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Reviews and syntheses: Greenhouse gas emissions in natural and agricultural lands in sub-Saharan Africa: synthesis of available data and suggestions for further studies

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Abstract

This paper summarizes currently available data on greenhouse gas (GHG) emissions from African natural and agricultural lands, outlines the knowledge gaps and suggests future directions and strategies for GHG emission studies. GHG emission data were collected from 73 studies conducted in 22 countries in sub-Saharan Africa (SSA). Soil GHG emissions from African natural terrestrial systems ranged from 3.3 to 57.0 Mg carbon dioxide (CO_2) $ha^{-1}\,yr^{-1}$, −4.8 to 3.5 kg methane (CH_4) $ha^{-1}\,yr^{-1}$ and −0.1 to 13.7 kg nitrous oxide (N_2O) $ha^{-1}\,yr^{-1}$. Soil physical and chemical properties, rewetting, vegetation type, forest management and land-use changes were all found to be important factors affecting soil GHG emissions. Greenhouse gas emissions from African aquatic systems ranged from 5.7 to 232.0 Mg CO_2 $ha^{-1}\,yr^{-1}$, −26.3 to 2741.9 kg CH_4 $ha^{-1}\,yr^{-1}$ and 0.2 to 3.5 kg N_2O $ha^{-1}\,yr^{-1}$ and were strongly affected by discharge. Soil GHG emissions from African croplands ranged from 1.7 to 141.2 Mg CO_2 $ha^{-1}\,yr^{-1}$, −1.3 to 66.7 kg CH_4 $ha^{-1}\,yr^{-1}$ and 0.05 to 112.0 kg N_2O $ha^{-1}\,yr^{-1}$ and the N_2O emission factor (EF) ranged from 0.01 to 4.1 %. Incorporation of crop residues or manure with inorganic fertilizers resulted in significant changes in GHG emissions but these were different for CO_2 and N_2O. Soil GHG emissions in vegetable gardens ranged from 73.3 to 132.0 Mg CO_2 $ha^{-1}\,yr^{-1}$ and 53.4 to 177.6 kg N_2O $ha^{-1}\,yr^{-1}$ and N_2O EFs ranged from 3 to 4 %. Soil CO_2 and N_2O emissions from agroforestry were 38.6 Mg CO_2 $ha^{-1}\,yr^{-1}$ and 0.2 to 26.7 kg N_2O $ha^{-1}\,yr^{-1}$, respectively. Improving fallow with nitrogen (N)-fixing trees increased CO_2 and N_2O emissions compared to conventional croplands and type and quality of plant residue is likely to be an important control factor affecting N_2O emissions. Throughout agricultural lands, N_2O emissions slowly increased with N inputs below 150 kg N $ha^{-1}\,yr^{-1}$ and increased exponentially with N application rates up to 300 kg N $ha^{-1}\,yr^{-1}$. The lowest yield-scaled N_2O emissions were reported with N application rates ranging between 100 and 150 kg N ha^{-1}. Overall, total CO_2 equivalent (eq)

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emissions from African natural and agricultural lands were $56.9 \pm 12.7\,Pg\,CO_2\,eq\,yr^{-1}$ and natural and agricultural lands contributed 76.3 and 23.7 %, respectively. Additional GHG emission measurements throughout Africa agricultural and natural lands are urgently required to reduce uncertainty on annual GHG emissions from the different land uses and identify major control factors and mitigation options on emissions. There is also a need to develop a common strategy for addressing this data gap that may involve identifying priorities for data acquisition, utilizing appropriate technologies, and establishing networks and collaboration.

1 Introduction

Global greenhouse gas emissions were estimated to be 49 (±4.5) Gt CO_2 eq in 2010 (IPCC, 2014), with approximately 21.2–24 % (10.3–12 Gt CO_2 eq) of emissions originating from soils in agricultural, forestry and other land use (AFOLU) (Tubiello et al., 2015; IPCC, 2014). Annual non-CO_2 GHG emissions (primarily CH_4 and N_2O) from agriculture were estimated to be 5.2–5.8 Gt $CO_2\,eq\,yr^{-1}$ in 2010 (FAOSTAT, 2014; Tubiello et al., 2013), with approximately 4.3–5.5 Gt $CO_2\,eq\,yr^{-1}$ attributable to land use and land-use change activities (IPCC, 2014).

Greenhouse gas fluxes in Africa play an important role in the global GHG budget (Thompson et al., 2014; Hickman et al., 2014; Valentini et al., 2014; Ciais et al., 2011; Bombelli et al., 2009). In recent years, conversion rates of African natural lands, including forest, grassland and wetland to agricultural lands have increased (Gibbs et al., 2010; FAO, 2010). The dominant type of land use change has been the conversion of forest to agriculture with average deforestation rates of 3.4 million ha per year (FAOSTAT, 2014) (Fig. 1). This land-use conversion results in an estimated release of $0.32 \pm 0.05\,Pg\,C\,yr^{-1}$ (Valentini et al., 2014) or 157.9 ± 23.9 Gt CO_2 eq in 1765 to 2005 (Kim and Kirschbaum, 2015), higher than fossil fuel emissions for the continent (Valentini et al., 2014).

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Soil emissions of all the major GHGs from Africa can be potentially significant at global scales. For example, CO_2 eq emissions from 12 river channels in SSA and wetlands of the Congo River were about 0.9 Pg C per year, equivalent to about 25 % of the global terrestrial and ocean carbon sink (Borges et al., 2015). Nitrous oxide emissions in Africa contribute between 6–19 % of the global total, and changes in soil N_2O fluxes in Africa drive large inter-annual variations in tropical and subtropical N_2O sources (Thompson et al., 2014; Hickman et al., 2011). Nitrous oxide emissions from biogenic sources and fires in natural lands were estimated to contribute to 34 % of total N_2O emissions in the region (Valentini et al., 2014). Even with the low fertilizer rates used across the continent, agricultural GHG emissions in Africa are substantial; amounting to 26 % of the continent's total GHG emissions (Valentini et al., 2014). According to Lassaletta et al. (2014), mean N application rates in Africa were 34 kg N ha^{-1} in 2009 and only 16 kg N ha^{-1} in sub-Saharan African countries. Only Mauritius, Botswana and South Africa had average N application rates exceeding100 kg N ha^{-1}. However, use of synthetic fertilizers such as urea has increased in the last four decades as well as the number of livestock (and their manure and urine products) (Bouwman et al., 2009 and 2013) (Figs. 2 and 3). The increasing trend in N application rates is expected to cause a twofold increase in agricultural N_2O emissions in Africa by 2050 (from 2000) (Hickman et al., 2011). In the case of CH_4 emissions, there are important differences between ecosystems. Tropical humid forest, wetlands and termite mounds are likely sources of CH_4, while seasonally dry forests and savannahs are typically CH_4 sinks (Valentini et al., 2014).

Our current understanding of GHG emissions in Africa is particularly limited when compared to the potential the continent has as both a GHG sink and source. This lack of data on GHG emissions from African natural and agricultural lands and the lack of a comprehensive analysis of existing data hinder the progress of our understanding of GHG emissions on the continent (Hickman et al., 2014; Valentini et al., 2014; Ciais et al., 2011; Bombelli et al., 2009). In order to identify mitigation measures and other climate smart interventions for the region it is important to quantify baseline GHG emis-

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sions, as well as understand the impacts of different land-use management strategies on GHG emissions (e.g., Palm et al., 2010).

In this study our objectives are to synthesize currently available data on GHG emissions from African AFOLU; create an inventory of information from studies on emissions; and select priority topics for future GHG emission studies in natural and agricultural lands in SSA.

2 Methodology

2.1 Data collection

Data were acquired by searching existing peer-reviewed literature using the names of the sub-Saharan countries and the GHGs (i.e. CO_2, CH_4 and N_2O) as search terms (using Web of Science and Google Scholar; 1960–2015). We selected studies that reported in situ annual GHG emissions or those that provided enough information to estimate annual GHG emissions through unit conversion and/or extrapolation of given data. Data from 73 studies, conducted in 22 countries ($n = 244$) in SSA were used and were further categorized as GHG emission in natural lands ($n = 117$; Table S1 in the Supplement) and agricultural lands ($n = 127$; Table S2) (Fig. 4). The category of GHG emissions in natural lands were further divided into emissions from forest/plantation/woodland ($n = 55$), savannah/grassland ($n = 31$), streams/rivers ($n = 14$), wetlands/floodplains/lagoons/reservoirs ($n = 11$), termite mounds ($n = 5$), and salt pans ($n = 1$) (Table 1). The category of GHG emission in agricultural lands, were subdivided into emissions from cropland ($n = 105$), rice paddy ($n = 1$), vegetable garden ($n = 5$), and agroforestry ($n = 16$) (Table 1). Across all categories there were 174 CO_2, 201 CH_4 and 184 N_2O emissions measurements. Where N_2O emission studies included experimental data from control plots with no N fertilizer additions (i.e. for background N_2O emissions) and from plots with different levels of applied N, a N_2O emission factor (EF) was calculated following the IPCC (2006) Tier I methodology as

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follows:

$$N_2O\ EF\,(\%) = \frac{N_2O\ \text{emission}_{N\ \text{treatment}} - N_2O\ \text{emission}_{\text{control}}}{N\ \text{input}} \times 100 \quad (1)$$

where, N_2O EF (%) is N_2O emission factor, N_2O $\text{emission}_{N\ \text{treatment}}$ is N_2O emission in N input, N_2O $\text{emission}_{\text{control}}$ is control treatments with no N fertilizer additions, and N_{input} is the amount of added N.

It should be noted that our data compilation includes a wide variety of studies that were conducted under diverse biophysical conditions using a range of methodologies for quantifying GHG emissions (e.g., different sampling protocols, chamber design, and emission rate calculations), soil properties, and climatic factors. Therefore, the overall figures on GHG emissions shown are based on results achieved by different measurement techniques with inherent and contrasting sources of error.

2.2 Statistical analyses

The compiled datasets were used to examine the best model fit and to derive the corresponding model parameters for N_2O emissions and yield-scaled N_2O emissions as a function of the respective N input levels. Different data fitting models (linear, nonlinear, natural log, logarithm and sigmoidal) were tested for each dataset. The regression models were checked for violation of assumptions of normal distribution (Shapiro–Wilk test), homoscedasticity (Breusch–Pagan test), and constant variance (Durbin–Watson statistic) (Motulsky and Christopoulos, 2004). Separate t tests were used to assess significance of regression coefficients and intercepts in the fitted parametric models and adjusted coefficients of determination (adjusted R^2) of fitted parametric models were used as criteria for model selection: the model with the higher adjusted R^2 was selected. Statistical significance was considered at the critical level of 5 %. These statistical analyses were conducted using SAS® ver. 9.2 (SAS Institute, Cary, NC, USA) and SigmaPlot® ver. 11.0 (Systat Software Inc., San Jose, CA, USA).

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3 Results and discussion

3.1 Greenhouse gas emissions in natural lands

3.1.1 Terrestrial systems

Soil GHG emissions from African natural terrestrial systems such as natural forest, plantation, woodland, savannah, grassland, termite mounds and salt pans ranged from 3.3 to 130.9 $Mg\,CO_2\,ha^{-1}\,yr^{-1}$, −4.8 to 3.5 $kg\,CH_4\,ha^{-1}\,yr^{-1}$ and −0.1 to 13.7 $kg\,N_2O\,ha^{-1}\,yr^{-1}$ (Tables 1 and S1). The high variability in emission rates was likely related to differences in soil temperature, moisture content and physical-chemical properties as well as the type of natural vegetation present. Within forest systems, CO_2 emissions were strongly related to both soil moisture and temperature. For example, soil moisture explained about 50 % of the seasonal variability in soil CO_2 efflux in a *Croton macrostachys*, *Podocarpus falcatus* and *Prunus africana* forest in Ethiopia (Yohannes et al., 2011), as well as much of the seasonal variation in soil CO_2 efflux in a 3-year-old *Eucalyptus* plantation in Republic of Congo (Epron et al., 2004). Thomas et al. (2011) found that the Q_{10} of soil CO_2 efflux (a measure of the temperature sensitivity of efflux, where a Q_{10} of 2 represents a doubling of efflux given a 10 °C increase in temperature) was dependent on soil moisture at sites across the Kalahari in Botswana, ranging from 1.1 in dry soils, to 1.5 after a 2 mm rainfall event and 1.95 after a 50 mm event. Similarly, in a Zambian woodland, the main driving factor controlling CO_2 emissions at a seasonal time scale was a combination of soil water content and temperature (Merbold et al., 2011).

Soil physical (e.g., bulk density, porosity and soil texture) and chemical properties (e.g., pH, C and N) also affected soil GHG emissions (e.g., Saggar et al., 2013; Smith, 2010; Snyder et al., 2009). Soil CO_2 efflux was positively related to total soil C content in undisturbed *miombo* woodland in Zambia, although not in an adjacent disturbed woodland (Merbold et al., 2011). In a Kenyan rainforest, CO_2 emissions were negatively correlated with subsoil C and positively correlated with subsoil N concentrations,

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while N_2O emissions were negatively correlated with clay content and topsoil C : N ratios (Werner et al., 2007). However, soil bulk density and pH were the most influential factors driving spatial variation of in situ N_2O emissions in a tropical highland rainforest in Rwanda (Gharahi Ghehi et al., 2014). Similarly, a laboratory-based experiment using soils from 31 locations in a tropical mountain forest in Rwanda showed that N_2O emissions were negatively correlated with soil pH, and positively correlated with soil moisture, soil C and soil N (Gharahi Ghehi et al., 2012).

In many temperate systems, vegetation type also affects soil GHG emissions, likely because of differences in litter quality and production rate, amount of below-ground biomass, the structure of root systems as well as plant-mediated effects on soil microclimate (e.g., Díaz-Pinés et al., 2014; Masaka et al., 2014; Kim et al., 2010). This is consistent with findings from African systems where annual soil CO_2 efflux also varied with vegetation types. For example, annual soil CO_2 emissions were significantly lower in N-fixing acacia monocultures than in eucalypt monocultures and mixed-species stands in Republic of Congo (Epron et al., 2013). The differences were attributed to leaf area index in another study from the Republic of Congo where they found 71 % of seasonal soil CO_2 efflux variability was explained by the quantity of photosynthetically active radiation absorbed by the grass canopy (Caquet et al., 2012). Also in the Republic of Congo, it was found that litterfall accounted for most of the age-related trends after the first year of growth, and litter decomposition produced 44 % of soil CO_2 flux in the oldest stand (Nouvellon et al., 2012), strongly suggesting that the amount and quality of litter plays a major role in determining soil CO_2 flux. However, the effect of vegetation type can also interact with soil physical-chemical properties. For example in Benin, root respiration contributed to 30 % of total soil CO_2 efflux in oil palms when the soil was at field capacity and 80 % when soil was dry (Lamade et al., 1996).

Forest soils predominantly act as sinks for CH_4 (Werner et al., 2007). In Cameroon, the largest CH_4 oxidation rates were observed from relatively undisturbed near-primary forest sites (–14.7 to –15.2 $ng\,m^{-2}\,s^{-1}$) compared to disturbed forests (–10.5 to 0.6 $ng\,m^{-2}\,s^{-1}$) (Macdonald et al., 1998). Savannah and grassland were found to

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be both a sink and source of CH_4. In Mali, CH_4 uptake was observed in dry sandy savannah (Delmas et al., 1991), while a savannah in Burkina Faso was found to be both a CH_4 sink and source during the rainy season, although overall it was a net CH_4 source (Brümmer et al., 2009).

Soil rewetting typically has a large impact on GHG emissions. Two broad mechanisms responsible for changed soil GHG flux following rewetting have been hypothesized: (1) enhanced microbial metabolism by an increase in available substrate due to microbial death and/or destruction of soil aggregates (i.e. commonly known as the Birch effect, Birch, 1964), and (2) physical mechanisms that can influence gas flux, including infiltration, reduced diffusivity, and gas displacement in the soil (e.g., Kim et al., 2012). Consistent with this mechanisms of re-wetting effects in soils of other continents (e.g., Kim et al., 2012), soil CO_2 efflux increased immediately after rainfall in a sub-tropical palm woodland in northern Botswana, however the increase was short-lived (Thomas et al., 2014), while large pulses of CO_2 and N_2O, followed by a steady decline were also observed after the first rainfall event of the wet season in a Kenyan rainforest (Werner et al., 2007). Soil CO_2 efflux in a South African savannah was strongly stimulated by addition of rainfall (Fan et al., 2015; Zepp et al., 1996) and soil N_2O concentrations increased markedly 30 min after wetting and peaked between 2 and 5 h after rainfall in a semi-arid savannah (Scholes et al., 1997). In Zimbabwe, the release of N_2O from dryland savannahs was shown to constitute an important pathway of release for N, and emissions were strongly linked to patterns of rainfall (Rees et al., 2006). In Botswana, Thomas and Hoon (2010) reported large and short-lived pulses of soil CO_2 efflux after artificial wetting of dry soils: soil CO_2 efflux on dry soils was between 2.8–14.8 $mg\,C\,m^{-2}\,h^{-1}$ but increased to 65.6 $mg\,C\,m^{-2}\,h^{-1}$ in the hour after light wetting and 339.2 $mg\,C\,m^{-2}\,h^{-1}$ in the hour after heavy wetting.

Forest management such as burning, which is a common practice in SSA, and thinning, affects GHG emissions (Table 2). The IPCC Tier 1 methodology only calculates the amount of GHG emissions as a percentage of the carbon that is released through the burning; however it may also increase forest soil GHG emissions once the fire has

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passed. For example, soil CO_2 efflux immediately increased after burning of woodland in Ethiopia (Andersson et al., 2004); also, five days after burning rainfall resulted in a 2-fold increase in soil CO_2 efflux from the burned plots compared to the unburned plots. In contrast, 12 days after burning soil CO_2 efflux was 21 % lower in the burned plots (Andersson et al., 2004). However, contrasting impacts of fire on soil GHG emission were observed in a savannah/grassland in the Republic of Congo where fire did not change soil CO_2, CH_4 and N_2O fluxes (Castaldi et al., 2010; Delmas et al., 1991). Similarly, in South Africa, soil CH_4 efflux was not significantly affected by burning (Zepp et al., 1996). In contrast, annual fires decreased soil CH_4 oxidation rates in a Ghanaian savannah (Prieme and Christensen, 1999). These case studies demonstrate that fire impacts are not always consistent and this is likely the result of different fire characteristics (e.g., intensity or frequency), soil type (e.g., Kulmala et al., 2014; Kim et al., 2011) and post-fire weather conditions. Thinning forest cover also can increase soil CO_2 efflux. Yohannes et al. (2013) reported 24 and 14 % increases in soil CO_2 efflux in the first and second years following thinning of a 6 year old *Cupressus lusitanica* plantation in Ethiopia.

Termite mounds are known sources of CH_4 and CO_2, and a study in a Burkina Faso savannah found that CH_4 and CO_2 released by termites (*Cubitermes fungifaber)* contributed 8.8 and 0.4 % of total soil CH_4 and CO_2 emissions, respectively (Brümmer et al., 2009). In Cameroon, the mounds of soil-feeding termites (*Thoracotermes macrothorax* and *Cubitermes fungifaber*) were point sources of CH_4 ranging 53.4 to 636 $ng\,s^{-1}$ $mound^{-1}$, which at the landscape scale may exceed the general sink capacity of the soil (Macdonald et al., 1998). In Zimbabwe, it was found that *Odontotermes transvaalensis* termite mounds located in dambos (seasonal wetlands) were an important source of GHGs, and emissions varied with catena position for CO_2 and CH_4 (Nyamadzawo et al., 2012).

Compared to the other environments covered in this review there are very few studies from salt pans. Thomas et al. (2014) however, found soil CO_2 efflux increased with temperature and also increased for a few hours after flooding of the surface of the

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Makgadikgadi salt pan in Botswana. Annual CO_2 emissions in salt pan were estimated as 0.7 $Mg\,CO_2\,ha^{-1}\,yr^{-1}$ (Thomas et al., 2014).

3.1.2 Aquatic systems

Greenhouse gas emissions from African aquatic systems such as streams, rivers, wetlands, floodplains, reservoir, and lagoons ranged from 5.7 to 232.0 $Mg\,CO_2\,ha^{-1}\,yr^{-1}$, −26.3 to 2741.9 $kg\,CH_4\,ha^{-1}\,yr^{-1}$ and 0.2 to 4.5 $kg\,N_2O\,ha^{-1}\,yr^{-1}$ (Tables 1 and S1). In the Nyong River (Cameroon), CO_2 emissions (5.5 $kg\,CO_2\,m^{-2}\,yr^{-1}$) were four times greater than the flux of dissolved inorganic carbon (Brunet et al., 2009). In Ivory Coast, three out of five lagoons were oversaturated in CO_2 during all seasons and all were CO_2 sources (3.1–16.2 $g\,CO_2\,m^{-2}\,d^{-1}$) due to net ecosystem heterotrophy and inputs of riverine CO_2 rich waters (Koné et al., 2009). In the flooded forest zone of the Congo River basin (Republic of Congo) and the Niger River floodplain (Mali), high CH_4 emissions (5.16×10^{20} – 6.35×10^{22} g $CH_4\,m^{-2}\,d^{-1}$) were recorded on flooded soils (Tathy et al., 1992; Delmas et al., 1991). In Zimbabwe, dambos can be major or minor sources of GHGs depending on catena position. Upland dambos were important sources of N_2O and CO_2, and a sink for CH_4; while those in a mid-slope position were a major source of CH_4, but a weak source of CO_2 and N_2O; and those at the bottom were a weak source for all GHGs (Nyamadzawo et al., 2014a). In the Congo Basin (Republic of Congo), streams and rivers in savannah regions had higher CO_2 emissions (46.8–56.4 $g\,CO_2\,m^{-2}\,d^{-1}$) than swamps (13.7–16.3 $g\,CO_2\,m^{-2}\,d^{-1}$) and tropical forest catchments (37.9–62.9 $g\,CO_2\,m^{-2}\,d^{-1}$) (Mann et al., 2014). In the Okavango Delta (Botswana), the average CH_4 flux in river channels (0.75 $g\,CH_4\,m^{-2}\,d^{-1}$) was higher than that in floodplains and lagoons (0.41–0.49 $g\,CH_4\,m^{-2}\,d^{-1}$) (Gondwe and Masamba, 2014). In the Zambezi River (Zambia), while CO_2 and CH_4 concentrations in the main channel were highest downstream of the floodplains, N_2O concentrations were lowest downstream of the floodplains (Teodoru et al., 2015). Overall, 38 % of the total C in the Zambezi River is emitted into the atmosphere, mostly as CO_2 (98 %)

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(Teodoru et al., 2015). A recent study of 10 river systems in SSA estimated water-air CO_2, CH_4 and N_2O fluxes to be 8.2 to 66.9 g CO_2 m^{-2} d^{-1}, 0.008 to 0.46 g CH_4 m^{-2} d^{-1}, and 0.09 to 1.23 mg N_2O m^{-2} d^{-1}, respectively (Borges et al., 2015). The authors suggested that lateral inputs of CO_2 from soils, groundwater and wetlands were the largest contributors of the CO_2 emitted from the river systems (Borges et al., 2015).

The concentration and flux of GHGs are strongly linked to stream discharge but clear patterns have not yet been identified. In the Congo River, surface CO_2 flux was positively correlated with discharge (Wang et al., 2013), while in Ivory Coast, rivers were often oversaturated with CO_2 and the seasonal variability of partial pressure of CO_2 (pCO_2) was due to dilution during the flooding period (Koné et al., 2009). Similarly, CO_2 fluxes show a very pronounced seasonal pattern strongly linked to hydrological conditions in the Oubangui River in the Central African Republic (Bouillon et al., 2012). Although higher CH_4 concentrations were found during low-discharge conditions, N_2O concentrations were lowest during low-discharge conditions (Bouillon et al., 2012). In the Zambezi River (Zambia), inter-annual variability was relatively large for CO_2 and CH_4 and significantly higher concentrations were measured during wet seasons (Teodoru et al., 2015). However, inter-annual variability of N_2O was less pronounced and generally higher values were found during the dry season (Teodoru et al., 2015).

The relationship between GHG fluxes from aquatic systems and water temperature is not clear. In the Okavango Delta (Botswana), CH_4 emissions were highest during the warmer, summerrainy season and lowest during cooler winter season suggesting the emissions were probably regulated by water temperature (Gondwe and Masamba, 2014). However, Borges et al. (2015) found no significant correlation between water temperature and pCO_2 and dissolved CH_4 and N_2O in 11 SSA river systems, although there was a positive relationship between pCO_2 and dissolved organic C in six of the rivers. They also found the lowest N_2O values were observed at the highest pCO_2 and lowest $\%O_2$ levels, suggesting the removal of N_2O by denitrification (Borges et al., 2015).

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3.2 Greenhouse gas emissions from agricultural lands

3.2.1 Croplands

Soil GHG emissions reported from African croplands ranged from 1.7 to 141.2 $Mg\,CO_2\,ha^{-1}\,yr^{-1}$, −1.3 to 66.7 $kg\,CH_4\,ha^{-1}\,yr^{-1}$ and 0.05 to 112.0 $kg\,N_2O\,ha^{-1}\,yr^{-1}$ (Tables 1 and S1). The N_2O EF ranged from 0.01 to 4.1 % (Tables 1 and S1).

The effects of the amount and type of N input on N_2O emissions in croplands have been studied in several locations (Table 2). In western Kenya, the rate of N fertilizer application (0 to 200 $kg\,N\,ha^{-1}$) had no significant effect on N_2O emissions (620 to 710 $g\,N_2O\text{-}N\,ha^{-1}$ for 99 days) (Hickman et al., 2014), however another study from western Kenya, found a relationship between N input and N_2O emissions that was best described by an exponential model with the largest impact on N_2O emissions occurring when N inputs increased from 100 to 150 $kg\,N\,ha^{-1}$ (Hickman et al., 2015). An incubation study in Madagascar demonstrated that application of mixed urea and di-ammonium-phosphate resulted in lower N_2O emissions (28 vs. 55 $ng\,N_2O\text{–}N\,g^{-1}\,h^{-1}$ for 28 days, respectively) than a mixed application of urea and NPK fertilizer (Rabenarivo et al., 2014).

Incorporation of crop residues to the soil has frequently been proposed to increase soil fertility (Malhi et al., 2011), however incorporation of crop residues also affects CO_2 and N_2O emissions (Table 2). In Tanzania, incorporation of plant residue into soil increased annual CO_2 fluxes substantially (emissions rose from 2.5 to 4.0 and 2.4 to 3.4 $Mg\,C\,ha^{-1}\,yr^{-1}$ for clay and sand soils, respectively) (Sugihara et al., 2012), although a study in Madagascar showed that rice-straw residue application resulted in larger fluxes of CO_2 but reduced N_2O emissions due to N immobilization (Rabenarivo et al., 2014). In contrast, application of *Tithonia diversifolia* (tithonia) leaves led to greater N_2O emissions compared to urea application in maize fields in Kenya (Sommer et al., 2015; Kimetu et al., 2007). The higher N_2O emissions after application of

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Tithonia diversifolia were attributed to high levels of nitrate and available carbon in the soil caused by the application that subsequently enhanced denitrification rates. In incubation studies with cultivated soil from Ghana, N_2O emissions were significantly higher from soils amended with low C : N ratio clover residues compared to high C : N ratio barley residues (Frimpong et al., 2012) and increasing the proportion of maize in a cowpea-maize residue significantly decreased N_2O emissions compared to cowpea residue incorporation alone (Frimpong et al., 2011), again likely due to the higher C : N ratio of the maize residue compared with the cowpea. Another incubation study with cultivated soil from Ghana showed that N_2O emissions increased after addition of residues of three tropical plant species (*Vigna unguiculata*, *Mucuna pruriens* and *Leucaena leucocephala*) and emissions were positively correlated with the residue C : N ratio, and negatively correlated with residue polyphenol content, polyphenol : N ratio and (lignin + polyphenol) : N ratio (Frimpong and Baggs, 2010). It is rare for N_2O emissions to be positively correlated to C : N ratio and the authors of the study suggest that it was either because soil C was limiting denitrification rates or that release of N from the residues was slow (Frimpong and Baggs, 2010). The results demonstrate that the quality of residues (e.g., C : N ratio, N, lignin and soluble polyphenol contents) affect GHG emissions and further studies are needed to clearly identify the relationship between them (Snyder et al., 2009; Mafongoya et al., 1997).

Adding an additional source of N (mineral or organic) when crop residues are incorporated into the soil could stimulate mineralization of crop residues, increase N-use efficiency and produce higher yields (e.g., Garcia-Ruiz and Baggs, 2007) (Table 2). It was found that application of mixed crop residue or manure and inorganic fertilizers resulted in different response of CO_2 and N_2O emissions. In maize (*Zea mays* L.) and winter wheat (*Triticum aestivum* L.) fields in Zimbabwe, application of inorganic fertilizer (ammonium nitrate, NH_4NO_3–N) with manure increased CO_2 emissions (26 to 73 %), compared to sole application of manure (Nyamadzawo et al., 2014a). However, the mixed application resulted in lower N_2O emissions per yield (1.6–4.6 g N_2O kg^{-1} yield), compared to sole application of inorganic fertilizer (6–14 g N_2O kg^{-1} yield) (Nya-

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madzawo et al., 2014a). Similarly, in a maize field in Zimbabwe, N_2O emissions were lower after the application of composted manure and inorganic fertilizer (NH_4NO_3–N) compared to sole application of inorganic fertilizer. The same treatments, however, led to the opposite results for CO_2 emissions (Mapanda et al., 2011). In Mali, pearl millet (*Pennisetum glaucum*) fields treated with both manure and inorganic fertilizer urea emitted significantly less N_2O than plots receiving only urea fertilizer (Dick et al., 2008). The lower N_2O emissions in soils amended with manure were attributed to the initial slow release and immobilisation of mineral N and the consequently diminished pool of N available to be lost as N_2O (Nyamadzawo et al., 2014a, b; Mapanda et al., 2011; Dick et al., 2008). In an incubation study with cultivated soils from Zimbabwe, Ghana and Kenya, combining organic residue (maize, calliandra, and tithonia) and urea fertilizers decreased N_2O emissions in coarse-textured soils but it increased N_2O emissions in fine-textured soils due to the higher level of available N (Gentile et al., 2008).

The effects of crop type and management on GHG emissions have also been studied by several groups (Table 2). In Uganda, there were no significant differences in soil CO_2 effluxes from different crops (lettuces, cabbages, beans) (Koerber et al., 2009). However, in Zimbabwe, rape production resulted in greater N_2O emissions (0.64–0.93 % of applied N was lost as N_2O) than tomatoes (0.40–0.51 % of applied N was lost as N_2O) (Masaka et al., 2014). The results suggest that the effect of crop type on GHG emissions is difficult to predict and more research is needed to elucidate the relationship between crops, crop management and GHG emissions.

In Mali, growing N-fixing haricot beans in rotation did not significantly increase N_2O emissions (Dick et al., 2008). In Madagascar, N_2O emissions were not significantly affected by management practices such as direct seeding mulch-based cropping and traditional hand-ploughing after harvesting (Chapuis-Lardy et al., 2009). However, the authors admitted the lack of difference between treatments may be partially due to the short duration of the experiment and suggested more complete monitoring to validate the observation. In highland Tanzanian maize fields, GHG fluxes were similar from soils under conventional and various conservation agriculture practices (Kimaro et al.,

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2015). However, when fluxes were yield-scaled the global warming potential (Mg CO_2 eq Mg grain^{-1}) was lower from fields with reduced tillage plus mulch and leguminous trees (2.1–3.1) and from fields with reduced tillage plus mulch and nitrogen fertilizer (1.9–2.3) compared to fields under conventional agriculture (1.9–8.3) (Kimaro et al., 2015).

Croplands were found to be both a sink and a source of CH_4. In Burkina Faso, CH_4 flux rates from croplands ranged from −0.67 to 0.70 kg CH_4–C ha^{-1} yr^{-1} (Brümmer et al., 2009), while in Republic of Congo, CH_4 uptake was observed in cassava and peanut fields and a recently ploughed field (Delmas et al., 1991). However, cropped and fertilised dambos in Zimbabwe were consistently sources of CH_4 (13.4 to 66.7 kg CH_4 ha^{-1} yr^{-1}) (Nyamadzawo et al., 2014b).

3.2.2 Grazing grassland

Only one study measured GHG emissions in grazing grasslands. Thomas (2012) found that soil CO_2 efflux from a Botswana grazing land was significantly higher in sandy soils where the biological soil crust (BSC) was removed and on calcrete where the BSC was buried under sand. The results indicated the importance of BSCs for C cycling in drylands and indicate that intensive grazing, which destroys BSCs through trampling and burial, will adversely affect C sequestration and storage (Thomas, 2012).

3.2.3 Rice paddies

Rice paddies are known to be sources of CH_4 (e.g., Linquist et al., 2012). Experiments measuring GHG emissions in rice paddies were conducted in Kenya (Tyler et al., 1988) and Zimbabwe (Nyamadzawo et al., 2013). In Kenya, the range of δ^{13}C in CH_4 for rice paddies was from −57 to −63 ‰ and δ^{13} CH_4 did not show any seasonal trend and did not indicate appreciable variability among two different strains of rice (Tyler et al., 1988). In Zimbabwe, intermittently saturated dambo rice paddies were a source of GHG and annual emissions from these rice paddies (150 day growing season and 126 kg

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of applied N ha^{-1}) were estimated as 2680 kg CO_2 ha^{-1} yr^{-1}, 12.5 kg CH_4 ha^{-1}, and 0.12 kg N_2O ha^{-1} (Nyamadzawo et al., 2013). The IPCC (2006) use a CH_4 emission factor of 1.30 kg CH_4 ha^{-1} day^{-1} for rice cultivation. The CH_4 emissions in the dambo rice paddies referred to here are much lower than the IPCC estimate (195 kg CH_4 ha^{-1} = 1.3 kg CH_4 ha^{-1} day^{-1} × 150 days). The corresponding IPCC (2006) N_2O EF is 0.3 % for rice cultivation and thus the N_2O emissions in the dambo rice paddies are also much lower than the IPCC estimate (0.40 kg N_2O–N ha^{-1} = 126 kg N ha^{-1} × 0.003; 0.63 kg N_2O ha^{-1}).

3.2.4 Vegetable gardens

Greenhouse gas emissions from soils in vegetable gardens in peri-urban areas of Burkina Faso (Lompo et al., 2012) and Niger (Predotova et al., 2010) were much higher than all other land uses, ranging from 73.3 to 132.0 Mg CO_2 ha^{-1} yr^{-1} and 53.4 to 177.6 kg N_2O ha^{-1} yr^{-1} (Tables 1 and S1). In Burkina Faso annual CO_2 and N_2O emissions from the garden soils were 68 to 85 % and 3 to 4 % of total C and N input, respectively (Lompo et al., 2012). The N_2O EFs (3 to 4 %) were higher than the IPCC default value of 1.0 % for all cropping systems (IPCC, 2006) and the global N_2O EF of vegetable fields (0.94 %) (Rezaei Rashti et al., 2015). The high N_2O EFs may be attributed to the large amount of applied N in vegetable gardens (2700–2800 kg N ha^{-1} yr^{-1}) since surplus N will stimulate N_2O production and also indirectly promote N_2O production by inhibiting biochemical N_2O reduction (e.g., Shcherbak et al., 2014; Kim et al., 2013). In vegetable gardens of Niger, a simple plastic sheet roofing and addition of ground rock phosphate to stored ruminant manure decreased N_2O gaseous losses by 50 % in comparison to dung directly exposed to the sun (Predotova et al., 2010). The authors argued that a decreased evaporation rate was behind this abating effect.

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3.2.5 Agroforestry

Soil CO_2 and N_2O emissions from African agroforestry were 38.6 Mg CO_2 ha^{-1} yr^{-1} and 0.2 to 26.7 kg N_2O ha^{-1} yr^{-1}, respectively (Tables 1 and S1). In agroforestry homegardens in Sudan, CO_2 (16.6 Mg CO_2 ha^{-1} from June to December) and N_2O emissions (17.3 kg N_2O ha^{-1} from June to December) accounted for two-thirds of total C output and one-third of total N output, respectively and the CO_2 and N_2O fluxes were positively correlated with soil moisture (Goenster et al., 2015). Improving fallow with N-fixing trees is a common practice in several areas of Africa since it provides additional N to the soil that can be utilised by the subsequent cash crop (e.g., Makumba et al., 2007; Chikowo et al., 2004; Dick et al., 2001). However, the practice is also thought to increase CO_2 and N_2O emissions compared to conventional croplands (Table 2). In an intercropping system with a N-fixing tree (*Gliricidia*) and maize in southern Malawi, soil C was depleted as a result of enhanced CO_2 emissions, with over 67 % of soil C lost over the first 7 years of intercropping (Kim, 2012). In Zimbabwe, N_2O emissions in improved-fallow agroforestry systems were 7 times higher than emissions in maize monoculture (Chikowo et al., 2004). In Senegal, soil collected under the N-fixing tree (*Acacia raddiana*) emitted significantly more N_2O than soil collected under the N-fixing crop (*Arachis hypogaea*) and non-N fixing tree (*Eucalyptus camaldulensis*) (Dick et al., 2006). In western Kenya, N_2O emissions increased after incorporation of fallow residues and emissions were higher after incorporation of improved-fallow legume residues than natural-fallow residues (Baggs et al., 2006; Millar and Baggs, 2004; Millar et al., 2004). It was found that N_2O emissions were positively correlated with residue N content (Baggs et al., 2006; Millar et al., 2004) and negatively correlated with polyphenol content and their protein binding capacity (Millar and Baggs, 2004), soluble C-to-N ratio (Millar and Baggs, 2005) and lignin content (Baggs et al., 2006). While high residue N content likely leads to more available soil N and consequently increased N_2O production (Baggs et al., 2006; Millar and Baggs, 2005; Millar et al., 2004), polyphenols and lignins are both resistant to decomposition and could result in

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N immobilization resulting in less labile soil N and less N_2O production (Baggs et al., 2006; Millar and Baggs, 2004). The type and quality of plant residue is likely to be an important control factor affecting N_2O emissions.

As in natural systems, improved fallow with N-fixing trees also results in increased N_2O emissions following rainfall events. In an incubation experiment in Uganda, N_2O emissions following simulated rainfall were a least 4 times larger for soils from under N-fixing trees (*Calliandra calothyrsus*) compared to soils with non-N fixing trees (*Grevillea robusta*) (Dick et al., 2001). Similarly, in Mali, N_2O emissions were around six times higher from improved fallow with N-fixing trees (*Gliricidia sepium* and *Acacia colei*) following a simulated rainfall event, compared with the emissions from soil under traditional fallow and continuous cultivation (Hall et al., 2006). Replacing traditional natural fallow with improved-fallow systems in the humid tropics of Kenya also increased N_2O emissions by up to 3.9 kg N_2O–N ha^{-1} over a 122 day maize cropping season (Millar et al., 2004).

3.3 Greenhouse gas emissions from land use change

Land-use change affects soil GHG emissions due to changes in vegetation, soil, hydrology and nutrient management (e.g., Kim and Kirschbaum, 2015) and the effects of land-use change on soil GHG emissions have been observed in woodlands and savannah. In Zimbabwe, clearing and converting woodlands to croplands increased soil emissions of CO_2, CH_4 and N_2O (Mapanda et al., 2012) and soil CO_2 emissions from the converted croplands were higher than *Eucalyptus* plantations established in former natural woodlands (Mapanda et al., 2010). In Republic of Congo, early-rotation changes in soil CO_2 efflux after afforestation of a tropical savannah with *Eucalyptus* were mostly driven by the rapid decomposition of savannah residues and the increase in *Eucalyptus* rhizospheric respiration (Nouvellon et al., 2012).

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3.4 Summary of greenhouse gas emissions in natural and agricultural lands in Africa

3.4.1 CO_2 emissions

Carbon dioxide emissions ranged from 3.3 to 130.9 Mg CO_2 ha^{-1} yr^{-1} in natural terrestrial systems and from −11.9 to 232.0 Mg CO_2 ha^{-1} yr^{-1} in aquatic systems. The area weighted average was 27.6±17.2 Mg CO_2 ha^{-1} yr^{-1} (Tables 1 and S1). Aquatic systems such as water bodies or water submerged lands were the largest source of CO_2 followed by forest, savannah, termite mounds and salt pans (Table 1). Soil CO_2 emissions in agricultural lands were similar to emissions from natural lands and ranged from 6.5 to 141.2 Mg CO_2 ha^{-1} yr^{-1} with an area weighted average of 23.0 ± 8.5 Mg CO_2 ha^{-1} yr^{-1} (Tables 1 and S2). Vegetable gardens were the largest sources of CO_2 emission largely due to the large C inputs, followed by agroforestry, cropland and rice fields (Tables 1 S2).

3.4.2 CH_4 emissions

Forest/plantation/woodland were sinks of CH_4 (−1.5 ± 0.6 kg CH_4 ha^{-1} yr^{-1}) and savannah/grassland, crop lands, termite mounds, and rice fields were low to moderate CH_4 sources (0.5–30.5 kg CH_4 ha^{-1} yr^{-1}). Stream/river and wetland/floodplain/lagoon/reservoir were high CH_4 sources (766.0–950.4 kg CH_4 ha^{-1} yr^{-1}) (Tables 1 and S1). The area weighted averages of CH_4 emissions from natural and agricultural lands were 43.0±5.8 and 19.5±5.6 kg CH_4 ha^{-1} yr^{-1}, respectively.

3.4.3 N_2O emissions and emission factor (EF)

Nitrous oxide emissions in natural lands ranged from −0.1 to 13.7 kg N_2O ha^{-1} yr^{-1} and the area weighted average was 2.5 ± 0.8 kg N_2O ha^{-1} yr^{-1} (Tables 1 and S1). Our study

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reveals that forest, plantation and woodland were the largest source of N_2O followed by rivers and wetlands, savannah and termite mounds (Table 1). Soil N_2O emissions in agricultural lands ranged from 0.051 to 177.6 $kg\,N_2O\,ha^{-1}\,yr^{-1}$ and the area weighted average was $4.5 \pm 2.2\,kg\,N_2O\,ha^{-1}\,yr^{-1}$ (Tables 1 and S2). The largest N_2O source in agricultural lands was vegetable gardens followed by agroforestry, cropland and rice fields (Table 1). The N_2O EF was 0.5 ± 0.2 % and 3.5 ± 0.5 % for cropland and vegetable gardens, respectively (Tables 1 and S1). The results indicate that the N_2O EF of African cropland is lower and the N_2O EF of African vegetable gardens is higher than IPCC default N_2O EF (1 %, IPCC, 2006).

The relationship between N input and N_2O emissions varied depending on N input level (Fig. 4). N_2O emissions increase slowly up to 150 $kg\,N\,ha^{-1}\,yr^{-1}$, after which emissions increase exponentially up to 300 $kg\,N\,ha^{-1}\,yr^{-1}$ (Fig. 5a). Consistent with van Groenigen (2010) N inputs of over 300 $kg\,N\,ha^{-1}\,yr^{-1}$ resulted in an exponential increase in emission (Fig. 5b), slowing to a steady state with N inputs of 3000 $kg\,N\,ha^{-1}\,yr^{-1}$. Overall, the relationship between N input and N_2O emissions shows a sigmoidal pattern (Fig. 5c). The observed relationship is consistent with the proposed hypothetical conceptualization of N_2O emission by Kim et al. (2013) showing a sigmoidal response of N_2O emissions to N input increases. The results suggest that N inputs over 150 $kg\,N\,ha^{-1}\,yr^{-1}$ may cause an abnormal increase of N_2O emissions in Africa. The relationship between N input and N_2O emissions show that the lowest yield-scaled N_2O emissions were reported for N application rates ranging from 100 to 150 $kg\,N\,ha^{-1}$ (Fig. 6). The results are in line with the global meta-analysis of Philiber et al. (2012) who showed that from an N application rate $\sim$ 150 $kg\,N\,ha^{-1}$ the increase in N_2O emissions is not linear but exponential.

3.4.4 CO_2 eq emission

Carbon dioxide eq emission (including CO_2, CH_4 and N_2O) in natural lands ranged from 11.7 to 121.3 $Mg\,CO_2\,eq\,ha^{-1}\,yr^{-1}$ and the area weighted average of CO_2 eq emissions

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(excluding salt pans) was $29.9 \pm 22.5\,Mg\,CO_2\,eq\,ha^{-1}\,yr^{-1}$ (Table 1). Water bodies or water submerged lands such as rivers and wetlands were the largest source of CO_2 eq emissions followed by forest/plantation/woodland, savannah/grassland and termite mounds (Table 1). Carbon dioxide eq emissions in agricultural lands ranged from 7.3 to $26.1\,Mg\,CO_2$ eq $ha^{-1}\,yr^{-1}$ and had an area weighted average of CO_2 eq emissions (excluding vegetable gardens and agroforestry) of $25.6 \pm 12.4\,Mg\,CO_2\,eq\,ha^{-1}\,yr^{-1}$ (Table 1).

Total CO_2 eq emissions in natural lands (excluding salt pans) were $43.4 \pm 9.3\,Pg\,CO_2\,eq\,yr^{-1}$ with forest/plantation/woodland the largest source followed by savannah/grassland, stream/river, wetlands/floodplains/lagoons/reservoir, and termite mounds (Table 1). Total CO_2 eq emissions in agricultural lands (excluding vegetable gardens and agroforestry) were $13.5 \pm 3.4\,Pg\,CO_2\,eq\,yr^{-1}$ with crop land the largest source followed by rice fields (Table 1). Overall, total CO_2 eq emissions in natural and agricultural lands were $56.9 \pm 12.7\,Pg\,CO_2\,eq\,yr^{-1}$ with natural and agricultural lands contributing 76.3 and 23.7 %, respectively.

3.5 Suggested future studies

Despite an increasing number of published estimates of GHG emissions in the last decade, there remains a high degree of uncertainty about the contribution of AFOLU to emissions in SSA. To address this and reduce the uncertainty surrounding the estimates, additional GHG emission measurements across agricultural and natural lands throughout Africa are urgently required. Identifying controlling factors and their effects on GHG fluxes is a pre-requisite to enhancing our understanding of efflux mechanisms and a necessary step towards scaling up the field-scale data to landscape, national and continental scales. It is important to know how GHG fluxes can be affected by management practices and natural events such as logging (e.g., Yashiro et al., 2008), thinning (e.g., Yohannes et al., 2013), storms (e.g., Vargas, 2012), pest outbreaks (e.g., Reed et al., 2014), fires (e.g., Andersson et al., 2004), and wood encroachment (e.g., Smith

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and Johnson, 2004) in natural terrestrial systems and changing discharge (e.g., Wang et al., 2013) and water table (e.g., Yang et al., 2013) in aquatic systems. It is also important in agricultural lands to know how GHG fluxes are affected by management factors such as soil compaction (e.g., Ball et al., 1999), tillage (e.g., Sheehy et al., 2013), removal of crop residues (Jin et al., 2014), incorporation of crop residues and synthetic fertilizer (e.g., Nyamadzawo et al., 2014a), N input (e.g., Hickman et al., 2015) and crop type (e.g., Masaka et al., 2014). However, because management and soil physical/chemical interactions cause different responses in soil GHG emissions (e.g. Pelster et al., 2012), it is critical to measure these interaction effects in the African context. The effect of predicted climatic change in Africa such as increased temperature (e.g., Dijkstra et al., 2012), changing rainfall patterns (e.g., Hall et al., 2006), increase in droughts incidence (e.g., Berger et al., 2013), rewetting effects (e.g., Kim et al., 2012) and increased atmospheric CO_2 concentration (e.g., Lane et al., 2013) also require further testing using laboratory and field experiments.

Where possible studies should seek to identify and separate driving processes contributing to efflux of soil CO_2 (e.g., autotrophic and heterotrophic sources), CH_4 (e.g., methanogenesis and methanotrophy) and N_2O (e.g., nitrification, denitrification, nitrifier denitrification). This is important because the consequences of increasing GHG emissions depend on the mechanism responsible. For example, if greater soil CO_2 efflux is primarily due to autotrophic respiration from plant roots, then it simply reflects greater plant growth. If however, it is due to heterotrophic microbial respiration of soil organic carbon then it represents a depletion of soil organic matter and a net transfer of C from soil to the atmosphere. Currently there are very few studies that differentiate these sources making it impossible to truly determine the consequences and implications on changes in soil GHG efflux.

Land-use change has been recognized as the largest source of GHG emission in Africa (Valentini et al., 2014). Hence, various types of conversion from natural lands to different land-use types should be assessed to know how these changes may affect the GHG budget (e.g., Kim and Kirschbaum, 2015). The focus of the assessment should be

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on deforestation and wetland drainage, followed by a conversion to agricultural lands, since they are dominant types of land-use change in Africa (Valentini et al., 2014).

3.6 Strategic approaches for data acquisition

A strategic plan for acquisition of soil GHG emission data in sub-Saharan Africa is required. The success of any plan is dependent on long-term investment, stakeholder involvement, technical skill and supporting industries, which have not always been available in the region (Olander et al., 2013; Franks et al., 2012). A major challenge is to address the lack of consistency in the various methodologies used to quantify GHG emissions (Rosenstock et al., 2013). Relatively low cost and simple techniques can be used to determine GHG emission estimates in the first instance. Soil CO_2 fluxes can be quantified with a soda lime method (Tufekcioglu et al., 2001; Cropper et al., 1985; Edwards, 1982) or an infra-red gas analyzer (Bastviken et al., 2015; Verchot et al., 2008; Lee and Jose, 2003) and these do not require advanced technology or high levels of resource to undertake. Later, other GHG such as N_2O and CH_4 fluxes in addition to CO_2 flux can be measured with more advanced technology (e.g., gas chromatography, photo-acoustic spectroscopy, or laser gas analyzers). Initially, the measurement can be conducted using manual gas chambers with periodical sampling frequencies. The sampling interval can be designed so that it is appropriate to the particular type of land-use or ecosystem, management practices and/or for capturing the effects of episodic events (e.g., Parkin, 2008). For example, GHG measurement should be more during potentially high GHG emission periods following tillage and fertilizer applications and rewetting by natural rainfalls or irrigation. With more advance technology and utilisation of automatic chamber systems measurements can be conducted at a much high frequency with relative ease.

In order for the challenges associated with improving our understanding of GHG emissions from African soils it is critical to establish networks of scientists and scientific bodies both within Africa and across the world. Good communication and collaboration between field researchers and the modelling community should also be established

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during the initial stages of research, so results obtained from field scientists can be effectively used for model development and to generate hypotheses to be tested in the field and laboratory (de Bruijn et al., 2009).

Furthermore, lessons learned from scientific experiments can only really be successfully implemented by farmers if local stakeholders are involved from the start and throughout (see for example Stringer et al., 2012). Interviews, focus-groups, on-site or farm demonstrations, local capacity building training, local farmers and extension staff can all improve dialogue and understanding between local communities and scientists, ultimately improving the likelihood of successful GHG emission and mitigation strategies. These will equip local researchers and stakeholders (including farmers and extension staff) with state of art methodologies and help motivate them to develop their GHG mitigation measures and assist them in understand their roles and contributions to global environmental issues.

4 Conclusions

This paper synthesizes the available data on GHG emissions from African agricultural and natural lands. Emissions of CO_2, CH_4 and N_2O in a variety of environments (forests, savannahs, termite mounds, salt pans, agricultural areas and water bodies) were considered. Two broad conclusions can be drawn from the work. The first one is that African natural and agricultural lands may be a significant source of GHG and that the emissions may increase through land-use change and management strategies. Secondly, there are huge research gaps. Africa is a vast continent, with a multitude of land uses, climates, soils and ecosystems. Field-based data on soil GHG emissions from many areas, soil types and environments are extremely sparse and as a result our understanding of Africa's contribution to global GHG emissions remains incomplete and highly uncertain. There is an urgent need to develop and agree on a strategy for addressing this data gap. The strategy may involve identifying priorities for data

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acquisition, utilizing appropriate technologies, and establishing networks and collaboration.

Appendix: A blog for open discussion and web based open databases

We have created a blog entitled "Greenhouse gas emissions in Africa: study summary and database" (http://ghginafrica.blogspot.com/) and an open-access database, which can be modified by the users, entitled "Soil greenhouse gas emissions in Africa database" (linked in the blog) based on this review. In the blog, we have posted a technical summary of each section of this review, where comments can be left under the posts. The database contains detailed information on the studies reported on GHG emissions, such as ecosystem and land use types, location, climate, vegetation type, crop type, fertilizer type, N input rate, soil properties, GHGs emission measurement periods, N_2O EF, and corresponding reference. The database is hosted in web based spreadsheets and is easily accessible and modified. The authors do not have any relationship with the companies currently being used to host the blog and databases.

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Table 1. Summary of greenhouse gas carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) emissions and CO_2 equivalents (CO_2 eq) in natural and agricultural lands in sub-Saharan African countries. Mean ± standard error (number of data) are shown.

Type	Area (Mha)	CO_2 emission Mg CO_2 ha^{-1} yr^{-1}	CH_4 emission kg CH_4 ha^{-1} yr^{-1}	N_2O emission kg N_2O ha^{-1} yr^{-1}	N_2O emission factor %	CO_2 eq emission Mg CO_2 eq ha^{-1} yr^{-1}	Total CO_2 eq emission Pg CO_2 eq yr^{-1}
Forest/plantation/ woodland	740.6[a]	32.0 ± 5.0 (34)	−1.5 ± 0.6 (15)	4.2 ± 1.5 (10)	[d]	34.0 ± 5.7	25.2 ± 4.2
Savannah/grassland	638.9[a]	15.5 ± 3.8 (11)	0.5 ± 0.4 (18)	0.6 ± 0.1 (6)	[d]	15.8 ± 3.8	10.1 ± 2.4
Stream/river	28.2[a]	78.1 ± 13.2 (27)	436.3 ± 133.8 (24)	1.6 ± 0.3 (17)	[d]	93.4 ± 17.9	2.8 ± 1.0
Wetlands/floodplains/ lagoons/reservoir	43.8[a]	96.6 ± 31.0 (7)	950.4 ± 350.4 (5)	2.0 ± 1.5 (2)	[d]	121.3 ± 39.7	5.3 ± 1.7
Termite mounds	0.97[b]	11.6 ± 6.2 (3)	2.3 ± 1.1 (3)	0.01 (1)	[d]	11.7 ± 6.3	0.01 ± 0.01
Salt pan	[d]	0.7 (1)	[d]	[d]	[d]	[d]	[d]
Total natural lands[h]	1452.5	27.6 ± 2.9 [e]	43.0 ± 5.8 [e]	2.5 ± 0.4 [e]	[d]	29.9 ± 22.5 [e]	43.4 ± 9.3 (76.3 %)[g]
Cropland	468.7[a]	23.4 ± 5.1 (45)	19.3 ± 4.2 (26)	4.0 ± 1.5 (83)	0.5 ± 0.2 (24)	26.1 ± 6.0	12.2 ± 2.8
Rice field	10.5[c]	6.5 (1)	30.5 (1)	0.19 (1)	[d]	7.3	1.3 ± 0.6
Vegetable gardens	[d]	96.4 ± 10.2 (5)	[d]	120.1 ± 26.1 (5)	3.5 ± 0.5 (2)	[d]	[d]
Agroforestry	190[f]	38.6 (1)	[d]	4.7 ± 2.2 (15)	[d]	[d]	[d]
Total agricultural lands[i]	479.2	23.0 ± 8.5 [e]	19.5 ± 5.6 [e]	4.5 ± 2.2 [e]	[d]	25.6 ± 12.4 [e]	13.5 ± 3.4 (23.7 %)[g]
Total natural and agricultural lands[j]	1931.7						56.9 ± 12.7

[a] GlobCover 2009.
[b] 0.07 % of savanna and rainforest (Brümmer et al., 2009).
[c] FAO STAT (http://faostat3.fao.org/home/E), year 2012.
[d] No data available.
[e] Area weighted average.
[f] Zomer et al. (2009).
[g] Contribution to CO_2 eq emission in total natural and agricultural lands.
[h] Except salt pan.
[i] Except vegetable gardens and agroforestry.
[j] Except salt pan, vegetable gardens and agroforestry.

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Table 2. Summary of the effect of management practices on greenhouse gas (GHG) emissions.

Land use/ecosystem type	Management practices	Impact on GHG CO_2	N_2O	CH_4	Country (data source)
Forest/plantation/Woodland	Burning	+			Ethiopia; Andersson et al. (2004)
	Thinning	+			Ethiopia; Yohannes et al. (2013)
	Land use change (cleaning and conversion to croplands)	+	+	+	Zimbabwe; Mapanda et al. (2012, 2010)
	Flooding			+	Cameroon; Macdonald et al. (1998); Republic of Congo; Tathy et al. (1992); Mali; Delmas et al. (1991)
Savannah/grassland	Burning	•	•	•	Republic of Congo; Castaldi et al.(2010); Delmas et al. (1991); South Africa; Zepp et al. (1996)
	Land use change (cleaning and conversion to croplands)	+			[a]Republic of Congo; Nouvellon et al. (2012)
Croplands	Increase in N fertilization rate		+		Kenya; Hickman et al. (2015)
	Type of synthetic fertilizer		•		Madagascar; Rabenarivo et al. (2014)
	Application of plant residues		–		Tanzania; Sugihara et al. (2012); [b]Madagascar; Rabenarivo et al. (2014)
		+	+		Kenya; Kimetu et al. (2006); [c]Ghana; Frimpong et al. (2012)
	Crop residues + N fertilizer		+		[d]Zimbabwe; Nyamadzawo et al. (2014a, b)
			–		[e]Zimbabwe, Ghana and Kenya; Gentile et al. (2008)
	Combination of synthetic and organic fertilizers	+	–		[f]Zimbabwe; Mapanda et al. (2011)
			–		[g]Mali; Dick et al. (2008)
	Crop type	•			[h]Uganda; Koerber et al. (2009)
			–		[i]Zimbabwe; Masaka et al. (2014)
	Introducing N fixing crops in rotations		–		Mali; Dick et al. (2008)
	Direct seeding mulch-based		•		Madagascar; Chapuis-Lardy et al. (2009)
	Hand-ploughing after harvesting		•		Madagascar; Chapuis-Lardy et al. (2009)
	Intensive grazing	+			Botswana; Thomas (2012)
Vegetable gardens	Plastic cover for ruminant manure		–		Niger; Predotova et al. (2010)
	Incorporation of fallow residues		+		Kenya; Baggs et al. (2006); Millar and Baggs (2004); Millar et al. (2004)
Agroforestry	Improving fallow with N-fixing crops		+		Zimbabwe; Chikowo et al. (2004)
	Cover crops		+		Kenya; Millar et al. (2004)
	N-fixing tree species	+	+		Malawi; Kim (2012); Makumba et al. (2007); Senegal; Dick et al. (2006)

[a] U+DAP instead U+NPK.
[b] N_2O study.
[c] Low C : N ratio clover residues compared to high C : N ratio barley residues.
[d] Application of ammonium nitrate with manure to maize (*Zea mays* L.) and winter wheat (*Triticumaestivum* L.) plant residues.
[e] Plant residues of maize, calliandra, and tithonia + urea.
[f] Mixed application of composted manure and inorganic fertilizer (AN).
[g] Manure and urea.
[h] Lettuces vs. cabbages vs. beans.
[i] Tomatoes vs. rape.
+ indicates increasing, • indicates no change, and – indicates decreasing.

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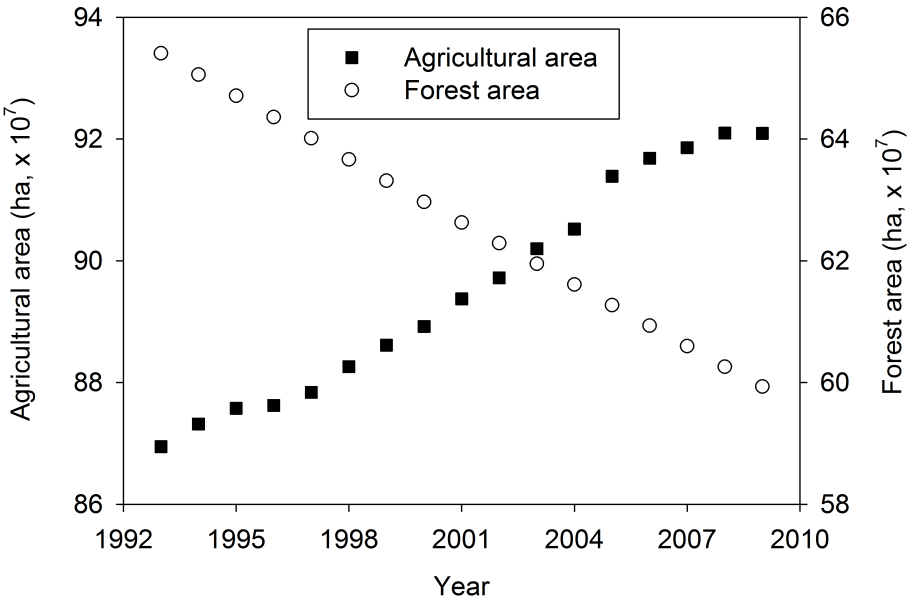

Figure 1. Change of areas of agricultural land and forest in Africa. Data source: FAOSTAT, http://faostat.fao.org/site/377/default.aspx#ancor, access 23 April 2015.

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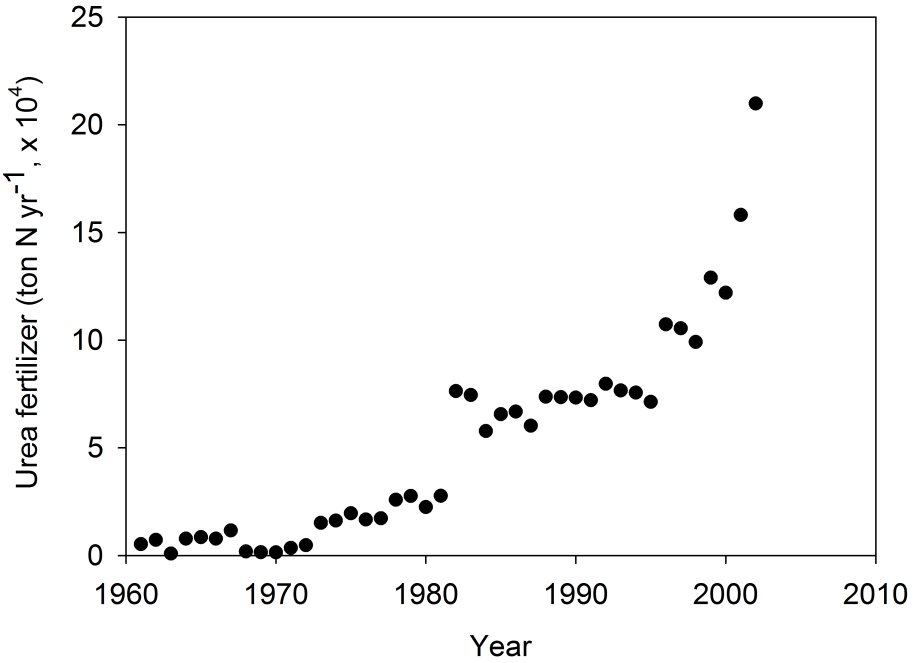

Figure 2. Change of use of urea fertiliser in Africa. Data source: FAOSTAT, http://faostat.fao.org/site/422/default.aspx#ancor, access 23 April 2015.

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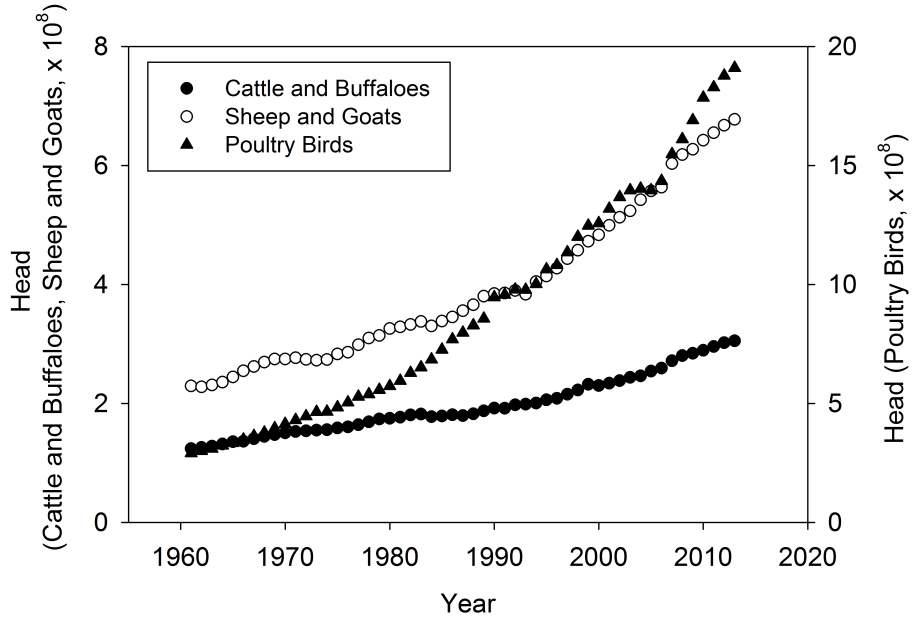

Figure 3. Trends of African livestock population. Data source: FAOSTAT, http://faostat3.fao.org/faostat-gateway/go/to/download/Q/QA/E, access 23 April 2015.

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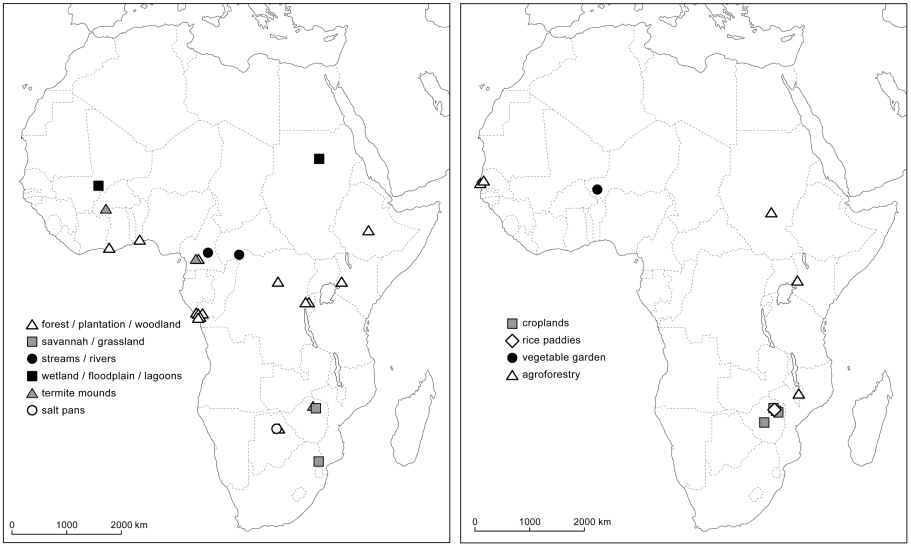

Figure 4. Maps showing study sites of CO_2, CH_4 and N_2O fluxes.

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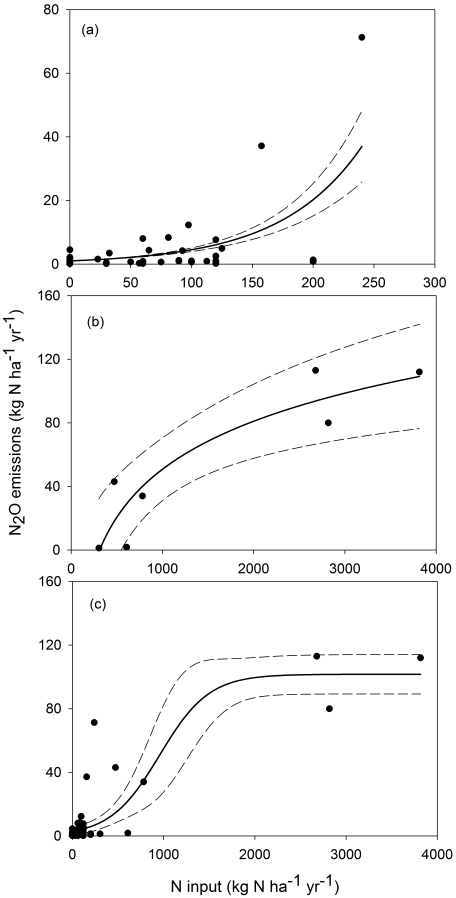

Figure 5. Relationship between nitrogen (N) input and nitrous oxide (N_2O) emissions observed in Africa. N input ranged from 0 to 300 **(a)**, 300 to 4000 **(b)** and 0 to 4000 kg N ha^{-1} yr^{-1} **(c)**. The dashed lines indicate 95 % confidence intervals.

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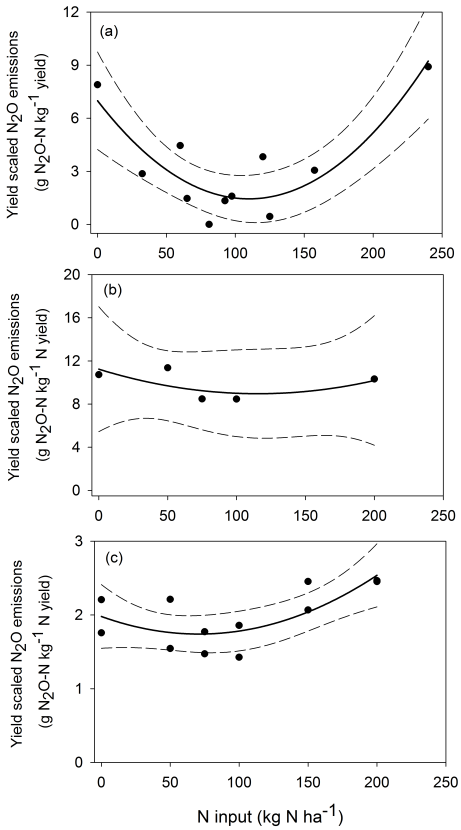

Figure 6. Relationship between nitrogen (N) input and yield scaled nitrous oxide (N_2O) emissions. Grain type: **(a)** rape (*Brassica napus*) and **(b)** and **(c)** maize (*Zea mays* L.). Data sources: **(a)** from Nyamadzawo et al. (2014), **(b)** from Hickman et al. (2014) and **(c)** from Hickman et al. (2015). The dashed lines indicate 95 % confidence intervals. Note the different scales across panels.

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