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Smallholder African farms in western Kenya have limited greenhouse gas fluxes

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Abstract

Few field studies examine greenhouse gas (GHG) emissions from African agricultural systems resulting in high uncertainty for national inventories. We provide here the most comprehensive study in Africa to date, examining annual CO_2, CH_4 and N_2O emissions from 59 plots, across different vegetation types, field types and land classes in western Kenya. The study area consists of a lowland area (approximately 1200 m a.s.l.) 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 ha^{-1}). Grazing plots had the highest CO_2 fluxes ($P = 0.005$); treed plots 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 the 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 2/3rds 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 in Asia and Sub-Saharan Africa (SSA), smallholder farms (farm size < 10 ha) comprise almost 80 % of farmland (Altieri and Koohafkan, 2012). Thus 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 lack of data makes it impossible for many developing countries to assess accurately emissions from soils used for agriculture or to use Tier II methodology, which requires the development and documentation of country specific emission factors, to calculate GHG inventories (IPCC, 2006). Also, because most of the research behind the development of the Tier I methodology has been completed in temperate zones, the differences in climate, soils, and farm management seem to result in consistent overestimates of fluxes (Hickman et al., 2014; Rosenstock et al., 2013b) that likely translate to inflated national agricultural GHG inventories in Africa.

Soil greenhouse gas emission potentials have been related to many soil properties such as pH (Khan et al., 2011), soil organic carbon (SOC) content (Chantigny et al., 2010), soil texture (Rochette et al., 2008), vegetation (crop) type (Stehfest and Bouwman, 2006) and management operations such as tillage, fertilizer type, crop rotation, amongst others (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, small-

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holder 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 a top-down approach like "Normalized Difference Vegetation Index" (NDVI), which uses remote sensing to determine the magnitude and temporal variability of primary productivity (Paruelo et al., 2001). Differences in fertility can also be predicted using a bottom-up approach 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 in general, in Kenya in particular and 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 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 × 10 km landscape in Kisumu county of Western Kenya (centered at 35.023° E, 0.315° S); just north of the town of Sondu (Fig. 1), and ranges from a lowland area at approximately 1200 m a.s.l. to a highland plateau at approximately

1800 m a.s.l. This region was found to be broadly representative of demographics and agro-ecological characteristics of other East African tropical highlands (Braun et al., 1997). The climate is humid with a mean temperature of approximately 23 °C and an average annual rainfall of about 1150 mm. 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 typically 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. Approximately 50 % of farmers applied fertilizers to their annual crops, although application rates were very low. For manure, application rates were approximately 100 kg manure ha^{-1} while application rates for synthetic fertilizer (two farmers applied diammonium phosphate and one applied urea) were < 50 kg fertilizer ha^{-1}. Pasture for grazing and degraded shrubland are also common within the area.

Typical soils in the area are well drained, deep, acidic, humic nitisols (about 3 % C) on the highland plateau derived from tertiary volcanic deposits and imperfectly to poorly drained, deep to very deep eutric and verto-eutric planosols with calcareous subsoils (1.7 % C) derived from quaternary sediments in the lowland areas (Sombroek et al., 1980). The soil 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. These metrics show differences between land cover types (e.g. cultivated vs. uncultivated) and between different cultivation management approaches (e.g. agroindustrial vs. subsistence) (Baldi et al., 2014). 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 classes, we ended up with 5 classes, generally consisting of the following landclasses: (1) lowland subsistence farms with degradation signs, (2) highland, moderate sized mixed farms, (3) upper slopes, moderate sized mixed farms, (4) midslopes, subsistence farms and shrubland; and (5) lowland pastures.

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.

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Plots were scored based on the preceding information and those with a higher score were considered field type 1, those with a low score were considered field type 3 and those intermediate plots were assigned a field type 2. It is assumed that field type 1 was the most highly managed and field type 3 the least managed. 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, perennial grasses/grazing, and annual cropping). 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/crop type combination and partly on logistical constraints (i.e. access). One plot however, was converted into a construction site in late 2013, when a house was built on the site, resulting in only 59 plots being measured for the full year.

2.2 Soil core incubation

A soil core incubation study was conducted to examine the effect of soil water content on soil GHG fluxes from the different land-classes, field types and cover types 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 h and then oven-dried (24 h 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, 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, 1958) 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).

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). Briefly, rectangular (0.35 m × 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. On each sampling date, an opaque, vented and insulated lid (0.125 m height) was tightly fitted to the base. 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. To increase 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

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et al., 2013) to form a composite air sample of 60 mL. 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 chamber temperature and air pressure) for both the soil core incubation and the field survey. We validated the data for each chamber/incubation jar measurement by examining the CO_2 concentrations over the 45 min. Chambers that experienced a decrease in CO_2 greater than 10 % between any of the measurement times were assumed to have a leak and when possible, only the final measurement was thrown out. In cases where the change in concentration was lower than the precision of the instrument, we assumed zero flux. 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. 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. 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 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

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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). 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 (0.35156° S, 35.05590° E, 1676 m a.s.l.) and the other in the lowlands (0.30847° S, 34.98769° E, 1226 m a.s.l.). Air temperature was measured using a Decagon ECT air temperature sensor (measurement every 5 min), 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. Data were logged on a Decagon Em50 data collection system and downloaded periodically (typically monthly).

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2.7 Plant production

To estimate crop yields and crop N content of annual crops in the region, we randomly selected and sampled 9 of the plots where we measured gas fluxes for biomass at the end of June 2013. All the plants within a 2.5 m × 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 h 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 Costech elemental combustion system described above for soil analysis.

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 crop type, land class, and field type as fixed factors. 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 *in vivo* measurements were tested using Spearman Rank Correlation, while correlations between GHG fluxes and soil properties were tested using Pearson Correlation. The cumulative in situ 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 vs. not ploughed for CO_2 and fertilized vs. not fertilized for N_2O) as fixed factors along with the two-way interaction terms. Cumulative in situ annual fluxes were compared with ANOVA using an un-balanced design and crop type, land class and field type as fixed factors. In all cases, the distributions of flux measurements were tested for normality using Shapiro–Wilks test and were transformed when necessary. 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 activity (maximum of 7.5 mg CO_2-C m^{-2} h^{-1}) when the soils were air-dried, with increased soil respiration, N_2O fluxes and CH_4 uptake 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. 2–4) 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 crop 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).

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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× the long-term mean.

3.3 Field scale soil GHG fluxes

Soil CO_2 fluxes during August 2013 ranged from 50 to 200 mg CO_2-C h^{-1} m^{-2}, slowly decreased through to November and remained low (< 100 mg CO_2-C h^{-1} m^{-2}) 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} two weeks later (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 h^{-1} m^{-2}; 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 h^{-1} m^{-2} from August 2013 until April 2014, after

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which the variability decreased varying between 0 and 50 µg CH_4-C h^{-1} m^{-2} (Fig. 3). Soil N_2O fluxes were low (generally < 10 µg N_2O-N h^{-1} m^{-2}) for most of the year; with fluxes increasing from a mean of 1.6 µg N_2O-N h^{-1} m^{-2} for the period from October 2013 to March 2014 to a mean of 10.5 µg N_2O-N h^{-1} m^{-2} 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 crop 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

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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 3× greater in treed plots vs. 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 mung beans (*Vigna radiata*) ($n = 1$) 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 ($\rho = -0.145$, $P = 0.471$).

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4 Discussion

The CO_2 fluxes tended to be seasonal in nature, and it was thought that management events, such as ploughing fields or fertilizer applications, would affect the flux rates throughout the year. However, ploughing during the commencement of the rainy season in March 2014 did not significantly increase soil respiration rates above the increases due to the soil re-wetting measured in uncultivated fields. Increased soil respiration due to ploughing however are short-term, usually lasting less than 24 h (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 missed. 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.

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. In general, the CH_4 flux at the soil surface is the result of the balance between production and consumption (Le Mer and Roger, 2001), so it could be that the low rates of atmospheric CH_4 uptake during the long rains was caused by greater CH_4 production in the soil overriding the existing methanotropic activity since the rainfall causing higher soil moisture and likely anaerobic conditions at depth (e.g. Butterbach-Bahl and Papen, 2002).

Seasonal effects were also apparent for the N_2O fluxes. Flux rates remained below 20 $\mu g\, m^{-2}\, h^{-1}\, m^{-2}$ 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, given a mean bulk density of 0.9 $g\,cm^{-3}$ and assuming particle density of 2.65 $g\,cm^{-3}$), 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 as well as March 2014, but it was only the latter period that corresponded to large increases in N_2O fluxes. Unlike September/October though, the high amounts of soil moisture in March coincided with an increase in inorganic N likely caused by the drying–rewetting cycle (Birch, 1960). The stimulation of N_2O fluxes during drying-rewetting cycles is also documented in previous studies (Butterbach-Bahl et al., 2004; Davidson, 1992; Ruser et al., 2006) However, commencement of the rainy season was also when farmers applied fertilizers. Fertilizer applications though, were low (1–20 $kg\,N\,ha^{-1}$) and were did not have a detectable affect on soil NO_3, NH_4 or total inorganic N concentrations ($P = 0.384$, 0.113 and 0.984 respectively). There was however, higher soil inorganic N concentrations at the start of the re-wetting period, (Fig. 5), confirming the release of NO_3 and NH_4 due to the rewetting of the soils.

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 rewetting compared to other sites is not yet understood as the topsoil C and N contents were 1.45 and 0.12 % respectively, pH was 6.3 and bulk density was 1.09 $g\,cm^{-3}$; all of which were well within the range of values for that land class (Table 2). The presence of N_2O emission hotspots are quite common though as denitrification activity can vary dramatically across very small scales (Parkin, 1987). However, we used the gas pooling

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technique (Arias-Navarro et al., 2013), which should have reduced small-scale flux variability.

The soil core incubations and field studies were consistent in that, contrary to expectations, there were no detectable differences in GHG fluxes between the different field types. We had expected differences in fluxes because a previous study in the same region indicated that there were differences in input use, food production, partial N and C balances and soil fertility (Tittonell et al., 2013); and these variables often affect soil GHG fluxes (Buchkina et al., 2012; Jäger et al., 2011). However, differences between field types in total soil C and N were only important when considering each farm individually (Tittonell et al., 2013), which, in our study, may have resulted in greater within-type variation that masked differences between the field types.

We had hypothesized that field type, which is related to the degree of inputs and labour, would be a significant predictive factor since field type 1 would have much more manure added than, for example, field type 3 and this may result in greater CO_2 fluxes. Also, since previous studies found that a significant amount of the variability in soil CO_2 fluxes in agro-ecosystems can be explained by NDVI (Sánchez et al., 2003) and crop type (Mapanda et al., 2010) we expected that both land class, which was based on NDVI, and cover type would also have significant effects on soil CO_2 fluxes. However, annual soil CO_2 fluxes were not related to land class or field type ($P = 0.229$ and 0.540 respectively; Fig. 4), although the cumulative fluxes, (2.7 to 14.0 Mg CO_2-C ha^{-1} yr^{-1}), were well within the range determined for other African studies (Table 1). Higher soil respiration rates from grazing land was inconsistent with a previous study that found similar rates between rain-fed perennial tropical grasslands, croplands and eucalyptus plantations in Zimbabwe (Mapanda et al., 2010). However, because we did not differentiate between root and microbial respiration it could be that the continual vegetation cover contributed more root respiration over the year than was found in the annual crops and treed plots.

As indicated earlier, there was a strong correlation between soil C and N content and cumulative CO_2 fluxes, and while manure inputs have previously been found to

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increase soil C content (Maillard and Angers, 2014), inputs in our study area were very low (between 4 and 6 wheelbarrow loads or approximately 90 to 135 kg C ha^{-1}), which may be too little to cause field-level differences in soil C content. Other factors may also affect soil organic C content such as soil texture (Burke et al., 1989; Franzluebbers et al., 1996), clay mineralogy (Powers et al., 2011) and land use history. Considering the strong correlation between soil C and N and 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 than croplands (Guo and Gifford, 2002), time since conversion could play a larger part in determining the soil organic C content that masks any effects that management activities may have on soil respiration rates in these low input systems.

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). Unlike the CO_2 fluxes, although there were no differences in cumulative CH_4 uptake between field types, there were differences between cover types as grazing plots took up less CH_4 than treed plots and also between land classes with land class 1 taking up less CH_4 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). However, given the higher bulk density in land class 2 (Table 2) and the propensity for denser soils to have reduced CH_4 oxidation rates (Hansen et al., 1993; MacDonald et al., 1996; Teepe et al., 2004), we expected that land class 2 would have uptake less CH_4 than class 1.

Annual N_2O fluxes (between 0.15 and 0.58 kg N_2O-N ha^{-1} yr^{-1}), were low when compared with fertilized plots in sub-tropical Brazil (Piva et al., 2014) or China (Chen et al., 2000), where fluxes ranged up to 4.26 kg N_2O-N ha^{-1} yr^{-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, although low soil moisture may have limited denitrification rates through much of the year as well. 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 and high microsite variability typical of N_2O fluxes (Parkin, 1987).

As noted above, the maximum N_2O and CO_2 fluxes from intact soil cores were correlated with cumulative annual fluxes from the farm sites indicating that these incubations can be used as a quick and relatively inexpensive method to determine which soils have a higher likelihood of being emission hotspots. However, there was no correlation with the CH_4 measurements, which is likely because the 5 cm long soil cores were too short to capture the activity of methanotrophic bacteria required for this process (Butterbach-Bahl and Papen, 2002).

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 measured at 7 sites across SSA in a previous study (Sanchez et al., 2009). As indicated earlier, 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 very common within this region. Increased nutrient inputs and water management are likely required to increase yields, which may result in increased GHG fluxes, however a previous study in low-yielding, semi-arid systems in Mali showed that increased GHG fluxes associated with larger nutrient inputs was less than the corresponding increase in crop yields (Dick et al., 2008). The mean yield scaled fluxes calculated for the eight maize and sorghum sub-samples was 26.6 g N_2O-N kg^{-1} above-ground N uptake (range = 2.9 to 67.0), approximately three times 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 and pest management in East Africa would likely result in lower yield-scaled fluxes from these agricultural systems.

This study was completed to gain an understanding of GHG flux rates from smallholder farms in East Africa and determine if there is an easy way to stratify the landscape to enable upscaling to national and regional scales. We had expected differences between the different field types due to the differences in soil fertility found by Tittonell et al. (2005). Separate NDVI classes as well, typically indicate differences in primary productivity (Xiao et al., 2004), which was also expected to result in differences in GHG fluxes. While there were some differences between the land classes and the crop types for the soil cores, differences were not detected in the field plots suggesting that although there may be the potential for greater fluxes from certain land classes, these may not express themselves in the field due to other factors such as water content.

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). This type of production system is the predominant agricultural production system in sub-Saharan Africa, suggesting that our findings might be valid even at a continental scale. Moreover, because input intensity is low, GHG fluxes were not related to management activities at the farm level. Increased use of mineral fertilizers to improve crop productivity is expected to have a positive impact on reducing soil GHG flux intensities.

5 Conclusions

Fluxes of GHG from low-input, rain-fed agriculture in western Kenya were low with no discernable difference between field types (proxy for management), with only minor differences between different land classes and crop types. The lack of differences between management activities was likely due to the low input rates and is likely representative of low input smallholder farming across much of sub-Saharan Africa. We suggest that time since conversion may be a significant factor in soil respiration rates for this region, masking the effects of management, that needs to be investigated further. Given the low yields common in western Kenya, yield-scaled fluxes can likely

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be reduced through various interventions to increase yields (e.g. increased fertilizer), which would also reduce the depletion of soil nutrients. However further studies that examine how intensification affects yields and GHG fluxes are required in order to minimize any increases in GHG fluxes from the much-needed intensification of agriculture in this region.

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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	Number of sites	Time of measurement	Sampling frequency	Flux rates[d]
Baggs et al. (2006)	Kenya	1	Feb–Jun 2002	Weekly	N_2O: 0.2–0.6 kg ha^{-1} CO_2: 1.8–2.3 Mg ha^{-1} CH_4: 0.1–0.3 kg ha^{-1}
Brümmer et al. (2008)	Burkina Faso	4	Jun–Sept 2005 Apr–Sept 2006	1–3× per week	N_2O: 0.19–0.67 kg ha^{-1} a^{-1}
Brümmer et al. (2009)	Burkina Faso	4	Jun–Sept 2005 Apr–Sept 2006	1–3× per week	CO_2: 2.5–4.1 Mg ha^{-1} a^{-1} CH_4: −0.67–−0.7 kg ha^{-1} a^{-1}
Chapuis-Lardy et al. (2009)	Madagascar	1	Nov 2006–Apr 2007	Weekly	N_2O: 0.3 kg ha^{-1}
Chikowo et al. (2004)	Zimbabwe	1	Dec 2000–Feb 2001	Weekly	N_2O: 0.1–0.3 kg ha^{-1}
Dick et al. (2008)[a]	Mali	3	Jan 2004–Feb 2005	Monthly	N_2O: 0.9–1.5 kg ha^{-1} a^{-1}
Hickman et al. (2015)[a]	Kenya	1	Mar 2011–Jul 2011 Apr 2012–Jan 2013	Daily to weekly	N_2O: 0.1–0.3 kg ha^{-1} a^{-1}
Kimetu et al. (2007)	Kenya	1	4 weeks	3× per month	N_2O: 1.3–12.3 µg m^{-2} h^{-1}
Koerber et al. (2009)[b]	Uganda	24	Jul 2005–Sept 2006	Monthly	CO_2: 30.3–38.5 Mg ha^{-1} a^{-1}
Lompo et al. (2012)[c]	Burkina Faso	2	Mar 2008–Mar 2009	2× per day	N_2O: 80.5–113.4 kg ha^{-1} a^{-1} CO_2: 22–36 Mg ha^{-1} a^{-1}
Makumba et al. (2007)	Malawi	1	Oct 2001–Apr 2002	Weekly	CO_2: 2.6–7.8 Mg ha^{-1} a^{-1}
Mapanda et al. (2010)[b]	Zimbabwe	12	Nov 2006–Mar 2007	2× per month to 1×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}
Mapanda et al. (2011)[b]	Zimbabwe	2	Nov 2006–Jan 2007 Nov 2007–Apr 2008 Nov 2008–Apr 2009	1× 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}
Predotova et al. (2010)[b]	Niger	3	Apr 2006–Feb 2007	2× per day for 6 days (repeated 8–9× per year)	N_2O: 48–92 kg ha^{-1} a^{-1} CO_2: 20–30 Mg ha^{-1} a^{-1}
Sugihara et al. (2012)[b]	Tanzania	2	Mar 2007–Jun 2010	1–2× per month	CO_2: 0.9–4.0 Mg ha^{-1} a^{-1}
Thomas (2012)	Botswana	2	Feb, Apr, Jul, Nov 2010	7× per day; 12 separate days only	CO_2: 1.1–42.1 mg m^{-2} h^{-1}

[a] Study includes fertilization up to 200 kg N ha^{-1}.
[b] Sampling is too infrequent for accurate estimates of cumulative fluxes (Barton et al., 2015).
[c] Uses photoacoustic spectroscopy, which has recently had questions raised about its accuracy (Iqbal et al., 2013; Rosenstock et al., 2013a).
[d] Note: flux rates are given as N-N_2O, C-CO_2 and C-CH_4.

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Table 2. Soil properties (±1 SEM) for the different land classes.

Land class	C content (%)	N content (%)	CN ratio	pH	Bulk Density ($m^3 m^{-3}$)
(1) Lowland small mixed farms with degradation signs	1.38 ± 0.13	0.10 ± 0.01	13.18 ± 0.51	6.61 ± 0.09	0.86 ± 0.03
(2) Lower slopes, moderate sized mixed farms with degradation signs	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	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	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	2.83 ± 0.36	0.24 ± 0.02	13.02 ± 0.81	5.84 ± 0.20	0.71 ± 0.04

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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 Annual Crop	Other	Wet Season Annual Crop	Other	*P* values Season	Management*	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

* Management refers to ploughing vs. no ploughing for the CO_2 and CH_4 and to fertilized vs. no fertilizer for the N_2O.

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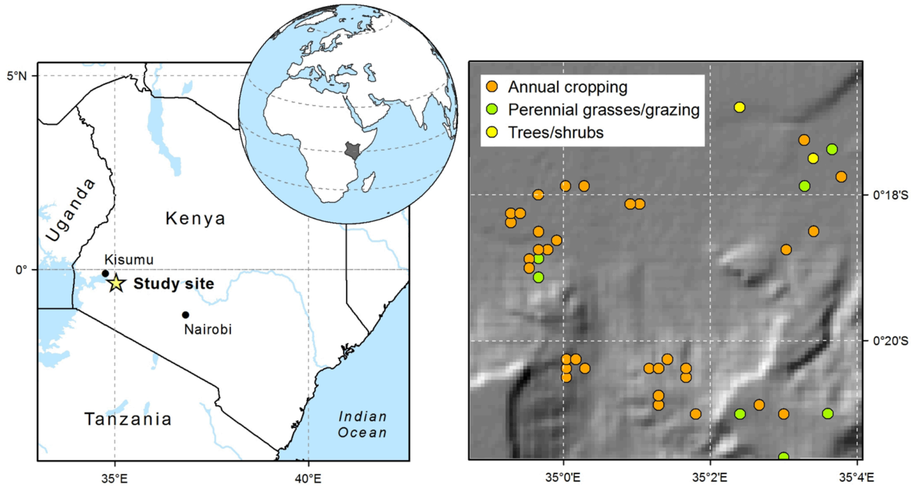

Figure 1. Map of study area showing the sampling location by the different vegetation cover types.

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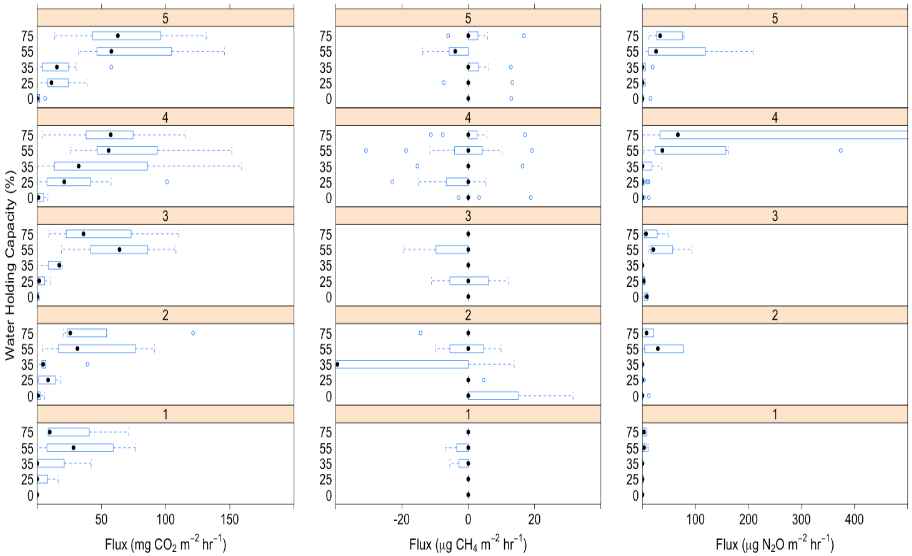

Figure 2. CO_2 (mg CO_2-C m^{-2} h^{-1}), CH_4 (µg CH_4-C m^{-2} h^{-1}), and N_2O (µg N_2O-N m^{-2} h^{-1}) flux rates from intact soil cores taken from 36 sites across 5 different land classes in western Kenya incubated at 20 °C and 5 different water content (0 [air dried], 25, 35, 55, and 75 % WHC).

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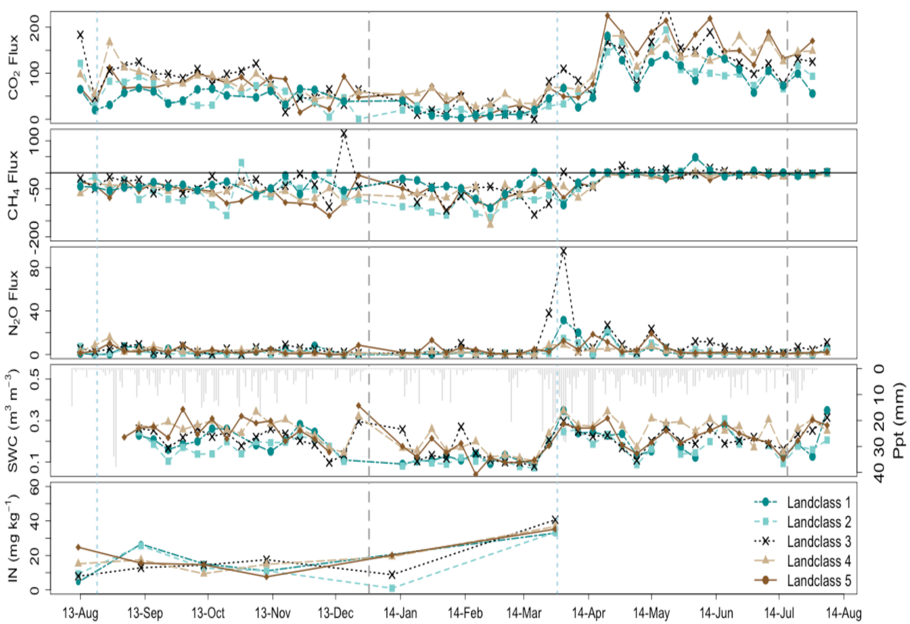

Figure 3. CO_2 ($mg\,CO_2\text{-}C\,m^{-2}\,h^{-1}$), CH_4 ($\mu g\,CH_4\text{-}C\,m^{-2}\,h^{-1}$), and N_2O ($\mu g\,N_2O\text{-}N\,m^{-2}\,h^{-1}$) fluxes over 1 year, as well as precipitation (mm), soil moisture content at 5 cm depth ($m^3\,m^{-3}$) and inorganic N (NO_3 + NH_4) soil concentrations 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).

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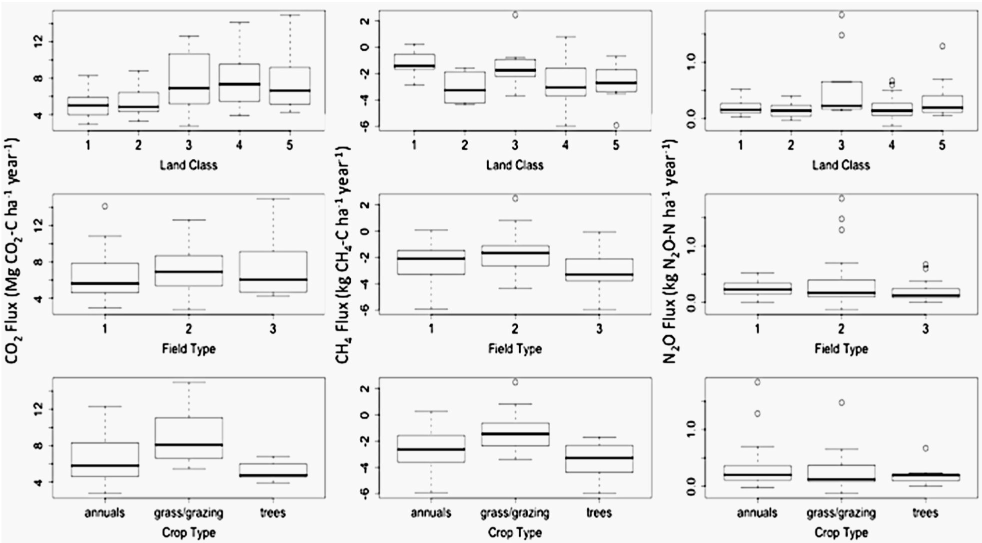

Figure 4. Box and whisker plots of cumulative annual fluxes of CO_2 ($Mg\,CO_2$-$C\,ha^{-1}\,year^{-1}$), CH_4 ($kg\,CH_4$-$C\,ha^{-1}\,year^{-1}$) and N_2O ($kg\,N_2O$-$N\,ha^{-1}\,year^{-1}$) from 59 different fields in western Kenya split by land class.

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