough to catch differences between fields. For instance, the fixed sampling frequency may have caused us to miss some short-lasting emission peaks following fertilization, resulting in an underestimation of cumulative emissions. However, sampling during an emission pulse would result in an overestimate of emissions due to the extrapolation. Previous studies have found that weekly sampling resulted in an average uncertainty of ± 30% of the “best estimate” (Barton et al., 2015; Parkin, 2008) and that this uncertainty changes with the coefficient of variation in measured emission rates. However, the fertilizer was applied at a low rate (< 25 kg N ha^{-1}). Application of synthetic fertilizers up to 70 kg N ha^{-1} at planting in the region had no detectable effect on annual N_2O emissions (Hickman et al., 2015), while another nearby study found no effect of N fertilization on annual N_2O emissions (Rosenstock et al., 2016), suggesting that our weekly sampling did not miss relevant N_2O /GHG pulses.

There was a much larger response to re-wetting in land class 3 (mid-slopes, grazing land; Fig. 5) compared to land class 4 (upper slopes/plateau, mixed farms), which was primarily due to two (of 10) plots, both located on the same farm that emitted around 4 to 6 times more N_2O than the rest of the landclass 3 plots and 15 to 23 times more N_2O than the average for all other plots. The reason for the much higher fluxes after the re-wetting compared to other sites is not yet understood as the topsoil C and N contents were 1.45 and 0.12% respectively, well within the range of values for that land class (Table 2). The presence of livestock on these plots could have resulted in additions of N through either urine or manure deposition that we did not notice, causing these pulses of N_2O. However, the presence of N_2O emission hotspots in general is quite common as denitrification activity can vary dramatically across small scales (Parkin, 1987).

Annual N_2O fluxes were low (<0.6 kg N ha^{-1} y^{-1}) when compared with other tropical and sub-tropical studies, such as a fertilized field in Brazil (Piva et al., 2014) or China (Chen et al., 2000), with fluxes up to 4.3 kg N_2O-N ha^{-1} y^{-1}. However our results were similar to previous studies in low input African agro-ecosystems (Table 1). The low cumulative fluxes were most likely a result of low substrate (inorganic N) availability, in addition to low soil moisture limiting denitrification through much of the year. Similar to the CO_2 fluxes, the cumulative N_2O fluxes did not differ by cover type, field type or by land class. However, it is possible that differences between the classes could be too small to detect given the low cumulative N_2O fluxes, high microsite variability typical of N_2O fluxes (Parkin, 1987) and weekly sampling (Barton et al., 2015; Parkin, 2008).

There are additional sources of uncertainty associated with the sampling methods (chamber architecture, instrumentation sensitivity, etc). According to Levy et al. (2011), the uncertainty of the methods then would be about 20%, which when combined with the uncertainty around the weekly sampling would be about 50%. Although this may sound high, this is similar to the majority of other studies (e.g. see Helgason et al. (2005)) measuring GHG emissions and better than many of the studies so far in Africa (Table 1).

Soil core incubations do not reflect site conditions and should not be used to predict baseline emissions on the field. Still, the rankings for the maximum soil core N_2O and CO_2 fluxes were correlated with in-situ cumulative annual fluxes indicating that, they can be used as a quick and relatively inexpensive method to compare potential emissions from different land-uses that are not already well understood. On the contrary, 5 cm long soil cores were probably too short to properly capture the activity of methanotrophic bacteria (Butterbach-Bahl and Papen, 2002), which is a requisite to infer net CH_4 soil-atmosphere exchange rates.

Both the soil core incubations and field studies showed no detectable differences in GHG fluxes between the different field types, contrary to our expectations. We had anticipated differences in GHG fluxes because of differences among field types in input use, food production, partial N and C balances and soil fertility as previously reported in the region (Tittonell et al., 2013); and these variables often affect soil GHG fluxes (Buchkina et al., 2012; Jäger et al., 2011). We further hypothesized that land class and cover type would also have significant effects on soil CO_2 fluxes since a significant amount of the variability in soil CO_2 fluxes in agro-ecosystems can be explained by NDVI (Sánchez et al., 2003) and cover type (Mapanda et al., 2010), while differences in NDVI also indicate differences in primary productivity (Xiao et al., 2004). We found however no clear effect of field or land type on soil GHG fluxes.

Tittonell et al. (2013) reported important differences between field types only at each farm individually (Tittonell et al., 2013), which in our case, may have resulted in greater within-type variation that masked differences between the field types. Moreover, the small differences in the degree of inputs and labour may have not been enough to provoke distinct GHG fluxes, because the whole region/study site is characterized by low nutrient availability. For example, manure inputs have previously been found to increase soil C content (Maillard and Angers, 2014), but the inputs in our study area were very low (4-6 wheelbarrow loads or approximately 95 kg C ha^{-1}) and probably not enough to cause field-level differences. Further, considering that a previous study found that N is being rapidly mined from soils in the Lake Victoria basin (Zhou et al., 2014), it is likely that soil C is also being lost across the landscape. As most of this area has been converted from natural forests, and forests generally have higher SOC stocks than croplands (Guo and Gifford, 2002), time since conversion could play a larger part in determining the SOC content, which could mask any effects that management activities have on soil respiration rates in these low input systems.

Crop yields from the annual cropping systems (100 – 750 kg ha^{-1} for one growing season) were lower than the range (600 to 2800 kg ha^{-1}) for rain-fed smallholder farms previous reported across SSA (Sanchez et al., 2009). The farmers complained of poor timing of the rains that caused lower yields than normal. However, the results of the two studies suggest that low yields are common within this region. Increased nutrient inputs and water management are likely required to increase yields (Quiñones et al., 1997), which may result in increased GHG fluxes. However, it is expected that increases in GHG fluxes will be lower than the corresponding increase in crop yields following addition of nutrients (Dick et al., 2008), particularly at lower application rates (Shcherbak et al., 2015), thus resulting in lower GHG intensities. The mean yield scaled fluxes calculated for the eight maize and sorghum sub-samples was 14.9 g N_2O-N kg^{-1} above-ground N uptake (range = 1.1 to 41.6), approximately 77% higher than the 8.4 g N_2O-N kg^{-1} above-ground N uptake for plots fertilized at 180 – 190 kg N ha^{-1} in a European meta-analysis (van

Groenigen et al., 2010). These data suggest that intensification and N fertilization along with improved agronomic performance through better nutrient, water management in East Africa has a strong potential to lower yield-scaled fluxes from smallholder farms in SSA.

5 CONCLUSION

This study indicates that GHG fluxes from low-input, rain-fed agriculture in western Kenya are lower than fluxes from other agricultural systems with greater management intensities (e.g. sub-tropical systems in China and Latin America). The input intensity for these farming systems is currently low, and so GHG fluxes were not related to management activities at the farm level. Given that this type of smallholder, low-input farming is very common across SSA, it is likely that our findings are valid at a much wider scale, although additional studies are required to confirm this hypothesis. Given that GHG emissions are often associated with soil moisture and that much of East Africa is drier than the climate at this study site, baseline emissions across East Africa may be extremely low. However, even though absolute emissions were low, high yield-scaled GHG fluxes in western Kenya could be reduced through interventions to increase yields (e.g. increased fertilizer, improved soil and water management). As far as we know, this study provides the most comprehensive estimate of GHG emissions from smallholder African farms, in terms of number of sites, monitoring duration and temporal frequency of the measurements. However, more studies are needed to capture annual variability as well as examining baseline emissions in other regions of the continent. These baseline studies are required to compare with proposed low emission development strategies to ensure that improvements in agricultural production continue to minimize GHG emissions, while also examining how intensification affects yields and GHG fluxes.

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Table 1. List of *in situ* empirical studies of greenhouse gas fluxes from agricultural systems in sub-Saharan Africa

Reference	Location (& crop type / treatment)	Sites	Time of measurement	Sampling frequency	Flux rates[4]
Annual Flux Estimates					
(Brümmer et al., 2008; Brümmer et al., 2009)	Burkina Faso (sorghum, cotton or peanut)	4	June – Sept 2005 April – Sept 2006	1 – 3X per week	N_2O: 0.19 – 0.67 kg ha^{-1} y^{-1} CO_2: 2.5 – 4.1 Mg ha^{-1} y^{-1} CH_4: -0.67 – -0.7 kg ha^{-1} y^{-1}
(Dick et al., 2008)[1]	Mali (pearl millet with / without legume intercropping)	3	Jan 2004 – Feb 2005	Monthly	N_2O: 0.9 – 1.5 kg ha^{-1} y^{-1}
(Hickman et al., 2015)	Kenya (maize)	1	Mar 2011 – July 2011 Apr 2012 – Jan 2013	Daily to weekly	N_2O: 0.1 – 0.3 kg ha^{-1} y^{-1}
(Koerber et al., 2009)[2]	Uganda (vegetables)	24	July 2005 – Sept 2006	Monthly	CO_2: 30.3 – 38.5 Mg ha^{-1} y^{-1}
(Lompo et al., 2012)[3]	Burkina Faso (urban gardens)	2	Mar 2008 – Mar 2009	2X per day (“several” times per cropping period)	N_2O: 80.5–113.4 kg ha^{-1} y^{-1} CO_2: 22-36 Mg ha^{-1} y^{-1}
(Makumba et al., 2007)	Malawi (maize with agroforestry)	1	Oct 2001 – Apr 2002	Weekly	CO_2: 2.6 – 7.8 Mg ha^{-1} y^{-1}
(Predotova et al., 2010)[3]	Niger (urban and peri-urban gardens)	3	Apr 2006 – Feb 2007	2X per day for 6 days (repeated 8 - 9X per year)	N_2O: 48 – 92 kg ha^{-1} y^{-1} CO_2: 20 – 30 Mg ha^{-1} y^{-1}
(Sugihara et al., 2012)[2]	Tanzania (maize, with / without residue)	2	Mar 2007 – June 2010	1 – 2X per month	CO_2: 0.9 – 4.0 Mg ha^{-1} y^{-1}
Seasonal Flux Estimates					
(Baggs et al., 2006)	Kenya (maize with agroforestry, till / no	1	Feb – June 2002 (Rainy Season)	Weekly	N_2O: 0.2 – 0.6 kg ha^{-1} CO_2: 1.8 – 2.3 Mg ha^{-1}

	till)				CH_4: 0.1 – 0.3 kg ha^{-1}
(Chapuis-Lardy et al., 2009)	Madagascar (maize with soybean)	1	Nov 2006 – April 2007 (Rainy Season)	Weekly	N_2O: 0.3 kg ha^{-1}
(Chikowo et al., 2004)	Zimbabwe (maize / improved fallow)	1	Dec 2000 – Feb 2001 (Rainy Season)	Weekly	N_2O: 0.1 – 0.3 kg ha^{-1}
(Mapanda et al., 2011)[2]	Zimbabwe (maize, with different fertilizer rates and types)	2	Nov 2006 – Jan 2007 Nov 2007 – Apr 2008 Nov 2008 – Apr 2009 (Rainy Seasons)	1X per 2 months	N_2O: 0.1-0.5 kg ha^{-1} CO_2: 0.7 – 1.6 Mg ha^{-1} CH_4: -2.6 - +5.8 kg ha^{-1}
(Millar et al. 2004)	Kenya (maize with regular and improved fallow)		Sep 1999 – Dec 1999 Mar 2000 – Jun 2000 (Rainy Seasons)	1 – 2X per week	N_2O: 0.1-4.1 kg ha^{-1} CO_2: 0.7 – 1.7 Mg ha^{-1}
Mean Flux Rates from Short Duration Studies					
(Kimetu et al., 2007)	Kenya (maize)	1	Mar 2000 – June 2000 (Rainy Season)	3X per month	N_2O: 1.3 – 12.3 μg m^{-2} h^{-1}
(Mapanda et al., 2010)[2]	Zimbabwe (grassland/grazing, tree plantations and maize)	12	Nov 2006 – Mar 2007 (Rainy Season)	2X per month to 1X per 2 months	N_2O: 1.0 – 4.7 μg m^{-2} h^{-1} CO_2: 22.5 – 46.8 mg m^{-2} h^{-1} CH_4: -9.4 - +6.9 μg m^{-2} h^{-1}
(Thomas, 2012)	Botswana (grazing)	2	Feb, April, July, Nov 2010 (Both Rainy and Dry Season)	7X per day; 12 separate days only	CO_2: 1.1 – 42.1 mg m^{-2} h^{-1}

[1] Study includes fertilization up to 200 kg N ha^{-1}

[2] Sampling is too infrequent for accurate estimates of cumulative fluxes (Barton et al., 2015)

[3] Uses photoacoustic spectroscopy, which has recently had questions raised about its accuracy (Rosenstock et al., 2013a); also, these studies used exceptionally high N application rates, from 473 to approximately 4000 kg N ha^{-1} y^{-1}

[4] Note: flux rates are given as the range of values from the various replicates used in the studies (i.e. the spatial variability and where available [Mapanda et al. 2011 and Thomas 2012], the temporal variability as well), and are reported as N- N_2O, C- CO_2

and C- CH_4.; Please also note units: where possible, annual cumulative fluxes are presented, however in cases with insufficient data to estimate cumulative annual fluxes, we present either mean flux rates, or the cumulative for the given period.

Table 2: Soil properties (± 1 SEM) for 0 to 20 cm depth, sampled immediately before initiation of gas sampling for the different land classes

Land class	C[2] content (%)	N content (%)	CN ratio	pH	Bulk Density (g cm^{-3})
(1) Lowland small (<2 ha) mixed farms with degradation[1] signs (n = 7)	1.38 ± 0.13	0.10 ± 0.01	13.18 ± 0.51	6.61 ± 0.09	0.86 ± 0.03
(2) Lower slopes[3], moderate (2-5 ha) sized mixed farms with degradation signs (n = 8)	1.18 ± 0.14	0.10 ± 0.01	11.60 ± 0.58	6.58 ± 0.16	1.14 ± 0.08
(3) Mid-slopes, moderate sized grazing land (n = 10)	2.27 ± 0.37	0.18 ± 0.03	12.16 ± 0.42	6.02 ± 0.21	0.98 ± 0.07
(4) Upper slopes/highland plateau, mixed farms (n = 22)	2.67 ± 0.17	0.21 ± 0.02	12.69 ± 0.52	5.46 ± 0.24	0.80 ± 0.06
(5) Mid-slopes, isolated moderate sized farms (n = 12)	2.83 ± 0.36	0.24 ± 0.02	13.02 ± 0.81	5.84 ± 0.20	0.71 ± 0.04

[1] degradation signs were bare soil and evidence of erosion visible on MODIS images.
[2] due to lack of carbonates, total C equals organic C
[3] Sloped areas went from the lowlands (approx. 1200 masl) up to the highlands (approx. 1800 masl) ranging from 10 – 30%.

Table 3: Comparison of mean (± 1 SEM) cumulative CO_2-C, CH_4-C and N_2O-N fluxes for four weeks during the dry season (February 2014) and rainy season (April 2014) for differently managed sites in western Kenya.

GHG	Dry Season		Wet Season		P values		
	Annual Crop	Other	Annual Crop	Other	Season	Management[1]	Interaction
CO_2-C (g m^{-2})	19.4 ± 2.8	20.0 ± 3.8	76.6 ± 5.0	62.7 ± 5.7	< 0.0001	0.393	0.204
CH_4-C (mg m^{-2})	-7.4 ± 4.4	2.2 ± 6.7	-3.7 ± 3.6	-15.0 ± 3.5	0.610	0.873	0.044
	Fertilized	Not Fertilized	Fertilized	Not Fertilized			
N_2O-N (mg m^{-2})	0.52 ± 0.23	1.44 ± 0.40	9.87 ± 4.23	5.35 ± 1.14	< 0.0001	0.562	0.112

[1] Management refers to ploughing versus no ploughing for the CO_2 and CH_4 and to fertilized versus no fertilizer for the N_2O

Figures:

Fig. 1. Map of study area showing the sampling location by the different vegetation cover types

Fig. 2. CO_2 (mg C- CO_2 m^{-2} h^{-1}), CH_4 (μg C- CH_4 m^{-2} h^{-1}), and N_2O (μg N_2O-N m^{-2} h^{-1}) flux rates from intact soil cores taken from 36 sites across 5 different land classes in western Kenya incubated at 20°C and 5 different water content (0 [air dried], 25, 35, 55, and 75% WHC).

Fig. 3. CO_2 (mg C- CO_2 m^{-2} h^{-1}), CH_4 (μg C- CH_4 m^{-2} h^{-1}), and N_2O (μg N_2O-N m^{-2} h^{-1}) fluxes over 1 year, as well as precipitation (mm), soil moisture content at 5 cm depth (m^3 m^{-3}) and inorganic N (NO_3 + NH_4) soil concentrations for 59 different fields in western Kenya by land class. Note: Vertical dotted lines correspond to planting and vertical dashed lines correspond to harvesting of annual crops. (Land class 1 = degraded lowland farms; class 2 = degraded farms, lower slopes; class 3 = mid slopes, grazing; class 4 = upper slopes/plateau, mixed farms; and class 5 = mid slopes moderate sized farms)

Fig. 4. Box and whisker plots of cumulative annual fluxes of CO_2 (Mg CO_2-C ha^{-1} $year^{-1}$), CH_4 (kg CH_4-C ha^{-1} $year^{-1}$) and N_2O (kg N_2O-N ha^{-1} $year^{-1}$) from 59 different fields in western Kenya split by land class, field type or vegetation type.

Fig. 1

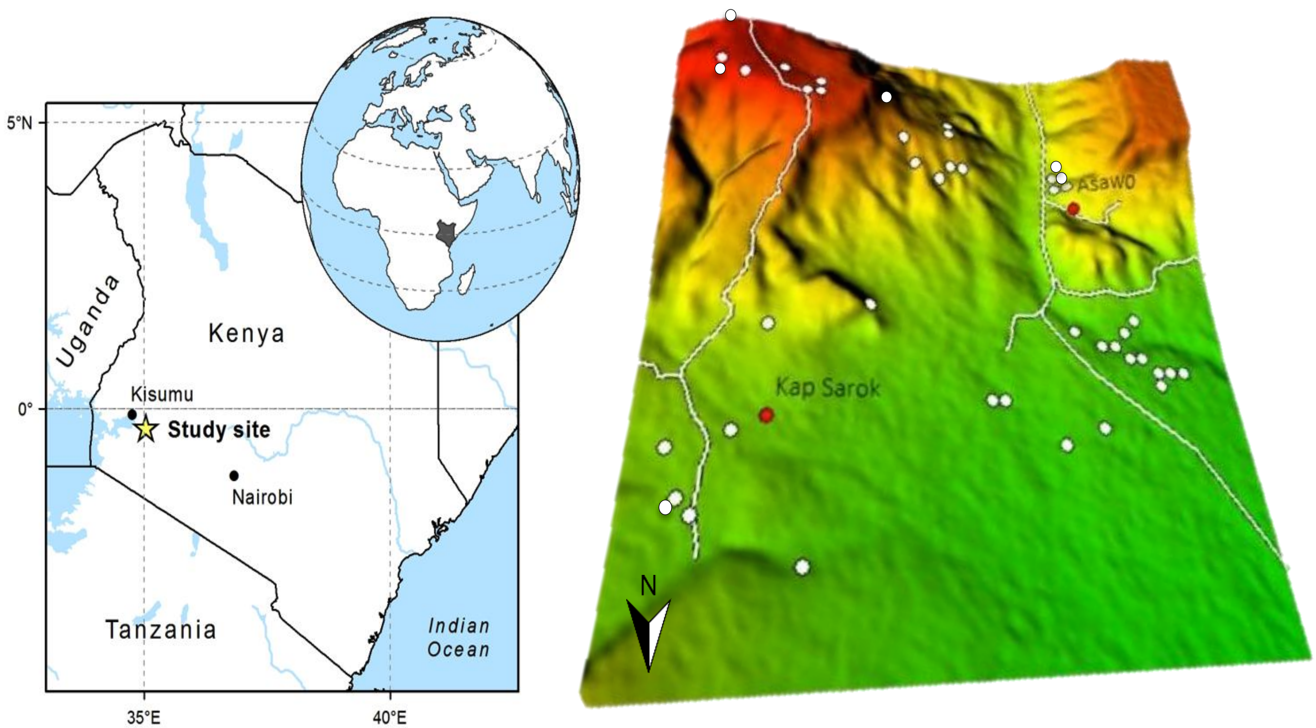
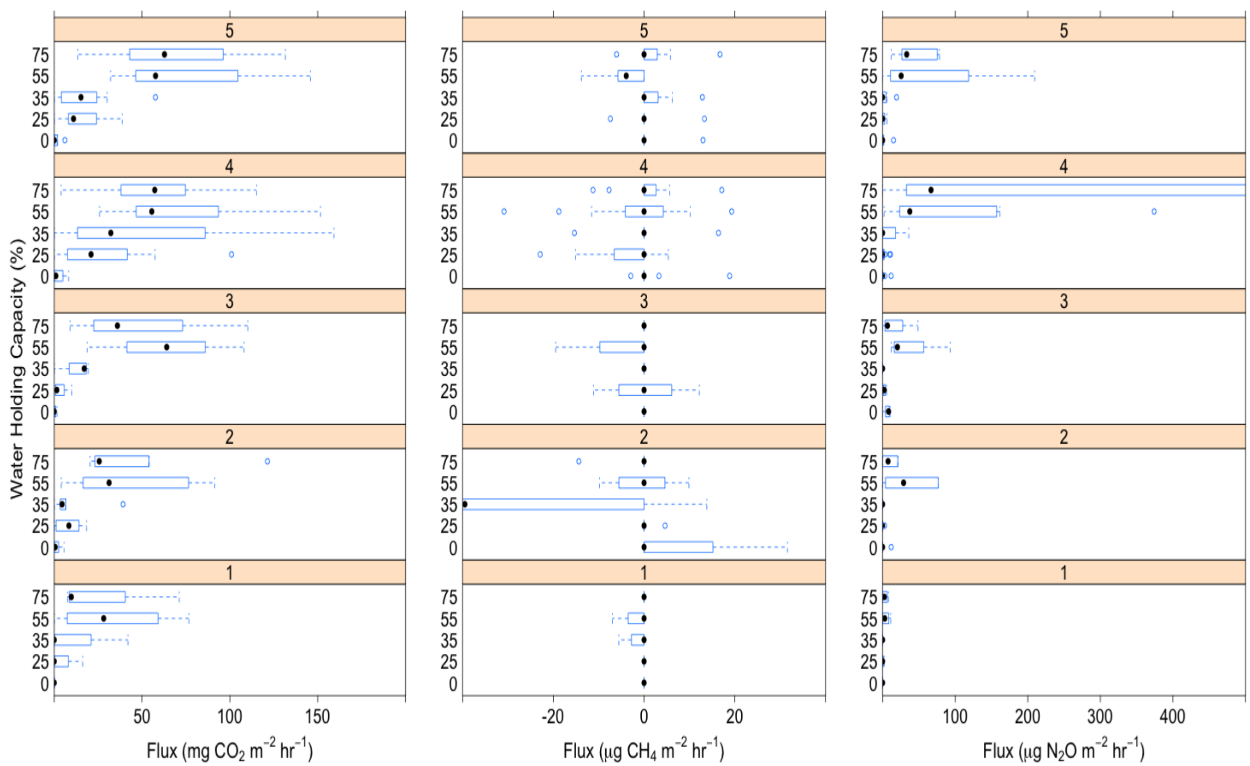

5°N
0°
35°E
40°E
Uganda
Kenya
Kisumu
Study site
Nairobi
Tanzania
Indian Ocean
Kap Sarok
Asawo
N

Fig. 2



5
4
3
2
1
75
55
35
25
0
Water Holding Capacity (%)
50
100
150
Flux (mg CO_2 m^{-2} hr^{-1})
-20
0
20
Flux (µg CH_4 m^{-2} hr^{-1})
100
200
300
400
Flux (µg N_2O m^{-2} hr^{-1})

Fig. 3

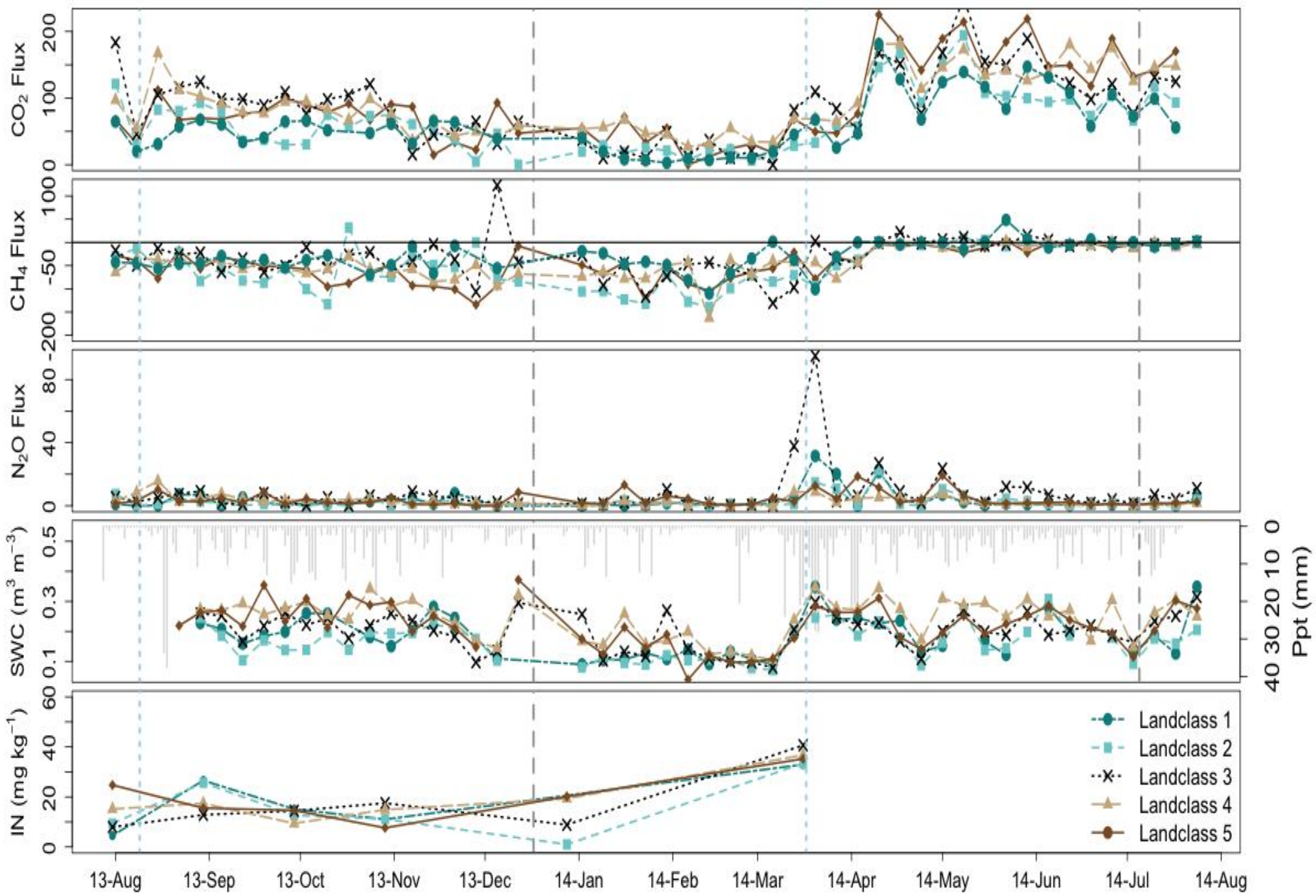
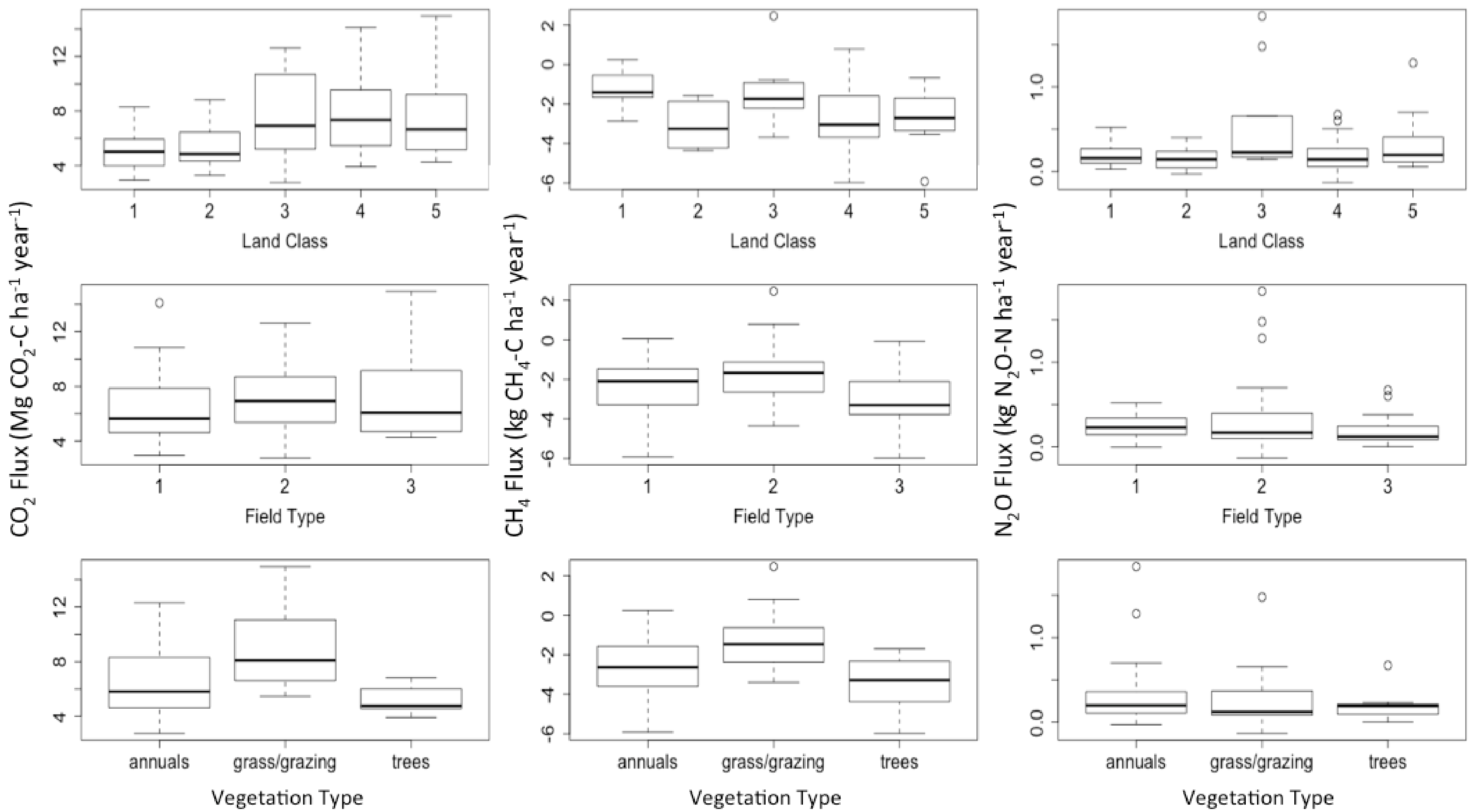

CO2 Flux
0
100
200
CH4 Flux
-200
-50
100
N2O Flux
0
40
80
SWC (m3 m-3)
0.1
0.3
0.5
Ppt (mm)
0
10
20
30
40
IN (mg kg-1)
0
20
40
60
Landclass 1
Landclass 2
Landclass 3
Landclass 4
Landclass 5
13-Aug
13-Sep
13-Oct
13-Nov
13-Dec
14-Jan
14-Feb
14-Mar
14-Apr
14-May
14-Jun
14-Jul
14-Aug

Fig. 4



CO2 Flux (Mg CO2-C ha-1 year-1)
CH4 Flux (kg CH4-C ha-1 year-1)
N2O Flux (kg N2O-N ha-1 year-1)
Land Class
Field Type
Vegetation Type
annuals
grass/grazing
trees