Title

Greenhouse gas emissions from natural ecosystems and agricultural lands in sub-Saharan Africa: synthesis of available data and suggestions for further research

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Abstract

This paper summarizes currently available data on greenhouse gas (GHG) emissions from African natural ecosystems and agricultural lands. The available data are used to synthesise current understanding of the drivers of change in GHG emissions, outline the knowledge gaps and suggest future directions and strategies for GHG emission research. GHG emission data were collected from 75 studies conducted in 22 countries (n=244) in sub-Saharan Africa (SSA). Carbon dioxide (CO_2) emissions were by far the largest contributor to GHG emissions and global warming potential (GWP) in SSA natural terrestrial systems. CO_2 emissions ranged from 3.3 to 57.0 Mg CO_2 ha^{-1} yr^{-1}, methane (CH_4) emissions ranged from -4.8 to 3.5 kg ha^{-1} yr^{-1} [-0.16 to 0.12 Mg CO_2 equivalent (eq) ha^{-1} yr^{-1}], and nitrous oxide (N_2O) emissions ranged from -0.1 to 13.7 kg ha^{-1} yr^{-1} (-0.03 to 4.1 Mg CO_2 eq 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 from natural terrestrial systems. In aquatic systems, CO_2 was the largest contributor to total GHG emissions, ranging from 5.7 to 232.0 Mg CO_2 ha^{-1} yr^{-1}, followed by -26.3 to 2741.9 kg CH_4 ha^{-1} yr^{-1} (-0.89 to 93.2 Mg CO_2 eq ha^{-1} yr^{-1}) and 0.2 to 3.5 kg N_2O ha^{-1} yr^{-1} (0.06 to 1.0 Mg CO_2 eq ha^{-1} yr^{-1}). Rates of all GHG emissions from aquatic systems were affected by type, location, hydrological characteristics, and water quality. In croplands, soil GHG emissions were also dominated by CO_2, ranging from 1.7 to 141.2 Mg CO_2 ha^{-1} yr^{-1}, with -1.3 to 66.7 kg CH_4 ha^{-1} yr^{-1} (-0.04 to 2.3 Mg CO_2 eq ha^{-1} yr^{-1}) and 0.05 to 112.0 kg N_2O ha^{-1} yr^{-1} (0.015 to 33.4 Mg CO_2 eq ha^{-1} yr^{-1}). N_2O emission factors (EF) ranged from 0.01 to 4.1%. Incorporation of crop residues or manure with inorganic fertilizers invariably resulted in significant changes in GHG emissions but results were inconsistent as the magnitude and direction of changes were differed by gas. Soil GHG emissions from 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} (15.9 to 52.9 Mg CO_2 eq 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} y^{-1} and 0.2 to 26.7 kg N_2O ha^{-1} yr^{-1} (0.06 to 8.0 Mg CO_2 eq ha^{-1} yr^{-1}), respectively. Improving fallow with nitrogen (N)-fixing trees led to increased CO_2 and N_2O emissions compared to conventional croplands. The type and quality of plant residue in the fallow is an important control on how CO_2 and N_2O emissions are affected. 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 eq emissions from SSA natural ecosystems and agricultural lands were 56.9 ± 12.7 x 10^9 Mg CO_2 eq yr^{-1} with natural ecosystems and agricultural lands contributing 76.3% and 23.7%, respectively. Additional GHG emission measurements are urgently required to reduce uncertainty on annual GHG emissions from the different land uses and identify major control factors and mitigation options for low-emission development. A common strategy for addressing this data gap that may include identifying priorities for data acquisition, utilizing appropriate technologies, and involving international networks and collaboration.

Key words: Africa, greenhouse gas, carbon dioxide, methane, nitrous oxide, natural ecosystems, agricultural lands

1. Introduction

Global greenhouse gas (GHG) emissions were estimated to be 49 x 10^9 Mg CO_2 eq in 2010 (IPCC, 2014), with approximately 21.2 – 24% (10.3 – 12 x 10^9 Mg 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 x 10^9 Mg CO_2 eq yr^{-1} in 2010 (FAOSTAT, 2014; Tubiello et al., 2013), with approximately 4.3 – 5.5 x 10^9 Mg CO_2 eq yr^{-1} attributable to land-use change (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). For example, CO_2 eq emissions from 12 river channels in SSA and wetlands of the

Congo River were 3.3 x 10^9 Mg CO_2 eq per year, equivalent to *c.* 25% of the global terrestrial and ocean carbon sink (Borges et al., 2015). Nitrous oxide emissions in SSA contribute between 6 – 19% of the global total, and changes in soil N_2O fluxes in SSA drive large inter-annual variations in tropical and subtropical N_2O sources (Thompson et al., 2014; Hickman et al., 2011). 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) in Africa (Bouwman et al., 2009 and 2013). The increasing trend in N application rates is expected to cause a twofold increase in agricultural N_2O emissions in the continent 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, rice paddy fields and termite mounds are likely sources of CH_4, while seasonally dry forests and savannahs are typically CH_4 sinks (Valentini et al., 2014).

Interpretation of GHG emissions from soils and terrestrial water bodies is complex because of the multiple, sometimes competing, biological, chemical and physical processes affecting fluxes. Spatial and temporal variability in GHG fluxes is also high and challenging to capture with direct measurement. This in turn makes reliable annual GHG flux estimates from different soils, land uses and regions quite rare in SSA. Net soil CO_2 flux is largely a product of autotrophic respiration derived from plant roots and heterotrophic respiration of soil organic matter (Raich and Schlesinger, 1992). Soil CO_2 flux provides an integrated result of biological CO_2 production throughout the soil column, changes in soil CO_2 diffusivity in the soil profile, and in some areas geological processes (Raich and Schlesinger, 1992). Net CH_4 flux is the result of the balance between methanogenesis (microbial production under anaerobic conditions) and methanotrophy (microbial consumption) (Dutaur and Verchot, 2007). Methanogenesis occurs via the anaerobic degradation of organic matter by methanogenic archaea within the archaeal phylum *Euryarchaeota* (Thauer, 1988). Methanotrophy occurs by methanotrophs metabolizing CH_4 as their source of carbon and energy (Hanson and Hanson, 1996). Soil N_2O in produced through three main processes such as nitrification (Kowalchuk and Stephen, 2001), denitrification (Knowles, 1982) and nitrifier denitrification (Wrage et al., 2001). Identifying controlling factors and their effects on all GHGs is therefore complex and challenging but a pre-requisite to enhancing our understanding of efflux mechanisms and accurate quantification of GHG emissions. Environmental factors such as soil properties (e.g., soil type, carbon and nutrients; Pelster et al., 2012), climate characteristics (e.g., temperature, rainfall, drought; Dijkstra et al., 2012) and vegetation type (e.g., crop or forest types; Masaka et al., 2014) also affect GHG fluxes. Management practices can also play important roles in controlling GHG fluxes. The controlling management practices include land-use change (Kim and Kirschbaum, 2015), logging (Yashiro et al., 2008), changing water discharge (Wang et al., 2013), soil compaction (Ball et al., 1999), tillage (Sheehy et al., 2013), removal of crop residues (Jin et al., 2014) and N input (whether organic or inorganic) (Hickman et al., 2015).

Our current understanding of GHG emissions in SSA 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 (i.e. type of emission drivers: natural factors or anthropic ones) 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 emissions, 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 in natural ecosystems and agricultural lands in SSA; create an inventory of information from studies on emissions; and select priority topics for future GHG emission studies in natural ecosystems and agricultural lands in SSA.

2. Methodology

2.1. Data collection and analyses

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, N_2O) as search terms (using Web of Science and Google Scholar; 1960-2015). These criteria yielded 310 peer-reviewed papers. To produce the quantitative summary of GHG emissions, 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 75 studies, conducted in 22 countries (n=244) in SSA were used and were further categorized as GHG emission in natural ecosystems [n=117; Supplementary Information (SI) Table 1] and agricultural lands (n=127; SI Table 2) (Fig. 2). The category of GHG emissions in natural ecosystems were further divided into emissions from natural terrestrial systems [forest/ plantation/woodland (n=55), savannah/grassland (n=31), termite mounds (n=5), and salt pans (n=1)] and aquatic systems [streams/rivers (n=14), wetlands/ floodplains /lagoons/ reservoirs/lakes (n=11)] (Table 1). Greenhouse gas emissions in agricultural lands were subdivided into emissions from cropland (n=105), rice paddies (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. To allow comparison between different GHG emissions CH_4 and N_2O emissions were converted to CO_2 eq assuming a 100 year global warming potential and values of 34 and 298 kg CO_2 eq for CH_4 and N_2O, respectively (IPCC, 2013). 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 follows:

$$N_2O\ EF(\%) = \frac{N_2O\ emission_{N\ treatment} - N_2O\ emission_{control}}{N\ input} \times 100 \qquad [1]$$

where, *N_2O EF (%)* is N_2O emission factor, $N_2O\ emission_{N\ treatment}$ is N_2O emission in N input, $N_2O\ emission_{control}$ 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. To assess data quality of the cited studies we used the criteria (ranked from "very poor" to "very good") suggested by Rochette and Eriksen-Hamel (2008). These were originally intended for chamber N_2O measurements but are equally applicable to field-based CO_2 and CH_4 chamber measurements. We went through the methods of the papers used in the study (only those for terrestrial emissions, since these criteria are not applicable for aquatic systems) where there was sufficient detail in the methods section. We categorized the studies as three different groups: the methods are 1) *poor to very poor*, 2) *marginal* and 3) *good*. Studies that were ranked "poor" on 3 or more criteria, or "very poor" on 2 or more criteria were categorized as the methods were *poor to very poor*. In addition, we took into account the importance of sampling frequency (Barton et al., 2015) and sampling periods. Studies estimating annual GHG emissions with a sampling frequency lower than biweekly (i.e., less than 2 times per month) and sampling periods of less than 6 months (i.e., covering both rainy and dry seasons) were categorized as the methods were *poor to very poor*. Studies that were ranked as "poor" on 2 criteria, or "very poor" on 1 criterion, or with insufficient details on the methods were ranked as *marginal*. The *good* studies were those with only 1 "poor" ranking, sufficient detail and a sampling frequency of every 2 weeks or more frequent.

2.2. Statistical analyses

To determine the relationship between annual soil CO_2 emissions and edaphic and climatic factors (e.g., soil pH, soil bulk density, soil organic carbon (SOC), total N, and annual average air temperature and rainfall) in SSA natural terrestrial systems and agricultural lands, we used a Pearson correlation analysis. The compiled datasets were used to examine the best model fit selection 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. Adjusted coefficients of determination (adjusted R^2) of fitted parametric models were used as criteria for model selection: the model with the highest adjusted R^2 was selected. Statistical significance was considered at the critical level of 5%. 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).

3 Results and Discussion

3. 1 Summary of greenhouse gas emissions

3. 1. 1 CO_2 emissions

Carbon dioxide emissions ranged from 3.3 to 130.9 Mg CO_2 ha^{-1} y^{-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} (Table 1 and SI Table 1). 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} (Table 1 and SI Table 2). Vegetable gardens were the largest sources of CO_2 emission largely due to the excessive C inputs. However, this conclusion was based on two studies that used photoacoustic spectroscopy, which has been found have mixed results due to cross sensitivities between the various GHG and water vapour (Rosenstock et al., 2013; Iqbal et al., 2013), suggesting that vegetable gardens require further study. The next largest sources of emissions were agroforestry, cropland and then rice production systems (Table 1 and SI Table 2).

Observed annual soil CO_2 emissions in African natural terrestrial systems and agricultural lands showed significant correlations with annual mean air temperature (r=-0.322, P=0.01), annual rainfall (r=0.518, P <0.001), SOC (r=0.626, P<0.001), and soil total N content (r= 0.849, P <0.001) (Table 2). The negative relationship between annual soil CO_2 emissions and annual mean air temperature was unexpected since positive correlations between soil CO_2 flux and temperature are well established (e.g., Bond-Lamberty and Thomson, 2010). We speculate that the generally high temperatures, and poor quality, of many African soils mean that air temperature increases frequently result in vegetation stress and/or soil aridity, hindering root and soil microbial activities (root and microbial respiration) and subsequent soil CO_2 flux (e.g., Thomas et al., 2011). This would account for the negative relationship observed between annual mean air temperature and annual soil CO_2 emissions but this remains a largely untested hypothesis that deserves further exploration.

Many of these estimates are based on short-term, infrequent or poor quality sampling (Table S1) suggesting that the uncertainties are likely much greater than the provided standard error. This is not meant as a critique of these studies, as many of them were specifically designed to answer specific research questions about the effects of various factors on emission rates rather than determining the cumulative annual emissions. However, given the lack of other data, these still provide the “best guess” for cumulative emissions.

3. 1. 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}) (Table 1 and Table 1 in supplementary material). 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. As with studies on CO_2 emissions, many of these studies used only infrequent or poor sampling methodologies (Table S1), and there is a high degree of uncertainty surrounding the estimates.

3. 1. 3 N_2O emissions and emission factor (EF)

Nitrous oxide emissions in natural ecosystems 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} (Table 1 and SI Table 1). Our study reveals that forest, plantation and woodland were the largest source of N_2O followed by rivers and wetlands, savannah and termite mounds in natural ecosystems (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 ± 2.2 kg N_2O ha^{-1} yr^{-1} (Table 1 and SI Table 2). 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 (Table 1 and SI Table 1). The N_2O EF of cropland is lower and the N_2O EF of vegetable gardens is higher than IPCC default N_2O EF (1%, IPCC, 2006). The number of studies on N_2O emissions in SSA is, however, particularly low (n=14), with some questions regarding the quality of the methods (Table S1) in some of these studies, and there are significant regional gaps leading to large uncertainties in the conclusions that can be currently drawn.

N_2O emissions were significantly affected by N input levels (Fig. 3). 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. 3 (A)). Consistent with earlier work by van Groenigen (2010) N inputs of over 300 kg N ha^{-1} yr^{-1} resulted in an exponential increase in emission (Fig. 3 (B)), 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. 3 (C)). 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 SSA. 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. 4). The results are in line with the global meta-analysis of Philiber et al. (2012) who showed that from an N application rate ~150 kg N ha^{-1} the increase in N_2O emissions is not linear but exponential.

3. 1. 4 CO_2 eq emission

Carbon dioxide eq emission (including CO_2, CH_4 and N_2O) in natural ecosystems 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 (excluding salt pans) was 29.9 ± 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 equivalent 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 due to lack of data) of 25.6 ± 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 ± 9.3 x 10^9 Mg 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 ± 3.4 x 10^9 Mg 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 ecosystems and agricultural lands were 56.9 ± 12.7 x 10^9 Mg CO_2 eq yr^{-1} with natural ecosystems and agricultural lands contributing 76.3% and 23.7%, respectively.

3.1.5 Data quality assessment

For the purposes of this study more than half of the 75 studies cited in the study were categorized as having *poor to very poor* methods, 19 studies were *marginal* and 14 studies were *good* (Table S1 and S2). The primary reasons the studies were ranked as *poor to very poor* were because sampling periods were too short for calculating annual emissions (i.e., less than or only one season of data), sampling frequency was too low (i.e., monthly or less), or a combination of poor methods with the sample collection, primarily insufficient samples per gas collecting chamber and very long chamber deployment times. As mentioned earlier, many of these studies were undertaken to address a specific research question, not determine annual, cumulative emissions. Therefore, the degree of uncertainty around cumulative emissions is likely higher than what is indicated in Table 1.

3.2 Sources and drivers of greenhouse gas emissions in Africa

3.2.1 Greenhouse gas emissions in natural ecosystems

Natural terrestrial systems

A range of factors affect direct emissions of soil CO_2 in SSA natural terrestrial systems such as natural forest, plantation, woodland, savannah, grassland, termite mounds and salt pans. These factors can be grouped into i) climatic, ii) edaphic, iii) vegetation and iv) human interventions via land management (Table 3 and 4). Data on the effects of these factors on GHG emissions are variable, with some factors much less well understood than others. In almost all cases data are limited to a few studies, and there are areas where there has been no research. This lack of data hinders our ability to estimate the contribution of African landscapes to global GHG emissions.

Soil CO_2 emissions were related to both soil moisture and temperature in forest systems (Table 3). 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℃ 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 2mm rainfall event and 1.95 after a 50mm 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).

Increased GHG emissions following rewetting of dry soil were observed in various regions in SSA (Table 3). Two broad mechanisms responsible for increased soil GHG flux following rewetting of dry soil 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 influencing gas flux, including infiltration, reduced diffusivity, and gas displacement in the soil (e.g., Kim et al., 2012b). 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). 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 was strongly stimulated by addition of rainfall in a South African savannah (Fan et al., 2015; Zepp et al., 1996). 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). The results suggest that soil rewetting has a significant impact on GHG emissions in SSA.

Soil physical (e.g., bulk density, porosity and soil texture) and chemical properties (e.g., pH, C and N) also affect soil GHG emissions (e.g., Saggar et al., 2013; Smith, 2010; Snyder et al., 2009) (Table 3). 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, 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 (Table 3). 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 savannah grasslands in 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 in forests, litterfall accounted for most of the age-related trends after the first year of growth, with litter decomposition producing 44% of soil CO_2 flux in the oldest stand (Nouvellon et al., 2012), 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 management such as burning, which is a common practice in SSA, and thinning, affects GHG emissions (Table 4). 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 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 can also 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.

There is a particular paucity of data on sources and sinks of CH_4 in African natural terrestrial systems. 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 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). Termite mounds are known sources of CH_4 and CO_2 (Nyamadzawo et al., 2012; Brümmer et al., 2009). 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, 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 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).

Aquatic systems

African aquatic systems such as streams, rivers, wetlands, floodplains, reservoirs, lagoons, and lakes can be significant sources of GHG (Table 1 and SI Table 1). Differences in regional setting and hydrology mean that emissions are highly spatially and temporally variable and when combined with the paucity of studies, it is challenging to identify clear control factors (Table 3).

Studies found SSA aquatic systems can be significant sources of GHG emissions. 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 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 the Zambezi River (Zambia), 38% of the total C in the River is emitted into the atmosphere, mostly as CO_2 (98 %) (Teodoru et al., 2015). The source of CH_4 to the atmosphere from Lake Kivu corresponded to ∼60% of the terrestrial sink of atmospheric CH_4 over the lake's catchment (Borges et al., 2011). 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 magnitude of GHG emissions from SSA aquatic systems varied with type and location (Table 3). 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}) in the Congo Basin (Mann et al., 2014). 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}) in the Okavango Delta (Botswana) (Gondwe and Masamba, 2014). Methane emissions from river deltas were substantially higher (~103 mg CH_4 m^{-2} d^{-1}) than those from non-river bays (<100 mg CH_4 m^{-2} d^{-1}) in Lake Kariba (Zambia/Zimbabwe). Methane fluxes were higher in river deltas (~103 mg CH_4 m^{-2} d^{-1}) compared to non-river bays (<100 mg CH_4 m^{-2} d^{-1}) in Lake Kariba (Zambia/Zimbabwe) (DelSontro et al., 2011). 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 in the Zambezi River (Zambia and Mozambique) (Teodoru et al., 2015). Greenhouse gas emissions from Dambos in Zimbabwe varied with catena position (Nyamadzawo et al., 2014a). 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 of all GHGs (Nyamadzawo et al., 2014a).

The concentration and flux of GHGs are strongly linked to hydrological characteristics such as discharge (Table 3), but clear patterns have not yet been identified. Surface CO_2 flux was positively correlated with discharge in the Congo River (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 Lake Kivu, seasonal variations of CH_4 in the main basin were driven by deepening of the mixolimnion and mixing of surface waters with deeper waters rich in CH_4 (Borges et al., 2011). In the Zambezi River (Zambia and Mozambique), 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). In Kampala, Uganda, precipitation was a major driver for seasonal variation of CO_2, CH_4 and N_2O fluxes in subsurface flow wetland buffer strips due to its potential influence on hydraulic saturation affecting oxygen fluctuation (Bateganya et al., 2015).

Studies found the concentration and flux of GHGs are also strongly linked to environment and water quality (Table 3) but clear patterns have not yet been identified. In the Okavango Delta (Botswana), CH_4 emissions were highest during the warmer, summer rainy 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, but 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). In Lake Kivu (East Africa), the magnitude of CO_2 emissions to the atmosphere seems to depend mainly on inputs of dissolved inorganic carbon from deep geothermal springs rather than on the lake metabolism (Borges et al., 2014).

3.2.2. Greenhouse gas emissions from agricultural lands

Agricultural GHG emissions in SSA are substantial; amounting to 26% of the continent's total GHG emissions (Valentini et al., 2014) compared to 8.4% of total GHG emissions in the USA (US EPA, 2016). Identifying controls on the emission of GHG from SSA agricultural land is challenging because both natural variations associated with climate and soil type and management factors including nutrients (particularly fertilization) and crop type affect GHG emissions.

Croplands

The effects of the amount and type of N input on N_2O emissions in croplands have been studied in several locations (Table 4). In western Kenya, the rate of N fertilizer application (0 to 200 kg N ha^{-1}) on maize fields had no significant effect on N_2O emissions (620 to 710 g N_2O–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 to150 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–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 (tilling in crop residues following harvest) 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 3). In Tanzania, incorporation of maize straw and leaf 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 incorporation of rice-straw residue resulted in larger fluxes of CO_2 but reduced N_2O emissions due to N immobilization (Rabenarivo et al., 2014). In contrast, incorporation 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 incorporation of *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). 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 incorporation of residues of three tropical plant species (*Vigna unguiculata*, *Mucuna pruriens* and *Leucaena leucocephala*) and emissions were positively correlated with the C:N ratio of the residue, 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 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 4). 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 yield-scaled N_2O emissions (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) (Nyamadzawo 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 4). 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). In Kenya and Tanzania, Rosenstock et al. (2016) studied fluxes in four crop types and found large variation of CO_2 and N_2O flux both within and between crop types depending on environmental conditions and management. 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., 2016). 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 Mg CO_2 eq Mg grain^{-1}) and from fields with reduced tillage plus mulch and nitrogen fertilizer (1.9–2.3 Mg CO_2 eq Mg grain^{-1}) compared to fields under conventional agriculture (1.9–8.3 Mg CO_2 eq Mg grain^{-1}) (Kimaro et al., 2015). The results suggest that the effect of crop type and management on

GHG emissions is difficult to predict and more research is needed to elucidate the relationship between crops, crop management and GHG emissions.

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 fertilized 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).

Grazing grassland

We only found two studies reporting GHG emissions in pastoral grasslands and there remains a serious gap in our understanding of GHG emissions in these systems. 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 intensive grazing, which destroys BSCs through trampling and burial, will adversely affect C sequestration and storage (Thomas, 2012). Rosenstock et al. (2016) measured GHG fluxes from two pastures in western Kenya and found that CO_2 emissions ranged from 13.4–15.9 Mg CO_2 ha^{-1} yr^{-1}, similar to levels found in Amazon (Davidson et al., 2010).

Rice paddies

Rice paddies are well known 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, CH_4 fluxes 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 (150 day growing season and 126 kg of applied N ha^{-1}) were estimated as 2.7 Mg 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}). These results suggest the potential for large deviations from expected emissions based on existing information.

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) 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} (Table 1 and SI Table 1).

In Burkina Faso, annual CO_2 and N_2O emissions from vegetable 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 excessive amount of applied N (2700 – 2800 kg N ha^{-1} yr^{-1}) to get high yields in vegetable gardens since surplus N will stimulate N_2O production and inhibit 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).

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 (Table 1 and SI Table 1).

Improving fallow with N-fixing trees is a common agroforestry 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 4). Nitrous oxide 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 N immobilization resulting in less labile soil N and less N_2O production (Baggs et al., 2006; Millar and Baggs, 2004). Therefore, there may be potential to reduce N_2O emissions from agroforestry, but it may require ecological nutrient management (i.e., reduced inorganic fertilizer N inputs, accounting for N input from the legume trees; adding a C source such as a cover crop together with an N source) and rotation planning.

As in natural systems, N_2O emissions from agroforestry are also affected by rainfall events. In an incubation experiment in Uganda, N_2O emissions following simulated rainfall were at 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). In agroforestry homegardens in Sudan, CO_2 and N_2O fluxes were positively correlated with soil moisture (Goenster et al., 2015).

3.2.3 Greenhouse gas emissions from land use change

Land-use change has been recognized as the largest source of GHG emission in Africa (Valentini et al., 2014). Conversion rates of African natural lands, including forest, grassland and wetland, to agricultural lands have increased in recent years (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 additional release of 0.32 ± 0.05 x 10^9 Mg C yr^{-1} (Valentini et al., 2014) or 157.9 ± 23.9 x 10^9 Mg CO_2 eq in 1765 to 2005 (Kim and Kirschbaum, 2015), higher than fossil fuel emissions for Africa (Valentini et al., 2014). 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 African woodlands and savannah. 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). 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).

3.3 Suggested future research

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 (Table 5). To address this and reduce the uncertainty surrounding the estimates, additional GHG emission measurements that fully capture seasonal and annual variations across natural ecosystems and agricultural lands throughout SSA 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 (Table 5). 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 woody shrubs encroachment (e.g., Smith 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 (whether organic or inorganic) (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., 2012b) and increased atmospheric CO_2 concentration (e.g., Lane et al., 2013) also require further testing using laboratory and field experiments. Future research should consider the wider GHG budget of agriculture and include all the various (non-soil) components such as fuel use, and embodied emissions in chemical inputs.

Where possible studies should seek to identify and separate the 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 and nitrifier denitrification) and link new knowledge of microbial communities (e.g., functional gene abundance) to GHG emissions rates. 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 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).

Throughout the study, we identified various trade-offs including increased CO_2 emission following forest thinning management, increased GHG emissions in land-use changes, very high N_2O emissions in vegetable gardens due to excessive N input to get high yields, increased CO_2 and N_2O emission in incorporation of crop residues to the soil and agroforestry practices, and exponential increased of N_2O emission and yield-scaled N_2O emissions in excessive N input. More work is needed, however, before we have a clear picture of the net impact of the tradeoffs and drivers.

3.4 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 advanced technology and utilisation of automatic chamber systems measurements can be conducted at a much higher 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 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. Besides, data acquisition will not be only determined by technical but also by socio-political (and economic) barriers in sub-Saharan Africa.

These problems not only affect data acquisition but are also the driving forces for GHG emissions due to land-use change events.

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 needs to involve identifying priorities for data acquisition, utilizing appropriate technologies, and establishing networks and collaboration.

Appendix A

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 or 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 ecosystems and agricultural lands in sub-Saharan African countries. Mean ± standard error (number of data) are shown.

Type	Area (Mha)	CO_2 emission	CH_4 emission	N_2O emission	N_2O emission factor	CO_2 eq emission	Total CO_2 eq emission
		Mg CO_2 ha^{-1} yr^{-1}	kg CH_4 ha^{-1} yr^{-1}	kg N_2O ha^{-1} yr^{-1}	%	Mg CO_2 eq. ha^{-1} yr^{-1}	x 10^9 Mg CO_2 eq. yr^{-1}
Forest/ plantation/ woodland	740.6#	32.0 ± 5.0 (34)	-1.5 ± 0.6 (15)	4.2 ± 1.5 (10)	*	34.0 ± 5.7	25.2 ± 4.2
Savannah /grassland	638.9#	15.5 ± 3.8 (11)	0.5 ± 0.4 (18)	0.6 ± 0.1 (6)	*	15.8 ± 3.8	10.1 ± 2.4
Stream/river	28.2#	78.1 ± 13.2 (27)	436.3± 133.8 (24)	1.6 ± 0.3 (17)	*	93.4 ± 17.9	2.8 ± 1.0
Wetlands/floodplains/lagoons/reservoir	43.8#	96.6 ± 31.0 (7)	950.4 ± 350.4 (5)	2.0 ± 1.5 (2)	*	121.3 ± 39.7	5.3 ± 1.7
Termite mounds	0.97†	11.6 ± 6.2 (3)	2.3 ± 1.1 (3)	0.01 (1)	*	11.7 ± 6.3	0.01 ± 0.01
Salt pan	*	0.7 (1)	*	*	*	*	*
Total natural ecosystems[1]	1452.5	27.6 ± 2.9$	43.0 ± 5.8$	2.5 ± 0.4$	*	29.9 ± 22.5$	43.4 ± 9.3 (76.3%)††
Cropland	468.7#	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##	6.5 (1)	30.5 (1)	0.19 (1)	*	7.3	1.3 ± 0.6
Vegetable gardens	*	96.4±10.2 (5)	*	120.1 ± 26.1 (5)	3.5 ± 0.5 (2)	*	*
Agroforestry	190‡	38.6 (1)	*	4.7 ± 2.2 (15)	*	*	*
Total agricultural lands[2]	479.2	23.0 ± 8.5$	19.5 ± 5.6$	4.5 ± 2.2$	*	25.6 ±12.4$	13.5 ± 3.4 (23.7%)††
Total natural ecosystems and agricultural lands[3]	1931.7						56.9 ± 12.7

GlobCover 2009
†0.07% of savanna and rainforest (Brümmer et al., 2009)
FAO STAT (http://faostat3.fao.org/home/E), year 2012
* No data available
$Area weighted average
‡ Zomer et al., 2009
††Contribution to CO_2 eq. emission in total natural and agricultural lands
[1] except salt pan
[2] except vegetable gardens and agroforestry
[3] except salt pan, vegetable gardens and agroforestry

Table 2 Correlation between annual soil CO_2 emissions (Mg CO_2 ha^{-1} yr^{-1}) and environmental factors in African natural terrestrial systems

	Annual mean Air temperature (^oC)	Annual rainfall (mm)	Soil organic carbon (%)	Soil total nitrogen (%)
Correlation coefficient	-0.322	0.518	0.626	0.849
P- value	0.01	< 0.001	< 0.001	< 0.001
Number of samples	60	61	31	26

Table 3 Summary of environmental factors affecting greenhouse gas (GHG) emissions in land use/ecosystem type. O indicates GHG affected by environmental factor named

Land use/ecosystem type	Environmental factors	GHG			Location (data source)
		CO_2	N_2O	CH_4	
Forest/plantation/ Woodland	Temperature	O			Zambia (Merbold et al., 2011)
	Soil moisture	O			Ethiopia (Yohannes et al., 2011), Republic of Congo (Epron et al., 2004), Botswana (Thomas et al., 2011), Zambia (Merbold et al., 2011)
	Rewetting of dry soil/rainfall	O	O		Kenya (Werner et al., 2007)
	Soil carbon	O			Zambia (Merbold et al., 2011), Kenya (Werner et al., 2007)
	Soil nitrogen	O			Kenya (Werner et al., 2007)
	Soil C:N		O		Kenya (Werner et al., 2007)
	Soil clay		O		Kenya (Werner et al., 2007)
	Soil bulk density		O		Rwanda (Gharahi Ghehi et al., 2014)
	Soil pH		O		Rwanda (Gharahi Ghehi et al., 2014)
	Vegetation type	O			Republic of Congo (Epron et al., 2013; Caquet et al., 2012; Nouvellon et al., 2012)
Savannah	Rewetting of dry soil/rainfall	O	O		South Africa (Fan et al., 2015; Zepp et al., 1996), Zimbabwe (Rees et al., 2006)
Salt pans	Temperature	O			Botswana (Thomas et al., 2014)
	Flooding	O			Botswana (Thomas et al., 2014)
Streams/ rivers/ wetlands/ floodplains/ reservoirs/ lagoons/ lakes	Type	O		O	Congo Basin (Mann et al., 2014), Okavango Delta, Botswana (Gondwe and Masamba, 2014), Lake Kariba, Zambia/Zimbabwe (DelSontro et al., 2011)
	Location	O	O	O	Zambezi River, Zambia/Mozambique (Teodoru et al., 2015), Zimbabwe (Nyamadzawo et al., 2014a)
	Discharge	O		O	Congo River (Wang et al., 2013), Ivory Coast (Koné et al., 2009), Oubangui River, Central African Republic (Bouillon et al., 2012), Lake Kivu (Borges et al., 2011), Zambezi River, Zambia/Mozambique (Teodoru et al., 2015)
	Precipitation	O	O	O	Uganda (Bateganya et al., 2015)
	Water temperature			O	Okavango Delta, Botswana (Gondwe and Masamba, 2014)
	Dissolved inorganic carbon	O			Lake Kivu, East Africa (Borges et al., 2014)

Table 4 Summary of the effect of management practices on greenhouse gas (GHG) emissions. + indicates increasing, · indicates no change, and – indicates decreasing.

Land use/ecosystem type	Management practices	Impact on GHG			Country (data source)
		CO_2	N_2O	CH_4	
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; Mapanda et al., 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)	+			[1]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); [2]Madagascar (Rabenarivo et al., 2014)
		+	+		Kenya (Kimetu et al., 2006); [3]Ghana (Frimpong et al. 2012)
	Crop residues + N fertilizer		+		[4]Zimbabwe (Nyamadzawo et al., 2014a,b)
			–		[5]Zimbabwe, Ghana and Kenya (Gentile et al., 2008)
	Combination of synthetic and organic fertilizers	+	–		[6]Zimbabwe (Mapanda et al., 2011)
			–		[7]Mali (Dick et al., 2008)
	Crop type	·			[8]Uganda (Koerber et al., 2009)
			–		[9]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)
	Reduced tillage + mulch, leguminous crop/tree, or N fertilizer	+	+	+; –	Tanzania (Kimaro et al., 2015)

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)

[1]U+DAP instead U+NPK; [2]N_2O study; [3]Low C:N ratio clover residues compared to high C:N ratio barley residues; [4]Application of ammonium nitrate with manure to maize (*Zea mays* L.) and winter wheat (*Triticumaestivum* L.) plant residues; [5]Plant residues of maize, calliandra, and tithonia + urea; [6]Mixed application of composted manure and inorganic fertilizer (AN); [7]Manure and urea; [8]Lettuces vs cabbages vs beans; [9]Tomatoes vs rape

Table 5 Summary of status of knowledge of greenhouse gas emission processes, control factors, estimation and model prediction.

	CO_2			N_2O			CH_4		
	Processes & Control Factors	Annual Flux Estimate	Model prediction	Processes & Control Factors	Annual Flux Estimate	Model prediction	Processes & Control Factors	Annual Flux Estimate	Model prediction
Forest/ plantation/ woodland									
Savannah									
Aquatic systems									
Cropland									
Agroforestry									
Vegetable gardens									
Rice field									
Grazing grassland									

	Robust knowledge – incremental work needed
	Existing knowledge base - co-ordinated work needed
	A growing knowledge base - with many gaps and comprehensive work still needed
	Emerging knowledge base – significant research effort required

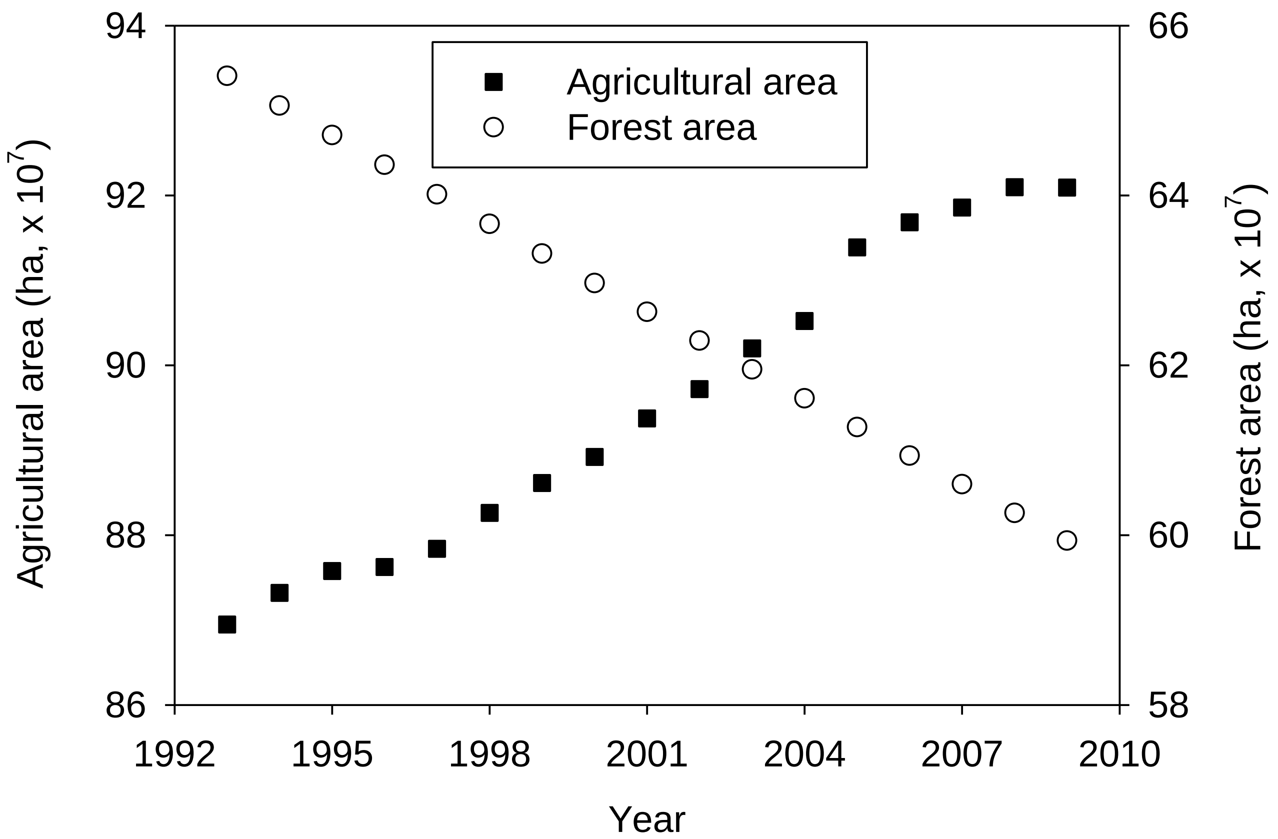
Figure captions

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.

Figure 2. Maps showing study sites of CO_2, CH_4 and N_2O fluxes

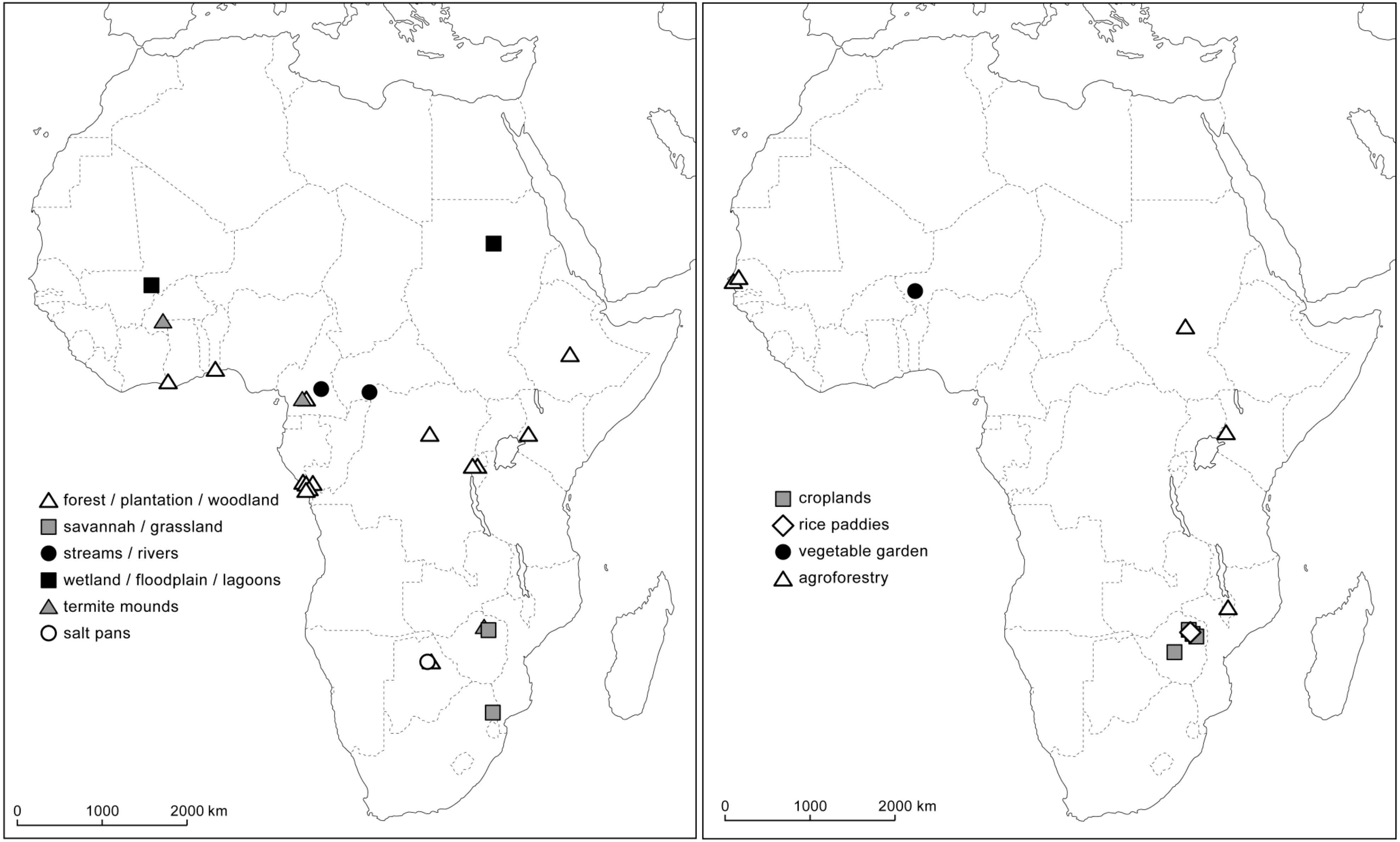
Figure 3. Relationship between nitrogen (N) input and nitrous oxide (N_2O) emissions observed in Africa. N input ranged from 0 to 300 (A), 300 to 4000 (B) and 0 to 4000 kg N ha^{-1} yr^{-1} (C). The dashed lines indicate 95% confidence intervals. *Control* indicates no fertilizer application, *Organic fertilizer* is manure, *Inorganic fertilizer* includes NPK, ammonium nitrate and urea fertilizers, and *Mixture* indicated mixed application of organic and inorganic fertilizers.

Figure 4. 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.

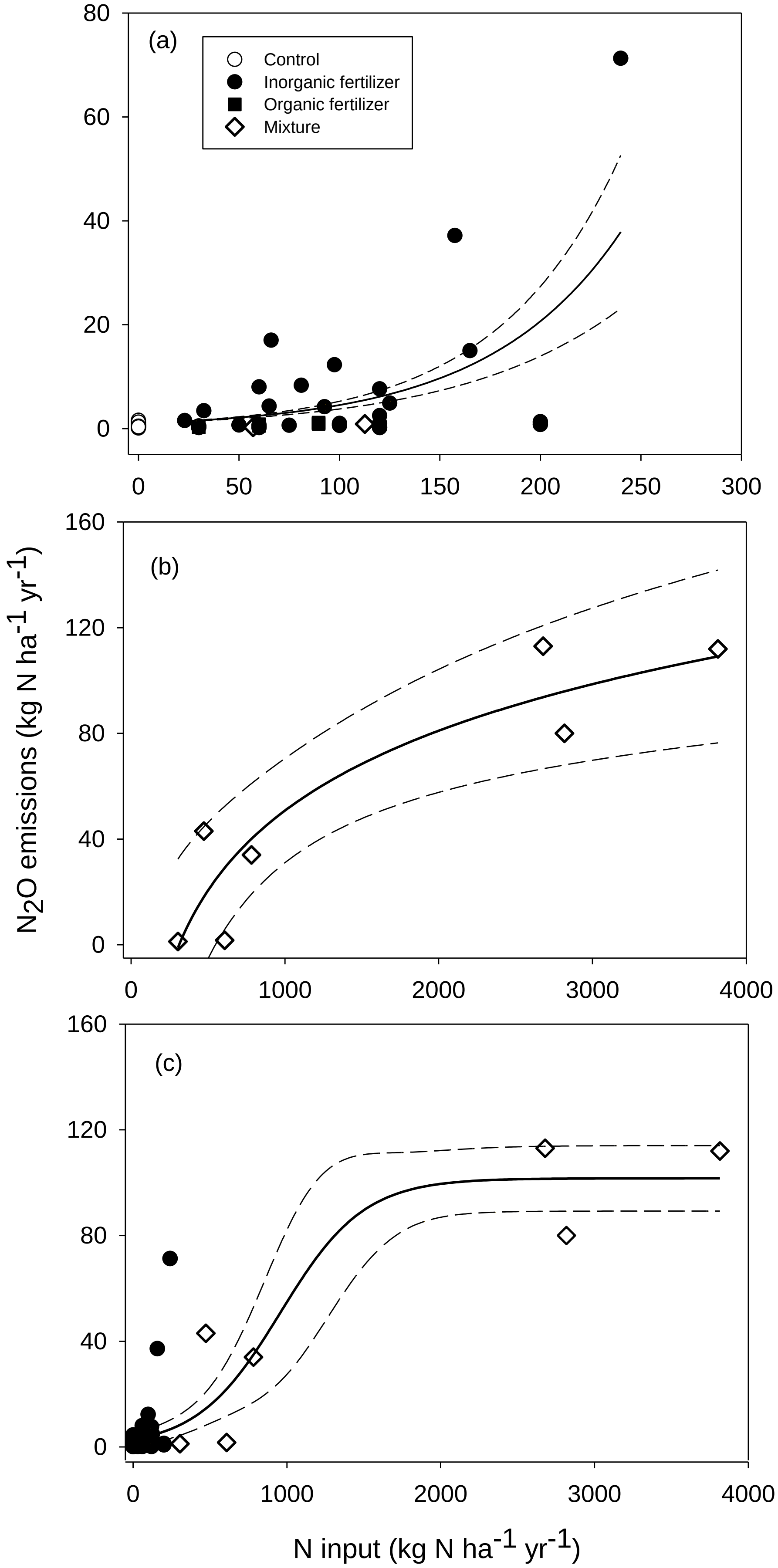


Agricultural area
Forest area
Agricultural area (ha, x 10^7)
Forest area (ha, x 10^7)
Year
94
92
90
88
86
66
64
62
60
58
1992
1995
1998
2001
2004
2007
2010

Figure 1


forest / plantation / woodland
savannah / grassland
streams / rivers
wetland / floodplain / lagoons
termite mounds
salt pans
0
1000
2000 km
croplands
rice paddies
vegetable garden
agroforestry
0
1000
2000 km

1 Figure 2



(a)
Control
Inorganic fertilizer
Organic fertilizer
Mixture
80
60
40
20
0
0
50
100
150
200
250
300
(b)
160
120
80
40
0
0
1000
2000
3000
4000
(c)
160
120
80
40
0
0
1000
2000
3000
4000
N$_2$O emissions (kg N ha^{-1} yr^{-1})
N input (kg N ha^{-1} yr^{-1})

Figure 3

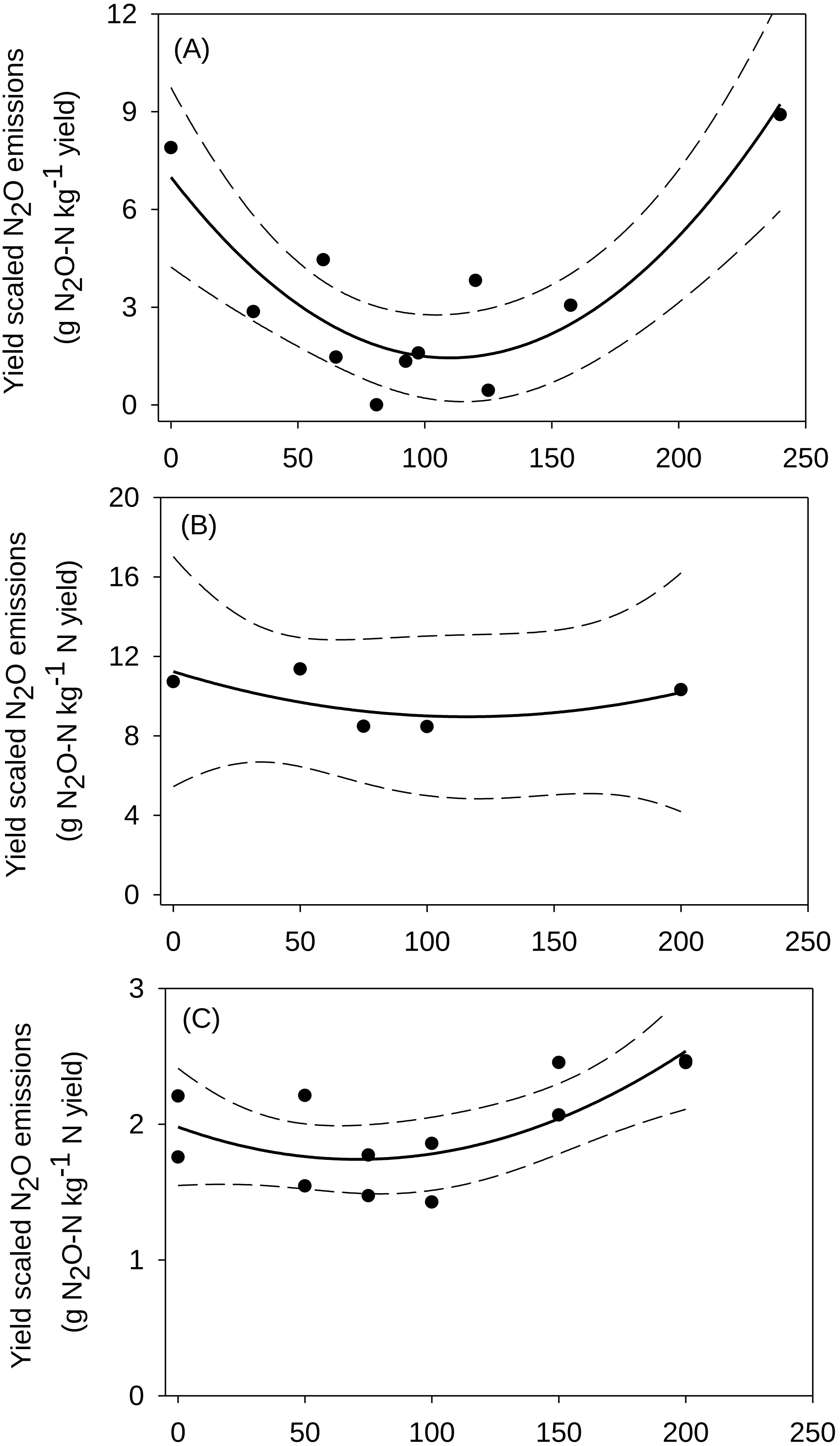

(A)
(B)
(C)
Yield scaled N2O emissions (g N2O-N kg-1 yield)
Yield scaled N2O emissions (g N2O-N kg-1 N yield)
Yield scaled N2O emissions (g N2O-N kg-1 N yield)
12
9
6
3
0
20
16
12
8
4
0
3
2
1
0
0
50
100
150
200
250
N input (kg N ha-1)

Figure 4

Supplementary Information (SI)

Table S1 Summary of greenhouse gas carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions and N_2O emission factor (%) in natural ecosystems. More detail information is available in 'Soil greenhouse gas emissions in Africa database' (http://ghginafrica.blogspot.com/).

Ecosystem type	Quality check†	Location	Temp	Rainfall	CO_2 (Mg CO_2 ha^{-1} y^{-1})	CH_4 (kg CH_4 ha^{-1} y^{-1})	N_2O (kg N_2O ha^{-1} y^{-1})	Reference
Forest/ plantation/ woodland	-	Benin	27	1200	57	*	*	Lamade et al., 1996
Forest/ plantation/ woodland	-	Botswana	*	*	13.8	*	*	Thomas et al., 2014
Forest/ plantation/ woodland	-	Cameroon	23.8	1513	*	-4.8 to 0.2	*	Macdonald et al., 1998
Forest/ plantation/ woodland	+	Ethiopia	15	1200	15.7 to 19.4	*	*	Yohannes et al., 2011
Forest/ plantation/ woodland	*	Ghana	25	1750	*	*	3.6	Castaldi et al., 2013
Forest/ plantation/ woodland	+	Kenya	23.3	1662	20.2	-2.9	4.1	Werner et al., 2007
Forest/ plantation/ woodland	?	Republic of Congo	24.4	1875	11.4 to 15.2	*	*	Maldague and Hilger, 1963
Forest/ plantation/ woodland	-	Republic of Congo	25	1200	*	-3.7 to 3.4	-0.1 to 0.1	Castaldi et al., 2010
Forest/ plantation/ woodland	-	Republic of Congo	25.3	1400	*	*	4.6	Serca et al., 1994
Forest/ plantation/ woodland	*	Republic of Congo	25	1200	13.3	*	*	Epron et al., 2004
Forest/ plantation/ woodland	*	Republic of Congo	25	1400	9.1 to 15.7	*	*	Epron et al., 2006
Forest/ plantation/ woodland	*	Republic of Congo	25	1274	23.9 to 24.3	*	*	Nouvellon et al., 2008
Forest/ plantation/ woodland	*	Republic of Congo	25	1266	17.2 to 27.1	*	*	Nouvellon et al., 2012
Forest/ plantation/ woodland	*	Republic of Congo	25.7	1430	50.6 to 74.1	*	*	Epron et al., 2013
Forest/ plantation/ woodland	*	Republic of Congo	25	1350	29.3 to 130.9	*	*	Versini et al., 2013
Forest/ plantation/ woodland	*	Rwanda	21	1246	11.8 to 14.8	*	*	Nsabimana et al., 2009
Forest/ plantation/ woodland	-	Rwanda	17	1660	*	*	6.4 to 13.7	Gharahi Ghehi et al., 2012
Forest/ plantation/ woodland	-	Zimbabwe	*	*	*	-1.2 to 2.0	*	Mapanda et al., 2010
Savannah/grassland	-	Botswana	*	*	8.0	*	*	Thomas et al., 2014
Savannah/grassland	+	Botswana	21.0 to 23.5	331	3.3 to 6.4	*	*	Thomas, 2012
Savannah/grassland	+	Burkina Faso	29.5	926	14.1 to 21.3	2.8 to 3.5	*	Brümmer et al., 2009
Savannah/grassland	-	Ghana	26.5	787	*	-1.1 to 0.3	*	Prieme and Christensen, 1999
Savannah/grassland	-	Mali	27.6	1100	*	-0.2	*	Delmas et al., 1991
Savannah/grassland	*	Republic of Congo	25	1200	32.5 to 39.7	*	*	Caquet et al., 2012
Savannah/grassland	-	Republic of Congo	23.6	1600	3.7 to 4.3	-2.2 to -2.3	*	Delmas et al., 1991
Savannah/grassland	*	South Africa	17.9	740	*	0.3 to 2.5	*	Zepp et al., 1996

Savannah/grassland	*	South Africa	*	550	12.9 to 24.2	*	*	Fan et al., 2015
Savannah/grassland	-	Zimbabwe	17.5 to 18.5	760 to 840	*	*	0.3 to 0.8	Reese et al., 2006
Streams/rivers	nd	Okavango Delta, Botswana	*	*	*	2741.9	*	Gondwe and Masamba, 2014
Streams/rivers	nd	Nyong basin, Cameroun	*	*	54.5 to 66.0	*	*	Brunet et al., 2009
Streams/rivers	nd	Oubangui River (Congo River basin)	*	1500	5.7	5.7	0.2	Bouillon et al., 2012
Streams/rivers	nd	Ivory Coast	*	*	7.9 to 27.3	58.6 to 97.4	*	Kone et al., 2009; Borges et al., 2015
Streams/rivers	nd	Ivory Coast	1500 to 1800	*	*	8.8 to 16.4	*	Koné et al., 2010
Streams/rivers	nd	Gabon	*	*	*	123.5 to 272.6	2.1 to 4.5	Borges et al., 2015
Streams/rivers	nd	Kenya	*	*	29.9 to 49.1	33.2 to 80.2	1.0 to 4.5	Borges et al., 2015
Streams/rivers	nd	Madagascar	*	*	31.2 to 84.0	76.2 to 265.0	0.6 to 1.9	Borges et al., 2015
Streams/rivers	nd	Congo River	*	*	49.5 to 228.9	29.3 to 1082.4	0.6 to 3.1	Wang et al., 2013; Mann et al., 2014; Borges et al., 2015
Streams/rivers	nd	Zambezi River	*	1450	39.6 to 67.6	97.3 to 793.0	0.3	Teodoru et al., 2015; Borges et al., 2015
Wetlands/floodplains/lagoons /reservoir/lake	nd	Zambezi River	*	1450	-4.8 to 9.9	6.8 to 125.6	*	Teodoru et al., 2015
Wetlands/floodplains/lagoons /reservoir/lake	nd	Okavango Delta, Botswana	*	*	*	1480.4 to 1787.0	*	Gondwe and Masamba, 2014
Wetlands/floodplains/lagoons /reservoir/lake	nd	Ivory Coast	*	*	-11.9 to 161.7	*	*	Koné et al., 2009
Wetlands/floodplains/lagoons /reservoir/lake	nd	Congo River	*	*	*	246.4	*	Tathy et al., 1992
Wetlands/floodplains/lagoons /reservoir/lake	nd	Mali	*	*	*	3.1	*	Delmas et al., 1991
Wetlands/floodplains/lagoons /reservoir/lake	nd	Zimbabwe	*	*	65.0 to 232.0	-26.3 to 1235.2	0.5 to 3.5	Nyamadzawo et al., 2014
Wetlands/floodplains/lagoons /reservoir/lake	nd	Lake Kivu	*	*	1.7 to 85.8	2.1 to 6.0	*	Borget et al., 2011 and 2014
Wetlands/floodplains/lagoons /reservoir/lake	nd	Lake Kariba	*	*	*	11 to 7665	*	Delsontro T et al., 2011
Wetlands/floodplains/lagoons /reservoir/lake	nd	Ivory Coast	*	1500 to 1800	*	4.4 to 19.3	*	Koné et al., 2010
Termite mound	+	Burkina Faso	29.5	926	13.5 to 21.3	3.0 to 3.7	*	Brümmer et al., 2009
Termite mound	-	Zimbabwe	18	850	0.002	0.1	0.01	Nyamadzawo et al., 2012
Salt pan	-	Botswana	*	*	0.7	*	*	Thomas et al., 2014

Symbols: +: methods are *good*; *: methods are *marginal*; -: methods are *poor to very poor*; ?: methods are unclear; nd: cannot comment due to no available criteria

Table S2 Summary of *in situ* carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) fluxes and N_2O emission factor (%) in agricultural ecosystems. More detail information is available in 'Soil greenhouse gas emissions in Africa database' (http://ghginafrica.blogspot.com/).

Ecosystem type	Quality check†	Location	Temperature (℃)	Rainfall (mm)	CO_2 (Mg CO_2 ha^{-1} y^{-1})	CH_4 (kg CH_4 ha^{-1} y^{-1})	N_2O (kg N_2O ha^{-1} y^{-1})	N_2O emission factor (%)	Reference
Croplands	+	Burkina Faso	29.5	926	9.2 to 16.5	-0.9	*	*	Brümmer et al., 2009
Croplands	+	Kenya	*	1750	*	*	1.0 to 1.3	*	Hickman et al., 2014
Croplands	+	Kenya	*	1750	*	*	0.2 to 0.5	0.01 to 0.1	Hickman et al., 2015
Croplands	+	Madagascar	16	1300	*	*	0.4	0.47	Chapuis-Lardy et al., 2009
Croplands	+	Malawi	24	930	15.0	*	*	*	Kim, 2012
Croplands	-	Mali	27.6	1100	*	*	0.9 to 2.4	0.3 to 4.1	Dick et al., 2008
Croplands	-	Republic of Congo	23.6	1600	1.7 to 3.7	-1.3 to -1.8	*	*	Delmas et al., 1991
Croplands	-	Senegal	29.7	670	*	*	0.05 to 0.1	*	Dick et al., 2006
Croplands	+	Tanzania	24.5	626 to 905	3.4 to 14.8	*	*	*	Sugihara et al., 2012
Croplands	+	Tanzania	*	*	17.6 to 20.2	-1.7 to 5.6	0.6 to 1.1		Kimaro et al., 2015
Croplands	*	Uganda	21.9	1224	111.1 to 141.2	*	*	*	Koerber et al., 2009
Croplands	*	Zimbabwe	19.1	940	*	*	0.5 to 1.4	*	Rees et al., 2012
Croplands	*	Zimbabwe	*	*	*	*	0.9 to 7.1	*	Chikowo et al., 2004
Croplands	*	Zimbabwe	19.1	940	*	*	0.3 to 0.8	*	Rees et al. 2013
Croplands	-	Zimbabwe	*	*	*	*	0.5	*	Mapanda et al., 2010
Croplands	-	Zimbabwe	18.6	750	19.0 to 44.9	13.4 to 66.7	0.3 to 112.0	*	Nyamadzawo et al., 2014b
Croplands	-	Zimbabwe	18.9	748	1.9 to 10.4	-0.04 to 49.1	0.2 to 3.9	*	Mapanda et al., 2012
Croplands	*	Zimbabwe	21	725	*	*	0.5 to 2.7	0.3 to 1.0	Masaka et al., 2014
Croplands	-	Zimbabwe	*	*	*	3.2 to 11.9	0.8 to 3.5	*	Mapanda et al., 2010
Rice paddy	-	Zimbabwe	18.6	750	6.5	12.5	0.2	*	Nyamadzawo et al., 2013
Vegetable gardens	*	Burkina Faso	27	900	80.7 to 132.0	*	125.7 to 177.6	*	Lompo et al., 2012
Vegetable gardens	*	Niger	30.3	542	73.3 to 100.8	*	53.4 to 176.0	*	Predotova et al., 2010
Agroforestry	-	Senegal	25.8	370	*	*	0.2 to 2.7	*	Dick et al., 2006
Agroforestry	*	Sudan	28.2	698	*	*	23.6 to 26.7	*	Goenster et al., 2014
Agroforestry	+	Kenya	24	1880	*	*	0.3 to 6.4	*	Millar et al., 2004
Agroforestry	+	Malawi	24	930	38.6	*	*	*	Kim, 2012

†Symbols: +: methods are *good*; *: methods are *marginal*; -: methods are *poor to very poor*.