Impact Assessment of Land-Use Change and Agricultural Treatments on Greenhouse Gas Emissions from Wetlands of Uganda and Tanzania

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Zusammenfassung

Feuchtgebiete sind von entscheidender Bedeutung für die Regulierung des globalen Klimas, da sie eine maßgebliche, globale Kohlenstoffsenke darstellen. Außerdem bieten Feuchtgebiete optimale Bedingungen für den Anbau von Nahrungsmitteln und unterstützen somit die Lebensmittelversorgung vieler Menschen in Regionen Afrikas südlich der Sahara. Die Umwandlung von natürlichen Feuchtgebieten zu Ackerland beeinträchtigt jedoch viele wertvolle Ökosystemdienstleistungen, wie beispielsweise die Regulierung des globalen Klimas. Veränderte und erhöhte Treibhausgas-(THG)-Emissionen können die Folge sein. Eine der größten Herausforderungen hinsichtlich des nachhaltigen Managements von Feuchtgebieten besteht in der Schlichtung von Interessenkonflikten und einer Kompromissfindung zwischen Nahrungsmittelanbau und verminderten THG-Emissionen. Um nachhaltige Managementempfehlungen entwickeln zu können, werden dringend THG-Emissionsdaten von afrikanischen Feuchtgebieten benötigt. Die Zahl der derzeit existierenden Studien ist gering, wodurch die Datenlange große Lücken aufweist. Die vorliegende Studie verfolgte das Ziel, einen Beitrag zur Verringerung dieser Datenlücken zu leisten. Im Rahmen des Vorhabens wurden THG-Emissionsdaten von Feuchtgebieten in Ost-Afrika in Hinblick auf unterschiedliche Feuchtgebietstypen, Landnutzungsformen und Positionen im Feuchtgebiet erhoben. Zudem wurden verschiedene Anbaumethoden untersucht und hinsichtlich ihrer mindernden Wirkung auf THG-Emissionen gemäß ihres Ertrags beurteilt. Zwei Feldexperimente wurden in unterschiedlichen Feuchtgebietstypen Ost-Afrikas etabliert. Bei dem Ersten handelte es sich um ein Feuchtgebiet im Landesinneren von Uganda, das Zweite war ein Überflutungsgebiet am Kilombero-Fluss in Tansania. CH_4-, CO_2- und N_2O-Emissionsdaten wurden mit statischen Kammermessungen über einen Zeitraum von zwei aufeinanderfolgenden Anbau- und Brachperioden erhoben. Während der Datenanalyse wurde ein Mangel an systematischen Methoden zur Qualitätskontrolle von Emissionsdaten aus Kammermessungen deutlich. Aufgrund dessen wurde ein achtstufiges Daten-Qualitäts-Management-System entwickelt, welches auf objektiven Kriterien basiert und die Datenzuverlässigkeit, ebenso wie die Datenakzeptanzrate verbessert. Die qualitätsgeprüften Ergebnisse dieser Studie bestätigten, dass Landnutzungsänderungen einen signifikanten Einfluss auf THG-Emissionen haben. Die ermittelten Emissionswerte zeigten eine deutliche Erhöhung des Treibhausgaspotentials nach der Umwandlung von natürlichen Feuchtgebieten zu Ackerland. Außerdem ergab diese Studie, dass intensive Anbaumethoden mit hohem Ertragspotential nicht zu signifikant erhöhten ertrags-basierten TGH-Emissionen führten. Die berechneten Treibhausgaspotential-Indizes intensiver Anbaumethoden mit Düngereinsatz zeigten gleich hohe oder sogar geringere Werte verglichen mit ungedüngten Anbaumethoden. Intensive Anbaumethoden mit hohen Erträgen stellen somit einen möglichen Kompromiss zwischen Nahrungsmittelanbau und THG-Emissionen dar. Für eine Minderung der THG-Emissionen ist es allerdings zwingend notwendig, natürliche Feuchtgebietsflächen

anderorts zu schonen und in ihrem natürlichen Zustand zu belassen.

Abstract

Wetlands play an important role in global climate regulation as they represent a great global carbon sink. Moreover, wetlands provide optimal conditions for food production and support the livelihoods of many people in Sub-Saharan Africa with food supply. The conversion of natural wetland areas to farmland seriously affects valuable ecosystem services, including global climate regulation, and can result in altered greenhouse gas (GHG) emissions. Therefore, a main challenge of sustainable wetland management is to find a reconciliation between food production and mitigation of GHG emissions. For the development of management recommendations, GHG emission data from wetlands in Sub-Saharan Africa are highly needed, because the numbers of GHG studies conducted in this region are low. This study aimed to reduce this knowledge gap and assessed GHG emissions from wetlands in East Africa with consideration of contrasting wetland types, different types of land use and different hydrological positions within the wetland. Moreover, different agricultural treatments were evaluated with respect to their effects on yield-based GHG emissions. Two field experiments were established in different wetland types in East Africa. The first test site was located in an inland valley wetland in Uganda, while the second one was located in a floodplain of the Kilombero river in Tanzania. CH_4, CO_2 and N_2O emission data were collected with static chambers for a total sampling period of two consecutive cropping and fallow periods. During data analysis, a lack of systematic quality assurance of GHG data from static chamber measurements became apparent. Thus, an eight-step data quality management system based on objective criteria was developed to ensure data reliability and improve data acceptance rates. The quality-checked results of this study confirmed that land-use change had a significant impact on GHG emissions, as the global warming potential (GWP) considerably increased after the conversion of natural wetlands to farmland. Moreover, this study showed that intensification of food production did not result in significantly higher yield-based GHG emissions. Intensive cropping treatments with fertilizer application showed equally high or even lower global warming potential indexes (GWPI) compared to non-fertilized treatments. In conclusion, intensive cropping management practices with high yield potentials represent a possible trade-off between food production and GHG emissions. However, to achieve GHG emission mitigation, a combination with natural wetland areas spared from agricultural production is essential.

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List of Abbrevations

ANOVA analysis of variance
DOC dissolved organic carbon
DON dissolved organic nitrogen
DQMS data quality management system
EC electrical conductivity
ECD electron capture detector
Eh redox potential
FID flame ionization detector
GC gas chromatograph
GHG greenhouse gas
GHGs greenhouse gases
GWP global warming potential
GWPI global warming potential index
IPCC international panel on climate change
MBC microbial biomass carbon
MBN microbial biomass nitrogen
POC particulate organic carbon
PON particulate organic nitrogen
pp percentage points
SSA Sub Saharan Africa

1. Introduction

Approximately 1 billion people worldwide suffer from insufficient dietary energy availability. Thus, the concern about global food security is growing (Barrett, 2010). Scientists and politicians aim to improve food security and enhance food production by improving land use and developing optimized cropping systems. The demand for increased food production has led to an increasing pressure of expanding agriculture into wetland areas (Rebelo et al., 2009b). Wetlands have an excellent potential for food production due to a good water availability and fertile soils which makes them attractive as farmland (Reddy and DeLaune, 2008). It is estimated that 50% of all wetlands worldwide were converted for the purpose of agriculture and urbanization during the twentieth century (Rijsberman and Silva, 2006). In Sub Saharan Africa (SSA), wetlands facilitate the livelihoods of many people by providing a large number of ecosystem services, including food supply (Bikangaga et al., 2007; Rebelo et al., 2009b). However, wetlands also feature unique environmental conditions and a wide range of natural ecosystem services, such as climate regulation, which are heavily influenced by the conversion of natural wetlands into agricultural areas (Millennium Ecosystem Assessment, 2005; Reddy and DeLaune, 2008).

Many federal wetland policies shifted their focus from promoting wetland conversion to supporting wetland protection within the last decades. This often involved a conflict of interests between private incentives and public benefits, as land owners do not profit from ecosystem services of unused wetlands (Heimlich et al., 1998). To solve this conflict a reconciliation of enhanced food production with environmental protection is desirable.

For the development of sustainable food security solutions, it is essential to consider technical options for enhancing food production together with possibilities to preserve ecosystem service benefits provided from natural wetlands. A good understanding of the spatial and temporal dynamics of matter fluxes in wetlands is required to approach this aim. This study focusses on the importance of wetlands for climate regulation and investigates the effects of land-use change and management options on GHG emissions from agriculturally used wetland areas.

2. Wetlands

2.1. An Introduction to Wetlands and their Role in Global Climate Regulation

Wetlands are defined as areas with water saturated soils, including marshes, swamps, bogs, small lakes and floodplains (Reddy and DeLaune, 2008; Kayranli et al., 2010). They are typically located in regions with high water table and low elevation, and retain water well during rainy periods due to their usually poor drainage. Wetlands occur in all climates around the world and form unique habitats for plants and animals (Reddy and DeLaune,

2008). Despite their wide variety, all wetlands are characterized by unique hydrology, vegetation and soil features (Schlesinger and Bernhardt, 2013). Wetland hydrology is characterized by permanent or long-term water presence at the surface or close to it. Hydrophytic vegetation describe plants that are adapted to saturated or flooded soil conditions. Hydric soils are characterized as saturated or flooded soils with long term or permanent anoxic conditions (Reddy and DeLaune, 2008).

The proportion of wetlands on the earth's land surface accounts to only 2% to 7%, depending on the definition of inland waters included (Reddy and DeLaune, 2008; Kayranli et al., 2010; Schlesinger and Bernhardt, 2013). However, despite their small land cover area, wetlands can have considerable impacts on larger-scale processes (Schlesinger and Bernhardt, 2013). For example, nitrogen and carbon cycles in wetland areas influence processes on the local (plant growth and soil development), regional (water quality) and global scale (GHG emissions and carbon storage) (Reddy and DeLaune, 2008). Thus, they have a substantial importance for the regulation of the earth's biogeochemical cycles, especially for the global carbon cycle (Schlesinger and Bernhardt, 2013). Wetlands are one of the most productive ecosystem types on earth with an average net primary productivity that equals the one of tropical rain forests (Reddy and DeLaune, 2008). Approximately 7 to 15% of the terrestrial productivity is contributed by wetlands (Schlesinger and Bernhardt, 2013). The high productivity leads to an accumulation of organic matter in soils which turns them into a carbon sink (Reddy and DeLaune, 2008). It is estimated that more than 50% of the world's soil carbon is stored in wetland soils, therefore they represent an important global carbon sink (Gorham, 1991; Schlesinger and Bernhardt, 2013). The net carbon sink capacity is supposed to be 830 $Tg\,C\,yr^{-1}$ with an average net carbon retention of 118$g\,C\,m^{-2}\,yr^{-1}$. The highest carbon retention occurs in the tropics and subtropics (Mitsch et al., 2012).

However, wetlands also are a natural source of greenhouse gases (GHGs), especially concerning methane (CH_4) (Mitsch et al., 2012). Roughly 20 to 33 % of all CH_4 emissions worldwide originate from wetlands (Schlesinger and Bernhardt, 2013), with 52 to 58% originating from the tropics (Bloom et al., 2010). Mitsch et al. (2012) indicated an even higher CH4 emission fraction of 78% from tropical and subtropical wetlands. Moreover, a rise of 7 % in wetland CH_4 emissions was observed between 2003 and 2007 (Bloom et al., 2010). In summary, wetlands act as both sink and source of carbon. Their role in global climate change is discussed critically in the literature, while recommendations for the adaptation of policies and land use planning vary considerably. Recent studies indicated that age and environmental boundary conditions, such as climate and location, are essential for the evaluation of their role in the global carbon cycle (Kayranli et al., 2010). After a time period of 300 years, CH_4 emission and carbon sequestration balance each other and wetlands turn into net carbon sinks (Mitsch et al., 2012).

2.2. Soil Properties

Wetland soils are referred to as hydric soils which are characterized by water saturation or flooding (Mitsch and Gosselink, 2007; Reddy and DeLaune, 2008). They develop due to year-round or seasonal water availability and continuous or periodic saturation of the soil, which cause anoxic conditions in the upper soil horizons and favour the growth of hydrophytic vegetation (Mitsch and Gosselink, 2007; Schlesinger and Bernhardt, 2013). Based on their hydrologic conditions, wetland soils can be classified in three subtypes:

- flooded soils (water table is permanently above surface)
- saturated soils (water table is barely below surface, no excess floodwater)
- soils with a water table that is always below surface
- temporary flooded soils (with anoxic soil conditions during flooding and oxic conditions during the dry season, e.g. floodplains)

The last two types can feature the properties of upland as well as wetland soils, while the first two types are clearly hydric soils (Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). Wetland and upland soils differ from each other by their content of organic volume, as well as water and air content. In general, the water content in (flooded) wetland soils is much higher than in (drained) upland soils, while the air volume is zero (Reddy and DeLaune, 2008). Moreover, soil can be classified in mineral soil and organic soil. Typical organic wetland soils are histosols. The categorization depends on the amount of organic material. With a share of more than 20-35% of organic material the soil is classified as organic soil or histosol (Mitsch and Gosselink, 1986). Organic soils tend to have a higher water content, but lower solid volume than mineral soils (see also Figure 1) (Reddy and DeLaune, 2008).

The lack of oxygen in wetland soils is the main driver for many biogeochemical processes

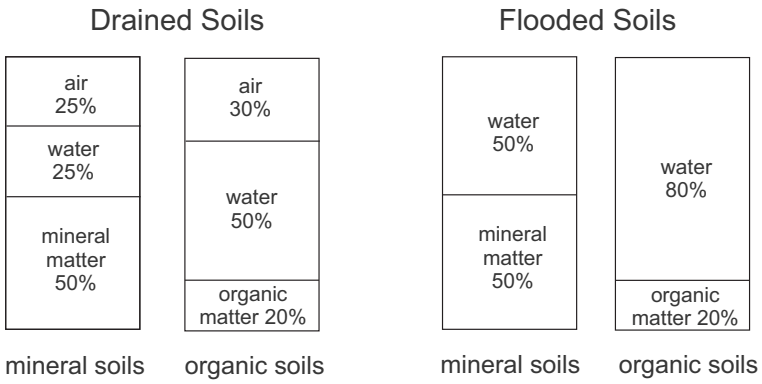

Figure 1: Differences in soil volume composition in flooded and drained soils, adapted from Reddy and DeLaune (2008).

(Reddy and DeLaune, 2008). Oxygen diffuses only a few millimeters below the water table in saturated wetland soils (Schlesinger and Bernhardt, 2013). Soil pores are filled with water instead of air. Consequently, wetland soils are characterized by reducing conditions and anaerobic processes, such as methanogenesis and nitrate reduction, which dominate the transformation of soil compounds (Reddy and DeLaune, 2008; Kayranli et al., 2010). Through microbial catabolic processes, oxidized forms are converted into reduced forms during flooding periods. The end products accumulate in soil (e.g. NH^+_4) (Reddy and DeLaune, 2008). Low chroma (<2) or grey soil colour usually indicate intense anoxic soil conditions, they are typical for soils which are strongly influenced by water (Vepraskas and Service, 1992; Scheffer and Schachtschabel, 2010). In contrast, upland soils have a good availability of oxygen. The key process of organic matter conversion is soil respiration. Oxidized chemical forms (e.g. NO_3^-) dominate the soil composition, while reduced soil compounds do not occur (Reddy and DeLaune, 2008). Drainage, e.g. for the purpose of agricultural usage, can cause a conversion of wetland soils to upland soils (Schlesinger and Bernhardt, 2013). As a result, water conditions and oxygen availability change significantly, and reduced soil compounds are converted into oxidized forms. In summary, the presence of reduced chemical forms is a valid indicator for the identification of anoxic hydric soils, while oxidized compounds verify oxic soil conditions (Reddy and DeLaune, 2008).

Soil flooding affects soil parameters including pH value, electrical conductivity (EC) and redox potential (Eh). Flooded soils tend to adjust to a neutral pH value, thus acid soils increase and alkaline soils decrease their pH value to reach this point. Depending on the soil characteristics, pH and EC generally increase under saturated conditions, while Eh decreases with time until it approaches a steady value (Reddy and DeLaune, 2008). The redox potential indicates soil wetness and evaluates anoxic conditions. The usual range of Eh values in wetlands varies from +700 to -300 mV (see Figure 2). Soils which are permanently water-logged are characterized by low or negative Eh values (-300 to 300 mV) and a high supply of reductants (electron donors, e.g. soil organic matter), while the supply of oxidants (electron acceptors) is strongly limited. Drained soils and upland soils feature opposite conditions. Oxidized forms dominate in upland soils and feature a high amount of oxidants resulting in a high redox potential ($>$ 300 mV) (Reddy and DeLaune, 2008). Positive Eh values are also typical for wetlands in a transition zone with moderate anaerobic conditions (Reddy and DeLaune, 2008).

2.3. The Carbon Cycle

2.3.1. Carbon Sequestration

Photosynthesis is the main process that fixes carbon from the atmosphere and adds it to terrestrial and wetland ecosystems. Wetlands feature five major reservoirs for organic carbon:

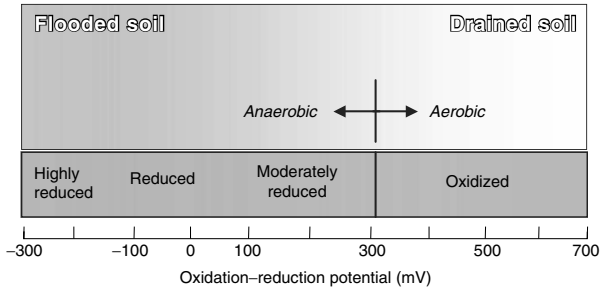

Figure 2: Development of the redox potential (Eh) during flooded and drained conditions (Reddy and DeLaune, 2008).

- plant biomass (standing stock)
- particulate organic carbon (POC)
- dissolved organic carbon (DOC)
- microbial biomass carbon (MBC)
- gaseous end products

The latter four are present in detritus, soil and water (Reddy and DeLaune, 2008).

The decomposition of dead organic matter is driven by the activity of various decomposing organism, including fungi and bacteria. After the physical fragmentation of particulate organic matter by grazers and the breakdown by extracellular enzymes, aerobic and anaerobic heterotrophic microorganisms convert monomers into CO_2 or CH_4 through their catabolic activities. The accumulation of carbon in soils is determined by the balance between organic carbon fixation due to net primary production and decomposition via heterotrophic metabolism (Reddy and DeLaune, 2008). Organic matter accumulates in soil when the decomposition rate is slower than the primary productivity (Mitsch and Gosselink, 2007). The productivity of wetlands varies depending on the time of year, geographic location, nutrient status and type of vegetation (Kayranli et al., 2010). However, natural wetland environments provide optimal conditions for the sequestration of carbon (Kayranli et al., 2010; Mitsch et al., 2012). High water tables, high productivity and low decomposition rates effect the storage of carbon within wetland soil and detritus because the decomposition process is impeded in saturated or flooded soils. Thus, the primary production in wetlands often exceeds decomposition and causes a net accumulation of organic matter (Gorham, 1991; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). The completeness and speed of plant detritus turnover is mainly determined by the availability of oxygen. In wetlands soils, decomposition is often incomplete and occurs at slower rates under anoxic conditions, while upland soils feature predominantly oxic soil conditions and show rapid decomposition rates of organic matter with a minimal net accumulation of organic carbon (Reddy and DeLaune, 2008; Kayranli et al., 2010). Due to the slow decomposition rates, several strata are built up and compressed,

forming organic soil layers over time, which are characterized by partially decomposed plant residues and a low degree of humification. Thereby, organic matter is accumulated, decomposed and subjected to burial, which results in a shift from aerobic to anaerobic processes, as shown in Figure 3). In addition, the diffusion of gases produced in deeper soil layers is slowed due to water saturation (Schlesinger and Bernhardt, 2013).

2.3.2. Aerobic and Anaerobic Carbon Turnover Processes

Organic carbon can either be stored in plants, microorganisms and detritus (see chapter 2.3.1), or transformed into several gaseous components, such as CO_2 or CH_4, by microbial organism and returned to the atmosphere (Kadlec et al., 2001). The transformation of organic matter in wetland soils involves aerobic and anaerobic processes (Reddy and DeLaune, 2008; Kayranli et al., 2010). The main components of the carbon cycle in wetlands with its gaseous end products are shown in Figure 4.

Soil Respiration

The most important processes in the aerobic zone is soil respiration. Thereby, organic carbon and oxygen are converted to CO_2 by root respiration, fauna respiration and microbial decomposition of soil organic matter. Energy is generated by the oxidation of sugars, and CO_2 is released as a result of metabolism. O_2 is the major electron acceptor in this process (see equation 1) (Kadlec et al., 2001; Yiqi and Zhou, 2006; Kayranli et al., 2010; Mitsch and Gosselink, 2015).

$$C_6H_{12}O_6 + 6\,O_2 \rightarrow 6\,CO_2 + 6\,H_2O + 12\,e^- + energy \tag{1}$$

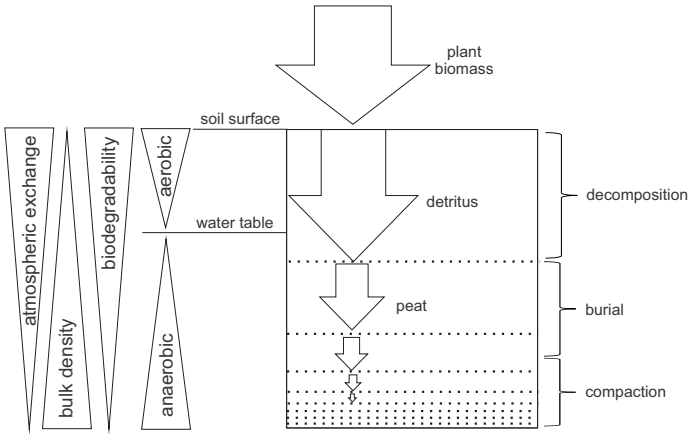

Figure 3: Decomposition of organic matter, adapted from Reddy and DeLaune (2008) and Schlesinger and Bernhardt (2013).

Methanogenesis

Anaerobic processes are less efficient in terms of energy transfer compared to aerobic respiration. However, due to the fact that wetlands naturally feature anoxic conditions, anaerobic processes, such as methanogenesis, are of great importance (Kayranli et al., 2010). Under most reduced conditions (redox potential -200 mV), when other terminal electron acceptors, such as O_2, NO_3 and SO_4^{2-}, have already been reduced, methanogens (*Archaea*) use CO_2 (see equation 2) or low-molecular-weight organic compounds (see equation 3 and 4) as terminal electron acceptor, which results in the production of CH_4 (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015).

$$CO_2 + 8\,e^- + 8\,H^+ \rightarrow CH_4 + 2\,H_2O \tag{2}$$

$$CH_3COOH\,(acetic\,acid) \rightarrow CH_4 + CO_2 \tag{3}$$

$$3\,CH_3OH\,(methanol) + 6\,H^+ \rightarrow 3\,CH_4 + 3\,H_2O \tag{4}$$

Methane Oxidation

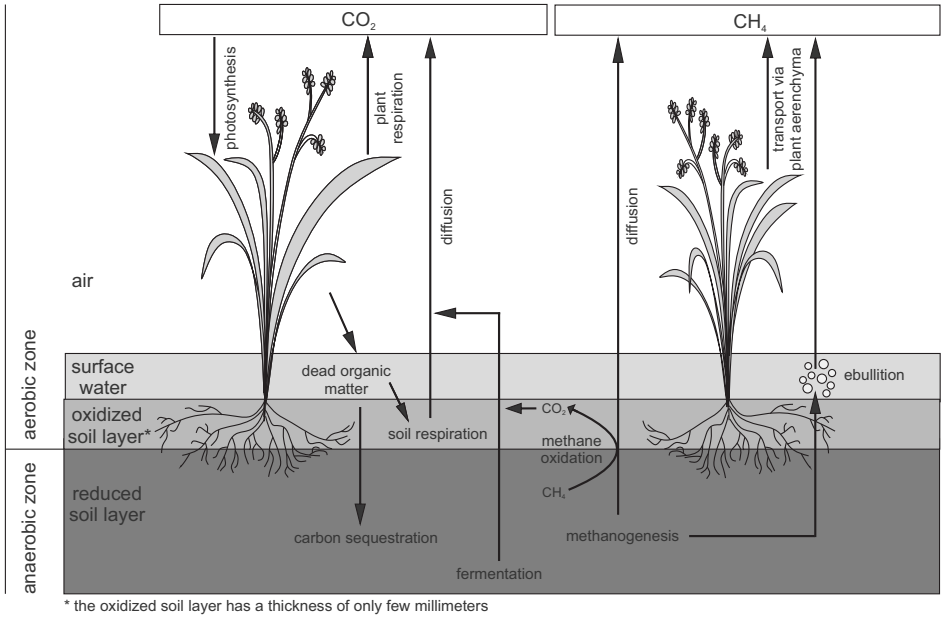

Figure 4: The carbon cycle with its major components in wetland soils.

Obligate methanotropic bacteria have the ability to convert CH_4 successively to methanol (CH_3OH), formaldehyde (HCHO), formic acid (HCOOH) and ultimately CO_2 (see equation5). Although methanotrophs are usually found in a nonflooded environment, they can also occur in the oxygenated surface zone of wetland soils with stratified oxic-anoxic horizons. Consequently, CH_4 can be converted into CO_2 by methanotrops during its way up from deep soil zones to the surface (Mitsch and Gosselink, 2015). About 20 to 40% of the CH_4 transported via diffusion is oxidized in the rhizosphere and in surficial oxic layers (Whalen, 2005).

$$CH_4 \rightarrow CH_3OH \rightarrow HCHO \rightarrow HCOOH \rightarrow CO_2 \tag{5}$$

Fermentation

Fermentation is often ignored as considerable source of CO_2 emissions from wetlands, although recent studies assumed that a substantial fraction of anaerobic carbon mineralization in wetland sediments might result from this process (Keller and Bridgham, 2007). Fermentation occurs in either facultative or obligate anaerobic bacteria which use various organic substances (e.g. sugars) for the conversion into CO_2 (Schlesinger and Bernhardt, 2013). Organic matter is used as terminal electron acceptor by microorganisms which produce different forms of low-molecular-weight acids (see equation 6) or alcohols together with CO_2 (see equation 7). Fermentation plays a key role in the provision of substrates, as high-molecular-weight organic compounds are broken down to low-molecular-weight carbohydrates which are available for other microorganism, such as methanogens (Mitsch and Gosselink, 2015).

$$C_6H_{12}O_6 \rightarrow 2\,CH_3CHOHCOOH\,(lactic\,acid) \tag{6}$$

$$C_6H_{12}O_6 \rightarrow 2\,CH_3CH_2OH\,(ethanol) + 2\,CO_2 \tag{7}$$

The carbon cycle in wetlands is essential for the regulation of CO_2 and CH_4 in the atmosphere. Hence, it is also important regarding recent questions concerning greenhouse effect and global warming, see chapter 2.6 (Reddy and DeLaune, 2008).

2.4. The Nitrogen Cycle

One of the most limiting nutrients in wetlands is nitrogen. It considerably regulates the productivity of the ecosystem, independent of whether it is a natural wetland or an agriculturally used one (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). Nitrogen input to wetlands involves external sources, such as fertilizer application to rice paddies,

as well as biological N_2 fixation. It can be stored in the following reservoirs (Reddy and DeLaune, 2008):

- plant and algal biomass nitrogen (living stock)
- particulate organic nitrogen (PON) (detritus, soil and water)
- microbial biomass nitrogen (MBN) (detritus, soil and water)
- dissolved organic nitrogen (DON) (detritus, soil and water)
- inorganic forms of nitrogen (water, soil pore water)
- gaseous end products (atmosphere, detritus, soil and water)

Nitrogen occurs in organic forms (e.g. proteins, nucleic acids, urea) and dissolved inorganic forms (e.g. ammonia NH_3, ammonium NH_4^+, nitrite NO_2^- , nitrate NO_3^-) (Reddy and DeLaune, 2008). Organic nitrogen can be converted into mineral nitrogen (e.g. NH_4^+) through nitrogen mineralization, also referred to as ammonification, which can proceed under oxic as well as anoxic conditions. Once NH_4^+ is formed, several pathways are possible (Mitsch and Gosselink, 2015). One important pathway for the production of GHGs is nitrification, an aerobic two step process carried out by *Nitrosomonas sp.* (see equation 8) and *Nitrobacter sp.* (see equation 9), which results in the formation of NO_2^- and finally NO_3^- (Kadlec et al., 2001; Mitsch and Gosselink, 2015). During this process the oxidation state of nitrogen increases stepwise from -3 in NH_4^+ to +3 in NO_2^- to +5 in NO_3^- (Reddy and DeLaune, 2008). N_2O is formed by decomposition of intermediates, e.g. NH_2OH (Hernandez and Mitsch, 2006).

$$2\,NH_4^+ + 3\,O_2 \rightarrow 2\,NO_2^- + 2\,H_2O + 4\,H^+ + energy \tag{8}$$

$$2\,NO_2^- + O_2 \rightarrow 2\,NO_3^- \tag{9}$$

NO_3^- is more mobile in soil solution compared to NH_4^+. It is not subjected to immobilization caused by negatively charged soil particles due to its own negative charge. Thus, it is easily lost through groundwater flow or immediately assimilated by plants or microbes. Alternatively, NO_3^- can be reduced to N_2O and gaseous molecular nitrogen (N_2) by denitrification and returned to the atmosphere in this way (Mitsch and Gosselink, 2015). It is a stepwise process as described in equation 10, 11, 12, 13.

Nitrate Reductase

$$NO_3^- + 2\,H^+ + 2\,e^- \rightarrow NO_2^- + H_2O \tag{10}$$

Nitrite Reductase

$$NO_2^- + 2\,H^+ + e^- \rightarrow NO + H_2O \tag{11}$$

Nitric Oxide Reductase

$$2\,NO + 2\,H^+ + 2\,e^- \rightarrow N_2O + H_2O \tag{12}$$

Nitrous Oxide Reductase

$$N_2O + 2\,H_+ + 2\,e^- \rightarrow N_2 + H_2O \tag{13}$$

Denitrification is a significant pathway of nitrogen loss from wetland ecosystems, because anoxic soil layers provide optimal conditions for nitrogen reducing microorganism. Nitrates are used as one of the first terminal electron acceptors by anaerobic microorganism after the depletion of oxygen (Schlesinger and Bernhardt, 2013; Mitsch and Gosselink, 2015). The approximate redox potential for nitrate reduction is +250 mV (between +100 and +300 mV), which is classified as moderately reduced (see Figure 2, page 21) (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015).

Figure 5 shows the major processes and components of the nitrogen cycle in wetlands. The zones of active nitrification-denitrification in wetlands are the aerobic-anaerobic interfaces at the flooded soil surface and the root rhizosphere of wetlands plants (Reddy and DeLaune, 2008). The entire process is induced by the formation of NH_4^+ and its diffusion into the aerobic zone where the conversion to NO_3^- via nitrification occurs. Afterwards NO_3^- diffuses into the anaerobic layer where the transformation to gaseous nitrogen forms (N_2 and N_2O) due to denitrification takes place. The diffusion rates of NH_4^+ and NO_3^- are regulated by the concentration gradients of the ions. The diffusion rate of NO_3^- is approximately seven times faster than for NH_4^+. Thus, the latter is the limiting factor for nitrification and subsequent denitrification, it constrains the whole process of nitrogen loss to the atmosphere (Mitsch and Gosselink, 2015).

2.5. Gaseous Transport

Gases formed from anaerobic processes accumulate in high concentrations in the upper soil layer and can be released into the atmosphere via diffusion, ebullition of gas bubbles (e.g. CH_4) or vascular transport through plant aerenchyma (Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). Diffusion is the transport of gases driven by their concentration gradient. This pathway is more important for CO_2 and N_2O emissions and less relevant for CH_4 because the gas has to pass the aerobic soil-floodwater interface where CH_4 can be oxidized into CO_2. Thus, the largest share of CH_4 reaches the atmosphere through other pathways (Yu et al., 1997; Reddy and DeLaune, 2008).

Gas ebullition of CH_4 only occurs in flooded sediments. The production of gas in soils combined with water inundation causes an impairment of gas exchange which leads to an increase of pressure and a rapid release of the accumulated gas in form of bubbles.

Atmospheric pressure changes and water level alterations can affect this process (Reddy and DeLaune, 2008). It is estimated that 30 - 85% of the total CH_4 emissions is released to the atmosphere through this pathway (Byrnes et al., 1995).

Vascular transport via plant aerenchyma is another important pathway for soil-borne gases to reach the atmosphere. Aerenchyma are air channels in stems, roots or leaves of wetland plants that allows them to exchange gases between roots and atmosphere (Reddy and DeLaune, 2008). More than 80% of the CH_4 and N_2O produced in rice paddies is emitted through rice plant aerenchyma (Yu et al., 1997). The remaining amount of gases (16%) is emitted directly through the soil/water/atmosphere interface by ebullition and diffusion. However, aerenchyma in wetland plants do not only stimulate CH_4 emissions, but also reduce them due to the delivery of oxygen to the rhizosphere leading to methane oxidation (Ding et al., 2005).

2.6. GHG Emissions and Climate Change

Between 1850 and 2000 the average atmospheric concentration increased from 270 ppm (v/v) to 360 ppm for CO_2 and from 0.7 ppm to 1.7 ppm for CH_4. While most parts of the CO_2 concentration increase is caused by fossil fuel burning since the industrial revolution, large parts of the increased CH_4 concentration arise from natural and anthropogenic sources, such as wetlands, rice paddies and livestock (Reddy and DeLaune, 2008). The

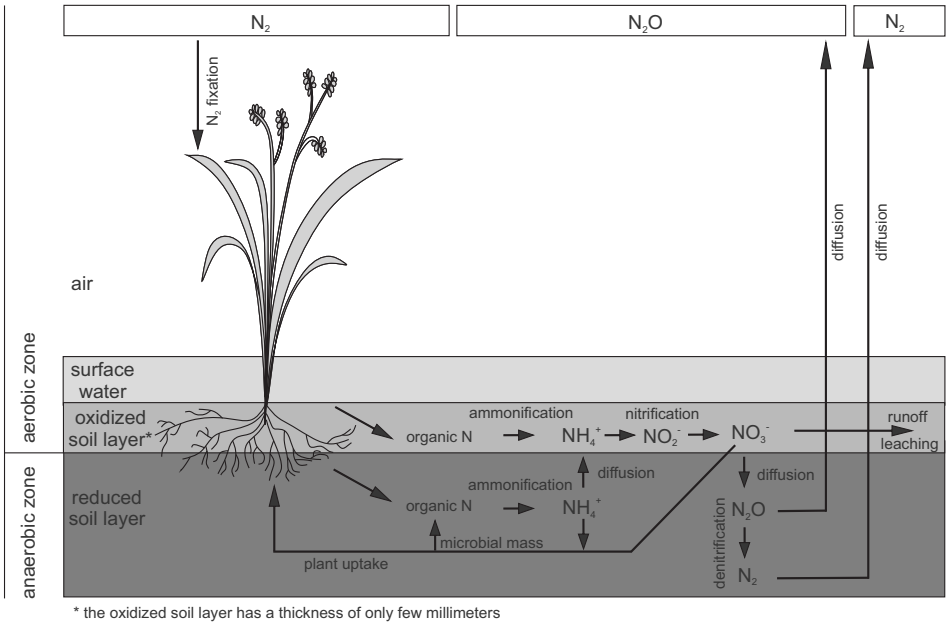

Figure 5: The nitrogen cycle in wetlands with its main processes and components.

role of wetlands on climate change is discussed widely in the literature. As described in chapter 2.3.1, 2.3.2 and 2.4, wetlands can be sinks as well as sources of carbon, nitrogen and GHGs (Whiting and Chanton, 2001; Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). Natural and constructed wetlands provide optimal environmental conditions for the production of the three most important GHGs (Whiting and Chanton, 2001; Mander et al., 2008): CO_2 is a gaseous end product of organic matter decomposition generated under oxic conditions, CH_4 is produced under pure anoxic conditions and N_2O can be produced under oxic as well as anoxic soil conditions. Due to enhanced microbial activities which result in an increased GHG production, wetlands can have negative effects on the global climate (Reddy and DeLaune, 2008). Wetlands and rice paddies are considered to be one of the most important sources of soil-borne CH_4 emissions (Reddy and DeLaune, 2008; Scheffer and Schachtschabel, 2010; Mitsch et al., 2012). Especially in rice paddies the amount of emitted CH_4 can be large due to high soil temperatures (Scheffer and Schachtschabel, 2010). The estimations for net CH_4 emissions from wetlands vary widely from 109 Tg CH_4-C (145 Tg CH_4) (Whalen, 2005), over 170 Tg CH_4-C (227 Tg CH_4) (Bloom et al., 2010) to 448 Tg CH_4-C per year (Mitsch et al., 2012). The estimated daily CH_4 emission rate from wetlands is 100 mg m^{-2} day^{-1} and represents the net effect of production and consumption (Whalen, 2005). Rice paddy areas account for approximately 49.5 Tg CH_4-C (66.0 Tg CH_4) (Bloom et al., 2010). In summary, 20 to 33% of the global CH_4 emissions from anthropogenic and natural sources arise from wetland areas (Whalen, 2005; Schlesinger and Bernhardt, 2013). Thereof, 55.5% to 78% arise from tropical and subtropical wetlands, which corresponds to 99 Tg CH_4-C (132 Tg CH_4) (Bloom et al., 2010; Mitsch et al., 2012). Rice paddies account for 29.1 ± 0.6% (Bloom et al., 2010).

N_2O is another GHG that is produced in wetlands and is known to have an effect on climate change (see chapter 2.4). N_2O influences the depletion of stratospheric ozone and contributes with 5-10% to the anthropogenic GHG effect. It has an atmospheric lifetime of 100-200 years (Reddy and DeLaune, 2008). The annual global emission of N_2O to the atmosphere is estimated to be 17.7 Tg N_2O-N yr^{-1} (Kroeze et al., 1999). Thereof, 37% of the total N_2O emissions arise from soils covered with natural vegetation (IPCC, 2013). However, the observed change in N_2O fluxes from 11 Tg N yr^{-1} in 1850 to 18 Tg N yr^{-1} in 1994 is mainly due to increased nitrogen fertilization in agriculture and combustion processes (Kroeze et al., 1999; Reddy and DeLaune, 2008). Estimates concerning the contribution of wetlands to the global N_2O emissions are insufficient and have a high degree of uncertainty (Mitsch and Gosselink, 2007; Reddy and DeLaune, 2008). There are major differences in the rate of nitrogen cycling and N_2O emissions regarding diverse wetland ecosystems. For example, rice paddies feature high nitrogen loadings and correspondingly high N_2O emissions compared to natural wetlands. Moreover, N_2O emissions from sites featuring a fluctuating redox potential are higher compared to soils with permanently flooded conditions and a continuously low redox potential (Reddy and DeLaune, 2008). In general, the N_2O emissions from wetlands are assumed to be smaller compared to

upland soils and farm fields (Hernandez and Mitsch, 2006; Hernandez and Mitsch, 2007; Mitsch and Gosselink, 2007).

In contrast to being a GHG source, wetlands also store great amounts of the terrestrial biogenic carbon and therefore play an important role in regulating atmospheric GHG concentrations. The estimated percentage of carbon stored in wetland soils in relation to the total amount of carbon stored in the world's soils ranges between 14.5% (202 Pg) (Reddy and DeLaune, 2008) to 20-30% (455-700 Pg) (Mitsch and Gosselink, 2015) and even more than 50% (Gorham, 1991; Schlesinger and Bernhardt, 2013). It is assumed that wetlands act as net carbon sink with an order of 830 to 1000 $Tg\,yr^{-1}$ (Mitsch et al., 2012; Mitsch and Gosselink, 2015). The average carbon net retention is 118 $g\,C\,m^{-2}\,yr^{-1}$ (Mitsch et al., 2012), while in tropical and subtropical wetlands the rate accounts to 150 to 250 $g\,C\,m^{-2}\,yr^{-1}$ (Mitsch and Gosselink, 2015). It is estimated that the carbon stored in peatlands reduced the global temperature by approximately 1.5-2° C within the past 10,000 years and therefore played a major role in climate control (Holden, 2005). However, during the recent years peatlands showed the trend of losing their storage properties due to drainage, fires and climate change and therefore turn into sources of carbon.

When considering the role of wetlands for climate change, the global warming potential (GWP) for each GHG has to be taken into consideration. CH_4 and N_2O have a much stronger effect on global warming than CO_2 due to their higher GWPs (Whiting and Chanton, 2001). The standard GWPs defined by the IPCC are summarized in Table 1 (IPCC, 2013). Mitsch et al. (2012) reported that the carbon sequestration of wetlands is not sufficiently high enough compared to the GWP_{100} of CH_4 (which was then assumed to be 25 according to IPCC (2007)), because CO_2 sequestration in most wetlands is not 25 times higher than the emitted amount of CH_4. Thus, they classify wetlands as source of climate warming. However, after a time period of 300 years, most wetlands turn into net carbon sinks because CH_4 decays in the atmosphere after 8-12 years making CH_4 emissions irrelevant compared to carbon sequestration (Mitsch et al., 2012). Whiting and Chanton (2001) also came to the result that the release of CH_4 contributes to the overall greenhouse effect because of the higher infrared absorptivity of CH_4 relative to CO_2. However, because the GWP of CH_4 decreases over longer time horizons (100 years), the authors assumed an attenuating effect especially of subtropical and temperate wetlands on global warming. Considering a 500-year time horizon, they conclude that wetlands are sinks of GHGs with positive effects on climate change. Kayranli et al. (2010) concluded that wetlands can be sources as well as sinks of carbon, depending on several factors such as age, management, location and climate. Mitsch and Gosselink (2015) did not make a final decision concerning the categorization of wetlands as GHG sink or source. However, they stated that the scale of opportunities for managing GHG emissions from wetlands is generally not large enough to have a major impact on the global carbon balance. Other studies claimed that the creation or restoration of wetlands can be a promising way to mitigate global warming (Whiting and Chanton, 2001; Kayranli et al., 2010; Mitsch et al.,

Table 1: GWP defined by the IPCC (IPCC, 2013).

GHG	Lifetime (years)	GWP_{20}	GWP_{100}
CH_4	12.4	84	28
N_2O	121.0	264	265

2012). With improved design, construction and operation of wetland areas, carbon storage can be enhanced and the release of GHGs, especially CH_4, will be reduced (Kayranli et al., 2010). Moreover, restored wetlands also have the potential to have a decreasing effect on N_2O emissions on landscape scale (Hernandez and Mitsch, 2006; Hernandez and Mitsch, 2007).

Global climate change is not only influenced by GHG emissions from wetlands, but also has significant effects on wetlands. The impacts of climate change on wetland ecosystems vary depending on type, magnitude or rate of change in precipitation and temperature. Changes in hydrology, biogeochemistry and biomass accumulation are anticipated for the future (Mulholland et al., 1997; Reddy and DeLaune, 2008). A rise of 7% in CH_4 emissions from wetlands is estimated for the period 2003 to 2007, caused by temperature increases at extratropical latitudes (45° to 67°N) (Bloom et al., 2010). The emissions from tropical wetlands remained mainly constant within this time period (Bloom et al., 2010). More information are required for a precise prediction of how the carbon balance of wetlands will respond to the anticipated global climate change. This includes a better understanding of the impacts of water level fluctuations from wetlands on carbon fluxes under changing climatic conditions (Kayranli et al., 2010).

2.7. Land-Use Change and Effects on the Ecosystem

Large wetland areas worldwide are attractive as farmland with great potential for food production because of their fertile soils and good water availability (Reddy and DeLaune, 2008). The pressure on wetlands for a conversion into farmlands is growing constantly due to global population growth, overexploitation of upland soils and rising food insecurity (Wood and van Halsema, 2008; Rebelo et al., 2009b; Barrett, 2010). Particularly in East Africa, government policies and the population put stress on land-use change, as the livelihoods of many people are supported by food supply from wetlands (Dixon and Wood, 2003; Bikangaga et al., 2007; Rebelo et al., 2009b). Thus, many wetland areas were turned into farmlands and lost their typical environmental characteristics (Reddy and DeLaune, 2008).

Until the middle of the twentieth century, drainage was the only considered policy for wetland management. Government programs encouraged landowners to drain wetlands

to make them suitable for agricultural usage (Mitsch and Gosselink, 2015). However, this attitude changed in the mid-1970s as the awareness for the importance of wetlands as wildlife habitat for water quality management and flood control increased. Today, wetland management involves the reconciliation of environmental aspects with various stakeholder interests (Mitsch and Gosselink, 2015). This is a great challenge because of the numerous ecosystem services that are provided by wetlands (e.g. food and water supply, water purification and climate regulation) and the sometimes conflicting interests of involved parties (Millennium Ecosystem Assessment, 2005). For example, the interest for increased agricultural production, which implies a conversion of natural wetland areas to farmlands or an intensification of land use, has to be reconciled with the positive effects of wetland services on the environment. As explained in chapter 2.6, natural wetlands feature beneficial conditions for the sequestration of carbon and thus have a great potential for the mitigation of global warming (Whiting and Chanton, 2001; Kayranli et al., 2010; Mitsch et al., 2012). However, after their conversion to agricultural areas, former wetlands can rather be classified as upland soils, due to the artificially controlled drainage that manipulates the water table for growing crops (Reddy and DeLaune, 2008). The drainage of wetland soils leads to an exposure of large carbon stocks to aerobic decomposition and an increase in carbon turnover rates. The decomposition of organic matter, which has been very slow under flooded anoxic conditions, changes into rapid aerobic decomposition and results in increased CO_2 emissions (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). On the other hand, drainage in wetlands has the positive effect that CH_4 emissions decrease significantly because methanogens are restricted to anoxic soil conditions and cannot produce CH_4 when oxygen is available. Moreover, land-use change also comes along with alterations in productivity. Because of slow turnover processes in wetland soils that result in low concentrations of available nutrients, a high proportion of nutrients is sequestered in the undecomposed soil organic matter. Areas which are less frequently flooded and show periodic drying feature a more rapid nutrient mineralization by aerobic microbes resulting in a higher productivity of vegetation (Schlesinger and Bernhardt, 2013). Moreover, drainage of wetlands is supposed to cause secondary water pollution because of the release of nutrients, as demonstrated in a peatland restoration project in Northern Germany (Scholz and Trepel, 2004a; Scholz and Trepel, 2004b; Kayranli et al., 2010). The biodiversity of wetlands, including wildlife and waterfowl, is also seriously affected by land-use change. Although wetland management practises try to maintain the natural conditions and hydrology as close as possible, it is difficult to accomplish this goal for wildlife conservation because planning and imitation of natural ecosystem conditions are challenging (Mitsch and Gosselink, 2015).

To prevent further degradation of wetland ecosystems the *Ramsar Convention on Wetlands* provided concepts for conservation measures and a wise use of wetlands (Matthews, 1993). The aim was to establish comprehensive national wetland policies and to integrate them into the national planning process. The guidelines included the following actions:

- improvement of institutional and organizational arrangements (identification of possibilities for wetland conservation and integration into the planning process)
- addressing of legislation and government policies (review of the existing situation, application of existing policies and adoption of new ones)
- increase of knowledge and awareness (information exchange with other countries, increase of awareness of decision-makers and the public, review of traditional techniques)
- review of the priorities for particular national wetland sites (identification of individual conservation and management needs)

(Matthews, 1993; Mitra et al., 2003). Some countries, such as Uganda, already established wetland protection policies, while others are in the process of doing so. For a successful realization of wise wetland use, it is important to consider poor farmer's interests and to integrate local and traditional agro-ecosystems into the sustainable management practises of wetlands. Several aspects, including yield and economic profitability as well as cultural factors, have to be taken into account when determining the sustainable productivity of agricultural systems. The key to successful wetland management is a participatory approach that aims for a reconciliation of all stakeholder interests (Mitra et al., 2003).

2.8. GHG Measurements in Africa

The role of Africa in the carbon cycle of the 21st century is of great importance and has potential implications for global climate change (Williams et al., 2007). However, the number of studies about carbon dynamics and greenhouse gas emissions in tropical regions, especially in Africa, are relatively rare. Oertel et al. (2016) reviewed soil emission studies from most relevant climate zones and land-cover types worldwide. They draw attention to the apparent bias of GHG research in the northern hemisphere and the urge to increase the number of studies conducted in the southern hemisphere. Especially sub-Saharan African is strongly underrepresented as shown in Figure 6. In comparison, Europe, China and the United States are well covered with GHG data (Caritat and Reimann, 2012; Oertel et al., 2016). Most assessments of Africa's carbon dynamics are model-based, but the estimations have large uncertainties. The network of regional site-based flux observations is relatively sparse and represents a remarkable limitation of the knowledge about Africa's carbon cycle (Williams et al., 2007). Especially CH_4 and N_2O emissions are insufficiently studied (Valentini et al., 2014). This gap in knowledge is a significant restraining factor for the development of wise resource management strategies for GHG mitigation (Williams et al., 2007).

Africa has 20% of the global land mass and 14% of the global population, but its contribution to the global fossil fuel carbon emissions is disproportionally small with 3% (UNFPA, 2004; Williams et al., 2007; Boden et al., 2010). Instead, fires and land conversion are the primary releasing factors for carbon (Williams et al., 2007). It is

estimated that >50% of the gross fire emissions worldwide and 17% of all global carbon emissions from land-use changes arise from Africa. The annual emission rate from wild fires is roughly 1.03 ± 0.22 Pg C yr^{-1}, whereas 0.32 ± 0.05 Pg C yr^{-1} are due to land-use change (Werf et al., 2006; Canadell et al., 2009; Valentini et al., 2014). Major regional differences in GHG emissions could be observed. South and North Africa contributed to almost the total fossil fuel emissions of the continent, while West and East Africa showed considerable emissions from agriculture, land-use change and forestry. Central Africa dominated the emissions from the land-use change and forestry sector, but was also observed to be a carbon sink (Valentini et al., 2014). In general, Valentini et al. (2014) suggested to regard Africa as small carbon sink on an annual scale. The average emission rate amounted to -0.61 ± 0.58 Pg C yr^{-1}. However, even though the overall and annual carbon budget indicates Africa to be a small carbon sink, this does not implicate it to be a net sink for radiative forcing as well. Taken N_2O and CH_4 emissions in terms of their CO_2 equivalent properties into account, Africa can rather be considered as a net source of radiative forcing. Nevertheless, Valentini et al. (2014) did not consider all aspects of the carbon cycle, for example they did not include the effects of wetlands in their study. It is estimated that Africa has approximately 34.5 million hectares of wetlands (Finlayson, 1991; Matthews, 1993). However, only few site-based research on GHG emissions from African wetlands have been conducted, for example by Jones (2001) (observing CH_4 emissions in Kenya),

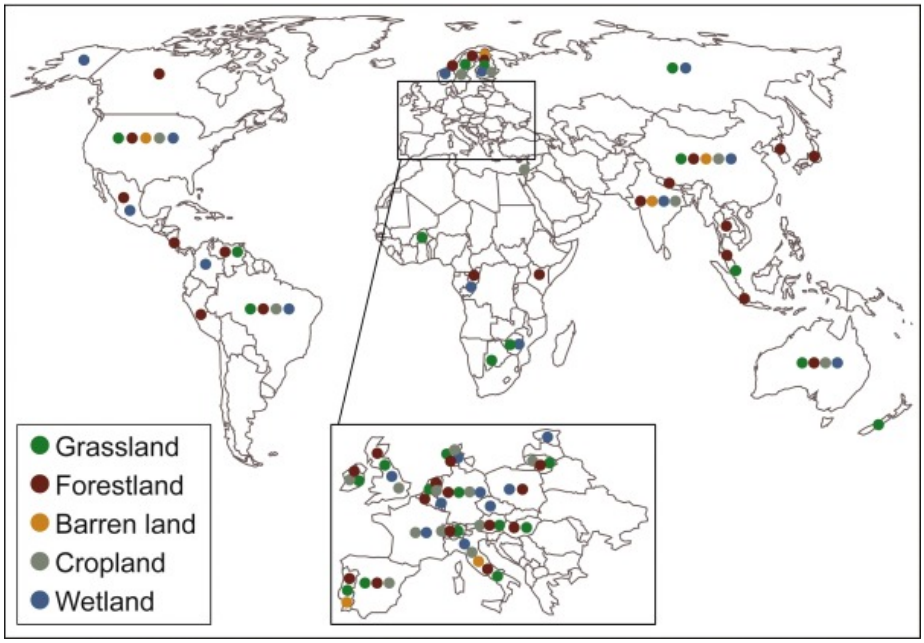

Figure 6: Worldwide distribution of GHG study sites and land-cover types reviewed by Oertel et al. (2016).

Jones and Humphries (2002) (CO_2 emissions in Kenya), Krüger et al. (2012) and Krüger et al. (2013) (CH_4, CO_2 and N_2O emissions in South Africa) and Nyamadzawo et al. (2014) (CH_4 and N_2O emissions in Zimbabwe). As described in chapter 2.6, wetlands can have the potential to influence the GHG budget not only regionally but also globally. For a better consideration of the impacts of wetlands on GHG emissions from Africa a great research gap needs to be filled.

3. Research Objectives

This thesis focuses on the impact assessment of land-use change and rice cropping methods on GHG emissions from agriculturally used wetlands in East Africa.

The main research objectives were:

- Development of standardized quality criteria for GHG data from static chamber measurements
- Assessment of GHG emissions with consideration of:
  - contrasting wetland types
  - different types of land use
  - different hydrological positions in the wetland
- Evaluation of several agricultural treatment methods with respect to environmental sustainability focussing on yield-based GHG emissions

A new data quality management system (DQMS) was developed to meet the need for a systematic quality control for GHG data collected with static chambers (chapter 4). For data collection, two long-term field studies were conducted in two East-African wetlands. The first field trial was implemented in an inland-valley wetland in central Uganda from September 2014 to September 2016 (chapter 5). The second field experiment was located in a lowland floodplain of the Kilombero river in Tanzania and ran between March 2015 and October 2016 (chapter 6).

4. Development of a Data Quality Management System for the Improvement of Manual Greenhouse Gas Flux Measurements with Static Chambers

Abstract

The most widely used method for estimating GHG emission fluxes from static chamber data is linear regression over time of the GHG concentration. The quality of the linear fit is generally quantifed by calculating the coefficient of determination R^2. Many studies report a decent fit of their data with this method with values of $R^2 \geq 0.9$ (e.g. Conen and Smith (2000)). However, there are also data sets with a less satisfying fit to the linear flux calculation approach, for example samples collected in remote areas with poor infrastructure and correspondingly long sample transport and storage periods. A quality control system only based on the R^2 criterion, which rejects all sample sets with $R^2 \leq 0.9$, can result in the exclusion of large parts of a data set and a deterioration of its informative value. To avoid a considerable loss in data quantity and to improve data validity, a quality management system consisting of eight control steps, that can save large parts of a data set by identifying and rejecting single outliers while ensuring a high data quality at the same time, was developed. The newly developed system improved the acceptance of three tested data sets considerably by up to 37.2 percentage points (pp) for CH_4, 11.7 pp for CO_2 and 48.3 pp for N_2O.

4.1. Introduction

4.1.1. Greenhouse Gas Flux Calculation

Greenhouse gas (GHG) flux calculation by linear regression of static chamber data is an established method in ecosystem and climate research (Gao and Yates, 1998; Yamulki and Jarvis, 1999; Conen and Smith, 2000; Forbrich et al., 2010). A fixed number of GHG samples, usually three to seven, is collected in regular time intervals during the chamber closure period. The fluxes of the measured gases can be calculated from this sample set by the concentration change in chamber headspace over time (C_t)(Arias-Navarro et al., 2013):

$$f(gas) = \frac{V \Delta C}{A \Delta t} \tag{14}$$

f is the gas flux (g gas m^{-2} s^{-1}) for either CH_4, CO_2 or N_2O, V is the headspace volume of the chamber (m^3), A is the soil surface area (m^2) and $\frac{\Delta C}{\Delta t}$ is the change in gas concentration per time unit within the chamber headspace.

The linear regression approach is based on the assumption that GHG fluxes between soil and atmosphere and the corresponding concentration gradients are constant due to a steady soil gas production (Kutzbach et al., 2007; Pihlatie et al., 2013). Numerous studies

are based on this assumption (e.g. Brümmer et al. (2009), Ma et al. (2013), Shi et al. (2013)) and several experimental studies confirm a good fit of the linear model for static chamber measurements (e.g. Yamulki and Jarvis (1999), Forbrich et al. (2010), Chadwick et al. (2014)).

4.1.2. Need for Data Quality Management System

During the comprehensive investigation for this study, it was noticed that there is a lack of a systematic quality assurance for GHG data which are collected with static chambers and measured via gas chromatography. Many studies that assess the quality of GHG measurements either deal with the effects of chambers on GHG emissions from soil and the technical improvement of chamber construction (e.g. Klein and Harvey (2012), Pihlatie et al. (2013)) or focus on the optimization of GHG flux calculation models (e.g. Kutzbach et al. (2007), Kroon et al. (2008), Pedersen et al. (2010), Pihlatie2013). From the state of knowledge, there are only few information about the quality control of GHG data obtained from gas chromatography and the controlled removal of outliers based on objective criteria. Many previous studies had no need for the development of a structured data quality management system which checks GHG data from static chamber measurements regarding their suitability for subsequent flux calculation, because the collected data either showed a good linear fit (Gao and Yates, 1998; Yamulki and Jarvis, 1999; Conen and Smith, 2000) or non-linear fit (Livingston et al., 2005; Livingston et al., 2006; Kutzbach et al., 2007; Kroon et al., 2008). Some studies, such as Kutzbach et al. (2007), assume that potential errors of static chamber measurements are negligible and violations of their physically based model assumptions can be minimised by careful planning and performance. However, stable conditions during sampling, such as headspace temperature, humidity, headspace turbulence and pressure perturbations during chamber placement as well as the avoidance of leaks, are difficult to implement under real field conditions (Kutzbach et al., 2007). Nowadays GHG research is also conducted in remote areas with poor infrastructure and eventually unreliable power supply. GHG samples from those regions often have to be stored for long periods and transported over long distances until analysis is performed. Despite utmost diligence during sample preparation and field work, the integrity of arriving samples can be questionable and obtained data can show an unsatisfying fit to the selected flux calculation approach.

An established tool to control the goodness of fit for the most preferred flux calculation model in climate research, i.e. linear regression, is the coefficient of determination R^2. For example, Ma et al. (2013) and Mitsch et al. (2012) rejected all flux sample sets that yielded a R^2 value of less than 0.90. However, the application of R^2 as only quality criterion can result in the exclusion of large parts of a data set, because the whole flux sample set, which consists of the data points of samples taken during chamber closure, is rejected completely. The consequence can be a considerable loss of flux results, especially

regarding data collected under challenging field conditions. For example, from the three data sets tested in this thesis, the Tanzanian Rice set showed the highest removal rate of up to 75.6% for CH_4, 65.9% for CO_2 and 81.1% for N_2O from the total data set, when using R^2 as the only criterion. In contrast, a systematic identification of outliers is beneficial for rejecting clearly erroneous data points from a flux sample set and preserving the remaining ones. This led to the realization that there is a strong need for the development of an automated DQMS which tests collected GHG data for their suitability for linear flux calculation and, if necessary, removes single outliers based on objective criteria to improve the reliability of the remaining data points and to preserve as many data points and flux sample sets as possible. The application of a DQMS benefits the quality and preservation of the total data set.

The following requirements were developed for the DQMS:

- removal of outliers based on objective criteria to avoid a subjective bias in the final data set
- acceptance of as many data points as possible without compromising the quality of the final data set
- automated rejection or acceptance of data in a time efficient process

4.1.3. Removal of Outliers

An essential demand for the DQMS is the selective rejection of outliers based on objective criteria. The rejection of a single outlier from a flux sample set can save the remaining data points of the set for final flux calculation. Not many studies have dealt with this issue until now. Mitsch et al. (2012) and Ma et al. (2013) used the R^2 as filter criterion. Kutzbach et al. (2007) rejected clearly erroneous samples of their Linnasuo data set based on visual inspection of the data curves. For exponential flux calculation Kutzbach et al. (2007) and Forbrich et al. (2010) used the standard deviation of residuals (σ_{Res}). Moreover, Forbrich et al. (2010) took unrealistic low offsets (based on the assumed atmospheric concentration before chamber deployment) as well as unusually high starting concentration values (caused by ebullition) as another quality criteria to exclude measurements. Saarnio et al. (1997) mentioned the rejection of erroneous samples without explaining the method in detail. Pihlatie et al. (2013) used the normalized root-mean-square-error (NRMSE), which is a relative measure of distances of individual measurement points from the fit, for the removal of outliers. Outliers were filtered from the data using defined NRMSE limits. Kroon et al. (2008) rejected concentration curves that were physically not explainable. In addition, they used R^2 as rejection criterion for linear regression and the relative standard deviation of estimates (σ_{Est}) for exponential models. Other studies did not include the removal of any data points at all (e.g. Pedersen et al. (2010), Kutzbach et al. (2007) for the Vaisjeäggi data set).

As described above, the application of R^2 as single criterion for rejection can lead to undesired frequent exclusions of sample sets from the total data set and is therefore inappropriate as single criterion, whereas outlier removal based on visual inspection is time-consuming and bears the risk of introducing a subjective bias. This approach aimed for an automated system built on objective criteria. Although Kutzbach et al. (2007) indicated that data rejection can lead to an underestimation of fluxes, this study takes the view that the rejection of disturbed data points is necessary to save large parts of the total data set and therefore improves the informative value of obtained GHG data.

The developed data quality management system (DQMS) has the purpose to reject single outliers, but also to save as many data points and sample sets as possible, without compromising the quality of the total data set. The remaining flux results were considered to be suitable for further (linear-) flux calculation.

4.2. Methods

4.2.1. Field Measurements

The described DQMS system was tested on three data sets from two agriculturally used wetland areas in Uganda (Namulonge, $00°31'30"N$, $32°36'54"E$) and Tanzania (Ifakara, $-08°07'60"S$, $36°40'60"E$). The test site in Uganda consisted of a rice- and maize cropping experiment which was located in a low mountain range area with an altitude of 1100 m above sea level and an average air pressure of 875.7 hPa. The climate is sub-humid with an average precipitation of 1350 mm (Leemhuis et al., 2016; Gabiri et al., 2018a). The focus area in Tanzania included a rice experiment located in a lowland floodplain of the Kilombero river 250 m above sea level with an average air pressure of 981.8 hPa. It is characterized by a sub-humid climate with an average precipitation of 1200 to 1400 mm (Koutsouris et al., 2015; Gabiri et al., 2018b). Each field trial was divided in three fields located in different hydrological zones: the fringe, mid-section and center position of the wetland. The positions differed regarding their topographic elevation, as well as lengths and intensity of their annual flooding period (see Figure 7). GHG samples were collected in four agricultural treatments and a (semi-) natural vegetation area with three replications per hydrological zone. Closed chamber measurements and manual sampling took place every two weeks for two cropping periods in each country, from September 2014 until August 2016 in the Ugandan Rice field, from October 2014 until September 2016 in the Ugandan Maize field and from March 2015 until September 2016 in the Tanzanian Rice field. The measurement chambers were made of robust non-transparent polypropylene boxes with an area of 0.6 m x 0.4 m. The chambers in the maize field had a height of 0.2 m, therefore only bare soil respiration was measured. In the rice fields and in the (semi-) natural vegetation area the whole plants were included in the chamber due to the presence of aerenchyma. Therefore, a height of 0.4 m or 0.8 m was needed, depending on the stage of plant growth. For GHG sampling the chambers were placed on polypropylene

frames, which were permanently installed in the soil, and fixed with 32 mm foldback clips to ensure gas tightness of the measuring chamber during the closure time of 60 min. In highly flooded areas, e.g. in the Tanzanian Rice field, the chambers were placed on styrodur rings that enabled them to float. In total, four GHG samples where collected in an interval of 20 min between each sampling ($t_1 = 0\,min, t_2 = 20\,min, t_3 = 40\,min, t_4 = 60\,min$).

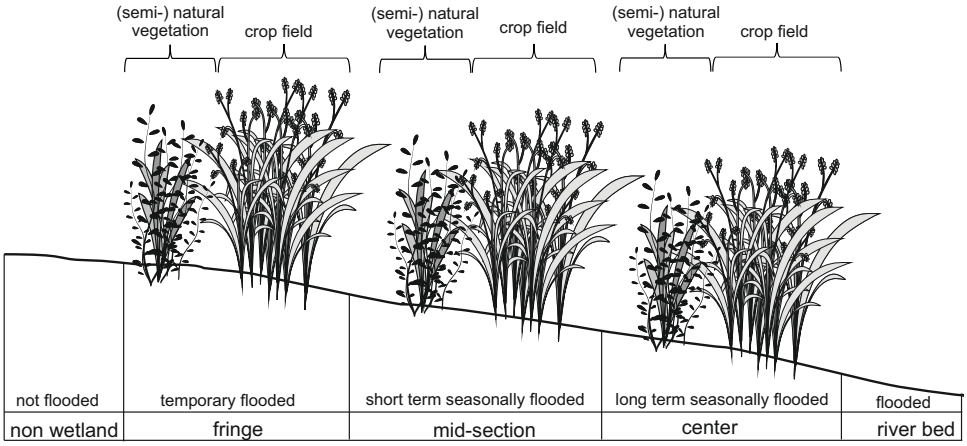

Figure 7: Hydrological positions in the central field trial

4.2.2. Sample Analysis

The collected gas samples were analysed with a gas chromatograph (GC) (European Greenhouse GC 8610C, SRI Instruments Europe GmbH, Bad Honnef, Germany) equipped with flame ionization detector (FID) for CO_2 and CH_4 measurements and electron capture detector (ECD) for N_2O measurements. The transfer of air samples from sample vial to the injection port of the GC was conducted with an autosampler syringe. A minimum overpressure of 0.16 bar in the sample vials was required to avoid ambient air to intrude into the syringe during sample transfer. The vials were checked after each measurement for their remaining overpressure.

4.3. Results

4.3.1. Data Quality Management System

The model is based on the assumption of steady production and emission of the target gas from soil to atmosphere. This implies a linear increase of gas concentration in samples taken during chamber closure period. The only exception is steady gas consumption which results in a linear decrease. The developed data quality management system consists of eight steps that test each flux sample set (t_1, t_2, t_3, t_4) for their compatibility to linear gas

flux calculation and removes outliers if necessary. Figure 8 shows the decision tree which was programmed in R version 3.3.1 (2016-06-21).

Step 1a - Linearity

The first quality control step tests for linearity of the data set by calculating the R^2 which is a main tool for goodness of fit control in linear regression (Khuri, 2013). The ${R_1}^2$ is calculated for each flux sample set (n = 4). A ${R_1}^2 > 0.9$ is desired and considered as suitable for further flux calculation (e.g. compare with Ma et al. (2013), Mitsch et al. (2012)) and forwarded to step 1b. The remaining flux series that do not meet the criterion are forwarded to step 2.

Step 1b - Negative CO_2 fluxes

CO_2 concentration values usually show an increase during chamber closure when non-transparent chambers are used. Due to the absence of light, any photosynthetic activity of plants within the chamber is inhibited, therefore no CO_2 uptake is expected and positive CO_2 fluxes are presumed. If a decrease in CO_2 concentration during chamber operation is observed, it very likely results from water vapour condensation within the chamber. CO_2 is soluble in water (Weiss, 1974; Myers, 2003). Thus, water condensation during chamber operation can cause serious disturbance of flux measurements, because CO_2 dissolves in condensed droplets. This results either in an uncontrolled decrease or increase of the CO_2 concentration, depending on whether condensation or subsequent evaporation is prevailing. Data series which show a decreasing CO_2 trend are considered to be unreliable in our model. In this case, the flux sample sets for all gases, including CO_2, CH_4 and N_2O, are removed completely because disadvantageous effects of condensation on the other two gases can not be excluded with absolute certainty. Data series with positive CO_2 fluxes and the corresponding CH_4 and N_2O results are kept and stored in the final data set.

Step 2 - Zero Fluxes

The precision of the analytical instrument for the particular gas concentration is an essential factor which has to be taken into account (Kutzbach et al., 2007). Zero fluxes are sample sets with very low flux rates whose measurements range within the mean standard deviation (2σ) of the measurement precision of the gas chromatograph (GC). Flux sample sets with measurements within this range provide acceptable results, even though their coefficients of determination (${R_1}^2$) might be smaller than 0.9 due to eventual measurement inaccuracies from the GC. To identify and preserve those flux sample sets, the measurement precision of the GC has to be taken into account. The GC used in this study had a 2σ range of 0.022 ppm for CH_4, 15.27 ppm for CO_2 and 0.0035 ppm for N_2O. Sample sets (n = 4) in which the average deviation of each data point from the mean is smaller than two times the standard deviation (2σ) of the GC are identified as zero fluxes (see equation 15) and stored in the final data set, while the remaining data are transferred

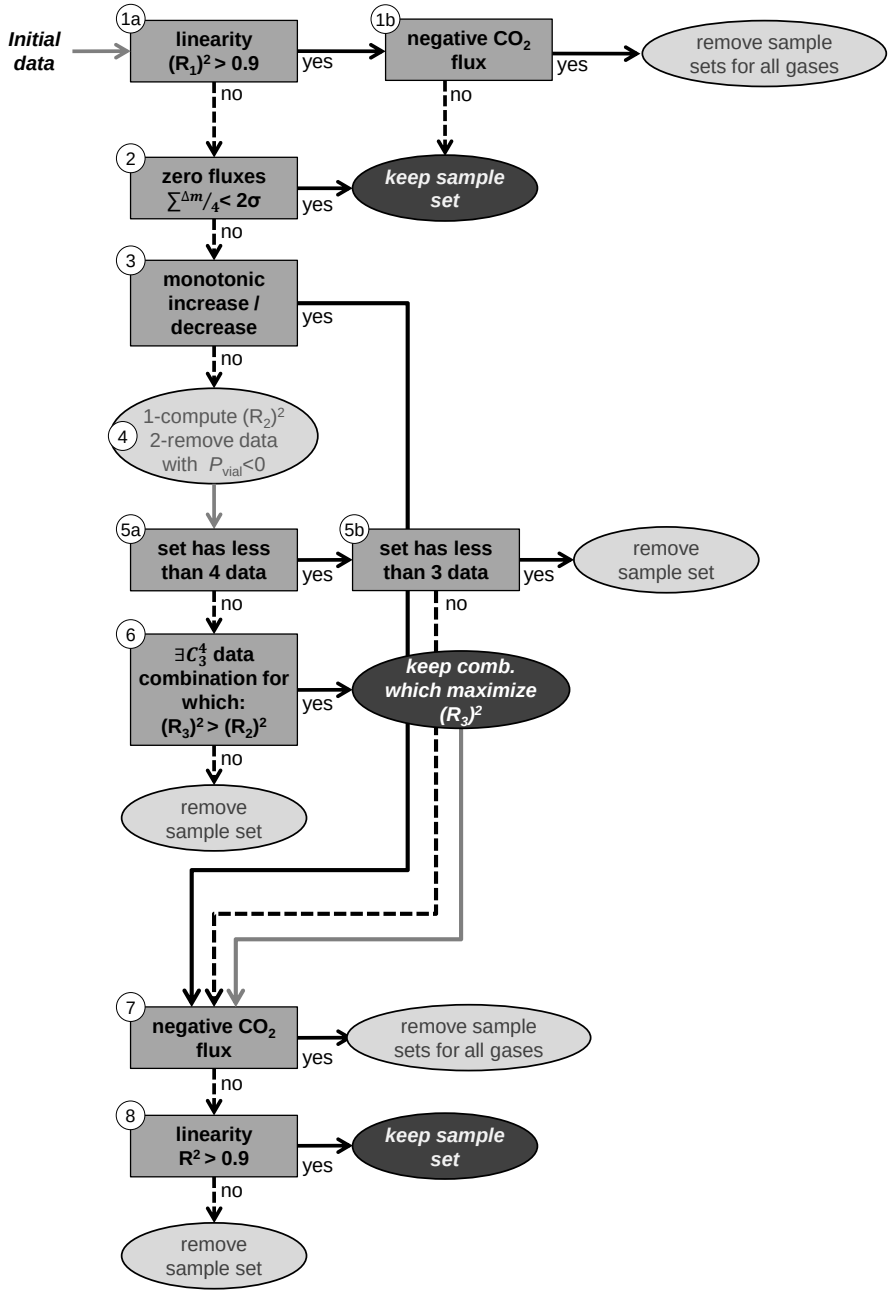

Figure 8: Data Quality Management System: each flux data series is tested according to eight quality criteria.

to step 3.

$$\sum_{1}^{4} \frac{|\Delta m|}{4} < 2\sigma \tag{15}$$

$|\Delta m|$ is the deviation of each data point from the mean concentration of the flux sample set. 2σ is the mean standard deviation of the measurement precision of the GC.

Step 3 - Monotonic Increase or Decrease

As the DQMS assumes a constant emission (or uptake) of GHG during chamber measurement, it also assumes a monotonic increase (or decrease) of concentration within a flux sample set. If this requirement is not fulfilled, it is taken as indication for an abnormality in the sample set, possibly caused by disturbance during sampling or leakage of individual samples. Thus, non-monotonic curve characteristics and divergent data points contribute to the identification of outliers. The removal of single outliers is essential for the preservation the other data points in the data set and therefore for the conservation of as many sample sets as possible in the final data set. Consequently, the removal of outliers benefits a better quality of the total data set, provided that it is based on objective criteria which allow a clear and non-biased identification of outliers. For more information about the rejection of values discussed in the literature see also chapter 4.1.3). The DQMS uses the universally valid criteria of monotonic increase or decrease to give a transparent and safe approach for the identification of erroneous values. If the gas concentration values of a flux sample set do not show a monotonic increase (or decrease), the set is forwarded to step 4. In the event that the criterion is fulfilled, the data are directly transferred to step 7.

$$\text{monotonic increase} \quad y_1 < y_2 < y_3 < y_4 \tag{16}$$

$$\text{monotonic decrease} \quad y_1 > y_2 > y_3 > y_4 \tag{17}$$

Step 4 - Removal of sample vials with negative pressure ($P_{vial} < 0$)

Step 4 is divided into two substeps:
(1) $R_2{}^2$ of all current flux sample sets is calculated again and will be further needed in step 6.
(2) An underestimated but major issue in analysing greenhouse gas emissions is the control of gas tightness of the used glass vials and validity of the gas samples. Especially, when research focuses on remote areas with inadequate infrastructure and unreliable power supply, the importance of proper vial sealing over long storage periods and during transport has to be taken into consideration. Transport via airplane with low ambient

pressure increases the risk of gas contamination through the septa of the storage vials which leads to uncertainties regarding the integrity of the gas sample (Glatzel and Well, 2007). Injection of 0.75 bar overpressure into the gas vial is recommended to counteract gas diffusion. This recommendation is based on pre-tests run before field work started. Moreover, it is an assurance for the integrity of the gas sample if overpressure is still present after transport. Many GCs, including one used in this study, require overpressure in the sample vials to work properly (e.g. 0.16 bar). The used GC is characterised by an open system without a direct connection between the gas sample and the detector, but with a syringe that transfers the sample from the vial to the injection port. Those systems require overpressure in the vials to prevent a contamination of the gas sample with ambient air. If the overpressure is insufficient, negative pressure will form in the vial while the sampling syringe sucks the required amount of sampling air from it. When the syringe is pulled out of the vial to transfer the sample air to the injection port, ambient air will be sucked into the syringe due to the present negative pressure, causing a contamination of the sample air. Therefore, each vial is checked for its remaining air pressure after GC analysis, the presence of negative pressure ($P_{vial}<0$) is an indicator that the gas sample was already leaky before measurement. In this case, the data point is removed from the corresponding sample set. The remaining data points of the sample set are forwarded to step 5a. If no sample features negative pressure, step 5a is carried out directly.

Step 5a - Number of remaining data ($n < 4$)

Sample sets with less than 4 data points ($n < 4$) are forwarded to step 5b, while sets which are still complete ($n = 4$) are transferred to step 6.

Step 5b - Number of remaining data ($n < 3$)

Sample sets that contain less than 3 data points ($n < 3$) are removed from the final data set because the number of remaining data points is considered to be too small to provide reliable flux results. If the data set consists of 3 data points ($n = 3$), it is forwarded to step 7.

Step 6 - Selective removal

Besides negative pressure in the sample vials ($P_{vial} < 0$), there are various other reasons for faulty data points. For example, change of physical conditions in the field during chamber measurement, including variations in soil and headspace air temperature, inconstant air pressure, headspace turbulence, unsteady photosynthetically active radiation (Kutzbach et al., 2007), or suddenly increased CH_4 emissions caused by ebullition (Forbrich et al., 2010; Leventhal and Guntenspergen, 2004). These occurrences usually lead to a disturbance of the assumed monotonic increase (or decrease) of data points, abnormal progression of the concentration curve and significant deterioration of the R^2 value. To identify biased

data points the concentration versus time curve of each sample set is tested for outliers which influence the goodness of fit of linear regression considerably in a negative way. It is assumed that the outlier of a flux sample set is the value which has the most unfavourable effect on the R^2 value compared to the other data points. Therefore, each element of the flux sample set is selectively removed one after the other. The respective coefficients of determination of the remaining three elements (R_3^2) are calculated and in each case compared with R_2^2 calculated in step 4 (R_2^2), which represents the valid R^2 value so far at this stage, until the most favourable option is found (see equation 18).

$$R_3{}^2 > R_2{}^2 \tag{18}$$

The element whose removal results in the largest improvement of $R_3{}^2$ is assumed to be an outlier and excluded from the flux series. The remaining data points of the sample set are forwarded to step 7. If no favourable combination can be found which surpasses the R_2^2 from step 4, the total flux sample set is removed.

Step 7 - Negative CO_2 fluxes

Like in step 1b (see page 41), the remaining flux sample sets are checked for negative CO_2 fluxes again. Flux sample sets with negative CO_2 fluxes as well as the corresponding CH_4 and N_2O results from the same flux sample set are removed from the final data set. The remaining data set is transferred to step 8.

Step 8 - Linearity

The last step of the DQMS repeats the test for linearity of step 1 (see page 41). If the tested flux sample set meets the conditions of $R^2 > 0.9$, it is stored in the final data set. If the R^2-criterion is still less than 0.9, the flux sample set is rejected.

Flux Computation

The final data set is used for the calculation of CH_4, CO_2 and N_2O fluxes. Taking temperature and pressure correction into account, the following formula is applied:

$$F = a\,h\frac{M}{V_0}\frac{p}{p_0}\frac{T_0}{T_\mathrm{C}}\,0.6 \tag{19}$$

with F as the final flux ($\mathrm{mg\,C\,m^{-2}\,h^{-1}}$) or ($\mathrm{mg\,N\,m^{-2}\,h^{-1}}$), a is the concentration gradient ($\mathrm{ppm\,min^{-1}}$), h is the chamber height (cm), M is the molar mass of C ($\mathrm{12\,g\,mol^{-1}}$) or N ($\mathrm{28\,g\,mol^{-1}}$), V_0 is the molar volume ($\mathrm{22.414\,L\,mol^{-1}}$), p is the air pressure (hPa), p_0 is the standard pressure (1013,25 hPa), T_0 is the standard temperature ($273.15\,K$) and T_C is the chamber temperature (K), 0.6 is the residual conversion factor that includes all unit conversions ($\mathrm{min\,h^{-1}}$).

4.3.2. Application

The application of the DQMS led to a clear improvement of the acceptance ratios in each of the three data sets. Figure 9 summarizes the outcome of the approach. The best improvement was obtained for the Ugandan Rice data set (2619 flux rate values) where the acceptance ratio for all three gases increased significantly by 37.2 percentage points (pp) for CH_4, 11.7 pp for CO_2 and 48.3 pp for N_2O. The final acceptance ratio was 82.0% for CH_4, 89.4% for CO_2 and 77.2% for N_2O. The Tanzanian Rice data set (1794 flux rate values) showed the second best optimization with an improvement of 25.0 pp for CH_4, 23.4 pp for CO_2 and 29.8 pp for N_2O. Although the acceptance ratios of this data set show the lowest share with 49.4% for CH_4, 57.5% for CO_2 and 48.7% for N_2O, it yet demonstrates the importance of the DQMS and its effectiveness. Without the decision tree, approximately one quarter of all collected flux data could be saved, however with the DQMS, at least half of the data could be saved with assurance of the reliability of the flux results, which is a great improvement of the results' informative value. The Ugandan Maize data set (2160 flux rate values) featured the lowest improvement with 16.8 pp for CH_4, 0.9 pp for CO_2 and 8.8 pp for N_2O. However, the total final acceptance ratio was comparatively good with 56.1% for CH_4, 91.6% for CO_2 and 49.4% for N_2O.

In summary, the mean improvement for CH_4 is 26.3 pp, for CO_2 12.0 pp and for N_2O 55.6 pp, while the average acceptance rate for CH_4 is 62.5%, for CO_2 79.5% and for N_2O 61.4%. The best improvement of all three data sets was observed for N_2O in the Ugandan Rice data set, while the maximum removal ratio was found in the N_2O data set for Tanzanian Rice. The CO_2 data in the Ugandan Maize data set showed the smallest improvement with 0.9 pp, however the acceptance ratio was already very good considering 90.7% without DQMS vs. 91.6% with DQMS.

4.4. Discussion

4.4.1. Evaluation of the DQMS

The DQMS is one of the first systematic multi-stage quality management systems for GHG data from static chamber measurement. When comparing the acceptance ratios without and with application of the DQMS, a strong improvement in each of the three data sets can be observed. In general, the acceptance ratio for CO_2 is higher compared to the other GHGs which is likely due to higher absolute CO_2 concentration values and thus a better robustness of CO_2 fluxes to measurement inaccuracies and outliers. CH_4 and N_2O flux sets are much more vulnerable for disturbances during sampling (e.g. ebullition in case of CH_4) and measurement uncertainties caused by lower absolute concentration values (mean CH_4 concentration = 2.89 ppm, mean N_2O concentration = 0.35 ppm) which probably explains the higher removal rates. Despite a clear improvement in all three data sets, there were noticeable differences when comparing the results for each test site. It is assumed that differences in the sampling conditions caused the variability of outcome. The data

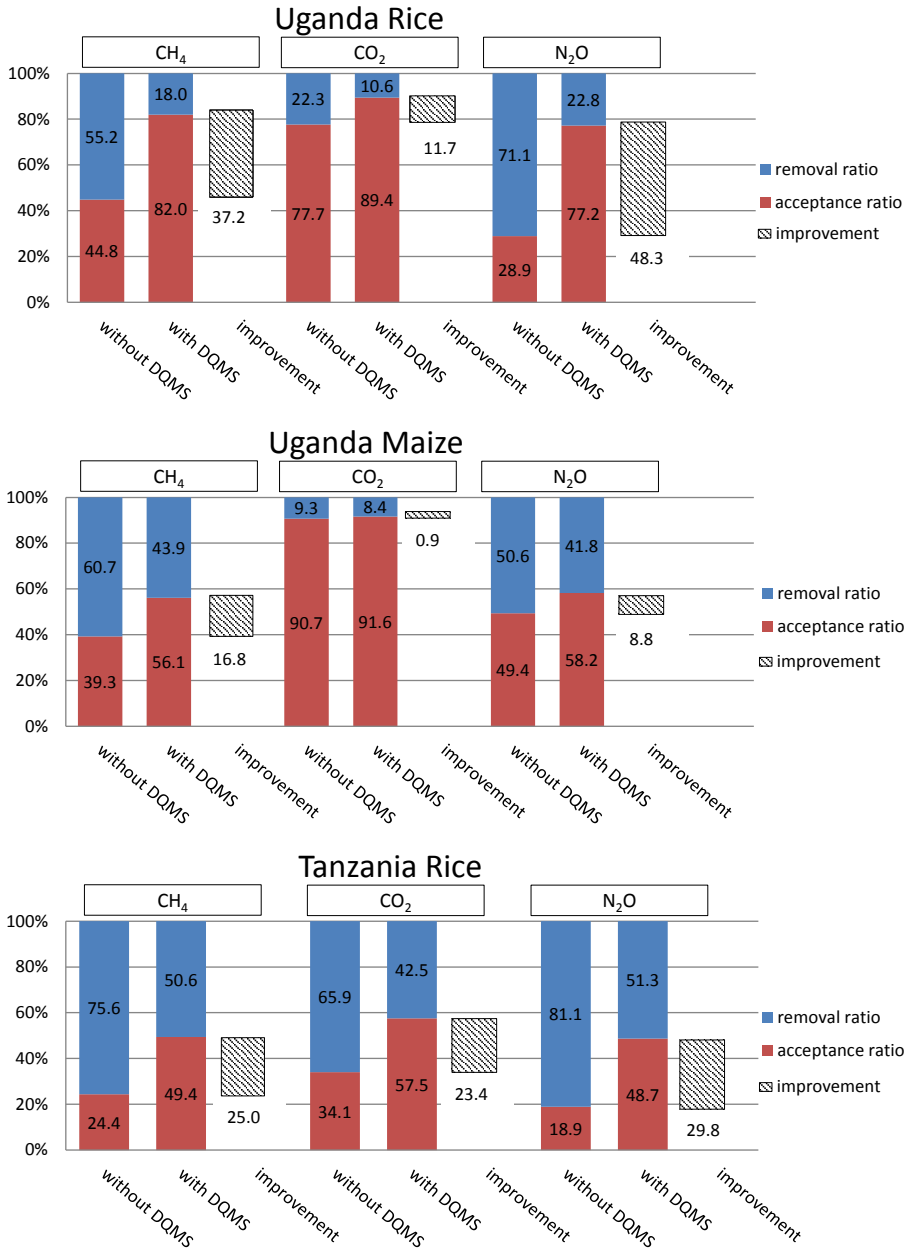

Figure 9: Results of the Data Quality Management System (DQMS).
Uganda Rice: 2619 flux rate values, Uganda Maize: 2160 flux rate values, Tanzania Rice: 1794 flux rate values

sets from the two rice fields included GHG data from the whole plant and also considered aerenchymal respiration, while the maize field data set contained bare soil respiration data. Moreover, the challenging sampling conditions in the Kilombero floodplain with longer flooding periods and larger flooding heights probably lead to generally low acceptance and improvement rates of the Tanzanian Rice data set.

The removal ratios in this study are relatively high compared to most literature results. However, a direct comparison is difficult, because the methods and their distinctness of presentation vary widely. Some studies do not provide detailed results about their proportion of rejected data at all. For example, Mitsch et al. (2012) and Ma et al. (2013) used $R^2 < 0.9$ as rejection criterion, but did not give any information about the removal or acceptance ratios. Kutzbach et al. (2007) applied visual inspection on the CO_2 data sets from the Linnasuo test site to identify erroneous data curves and rejected 6.1% of the data set, but no data rejection was done for the Vaisjeäggi data set. Kutzbach et al. (2007) also mentioned the standard deviation of residuals (σ_{Res}) from the applied exponential regression function as filter criterion, but do not report the removal ratio. Forbrich et al. (2010) used the σ_{Res} criterion and rejected all measurements with $\sigma_{\mathrm{Res}} > 0.2$ ppm from three CH_4 data sets. Moreover, they excluded measurements with unusually high or low starting concentration values. In total, the removal ratios accounted to 8.13% on the flark test sites (5.69% due to $_{\mathrm{Res}} > 0.2$ and 2.44% due to offsets < 1.6 ppm), 0.97% on hummocks and 1.20% on lawns (both due to offsets < 1.6 ppm). One single measurement was rejected based on an unusually high starting concentration value caused by an ebullition event, though it was not specified which test site was affected. Moreover, 20.3% of hummocks measurements were excluded due to fitting problems with the applied exponential model. Saarnio et al. (1997) mention a rejection of 1.78% of their CH_4 sample set because of clearly erroneous samples, though the exact filter criterion was not specified. Furthermore, 0.59% of the data set was rejected due to ebullition during sampling. Pihlatie et al. (2013) applied the normalized root-mean-square-error (NRMSE) for the removal of outliers. This filtering rejected 7%, 10%, 7% and 13% of the measured CH_4 fluxes in four data sets, respectively. Additionally, three exponential fluxes that showed negative flux values were removed manually. In total, 9% of all flux results were removed from their data set. Kroon et al. (2008) examined the concentration behaviour of N_2O in chamber headspaces and classified the concentration rise into several groups. Out of four data sets, 1%, 3%, 9% and 2% had to be removed because the concentration behaviours showed physically unexplainable curves. Moreover, they used $R^2 < 0.75$ as rejection criterion for linear regression and the relative standard deviation of the estimates ($\sigma_{\mathrm{Est}} < 150\%$) as filter criterion for exponential flux calculation, the latter primary was responsible for most of the data rejection. 87%, 75%, 82% and 55% of the fluxes were rejected and not used for further analysis. The removal ratios found in Kroon et al. (2008) were amongst the highest reported in the literature and comparable or even larger than the removal ratios from this study. The reasons for high or low proportions of data rejection depend

on various factors, such as the types of measured GHGs, sampling and field conditions, sample handling, transport, storage as well as type and number of used quality criteria for data filtering. Approaches how to deal with erroneous data are nearly as diverse as the influencing factors themselves. The review of rejection criteria applied in other studies and their outcome illustrates that a standardized quality method would be beneficial to guarantee an equal treatment of GHG data and to improve the informative value as well as the comparability of results. The DQMS developed in this study is a proposal for such a standardized quality control.

4.4.2. Risk of Over- or Underestimation

One focus of the DQMS is on the identification and inclusion of zero fluxes which provide valid results although they show low R^2 values. Other studies, such as Kroon et al. (2008), rejected many small fluxes based on their selected removal criteria and declared the effects to be negligible for cumulative fluxes. This justification is reasonable for this case, however the effects on flux mean values can be considerable. There is the risk of a serious overestimation of flux values if small values are removed systematically. Thus, the consideration of zero fluxes is essential to gain more realistic estimations of natural GHG emissions. In this study, it was expected to find a clear trend towards smaller flux mean values when evaluating the results with applied DQMS compared to the ones without. However, it was observed that the flux mean values with applied DQMS were higher than when R^2 was applied as only filter criterion, except for N_2O in the Tanzanian Rice data set (see Table 2). This indicates a systematic underestimation of values when R^2 is used as single quality criterion and, inter alia, no zero fluxes are considered. The average underestimation ratio over all three data sets was 7.74% for CH_4, 4.55% for CO_2 and 12.01% for N_2O. One possible explanation for the unexpected higher mean values with appliedDQMS is that the share of zero fluxes of the data sets might be relatively small and therefore their effect when included into flux mean calculation is of less consequence. However, Table 3 shows the percentage of zero fluxes of all three data sets and reveals an average share of 22.75% for CH_4, 8.45% for CO_2 and 26.28% for N_2O. The percentages for CH_4 and N_2O are higher than for CO_2 because soil respiration features considerably larger CO_2 emissions than for CH_4 and N_2O. Consequently, the numbers of zero fluxes identified and secured by the DQMS were much higher for CH_4 and N_2O. Because the share of zero fluxes in the three data sets is not significantly small, it does not support the previously made assumption to explain the unexpected trend towards higher values with applied DQMS. Alternatively, it is assumed that the range of GHG fluxes saved by the DQMS includes not only small fluxes (e.g. step 2: zero fluxes), but also large fluxes (e.g. step 6: selective removal) which have a stronger effect on the calculation of flux mean values than many zero fluxes. Consequently, the mean fluxes with applied DQMS are higher compared to the ones without DQMS.

Table 2: Comparison of mean flux values before and after the application of the DQMS

	Uganda Rice		Uganda Maize		Tanzania Rice	
mean flux	without DQMS	with DQMS	without DQMS	with DQMS	without DQMS	with DQMS
CH_4 [mg C m^{-2} h^{-1}]	0.672	0.711	0.110	0.121	0.148	0.162
CO_2 [mg C m^{-2} h^{-1}]	105.79	113.34	119.47	123.47	35.86	37.26
N_2O [mg N m^{-2} h^{-1}]	0.028	0.033	0.044	0.059	0.023	0.022

Table 3: Share of zero fluxes

GHG	Uganda Rice	Uganda Maize	Tanzania Rice
CH_4	14.53%	15.94%	37.79%
CO_2	0.954%	0.009%	24.382%
N_2O	27.85%	11.97%	39.02%

In summary, it was shown that the DQMS prevents from an underestimation of values. The application of a single quality criterion, such as R^2, is not recommended because it bears the risk of introducing a bias, either towards overestimation if small fluxes are rejected systematically, or towards underestimation if large flux values are removed. The DQMS takes a wide span of flux values into account without neglecting a certain range to ensure representative and reliable data.

4.4.3. Linear vs. non-linear flux calculation

Some recent studies question the constant gas exchange between soil and chamber headspace and report an altered concentration gradient caused by the accumulation of target gas within the chamber during measurement (Conen and Smith, 2000; Livingston et al., 2005; Kutzbach et al., 2007; Kroon et al., 2008). Kutzbach et al. (2007) indicated that the application of linear regression can lead to biased GHG emission rates, because of an underestimation of the initial slope of the *c(t)* curvature. Livingston et al. (2005) and Livingston et al. (2006) assumed that linear flux calculation might have caused a systematic underestimation of previous research results. Underestimation of fluxes by linear flux calculation ranges between 33% (Pihlatie et al., 2013), 16% (Conen and Smith, 2000), 15% to 25% (Livingston et al., 2005) and 2 to 31% (Hutchinson et al., 2000). Underestimation is not dependent on the flux level, but on chamber specifications. A large chamber volume is favourable to reduce the effect of underestimation significantly (Pihlatie et al., 2013). According to Pihlatie et al. (2013), non-linear calculation models,

which take the decrease of efflux during chamber operation into account, do not show any significant differences compared to the reference fluxes. Therefore, the use of non-linear flux calculation models (i.e. exponential, quadratic, or NDFE (non-steady state diffusive flux estimator models)), or a combination of linear and non-linear models is recommended by Livingston et al. (2005), Livingston et al. (2006), Kutzbach et al. (2007), Kroon et al. (2008), Pedersen et al. (2010), and Pihlatie et al. (2013). However, there are two essential factors which need to be considered when detecting non-linearity in a flux series with sufficiently statistical significance: the number of measurement points and the precision of the analytical instrument analysing the particular gas concentration (Kutzbach et al., 2007). A high temporal resolution during flux measurements is required to make exponential concentration changes observable (Kroon et al., 2008). However, due to the low temporal resolution of gas measurements using gas chromatography, non-linearity has to be clear to be visible at all (Forbrich et al., 2010). Most non-linear flux calculation models require a minimum of data points for flux calculation. The NDFE model, as described by Livingston et al. (2005) and Livingston et al. (2006), requires at least five observations ($n \geq 5$). Hutchinson et al. (2000) and Forbrich et al. (2010) promote at least six data points ($n \geq 6$). The exponential or quadratic flux calculation approach by Kutzbach et al. (2007) is based on a minimum of seven data points ($n \geq 7$). The only study which considers chamber measurements with any number and spacing of observations is the one by Pedersen et al. (2010). For practical and cost-saving reasons, a reduction of GHG samples per chamber deployment time is often required. Consequently, many studies do not meet the requirements for non-linear flux calculation models, and thus linear regression is the only applicable method for a sample size of smaller than five (Kroon et al., 2008). The DQMS was developed with the intention to consider the necessity for cost and sample reduction and to take studies with a sample size of four (n = 4) into account.

Another argument against the application of non-linear flux calculation is that many non-linear models require undisturbed physical field conditions during sampling and assume the effect of potential error to be negligible (Kutzbach et al., 2007). Though, the authors also reported a substantial percentage of tested data whose curvatures were inexplicable with non-linear models, possibly because some theoretical assumptions could not be verified in reality, such as constant soil and headspace air temperature, steady air pressure, no headspace turbulence and constant photosynthetically active radiation (Kutzbach et al., 2007). In this study, it was expected to struggle with challenging field conditions. GHG measurements in tropical areas, including sub-Saharan Africa, underlie difficult conditions including changing physical conditions during sampling. Thus, the approach considers the issue of uncontrollable sampling circumstances and deals with the challenge to identify and reject eventual outliers as a solution.

Many studies support the application of linear regression and justify linear flux calculation. For example, in case chamber closure time is short and uncertainty caused by spatial variation is larger than the effect of inappropriately used linear regression (Kroon et al.,

2008). Chadwick et al. (2014) confirmed linearity for a majority of N_2O flux calculations based on static chamber measurements. Only 0-14% of all tested measurements featured a non-linear headspace accumulation during a closure period of 40-60 minutes. Conen and Smith (2000) described a nearly perfect linear increase with R^2>0.99 for N_2O fluxes measured in closed chambers with a sampling interval of 20 min and a total measuring period of 5 hours. Forbrich et al. (2010) reported a linear increase for the majority of CH_4 measurements from closed chambers in peatlands with a sampling interval of 4 min during a total closure time of 24 min. Only 14% of the tested data fitted better to an exponential flux model. The accuracy of linear and non-linear flux calculation models converge under waterlogged soil conditions because a strong gas concentration gradient is expected in wet soils, while the time in which the concentration gradient vanishes is relatively large compared to the chamber deployment time (Livingston et al., 2006; Forbrich et al., 2010). Taking this into account, a nearly linear increase in gas concentrations during chamber deployment in the three wetland test sites of this study was assumed. Forbrich et al. (2010) reported an expected high concentration gradient and correspondingly linear increase in sedge-covered microsites caused by aerenchymatous plants. As the field experiments included rice and (semi-) natural vegetation featuring aerenchyma, the assumption of this study is considered to be strengthened. In conclusion, linear concentration change for static chamber measurements can be expected especially at wet sites (Forbrich et al., 2010). The only non-linear concentration change with stepwise increase is CH_4 emission in form of ebullition (Leventhal and Guntenspergen, 2004; Forbrich et al., 2010). In this case, both linear and non-linear regression are no appropriate method for the calculation of the gas flux. CH_4 measurement which are clearly affected by ebullition are usually rejected with consideration of R^2 (Saarnio et al., 1997; Forbrich et al., 2010). In this case, outliers caused by sudden ebullition were removed by the DQMS based on the R^2 criterion or because the requirement of monotonic increase was not fulfilled.

4.5. Conclusion

Not many studies focused on the quality control of GHG data obtained by gas chromatography. A good fit of the linear flux calculation model on the collected data was often reported in the literature (e.g. Gao and Yates (1998), Yamulki and Jarvis (1999), and Conen and Smith (2000)). However, recent GHG research also pays attention to areas with extreme or unstable physical conditions and poor infrastructure, such as sub-Saharan Africa, which involves a wide range of additional challenges, including long transport and storage periods for the gas samples. The difficult sampling conditions can result in a large number of disturbed data points and data sets of questionable quality. The application of the most established measure for quality control in linear regression i.e., R^2, can lead to a considerable loss of flux results. Hence, there was the need for the development of a quality control system that automatically reviews GHG data from static chambers for their suitability for flux calculation, identifies and rejects single outliers, and

saves reliable flux results in the final data set. This DQMS was based on eight objective and universally valid criteria. With the removal of single outliers, it was possible to save a considerable number of flux results which would have been lost otherwise. With the application of the DQMS, the acceptance increased considerably by a maximum of 37.2 pp for CH_4, 23.4 pp for CO_2 and 48.3 pp for N_2O, and a minimum of 16.8 pp for CH_4, 0.9 pp for CO_2 and 8.8 pp for N_2O in the tested data sets. The reasons for the variability of the result was assumed to be caused by the different sampling conditions at the three sites. While the data of two rice fields included gas emissions of the whole plant due to aerenchymal gas transport, bare soil respiration was measured in the maize field. The generally low acceptance and improvement rates of the Tanzanian Rice data set presumably occurred because of more challenging sampling conditions in the Kilombero floodplain caused by longer flooding periods and larger flooding heights. The application of the DQMS also showed, that it prevents an underestimation of values. The usage of R^2 as only filter criterion would lead to an underestimation bias of 7.74% for CH_4, 4.55% for CO_2 and 12.01% for N_2O. However, there is also the theoretical risk of overestimation if small fluxes, such as zero fluxes, are removed systematically. Therefore, the DQMS takes a wide span of flux values, including zero fluxes, into account, to prevent the risk of a systematic bias in the final data set.

For further development of the DQMS, it is recommended to introduce additional options. The risk of underestimation when linear flux calculation is applied, as indicated by Conen and Smith (2000), Hutchinson et al. (2000), Livingston et al. (2005), Kutzbach et al. (2007), and Pihlatie et al. (2013), should be taken into consideration. Moreover, many studies recommend to use non-linear flux calculation models, or a combination of linear and non-linear models (Livingston et al., 2005; Livingston et al., 2006; Kutzbach et al., 2007; Kroon et al., 2008; Pedersen et al., 2010; Pihlatie et al., 2013). An expansion of the DQMS towards a combined (linear/non-linear) flux calculation approach is probably feasible if the number of observations per flux measurement is increased ($n > 4$). Another aspect for improvement is that the DQMS does not consider outliers which fulfil the criterion of monotonic increase (step 3). It does not reject single outliers from a flux set whose first data point is too low or last data point is too high. Instead, the whole flux set is forwarded to step 7 and 8 and most likely rejected from the final data set, because there is no way of rejecting this single erroneous value from the flux set. Thus, there is still potential for another quality step that deals with this issue. In general, it is recommended for further studies and publications to pay more attention on the applied criteria for data filtering, this will improve the informative value of the data and benefit the comparability of results.

5. Effects of Land-Use Change and Agricultural Management on GHG Emissions from Rice Fields in Uganda

Abstract

The pressure on wetlands for land-use change is constantly increasing due to rising global food insecurity and a high demand for agricultural areas. Wetlands provide good conditions for food production, but they also play an important role in global climate regulation which is seriously affected by land-use change. Studies about GHG emission from wetlands in SSA are extremely rare, although emissions from this region are important for the global GHG budget. To gain detailed information about the effects of land-use change and different agricultural treatments on GHG emissions from typical wetlands in Africa, a field experiment was established in an inland valley wetland at Namulonge Research Station, Uganda, in August 2014. Five treatments were observed: a (semi-) natural vegetation reference (T0), the local farmer´s practice without mineral N fertilization or bunding (T2), an improved practice with bunding but without N fertilization (T3), intensive crop management with 120 kg urea-N/ha + 60 kg PK (T5) and an alternative management practice with green manure (T6). Measurements of GHG emissions were conducted over a total period of two consecutive years and revealed cumulative flux values ranging from -0.03 to 263.02 kg CH_4-C $ha^{-1}yr^{-1}$, 5946 to 16001 kg CO_2-C $ha^{-1}yr^{-1}$ and 0.16 to 9.54 kg N_2O-N $ha^{-1}yr^{-1}$. The observed GWP varied from 208 to 10095 kg CO_2eq ha^{-1} yr^{-1}, while the yield based global warming potential index (GWPI) ranged between 0.08 and 2.19 kg CO_2eq kg^{-1}grain yield yr^{-1}. The most considerable treatment differences were observed between the (semi-) natural vegetation reference and the agricultural treatments. In comparison, the differences between the individual cropping methods were less distinct. In conclusion, land-use change has a higher impact on changes in GHG emissions than diverse agricultural treatments. The GWP and GWPI results indicated that intensive cropping treatments with high yield potentials provide the best possible trade-off between optimized food production and preferably low GHG emission. However, the potential of highly productive systems for GHG mitigation requires land to be spared from agricultural production. The preservation of the largest possible natural wetland area provides the greatest opportunity for natural climate regulation.

5.1. Introduction

5.1.1. Wetlands and Agriculture

Rising global food insecurity due to world population growth and overexploitation of upland soils has led to great pressure on natural wetlands for a conversion into farmlands (Wood and van Halsema, 2008; Rebelo et al., 2009b; Barrett, 2010). Wetlands show a high potential for food production because they feature fertile soils and a good water availability

(Reddy and DeLaune, 2008). Especially in SSA wetlands support the livelihoods of many people through food supply (Bikangaga et al., 2007; Rebelo et al., 2009b). In East Africa the stress for expanding agriculture in wetlands areas is high due to pressure from government policies and population (Dixon and Wood, 2003). However, wetlands also provide other valuable ecosystem services besides from food supply, such as climate regulation and carbon cycling, which are seriously affected by land-use change (Millennium Ecosystem Assessment, 2005; Reddy and DeLaune, 2008). The trade-off between food production and preservation of natural ecosystem services is one of the main challenges in wetland management (Mc Cartney et al., 2011). For the achievement of this aim a deep understanding of matter fluxes in wetlands and their correlations with land-use change and agricultural activities is necessary.

5.1.2. Carbon Cycle in Wetlands

Wetlands are global carbon sinks and have a considerable importance for climate regulation. Even though their cover area on the earth's terrestrial surface is small with 2 to 7%, their internal biogeochemical processes have significant effects on global scale (Reddy and DeLaune, 2008; Kayranli et al., 2010; Schlesinger and Bernhardt, 2013). For example, the global temperature was reduced by 1.5 to 2° C within the last 10,000 years due to the amount of carbon accumulated in peatland soils (Holden, 2005). The great carbon sequestration rates in wetland soils are caused by a high primary productivity of wetland vegetation in combination with low decomposition rates of organic matter, which results in a net accumulation of carbon (Gorham, 1991; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013; Mitsch and Gosselink, 2007). The net primary production of wetlands amounts 7 to 15% of the terrestrial productivity worldwide (Schlesinger and Bernhardt, 2013) and reaches the same order of magnitude as the tropical rain forest (Reddy and DeLaune, 2008). The low decomposition rates are driven by a lack of oxygen in wetland soils which feature long-term or permanent anoxic conditions (Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013; Mitsch and Gosselink, 2015). Approximately 14.5% to more than 50% of the world's soil carbon is stored in wetlands (Gorham, 1991; Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015; Schlesinger and Bernhardt, 2013). The estimated average net carbon retention is $118\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ which sums up to a total net carbon sink capacity of $830\,\mathrm{Tg\,C\,yr^{-1}}$. The highest carbon retention rates were observed in the tropics and subtropics with 150 to $250\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$ (Mitsch et al., 2012).

On the other hand, wetlands are considered to be carbon sources because they feature favourable environmental conditions for the production of GHGs (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). For instance, they account for 20 to 33 % of all CH_4 emissions worldwide (Schlesinger and Bernhardt, 2013). The estimations range from $109\,\mathrm{Tg\,CH_4}$-C ($145\,\mathrm{Tg\,CH_4}$) (Whalen, 2005) to $448\,\mathrm{Tg\,CH_4}$-C per year (Mitsch et al., 2012). Thereof, 52 to 78% originate from tropical and subtropical wetlands (Bloom et al., 2010;

Mitsch et al., 2012).

In summary, wetlands have the ability to act as sink as well as source for carbon in the global carbon cycle. Recent studies indicate that wetland age and environmental boundary conditions (e.g. climate and location) have a key role for their evaluation in the carbon cycle (Kayranli et al., 2010; Mitsch et al., 2012). The time period during which CH_4 emission and carbon sequestration balance themselves is estimated to be 300 years. After this period wetlands turn from net carbon sources into net sinks (Mitsch et al., 2012).

5.1.3. Effects of Land-Use Change on GHG Emissions

Land-use change effects a disturbance in the natural balance of the carbon and nitrogen cycles in wetlands. The equilibrium among inflow and outflow is disturbed until a new equilibrium is attained. Thus, soil can either act as carbon sink or source depending on the ratio between inflow and outflow until the ecosystem stabilizes again (Guo and Gifford, 2002). The conversion of wetlands to agricultural areas implies artificially controlled drainage (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). This can cause an increase in carbon turnover rate because large organic matter stocks are suddenly exposed to oxygen and aerobic microorganism. Increased CO_2 emissions can be a result (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). The productivity of vegetation can increase because nutrients, that were sequestered in undecomposed soil organic matter, are exposed to a more rapid mineralization by aerobic microbes and thus show a better availability for plant uptake (Schlesinger and Bernhardt, 2013). Moreover, nitrogen fertilizer usage in agriculture can lead to additional external nitrogen input into wetlands and result in increased N_2O emissions (Kroeze et al., 1999; Reddy and DeLaune, 2008). However, a positive effect can be that CH_4 emissions decrease because methanogens are restricted to anoxic soil conditions and do not find optimal conditions in drained soil (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015).

5.1.4. GHG Research in Sub Saharan Africa

GHG research in tropical developing countries has large knowledge gaps. Several authors address the lack of available field data, especially in SSA (Ciais et al., 2011; Oertel et al., 2016; Kim et al., 2016; Boateng et al., 2017). Although GHG emissions from this region play an important role in the global GHG budget, uncertainties are large due to a lack of long-term field studies (Ciais et al., 2011; Valentini et al., 2014). Previous research indicates that CO_2 has the largest contribution to GHG emissions from natural ecosystems in SSA with 0.9 to 15.5 $\mathrm{Mg\,CO_2\text{-}C\,ha^{-1}yr^{-1}}$ (Kim et al., 2016), while CH_4 emissions ranged between -3.6 and 2.6 $\mathrm{kg\,CH_4\text{-}C\,ha^{-1}yr^{-1}}$ (-0.16 to 0.12 $\mathrm{Mg\,CO_2\,eq\,ha^{-1}yr^{-1}}$) (Kim et al., 2016). Forested and unforested wetlands emit up to 26.85% and 22.28% of the total CH_4 emissions from continental Africa, whereas paddy fields only account for

6.28% (Valentini et al., 2014). N_2O emissions were estimated to vary between -0.06 and 8.7 kg N_2O-N $ha^{-1}yr^{-1}$ (-0.03 to 4.1 Mg CO_2 eq $ha^{-1}yr^{-1}$) (Kim et al., 2016). It is estimated that 19% of the global N_2O emissions (Thompson et al., 2014) and 6% of the global anthropogenic N_2O emissions emerge from Africa (Hickman et al., 2011). Approximately 42% of the continent's N_2O emissions are due to agriculture (Hickman et al., 2011). Increased fertilizer use will most probably have an increasing effect on N_2O emissions. Future trends predict a doubling of agricultural N_2O emissions from Africa between 2000 and 2050 (Hickman et al., 2011). In summary, natural ecosystems in SSA contribute 76.3% of the total CO_2 equivalent emissions, while agricultural lands contribute only 23.7%. It is estimated that GHG emissions will increase in the future due to continuing land-use change (Kim et al., 2016).

5.1.5. Wetlands in Sub-Saharan Africa

The estimated wetland area in SSA covers 125 Mha to 131 Mha (10 to 14% of the total land surface), depending on whether or not lakes are included in the estimations (Finlayson et al., 1999; Lehner and Döll, 2004; Rebelo et al., 2009a). Kim et al. (2016) report an area of 43.8 Mha for wetlands, floodplains, lagoons and reservoirs and 10.5 Mha for rice fields. In Uganda, wetlands cover approximately 30,000 km^2 (13%) of the land's surface (Bikangaga et al., 2007). Because of their significant economic, ecological and social value that contributes to the rural livelihood and national economy, the Ugandan government recognized wetlands as areas worthy for conservation in 1989 (Bikangaga et al., 2007). However, the protection efforts and particularly the demarcation of areas for conservation provoked conflicts between policy makers and land owners. Although, conservation activities are generally considered to be positive, some doubts remain whether the efforts support or hinder the local population in accessing basic resources (Bikangaga et al., 2007).

5.2. Methods

5.2.1. Central Field Trial

The study was conducted in an inland valley wetland at Namulonge Research Station in central Uganda ($00°31'30"N$, $32°36'54"E$), approximately 30 km north of Kampala. The test site was located in a mid-hill area with an average altitude of 1100 m above sea level, an average airpressure of 875.7 hPa and a sub-humid climate with an average precipitation of 1350 mm [2] (Leemhuis et al., 2016). The climate was characterized by two dry and two rainy seasons per year. The first (longer) rainy season occurred between March and May, while the second (shorter) rainy season occurred between August and November. The average annual temperatures ranged from 22.14 to 24.0°C (Nsubuga et al., 2011). The soils of the test site consisted of Gleysols (Gabiri et al., 2018a). An agronomic field trial was established to investigate agricultural and environmental properties of different rice

cropping treatments in different hydrological zones within the wetland and to measure their effects on GHG emissions. The field trial consisted of three fields situated in three hydrological zones which differed regarding their lengths and intensity of annual flooding. Each field was named after their topographic elevation: fringe, mid-section and center (see Figure 10). The area was a (semi-) natural wetland which was converted to a rice cropping field for this study. However, a small area with (semi-) natural vegetation was left in each hydrological zone. The vegetation was described as (semi-) natural, because the fields were used as farmland and abandoned some years ago. The experimental plots of the field trial (5 m x 6 m) were arranged in a randomized complete block design. The GHG measurements were conducted in four agricultural treatments and one (semi-) natural vegetation area, with three replications per treatment and zone. Table 4 summarizes the applied agricultural treatments. T0 represented the natural reference with (semi-)natural vegetation. T2 reflected the local farmer´s practice without N fertilization. T3 introduced bunding for an improvement of water retention and prevention of nutrient leaching. Optimal crop management in form of an intensive crop management with application of mineral fertilizer was represented by T5. In addition, T6 was implemented to demonstrate a management alternative with green manure (lablab) as nitrogen input (60 kg/ha N) (Ziegler et al., 2015). The observed treatments only considered one rice cropping season per year, despite the two rainy seasons. The test site was established in August 2014. The beginning of the gas and soil sampling started simultaneously with the rice cropping season in September 2014 and was conducted for two consecutive years until August 2016.

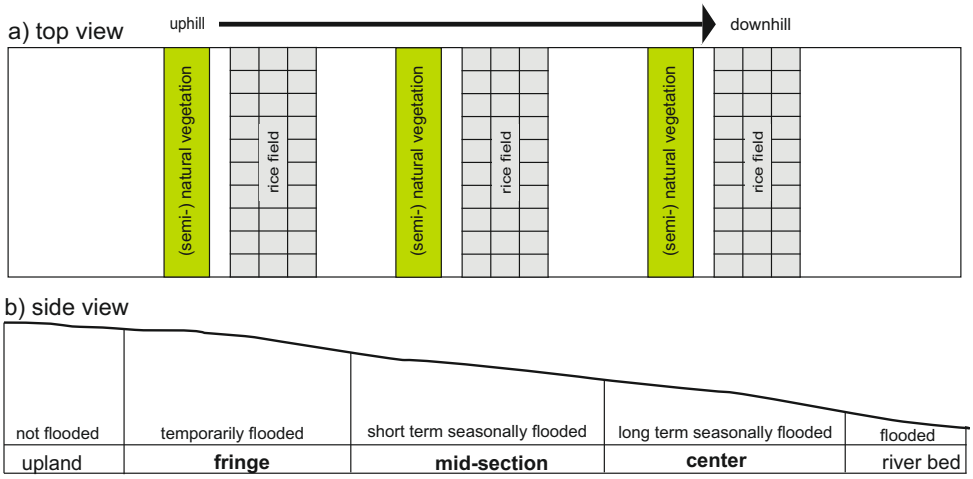

Figure 10: Central field trial

Table 4: Agricultural treatments, Namulonge

Treatment	Description
T 0	(semi-) natural vegetation
T 2	rice, no mineral N, no field bunds for rice
T 3	rice, no mineral N, plots bunded
T 5	rice, 120 kg urea-N/ha + 60 kg P/ha + 60 kg K/ha + supplementary irrigation, plots bunded
T 6	rice, no mineral N, pre-crop leguminous green manure (legume), plots bunded, 60 kg/ha N

5.2.2. GHG Sampling and Laboratory Analysis

GHG sampling was conducted every two weeks continuously for two years with the aim to monitor periodical changes between rainy and dry season, more specifically between cropping and fallow season, in which considerable fluctuations in GHG emissions were expected. The three main greenhouse gases CH_4, CO_2 and N_2O were sampled via static chamber measurements, which is a commonly used method for the determination of soil degassing into the atmosphere. The advantages are its simple practicability and low cost intensity. Thus, it can be implemented under various environmental conditions including remote wetland areas (Yu et al., 2013). In this study, fifteen chambers were operated simultaneously with a deployment duration of 60 min and a sampling interval of 20 min ($t_1 = 0\,min, t_2 = 20\,min, t_3 = 40\,min, t_4 = 60\,min$). The gas measurements were conducted in four agricultural treatments and in the reference plots with (semi-)natural vegetation, with three absolute replications ($n = 3$) in each hydrological zone. After gas sample collection, the samples were transported via air cargo to Germany and analysed with a GC (European Greenhouse GC 8610C, SRI Instruments Europe GmbH, Bad Honnef, Germany) that was equipped with FID for CO_2 and CH_4 measurements and ECD for N_2O measurements. Thereby, each air sample was transferred by an autosampler with a syringe from the sample vials to the injection port of the GC. To avoid any intrusion of ambient air into the syringe during sample transfer, a minimum overpressure of 0.16 bar in the sample vials was required. Each vial was checked for the remaining overpressure after each measurement.

5.2.3. Chamber Design

The gas flux chambers consisted of robust polypropylene boxes with an area of 0.6 m x 0.4 m, a height of 0.4 m and a total volume of 0.096 m^3. During measurements the chambers were placed on a polypropylene frame with the same dimensions as the chamber's base area. Each frame was inserted 7 cm to 10 cm into the soil and remained spatially invariable

in the field for the whole cropping season. The whole sampling design can be designated as "base-and-chamber" type, which is the most favourable type according to Rochette and Eriksen-Hamel (2008). Influences on gas emissions due to soil disturbances, e.g. associated with the insertion of the frames, are lower because of the one-time insertion. Each chamber was equipped with a rubber seal that enclosed the chamber base and was pressed on top of the polypropylene frame with 32 mm foldback clips to ensure gas tightness during closure time. Two connection ports were mounted on top of each chamber, one for the vent and one for gas sampling. The venting tube had the purpose to balance and compensate changes in atmospheric pressure within the chamber that arise during operation (Rochette and Eriksen-Hamel, 2008). It consisted of a 1/4 inch outer diameter BEV-A-LINE tube with a length of 1 m outside and a smaller 15 cm tube inside the chamber which ensured the pressure balance to develop from the headspace center. The gas sampling port consisted of a 1/4 inch brass tube fitting (Swagelok, Neuss, Germany), with an inserted silicone rubber septum (Agilent, Waldbronn, Germany, CrossLab inlet septum, gray, 9.5 mm, type 8010-0252). The GHG samples were collected with a 100 ml luer-lock polyethylene syringe (Ecomedis, Münster, Germany) and stored in a 20 mL glass vial (SRI, type 21-300044) sealed with a buthyl rubber septa (Macherey-Nagel GmbH & Co. KG, Düren, Germany, type 70237) and a fitting 20 mm diameter aluminium crimp cap. The outer colour of the chamber walls was white to enable a good reflection of sun radiation and to reduce warming, since temperature changes can affect the determination of gas concentration due to varying pressure effects and altered GHG production in the soil (Rochette and Eriksen-Hamel, 2008). In addition, one chamber was equipped with a temperature probe (KHXL-IM30U-RSC-300, OMEGA, Deckenpfronn, Germany) and a temperature datalogger (HH306A Type K, OMEGA, Deckenpfronn, Germany) to monitor temperature variations within the headspace during chamber closure. The data were applied to temperature correction during flux calculation. A special requirement for CH_4 measurements in a rice field during the cropping season is to include the whole plant in the chamber, due to the presence of aerenchyma inside the rice plant tissue. Aerenchyma are components of many wetland plant species that tolerate waterlogged soil conditions, including papyrus and rice. They serve for gas exchange between plant and atmosphere, and act as preferential escape paths for gas emissions, especially CH_4 (Wagatsuma et al., 1990; Jones, 2001). Consequently, during GHG measurement the total plant had to be included in the chamber to capture CH_4 emissions from the rice plant itself. For a later stage of rice growth when plant height exceeded 40 cm, the height of the normally used chambers, an extension frame of 40 cm height was placed between the frame and the normal chamber to extend the headspace to a total height of 80 cm. A schematic example of the chamber design used in this study is shown in Figure A.1 (page 127). The real chamber construction used in the field in shown in Figure A.2 (page 127).

5.2.4. Gas-Pooling and Vial Flushing

Two chamber frames were installed in each treatment replicate to double the sampled surface area. To keep the number of required vials low and minimize the transport and analysis costs, while maintaining the reliability of estimated GHG fluxes, the gas-pooling method described by Arias-Navarro et al. (2013) was applied for our sampling in each treatment replicate. Therefore, two 50 mL gas samples from both chambers within one replicate were pooled initially in a polyethylene syringe, resulting in a total sampling volume of 100 mL. Although information about the spatial variability of the replicate were lost due to pooling, the resulting single flux was considered to be spatially more representative as the sampled surface area was maximized. For this reason, gas sample pooling is considered to be a good opportunity to cope with spatial heterogenities (Arias-Navarro et al., 2013).

The usual procedure to prepare vials for gas sample collection is evacuation of the sample containers. This procedure effectively prevents a contamination of the sampling air with ambient air (Kelliher et al., 2012). Alternatively, for example if no vacuum pump for evacuation is available, the vials can be flushed with a sample air volume that is four times larger than the vial volume, as described by Klein et al. (2003). Accordingly, for this study the 20 mL glass vials were flushed with an air volume of 85 mL, which is equivalent to 4.25 times the vial volume. Afterwards, a quantity of 15 mL was additionally injected into the vial to create an overpressure of 0.75 bar within the vial, which was required for storage and GC analysis (see section 5.2.2).

5.2.5. Data Quality Management and Flux Calculation

Quality control of all GHG values was conducted with a newly developed DQMS. With the DQMS, all GHG data were checked regarding their applicability for linear flux calculation and removed outliers automatically, if necessary. The system has been described in detail in chapter 4. Approved flux results were stored in the final data set. CH_4, CO_2 and N_2O fluxes were calculated afterwards from the alteration rate of the gas concentration within the chamber headspace over time, corrected for air temperature and pressure:

$$F = a\,h\frac{M}{V_0}\frac{p}{p_0}\frac{T_0}{T_C}\,0.6 \tag{20}$$

F is the final flux ($mg\,C\,m^{-2}\,h^{-1}$) or ($mg\,N\,m^{-2}\,h^{-1}$, respectively), a is the concentration change during chamber closure time ($ppm\,min^{-1}$), h is the chamber height (cm), M is the molar mass of C ($12\,g\,mol^{-1}$) or N ($28\,g\,mol^{-1}$), V_0 is the molar volume ($22.414\,L\,mol^{-1}$), p is the air pressure (hPa), p_0 is the standard pressure (1013,25 hPa), T_0 is the standard temperature ($273.15\,K$) and T_C is the chamber temperature (K), *0.6* is the residual conversion factor which includes all unit conversions ($min\,h^{-1}$).

5.2.6. Soil Sampling and Laboratory Analysis

Soil samples for the analysis of nitrate (NO_3^-) and ammonium (NH_4^+) were collected concurrently with the GHG samples to analyse mineral nitrogen dynamics. Sampling was performed every two weeks and continued throughout two years to receive long-term monitoring data. The aim was to capture particular periods with considerable changes in mineral nitrogen activities which are expected to occur during the transitions period between the rainy and dry seasons. The soil samples consisted of five mixed samples from the same replicate plot and were collected with a small soil auger (Ackerlandbohrer, ø13 mm, working length 25 cm, total length 58 cm by Eijkelkamp Soil & Water, EM Giesbeek, Netherlands) from 0 to 10 cm depth. Since nitrate and ammonium are very sensitive to microbial degradation and temperature influences, the soil samples were cooled immediately in the field in a cooling box and stored in a freezer at −16°C for several weeks afterwards, until they were thawed at +7°C for analysis. After thawing and drying for 24 hours at 105°C, the dry weights of the soil samples were determined. The preparation of soil extracts was based on the presidedress soil nitrate test (PSNT) (Blume et al., 2015). 15 g dry soil material was extracted with 90 mL of a 0.01 M $CaCl_2$ solution (Merck, Darmstadt, Germany), shaken for 60 min with 180 revolutions per minute and filtered to get the soil extracts required for photometer measurements. One drop of concentrated sulfuric acid (H_2SO_4) was added to each extract to prevent microorganism from degrading the mineral nitrogen content. In addition to the soil extracts, three $CaCl_2$ blanks were prepared on each analysis day to detect and correct eventual contamination of the extraction solution. The blanks were prepared and measured with the same procedure like the soil extracts. After preparation, all soil extracts and $CaCl_2$ blanks were analysed photometrically. The soil samples from September 2014 until March 2015 were analysed in an external laboratory of Makerere University, Uganda, with a spectrophotometer according to local standards. The soil samples from April 2015 to August 2018 were analysed at Namulonge Research Station, Uganda, with a mobile photometer (pHotoFlex® STD, WTW, Weilheim, Germany) for NH_4^+ (using WTW reagent NH4-3TC(HR), analytical range 0.4-50.0 mg/L NH_4-N, program no. 313) and for NO_3^- (using WTW reagent NO3-1 TC, analytical range 0.2-30.0 mg/L NO_3-N, program no. 314 and WTW reagent 14542, analytical range 0.5-14.5 mg/L NO_3-N, program no. 17). The photometer results were given in mg/L NH_4-N and mg/L NO_3-N, respectively. In addition, nitrate was also measured with a Nitracheck Reflectometer (18.40 by Eijkelkamp, Giesbeek, Netherlands, analytical range 5 - 500 ppm (mg/L) NO_3^-). For the photometric NH_4^+ measurements it was required to adjust the pH value of the soil extracts and the $CaCl_2$ blanks to a desired value of pH ≈7 with 1 M NaOH solution (VWR, Langenfeld, Germany). The dilution effect caused by the addition of NaOH solution was corrected with a corresponding dilution factor. The mean value of the three daily $CaCl_2$ blank concentrations was subtracted from the NH_4^+ and NO_3^- results to correct eventually increased results due to a contamination of the extraction solution. The final results were

given in mg NH_4-N/kg and mg NO_3-N/kg.

5.2.7. Meteorological Data and Soil Moisture

Rainfall and soil moisture data were acquired in cooperation with Gabiri et al. (2018a). Daily rainfall was measured using a tipping bucket rain gauge with a 0.2 mm resolution (Stevens Water Monitoring Systems Inc., 2007) located at the fringe position. Daily temperature data were obtained from an automatic weather station of the National Crops Resources Research Institute (NaCRRI). The soil moisture content at the fringe position was measured with automatic soil Hydra probe SDI-12 sensors (Stevens Water Monitoring Systems Inc., Portland, Oregon, USA) connected to a data logger (Stevens Water Monitoring Systems Inc., 2007). The sensors recorded soil moisture at 10 and 30 cm soil depth every 15 minutes. The mid-section and center zone were equipped with 5TE soil moisture sensors connected to EM50 data loggers to monitor soil moisture (Decagon Devices Inc., 2016a; Decagon Devices Inc., 2016b). At these positions, soil moisture was measured at 10 and 30 cm hourly. All soil moisture data from each hydrological position were aggregated to a daily time resolution (Gabiri et al., 2018a).

5.2.8. Calculation of GWPI and GHGI

The calculation of GWP (kg CO_2 eq ha^{-1} yr^{-1}) was done according to Li et al. (2004), Nishimura et al. (2011) and Shi et al. (2013) with application of the cumulative CH_4 and N_2O flux results (kg C ha^{-1} and kg N ha^{-1}).

$$GWP = \frac{CH_4}{12} \cdot 16 \cdot 25 + \frac{N_2O}{28} \cdot 44 \cdot 298 \tag{21}$$

The GWPI (kg CO_2 eq kg^{-1} grain yield yr^{-1}) was calculated based on Shang et al. (2010) and Shi et al. (2013) by dividing GWP (kg CO_2 eq ha^{-1} yr^{-1}) by grain yield (kg C ha^{-1}).

$$GWPI = \frac{GWP}{grain\,yield} \tag{22}$$

5.2.9. Data Processing and Statistical Analysis

All statistical analyses were conducted using R version 3.3.1 (2016-06-21). GHG results were expressed as weighted seasonal mean values. Due to missing flux results (e.g. outliers removed by the DQMS), not all mean values were based on three replications (*n=3*). Values based on two or less replications received a smaller weight to lower their impact on the total seasonal mean value to prevent a bias. Mean values calculated out of three replications got a higher weight, because their reliability was assumed to be superior. For the calculation of cumulative fluxes, the weighted mean values of each cropping and fallow

season were multiplied with the respective number of days for each period and summed up to receive annual cumulative flux results. All results were expressed plus or minus one standard deviation. The total data set was tested for homogeneity of variance with Levene's test and for normal distribution. Because the pre-tests indicated the data set to be neither homogeneous nor normal distributed, the preconditions for an analysis of variance (ANOVA) were not given. Instead, the Kruskal-Wallis test with $p < 0.05$ was applied as non-parametric alternative to find significant differences among several groups, while the independent Mann-Whitney U Test ($p < 0.05$) was used for the comparison of two groups. Moreover, Spearman's rank correlation test was chosen to test relationships between mineral nitrogen dynamic and N_2O fluxes.

5.3. Results

5.3.1. Methane

The CH_4 flux results of both sampling years showed increased emissions during the two rice-cropping periods from September 2014 to January 2015 and from September 2015 to January 2016, while the emissions during the fallow periods were much lower and converged to zero (Figure 11). The weighted seasonal mean values confirmed this observation for all treatments in all hydrological zones (Figure A.3 and A.4). The mean values ranged from 0.006 to 8.31 mg CH_4-C $m^{-2}h^{-1}$ in the first cropping period and from 0.002 to 4.29 mg CH_4-C $m^{-2}h^{-1}$ in the second cropping period. In comparison, the results of the fallow periods ranged from -0.01 to 0.34 mg CH_4-C $m^{-2}h^{-1}$ in the first year and from -0.001 to 0.84 mg CH_4-C $m^{-2}h^{-1}$ in the second year. The emissions of the first year were considerably higher than in the second year, which can be seen in the weighted seasonal mean values as well as in the cumulative CH_4 fluxes (Figure 14 and15). The cumulative CH_4 fluxes showed a range from -0.03 to 263.02 kg CH_4-C $ha^{-1}yr^{-1}$ in the first year and 0.02 to 140.49 kg CH_4-C $ha^{-1}yr^{-1}$ in the second year. The Kruskal-Wallis test ($p < 0.05$) revealed significant differences regarding the cumulative CH_4 fluxes from different treatments. The emissions from T5 in the mid-section position were significantly increased compared to T0 in the center position during the first year. The second year showed a significant difference between the low emissions from T0 in the center zone and the high emissions from T5 in the fringe zone. The comparison of treatment specific hydrological zone effects showed a significant difference between the increased emissions of T5 in the fringe field and the decreased emissions of T5 in the center field during the second year (Figure 14). A summary of treatment differences which are independent from the hydrological position, as well as zone effects that occur regardless from the treatments are shown in Figure 15. A significant difference between the low emissions from the (semi-natural) vegetation treatment T0 and the high emissions from the two fertilized treatments T5 and T6 could be observed in the first year. However, this observation was not reproduced in the second year. The unfertilized treatments T2 and T3 did not show significant differences to any

other treatment in both years. The comparison of hydrological zone effects in the first year revealed the emissions from the mid-section position to be significantly higher than the emissions from the center position. In the second year, the cumulative fluxes from the fringe and mid-section position showed similar results and were marked as significantly higher than the fluxes from the center position.

5.3.2. Carbon Dioxide

The seasonal variation of CO_2 emissions showed a homogeneous pattern during the two sampling years, without clear differences between the cropping and fallow seasons (Figure 12). The weighted seasonal CO_2 mean values also did not reveal a consistent pattern. The results from the cropping periods were slightly higher or equal to the emissions from the fallow period (Figure A.3 and A.4). In the first cropping period, the weighted mean values ranged between 66.32 and 185.69 mg CO_2-C $m^{-2}h^{-1}$. In the second cropping period, the results varied from 62.24 to 177.54 mg CO_2-C $m^{-2}h^{-1}$. The emissions from the first and second fallow period ranged from 51.32 to 180.18 mg CO_2-C $m^{-2}h^{-1}$ and 68.75 to 154.83 mg CO_2-C $m^{-2}h^{-1}$, respectively. The weighted seasonal mean values and cumulative fluxes revealed the CO_2 emissions to be comparable in their order of magnitude during the first and second year. No increase or decrease in emissions strength was observed during the two years (Figure 14 and 15). The cumulative CO_2 emissions varied from 5946 to 16001 kg CO_2-C $ha^{-1}yr^{-1}$ in the first year and from 6391 to 14308 kg CO_2-C $ha^{-1}yr^{-1}$ in the second year. The Kruskal-Wallis test revealed significantly higher emissions from T0 in the fringe position compared to T3 in the same position during the first year. However, no differences were observed in the second year (Figure 14). Treatment effects which were independent from the influence of the hydrological position were found in both years. T0 showed significantly higher fluxes compared to all other cropping treatments (the only exception was T5 in the first year). In contrast, no significant differences were observed for hydrological position effects during the two sampling years (Figure 15).

5.3.3. Nitrous Oxide

The N_2O fluxes showed increased emissions during the two rice cropping periods in both sampling years with peaks between September and November 2014, as well as between September and October 2015. Additional periods of increased N_2O emissions were observed during both fallow periods from March to May 2015/2016 (see Figure 13). The weighted seasonal mean values showed a clear difference in emission magnitude between the cropping seasons and fallow seasons, as well as between the first and second year. The N_2O emissions during the two cropping seasons were considerably higher than during the fallow seasons (Figure A.3 and Figure A.4). Moreover, the results showed a decrease in magnitude from the first to the second year. During the first cropping period the weighted mean values varied between 0.003 and 0.215 mg N_2O-N $m^{-2}h^{-1}$ and

decreased to a level of 0.008 to 0.087 mg N_2O-N $m^{-2}h^{-1}$ in the second cropping period. In comparison, the weighted seasonal mean values from the first fallow period varied from 0.002 to 0.047 mg N_2O-N $m^{-2}h^{-1}$ and remained similar in the second fallow period with a range of 0.002 to 0.066 mg N_2O-N $m^{-2}h^{-1}$. The cumulative flux results underlined the decrease of N_2O from the first to the second year (Figure 14 and 15). The cumulative N_2O fluxes of the first year ranged from 0.19 to 9.54 kg N_2O-N $ha^{-1}yr^{-1}$, while the value from the second year decreased to a range between 0.16 and 6.14 kg N_2O-N $ha^{-1}yr^{-1}$. The comparison of treatment and zone interactions by the Kruskal Wallis test showed significant differences during the second year for T0 in the mid-section position which was significantly smaller than the emissions from T6 in the same position (Figure 14). However, no significant differences were observed in the first year. The focus on hydrological zone effects also showed no significant differences between the hydrological positions in the first year, although the emissions from the fringe field were smaller than from the mid-section and center field. During the second year, the emissions from the fringe position were marked as significantly smaller than emissions from the center position (Figure 15). Moreover, the cumulative N_2O emissions revealed significant treatment effects in both years. Emissions from the (semi-natural) vegetation treatment T0 were significantly lower compared to emissions from T2 and T5 during the first year. In the second year, the emissions from T0 were significantly lower than emissions from the fertilized treatments T5 and T6.

5.3.4. Nitrate and Ammonium

The concentration dynamics of NO_3^- and NH_4^+ in soil showed just slight seasonal differences (Figure 17). Only few NO_3^- concentration peaks and occasionally increased NH_4^+ concentration peaks were observed, mainly during the first sampling year. In general, the NH_4^+ values showed a decrease in concentration from the first to the second year regarding the cropping as well as the fallow seasons, while NO_3^- showed a decrease in concentration for the cropping periods only. The NO_3^- concentrations ranged between 0.50 and 994.37 mg NO_3^--N kg^{-1} during the first cropping period and decreased to a range of 0.00 to 561.52 mg NO_3^--N kg^{-1} during the first fallow period. In the second year, the concentrations of the cropping season decreased considerably to 0.07 to 199.42 mg NO_3^--N kg^{-1}, while the concentration range of the second fallow period was comparable to the first fallow period with values between 0.00 and 447.21 mg NO_3^--N kg^{-1}. In comparison, the NH_4^+ concentrations varied from 0.26 to 376.37 mg NH_4^+-N kg^{-1} during the first cropping period and remained in the same order of magnitude from 0.03 to 394.13 mg NH_4^+-N kg^{-1} during the first fallow period. In the second year, the NH_4^+ concentrations decreased considerably to 0.00 to 70.71 mg NH_4^+-N kg^{-1} during the second cropping period and a range of 0.02 to 183.07 mg NH_4^+-N kg^{-1} during the second fallow period. The total N concentrations varied from 0.00 to 1045.61 mg total N-N kg^{-1} during the first year and 0.00 to 458.18 mg total N-N kg^{-1} during the second year. In general, no clear pattern for NO_3^- and NH_4^+ concentration dynamics was observed during both

sampling years. Moreover, no significant differences in zone or treatment effects could be detected by the Kruskal Wallis test for the results of both years. Hence, the amount of fertilizer application was not reflected in the mineral nitrogen concentrations of the soil. The Spearman correlation test did not find a relationship between the time courses of N_2O fluxes with the temporal dynamics of NO_3^-, NH_4^+ and total N, respectively (Table A.2).

5.3.5. GWP and GWPI

All GWP and GWPI results are illustrated in Figure 16. The GWP outcomes show considerably higher results in the first year. The lowest GWP values appeared from T0 in all three hydrological positions during both years, while T5 represented the highest GWP values, but only during the first year. The Kruskal Wallis test revealed the GWP from T0 to be significantly lower compared to T3, T5 and T6 in both years (Figure 16, Picture 3 & 4). Statistical significance also occurred in the fringe field of the first year where T0 was proven to be significantly smaller than T5 (Figure 16, Picture 1). Hydrological zone effects were only observed during the first year, the GWP from the mid-section position was significantly higher than from the fringe position. This result was not repeated in the second year (Figure 16, Picture 3 & 4). Treatment and hydrological zone interactions were observed in the second year. Thereby, the results from T0 in the mid-section position were significantly lower than the results from T5 in the fringe position (Figure 16, Picture 2). The GWPI results showed neither significant differences for treatment and hydrological zone interactions (Figure 16, Picture 5 & 6), nor any treatment effects (Figure 16, Picture 7 & 8). The only statistical significances were observed for hydrological zone effects in the first year. The GWPI of T3 in the fringe position was significantly lower than the results from T3 in the mid-section position. In general, the GWPI of the fringe position was significantly smaller than the one of the mid-section position (Figure 16, Picture 5 & 7). No significant differences were revealed for the GWPI results of the second year (Figure 16, Picture 6 & 8).

5.4. Discussion

Available GHG data from wetlands of SSA are rare. The few existing studies encompass relatively short measurement periods of a few days, weeks or months (e.g. Jones (2001), Krüger et al. (2013), Nyamadzawo et al. (2014)). The present study considers a much longer measurement period of two consecutive years with two cropping and two fallow seasons. For a better comparison, the results of other studies where converted into the same units, if necessary ($mg\,C\,m^{-2}h^{-1}$, $mg\,N\,m^{-2}h^{-1}$, $kg\,C\,ha^{-1}$ and $kg\,N\,ha^{-1}$). The original flux values and units are given in the appendix in Table A.3.

5.4.1. Land-Use Change

Land-use change of natural wetlands in combination with artificial drainage commonly leads to a decrease in soil moisture and a change of anoxic to oxic soil conditions. This causes an increase in carbon turnover rates followed by an increase in CO_2 emissions and a decrease of CH_4 emissions (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013; Mitsch and Gosselink, 2015). However, the observed changes in GHG emissions were not as expected from the literature. All agricultural treatments (T2, T3, T5 and T6) showed higher CH_4 and lower CO_2 emissions compared to the (semi-) natural vegetation treatment (T0) which indicated an increase of CH_4 and a decrease of CO_2 emissions after land-use change (Figure 14 and 15). These conflicting observations can be explained by high soil moisture levels in the paddy field especially during the rainy period. The conditions favoured the production of CH_4 and inhibited CO_2 emission (Figure 18). The natural wetland of the research area was not permanently flooded, but periodically dry as indicated by conditions in the (semi-) natural wetland areas. Thus, the change in land use to a cultivated paddy field most likely did not reduce soil water content, but led to increased soil moisture levels. The effects of bunding on water retention and GHG emission was particularly obvious in the first year, when the CH_4 emissions of the bunded treatments (T3, T5 and T6) exceeded the emissions of the unbunded treatment (T2) and the natural reference treatment (T0) (Figure 15). An increase of N_2O emissions was observed in all agricultural treatments (T2, T3, T5 and T6), while the emissions from the (semi-) natural vegetation treatment (T0) remained low. This effect can be explained by fertilizer application in agriculture as well as by an increased availability of nutrients caused by land-use change (Kroeze et al., 1999; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013; Delkash et al., 2018).

5.4.2. Methane

The conversion of the research area from a (semi-) natural wetland to a paddy field led to an exposure of large carbon and nitrogen stocks (Schlesinger and Bernhardt, 2013), probably due to rapid mineralization of root litter. The high CH_4 emissions in the first year can be explained by this increase in nutrient availability, while the decrease of CH_4 fluxes in the second year suggested a depletion of the easy available carbon stocks. The seasonal pattern of CH_4 emissions followed the time course of the rice cropping seasons. The CH_4 fluxes increased when the cropping period started at the beginning of the long rainy season (September) and decreased with the beginning of the dry season shortly before harvest (December). However, no increase in CH_4 emissions were observed during the short rainy season from March to May. This indicated that the presence of rice plants and their favouring effect on CH_4 emissions through vascular transport by aerenchyma has a high measurable influence (Wagatsuma et al., 1990; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). The difference in CH_4 emissions between the (semi-)

natural vegetation treatment (T0) and the agricultural treatments (T2, T3, T5, T6) can also be explained by the advantageous effect of rice plants and agricultural soil cultivation on CH_4 production and emission. Hydrological zone effects revealed themselves in form of significantly lower CH_4 fluxes in the center field. Redox potential measurements in the center field showed moderately reduced to oxidized soil conditions between 100 to 800 mV (Figure 18), which is not optimal for CH_4 production as methanogenesis occurs under most reduced conditions (-200 mV) (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). Moreover, continuous oxidized soil conditions in the upper soil profile increased the possibility for methane oxidation (Whalen, 2005; Mitsch and Gosselink, 2015). In comparison, the redox potential conditions of the fringe and mid-section fields ranged from highly reduced to oxidized (-200 to 700 mV) with longer episodes of reduced soil conditions. This allowed methanogens to produce high quantities of CH_4 (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). Soil moisture measurements conducted by Gabiri et al. (2018a) in the same research area did not reveal any significant zone differences within between the mid-section and center field (compare transect 3: valley bottom vs. riparian zone in Gabiri et al. (2018a)).

The results of other studies conducted in wetlands of SSA showed a wide variation of CH_4 emissions. The CH_4 fluxes from this study can be classified in the low to medium range. In comparison, Nyamadzawo et al. (2014) observed cultivated seasonal wetlands in Zimbabwe and found relatively low CH_4 emissions of 0 to 1.35 mg CH_4-C $m^{-2}h^{-1}$. Krüger et al. (2013) also reported low emissions and small uptake rates of -0.07 to 1.76 mg CH_4-C m^{-2} h^{-1} from two natural South African wetlands. Jones (2001) reported flux results of 4.0 mg CH_4-C $m^{-2}h^{-1}$ from a papyrus dominated natural wetland in Kenya. Mitsch et al. (2012) found relatively high CH_4 emissions of 8.22 $\pm$ 1 mg CH_4-C $m^{-2}h^{-1}$ in a dry tropical wetland in Botswana. The widest range of values was reported by Gondwe and Masamba (2013) with diffusive CH_4 emissions ranging from 0.15 to 225 mg CH_4-C $m^{-2}h^{-1}$ from an inland wetland in Botswana. Kim et al. (2016) reported annual cumulative CH_4 emissions of 712.8 $\pm$ 262.8 kg CH_4-C $ha^{-1}yr^{-1}$ from wetlands, floodplains, lagoons and reservoirs, which is two to four times higher than the maximum of cumulative CH_4 values observed in this study. CH_4 emissions from African rice paddies ranged from 6.25 - 15.63 mg CH_4-C $m^{-2}h^{-1}$ (Cao et al., 1998) to 22.9 kg CH_4-C $ha^{-1}yr^{-1}$ (Kim et al., 2016). All studies confirmed that African wetlands and rice paddies are sources for CH_4, although the amount of emitted CH_4 varied considerably.

5.4.3. Carbon Dioxide

The change in land use from a natural wetland to a paddy rice field involved soil cultivation and field flooding, which had an inhibitory effect on CO_2 production (Liu et al., 2013; MacCarthy et al., 2018). This led to significantly lower CO_2 fluxes in the agricultural treatments (T2, T3, T5 and T6) compared to the (semi-) natural vegetation

treatment (T0) (see also Figure 15). Besides, no other treatment effects could be observed. The unbunded treatment (T2) did no show any significant differences in CO_2 emissions compared to the bunded agricultural treatments. According to the Mann-Whitney U Test, the soil moisture levels between unbunded and bunded treatments were also not significantly different within the first 10 cm in all hydrological zones. Therefore, the observations can be seen as indicator that soil cultivation and land-use change itself have a higher impact on CO_2 emissions than different cropping methods. No considerable hydrological zone effects were observed. The CO_2 emissions from the fringe field showed slightly higher fluxes, though this increase was not statistically significant in both years. Nevertheless, it can be explained by the significantly lower soil moisture content in the fringe field found by Gabiri et al. (2018a) which provides more advantageous conditions for soil respiration and CO_2 production. In general, the observed values of the present work are rather low, compared with results of other studies. Nyamadzawo et al. (2014) reported emissions of 136.36 to 567.27 mg CO_2-C $m^{-2}h^{-1}$ from a cultivated seasonal wetland in Zimbabwe, which were higher than the CO_2 results found in this study. MacCarthy et al. (2018) conducted CO_2 research with transparent chambers in a low land paddy rice field in Southern Ghana and measured 90 mg CO_2-C $m^{-2}h^{-1}$ during night time. Kim et al. (2016) reported emissions of 17727 kg CO_2-C $ha^{-1}yr^{-1}$ from rice fields in SSA which were within a comparable range.

5.4.4. Nitrous Oxide

The observed periods of N_2O emission peaks were synchronous to the rainy seasons in Uganda which occur each year between March and May as well as between September and November (Nsubuga et al., 2011). The increase in N_2O emissions can be explained by increased soil moisture levels during the rainy seasons providing favourable conditions for N_2O producing microorganism (Reddy and DeLaune, 2008). Moreover, the N_2O peaks occurred during the main cropping period from September to January and the second cropping period from March to June (the latter was only conducted in neighbouring plots within the same field). Fertilization took place shortly before and during the rice cropping period (see Figure 13). Nitrogen fertilizer input in wetlands is known to increase N_2O emissions (Yan et al., 2000; Groffman et al., 2000; Cooke et al., 2018). The seasonal mean values and cumulative flux results of this study confirm the enhancing effect of fertilization on N_2O emissions, as the emissions from the fertilized agricultural treatments (T5 and T6) were (partially significantly) higher than from the (semi-) natural vegetation treatment (T0) (see Figure 15). However, the emissions from the unfertilized agricultural treatments (T2 and T3) were also increased compared to T0, though not significant in most cases. The fact that the emission differences among the individual agricultural treatments were not significant, led to the conclusion that high amounts of fertilizer application do not necessarily result in high N_2O emissions. Several explanations are possible, for example a major proportion of applied fertilizer can be leached (Kimetu et al., 2006; Yeasmin et al.,

2012), effectively used for plant uptake (Snyder et al., 2009), reduced to N_2 (Groffman et al., 2000) or immobilized in organic soil substance and microbial biomass (see also section 5.4.5). Significant differences of hydrological zone effects were only found in the second year between the fringe and center field. However, the emission heights from the zones differed considerably in both years. Thus, it was neither possible to describe a general emission pattern nor formulate a management recommendation with regard to hydrological zone effects. The average N_2O emissions observed in this study were within the lower range of results reported by other studies. Nyamadzawo et al. (2014) found average emissions of 0.07 to 3.3 mg N_2O-N $m^{-2}h^{-1}$ in seasonal wetlands of Zimbabwe, while Krüger et al. (2013) reported fluxes between -0.02 to 3.70 mg N_2O-N m^{-2} h^{-1} from two South African wetlands during rewetting. According to Kim et al. (2016) wetlands belong to the largest N_2O sources in SSA after forests, plantations and woodlands. The authors reported cumulative N_2O emissions of 1.27 $\pm$ 0.95 kg N_2O-N $ha^{-1}yr^{-1}$ from wetlands, floodplains, lagoons and reservoirs and 0.12 kg N_2O-N $ha^{-1}yr^{-1}$ from rice fields, which are considerably lower than the cumulative flux results of this study.

5.4.5. Relationship of Mineral Nitrogen Dynamics and N_2O fluxes

Microbial transformation of nitrogen through nitrification and denitrification leads to the formation of N_2O (Reddy and DeLaune, 2008). Nitrification constitutes the oxidation of NH_4^+ to NO_3^- under oxic conditions, thereby N_2O is produced due to the decomposition of intermediates. Denitrification performs the reduction of NO_3^- to N_2O and N_2, respectively (Hernandez and Mitsch, 2006). Many studies report a positive correlation between NO_3^- concentration in water or soil and N_2O flux (e.g Smith et al. (1982), Eaton and Partiquin (1989), Yan et al. (2000), Groffman et al. (2000), and Cooke et al. (2018)). However, the N_2O results of the present study did not show any correlation with the dynamics of NO_3^-, NH_4^+ or total N (Table A.2). The periods of increased mineral nitrogen concentration emerged later than the periods of increased N_2O emissions. Instead of a synchronised occurrence of N_2O peaks and high mineral nitrogen contents, the NO_3^- and NH_4^+ concentrations started to rise after the N_2O fluxes decreased at the same time when the cropping period ended and the dry season started. Yan et al. (2000) also did not find a direct correlation between N_2O fluxes and NH_4^+ concentration. Due to the fact that nitrification rather depends on redox potential than on NH_4^+ content, N_2O emission is not necessarily interlinked to NH_4^+ concentration. The intensity of nitrification can vary as a function of changes in redox potential and soil moisture, while NH_4^+ concentration remain constant (Yan et al., 2000). Several explanations are possible for the non-correlated relationship between N_2O and NO_3^-. For example, N_2O can be further reduced to N_2 during denitrification. Although NO_3^- is the preferred electron acceptor during denitrification, limited NO_3^- availability and high denitrification rates can lead to a great demand for electron acceptors and a further reduction of N_2O to N_2 (Groffman et al., 2000; Hernandez and Mitsch, 2006; Cooke et al., 2018). Gale et

al. (1993) and Picek et al. (2007) reported complete denitrification of NO_3^- to N_2 from constructed wetlands. The observed mineral nitrogen concentrations of the present study were relatively low compared to other studies. For example, Nyamadzawo et al. (2014) found higher total N concentrations ranging from 1.0 to 3.0 g total N kg^{-1}. In conclusion, it is possible that large proportions of N_2O emitted from the test site were reduced to N_2. On the other hand, low pH and high O_2 levels can denature N_2O reductase enzymes and lead to higher N_2O/N_2 ratios (Groffman et al., 2000; Hernandez and Mitsch, 2006). Picek et al. (2007) reported N_2O emissions to be detectable at pH 5. The pH values in the research field of the present study were low with a mean value of 5.4 and a range between 4.1 to 6.5. Therefore, a conversion of N_2O to N_2 might have been impeded. Due to several factors that influence the reduction of N_2O to N_2, its assessment as well as the evaluation of the relationship between NO_3^- and N_2O is complicated. Other factors that can influence the correlation between NO_3^- concentration and N_2O flux are the loss of nitrate through leaching and runoff or plant uptake (Kimetu et al., 2006; Snyder et al., 2009; Yeasmin et al., 2012). During nitrification, immobile NH_4^+ is transformed to highly mobile NO_3^-. Due to the negative charge of NO_3^- and soil particles, the adsorptivity for NO_3^- is low, which promotes leaching of the same down the soil profile (Kimetu et al., 2006). Moreover, it also becomes highly available for plants. NO_3^- is usually the major uptake form of nitrogen (Snyder et al., 2009). The results of this study indicate that the soil mineral nitrogen was either immobilized, leached, taken up for plant growth or subjected to nitrification and denitrification, leading to the production of GHGs during the rainy season. During the dry season, the potential for leaching decreased due to lack of water, furthermore no crops were grown in the field and N_2O emissions decreased considerably. This resulted in an increase of mineral nitrogen concentration in soil.

5.4.6. GWP and GWPI

The GWP and GWPI results of this study were comparable with the findings of other studies, such as Ma et al. (2013), who reported values of 4720 to 12150 kg CO_2 eq ha^{-1} yr^{-1} and 0.51 to 0.72 kg CO_2 eq kg^{-1} yield yr^{-1} from a flooded rice fields in China. Shi et al. (2013) reported lower values ranging from 411.3 to 981.7 kg CO_2 eq ha^{-1} yr^{-1} and from 0.044 to 0.061 kg CO_2 eq kg^{-1} yield yr^{-1}. The assessment of GWP and GWPI from different cropping systems is of great importance for the mitigation of GHG emissions from agriculture. Robertson (2000) outlined that maximum mitigation of GHG emissions is only achievable when land is partially excluded from production. The GWP results from this study support this statement. The GWP from the (semi-) natural reference treatment (T0) is smaller than from the agricultural treatments which indicates that natural wetlands have a less negative effect on climate change than rice fields. However, rising global food insecurity and the resulting demand for agricultural areas impede an exclusion of wetlands from food production (Wood and van Halsema, 2008; Rebelo et al., 2009b). Instead, effective management practices are needed for a best possible trade-off between

increased biomass production and low GHG emissions (Cassman, 1999; Snyder et al., 2009). According to Adviento-Borbe et al. (2007), an optimized management for better exploitation of the yield potential can help to keep net GHG emissions from agricultural systems low. Intensification and high-yielding crops have the ability to increase soil carbon storage and thus to mitigate GHG emissions (Dobermann, 2007). The results of this study indicate that high production treatments (T5 and T6) do not feature significantly higher GWPs than low production treatments (T2 and T3). The combined consideration of rice yields and GHG emissions in form of GWPI confirms that intensive cropping does not necessarily result in negative environmental effects. High yields can compensate increased GHG emissions and even result in lower GWPIs than treatments with less GHG emissions but even lower yields. The potential of high production treatments for a net mitigation of GHGs is linked to land area spared from production (Snyder et al., 2009). With more crops produced on one land unit, more wetland areas can remain in a natural condition and thus maintain their positive effects on climate regulation.

5.5. Conclusion

The observed developments in GHG emissions induced by land-use change were not always as expected from the literature. CH_4 emissions increased while CO_2 emission decreased after land-use change. This was probably due to an assumed increase in soil water content after the transformation of the natural wetland to a cultivated paddy field, although land-use change usually involves drainage and a decrease in soil moisture. N_2O emissions also increased because of fertilizer application and increased nutrient availability (Kroeze et al., 1999; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013; Delkash et al., 2018). However, treatments with high amounts of fertilizer application did not show the highest N_2O emissions. The amount of applied fertilizers in each treatment was not reflected in the soil concentration regarding NO_3^- and NH_4^+. Moreover, a correlation between mineral nitrogen dynamics and N_2O emissions was also not observed. Possible explanations are that a major proportion of fertilized nitrogen was either leached, used for plant uptake or reduced to N_2 (Groffman et al., 2000; Kimetu et al., 2006; Snyder et al., 2009; Yeasmin et al., 2012). In general, the results of this study indicated that the effect of land-use change itself has a much stronger impact on GHG emissions than different treatment methods. The results of all three GHGs showed (partially significant) differences between the (semi-) natural vegetation reference and the agricultural treatments, while the differences among the individual cropping treatments were much less pronounced. The hydrological zone effects showed very variable results. A general statement about GHG reduction by effective zone management was not possible. According to the present results, the most effective method for the mitigation of GHG emissions is the exclusion of land from production and the restoration of wetlands to their natural state. Similar conclusions were made by Robertson (2000). Therefore, management practices are required which provide the best possible trade-off between maximized food production and minimized

GHG emission (Cassman, 1999; Snyder et al., 2009). Based on the results of this study low production treatments, such as the current farmer's practice (T2), are not recommendable because their GWP is in the same order of magnitude as high production treatments, but their yield is lower. Instead, intensive cropping treatments (e.g. T5 and T6) with good exploitation of their yield potential are advisable, because they have the ability to compensate increased GHG emissions and keep the resulting GWPI low. However, this approach is inevitably linked to the condition of land spared from agricultural activity (Snyder et al., 2009). Intensification of food productivity combined with the preservation of large natural wetland areas provides the best possible solution for the mitigation of GHG emissions and natural climate regulation.

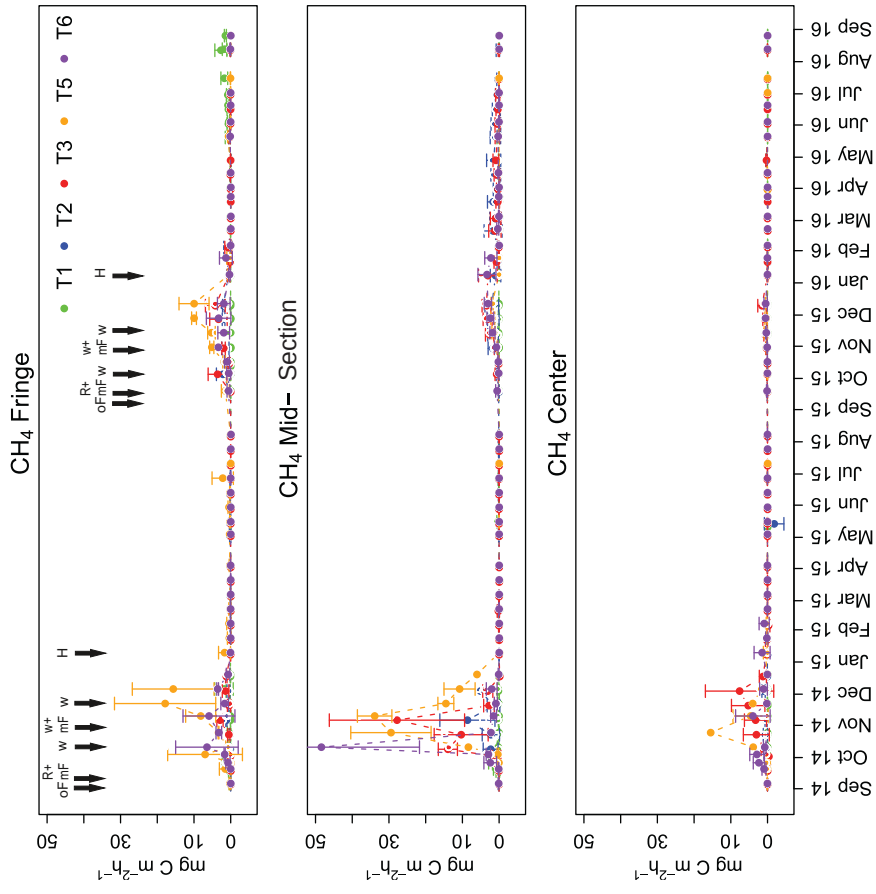

Figure 11: Seasonal variation of CH_4 fluxes with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF), weeding (W) and harvest (H).

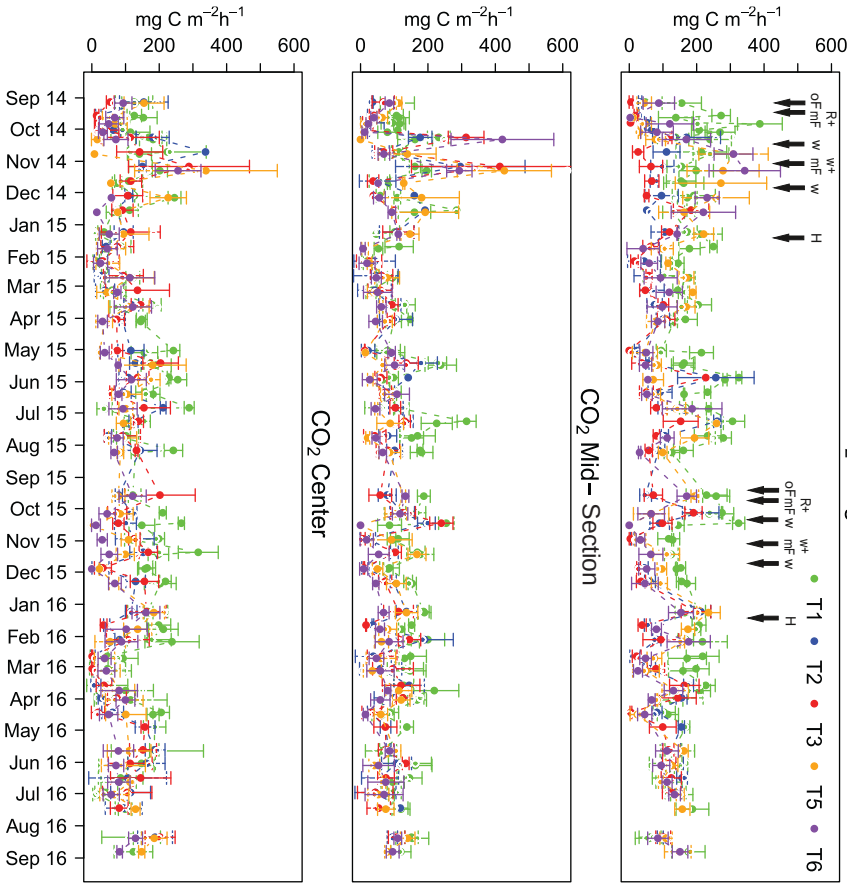

Figure 12: Seasonal variation of CO_2 fluxes with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF), weeding (W) and harvest (H).

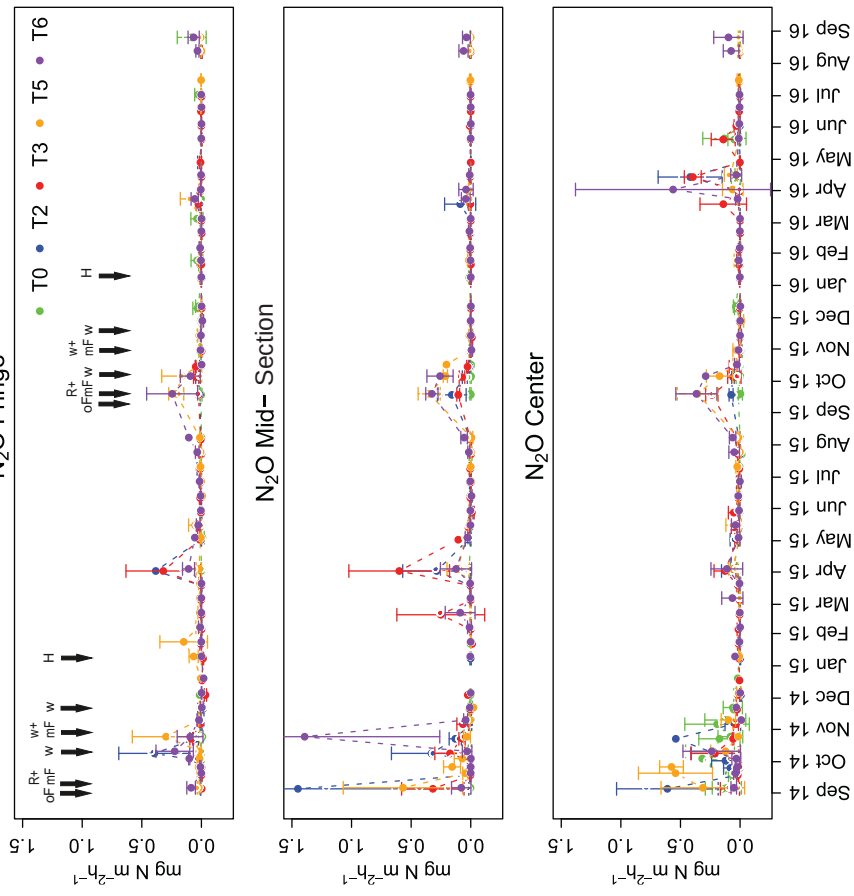

Figure 13: Seasonal variation of N_2O fluxes with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF), weeding (W) and harvest (H).

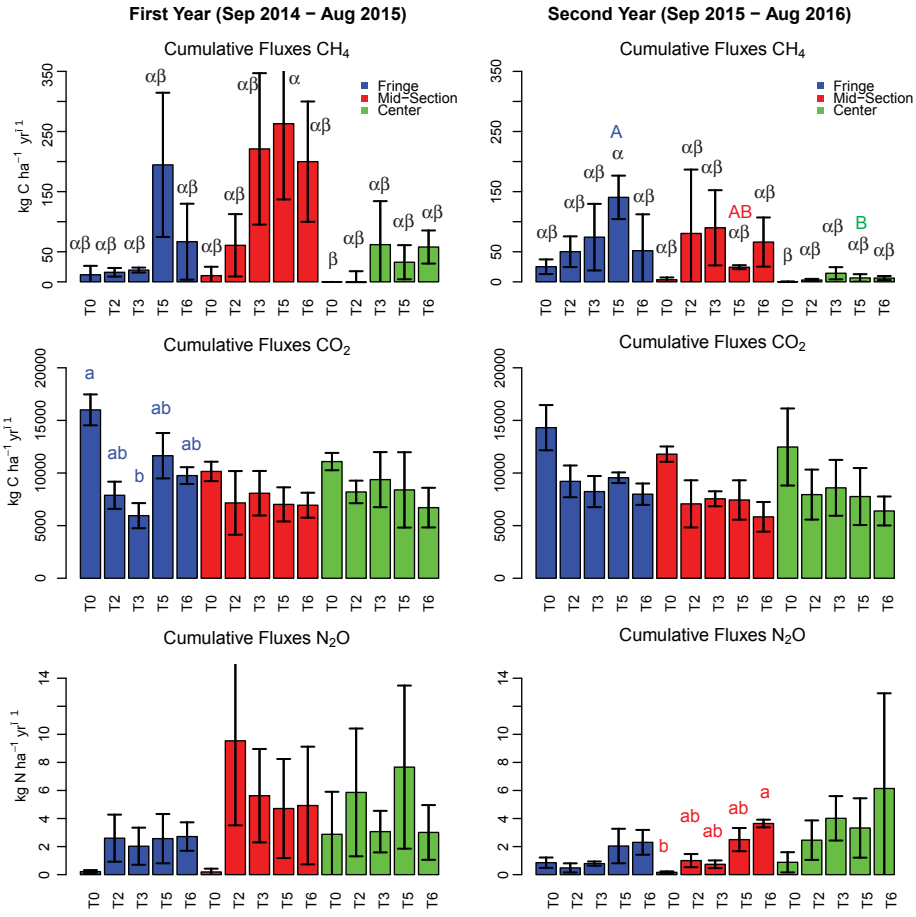

Figure 14: Cumulative GHG fluxes ± SD from the Namulonge field trial in Uganda. The letters indicate significant differences at p = 0.05 by the Kruskal-Wallis Test:
- α, β: significant differences for treatment and hydrological zone interactions
- a, b: significant differences for treatment effects within one hydrological zone
- A, B: significant differences for hydrological zone effects within one treatment
T0: (semi-) natural vegetation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application.

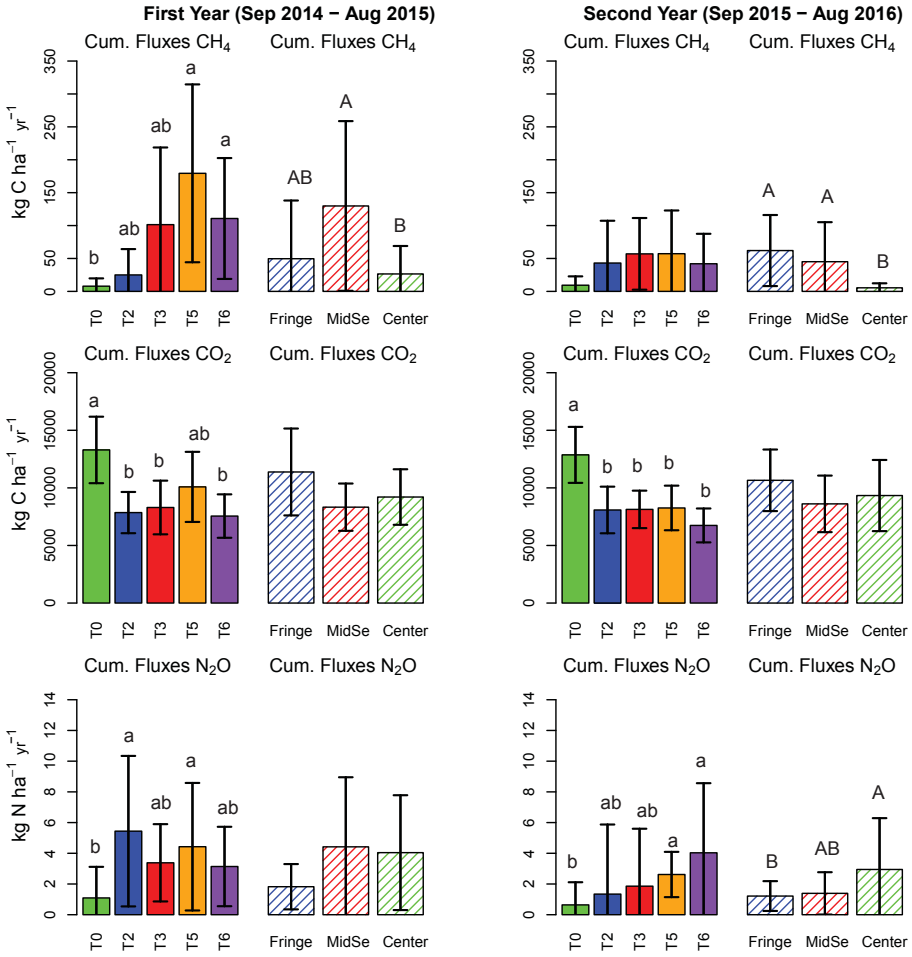

Figure 15: Treatment and hydrological zone summary for cumulative GHG fluxes ± SD from the Namulonge field trial in Uganda.
The letters indicate significant differences at p <0.05 by the Kruskal-Wallis Test:
- a, b: significant differences for treatment effects
- A, B: significant differences for hydrological zone effects
T0: (semi-) natural vegetation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application.

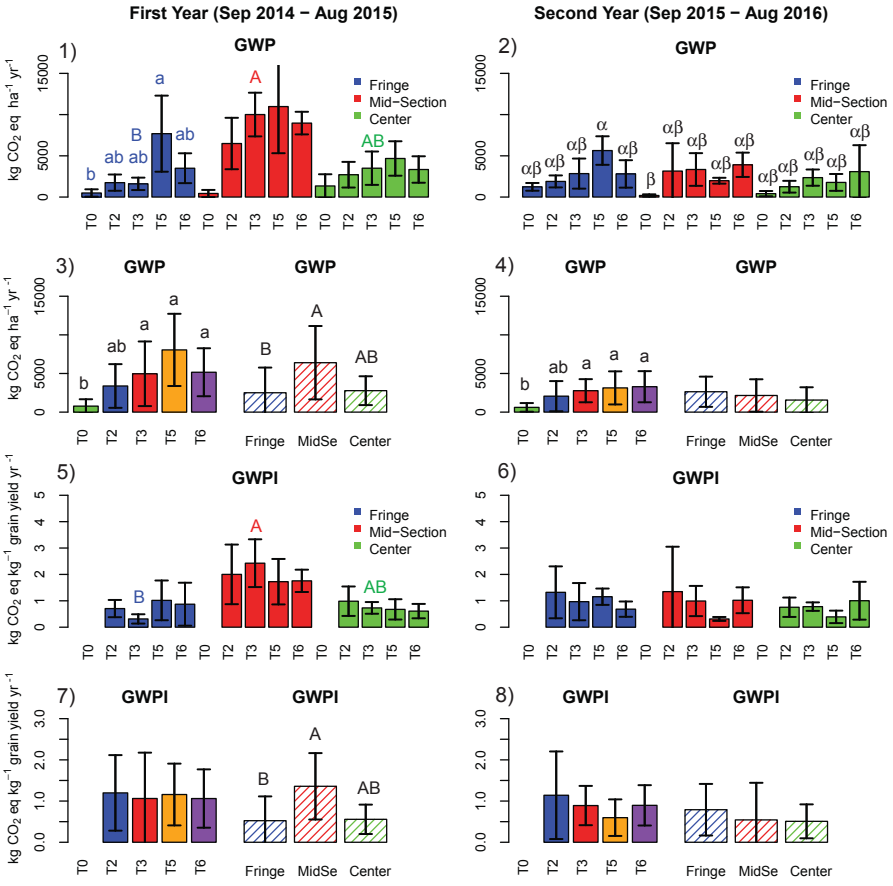

Figure 16: GWP and GWPI results from the Namulonge field trial in Uganda.
Picture 1, 2, 5 and 6 show all individual treatment-zone combinations. Picture 3, 4, 7 and 8 show summaries for each treatment and zone.
Letters indicate significant differences at p = 0.05 by the Kruskal-Wallis Test:
- α, β: significant differences for treatment and hydrological zone interactions
- a, b: significant differences for treatment effects
- A, B: significant differences for hydrological zone effects
GWP: global warming potential, GWPI: global warming potential index, T0: (semi-) natural vegetation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application.

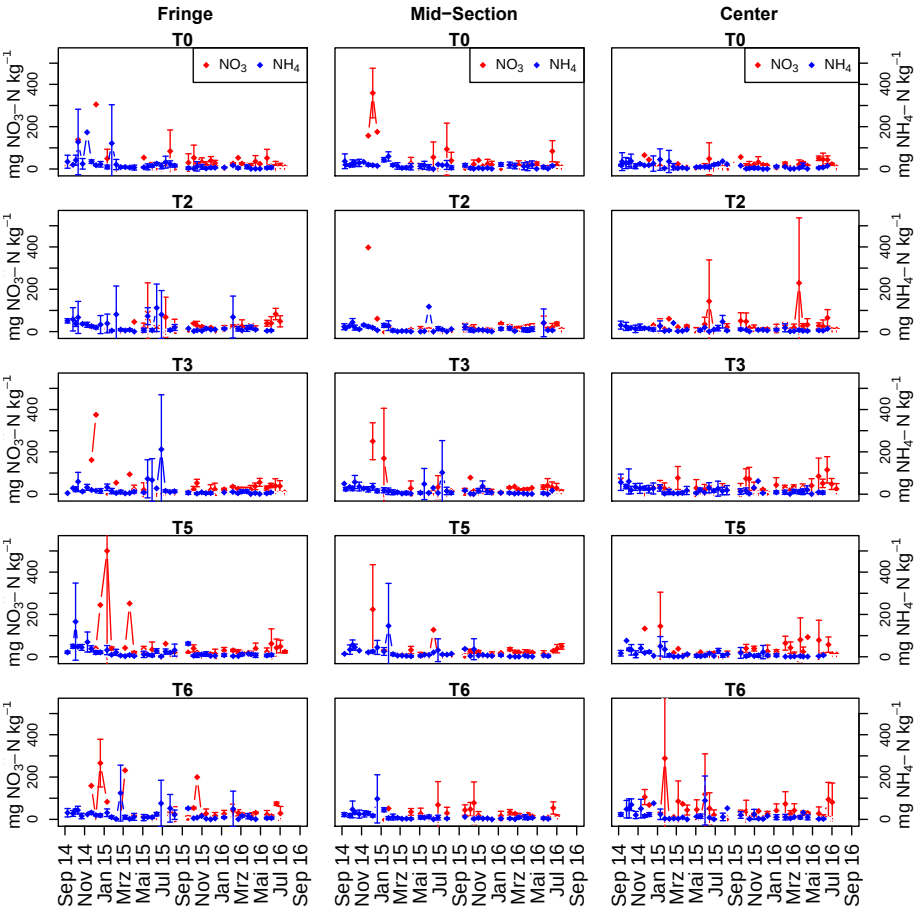

Figure 17: Nitrate and ammonium concentrations in soil, Namulonge.
T0: (semi-) natural vegetation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application.

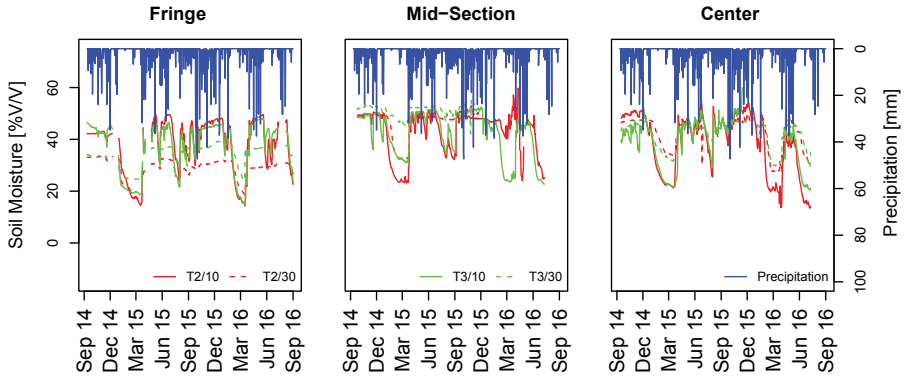

Figure 18: Rainfall and soil moisture data, Namulonge.
T2/10: Treatment 2, 10 cm depth. T2/30: Treatment 2, 30 cm depth. T3/10: Treatment 3, 10 cm depth. T3/30: Treatment 3, 30 cm depth.

6. Impacts of Agricultural Practices in Rice Fields on GHG Emissions from a Floodplain in Tanzania

Abstract

Wetlands are important space resources for food production in SSA, but agricultural activities are known to influence GHG emissions. Thus, sustainable wetland management has to reconcile food production with reduced GHG emissions. However, the numbers of studies dealing with GHG flux measurements from wetlands in SSA are low and long-term data about trace gas fluxes from agricultural used wetlands are rare. This study aimed to reduce the data gap by measuring CH_4, CO_2 and N_2O emissions from three locations with multiple plots in different hydrological zones of the Kilombero floodplain in Tanzania between March 2015 and October 2016. Five treatments were considered including renaturation (T1), no mineral N fertilization or bunding (T2), bunding without N fertilization (T3), intensive crop management with 120 kg urea-N/ha + 60 kg PK (T5) and green manure application (T6). The observed cumulative fluxes ranged from -13.55 to 26.33 kg CH_4-C $ha^{-1}yr^{-1}$, 1009 to 12747 kg CO_2-C $ha^{-1}yr^{-1}$ and -0.027 to 9.594 kg N_2O-N $ha^{-1}yr^{-1}$. The observed GWP results varied between -376 and 4902 kg CO_2eq $ha^{-1}yr^{-1}$ while the GWPI values ranged between 0.04 and 0.37 kg CO_2 eq kg^{-1}grain yield yr^{-1}. The results indicated that intensive cropping with fertilizer application did not necessarily lead to increased yield-based GHG emissions compared to local farmer's practise without fertilization. Although the fertilized treatments showed increased CH_4 emissions, the emissions were compensated by increased crop yields. In conclusion, sustainable wetland management can involve high-yielding practices if natural wetland areas are spared from food production elsewhere.

6.1. Introduction

Wetlands play an important role in carbon cycling and global climate regulation (Reddy and DeLaune, 2008). The mean global temperature within the last 10,000 years was reduced by 1.5 to 2° C because of the atmospheric carbon stored in peatland soils (Holden, 2005). Wetlands store about 14.5 to > 50% of the soil carbon worldwide (Gorham, 1991; Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015; Schlesinger and Bernhardt, 2013). The high carbon sequestration rates in wetlands are due to great net primary production rates that accounts for 7 to 15% of the world's terrestrial productivity (Schlesinger and Bernhardt, 2013). The net primary production surpass decomposition rates and as a result organic matter successively accumulates in wetland soils (Gorham, 1991; Whiting and Chanton, 2001). Especially in the tropics and subtropics, wetlands feature very high carbon retention rates with 150 to 250 g C m^{-2} yr^{-1} (Mitsch et al., 2012). Wetlands are also major GHG sources. Although their share on the earth's terrestrial surface only amounts to 2 to 7%, wetlands are responsible for 20 to 33 % of the world's CH_4 emissions (Reddy

and DeLaune, 2008; Kayranli et al., 2010; Schlesinger and Bernhardt, 2013; Mitsch and Gosselink, 2015). Thereof, tropical and subtropical wetlands have a share of 52 to 78% (Bloom et al., 2010; Mitsch et al., 2012). Wetland age, location and climate zone have a significant role regarding their evaluation in the carbon cycle (Kayranli et al., 2010; Mitsch et al., 2012). They have the ability to turn from net carbon sources into net sinks after 300 years. CH_4 emission and carbon sequestration balance themselves at the end of this period (Mitsch et al., 2012).

Wetlands are also valuable agricultural resources, because they feature good water supply and fertile soil properties. Thus, they provide the basis for the livelihood of many people in SSA (Bikangaga et al., 2007; Rebelo et al., 2009b; Mc Cartney et al., 2011). But land-use change seriously affects the unique natural ecosystem services provided by natural wetlands (Millennium Ecosystem Assessment, 2005; Reddy and DeLaune, 2008). For example, artificial drainage can cause the exposure of large organic matter and nutrient stocks to rapid mineralization by aerobic microorganism (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). Thus, the equilibrium among carbon and nitrogen inflow and outflow is heavily disturbed and can result in altered GHG emissions (Guo and Gifford, 2002). Nitrogen fertilization represents additional sources for nitrogen input and can lead to higher N_2O fluxes (Kroeze et al., 1999; Reddy and DeLaune, 2008). The pressure to expand agricultural activities on wetlands rose considerably within the last years due to population growth, rising global food insecurity and overexploitation of upland soils (Wood and van Halsema, 2008; Rebelo et al., 2009b; Barrett, 2010). Especially in East Africa, the stress is high because of government policies and demands by the population (Dixon and Wood, 2003).

A major challenge in wetland management is to find a trade-off between environmental sustainability and food production (Mc Cartney et al., 2011). Therefore, a comprehensive understanding of wetland processes and matter fluxes is needed. However, there is a great gap concerning GHG field data from SSA (Ciais et al., 2011; Oertel et al., 2016; Kim et al., 2016; Boateng et al., 2017). Uncertainties regarding available data are high, because of lacking long-term field studies (Ciais et al., 2011; Valentini et al., 2014). Generally, it is assumed that 42% of the N_2O emissions from SSA are caused by agricultural activities. A doubling of agricultural N_2O emissions is estimated for the period between 2000 and 2050 (Hickman et al., 2011), and an increase of GHG emissions due to continuing land-use change in the future is predicted (Kim et al., 2016). This study aims to analyse GHG emissions from different cropping treatments in rice fields of the Kilombero floodplain in Tanzania. The comparison of GHG fluxes, GWP and GWPI results from fertilized and unfertilized treatments can be used as sustainability indicators and will contribute to develop sustainable management recommendations and mitigate yield-based GHG emissions.

6.2. Methods

6.2.1. Research Area

The area was located in a lowland floodplain of the Kilombero river. The Kilombero valley has a great importance for food production in Tanzania as it became a major rice production area after the country's economic liberalisation. Paddy cultivation is the basis for subsistence farming as well as increased rice production and trading (Kato, 2006). The selected study site was located close to the city Ifakara ($8°07'59.99"S$, $36°40'59.99"E$) and characterized by a low elevation of 250 to 300 m above sea level, a sub-humid tropical climate with a mean annual temperature of 23–25° C and an annual rainfall of 1200 to 1400 mm (Koutsouris et al., 2015; Gabiri et al., 2018b). The rainy season was divided into two rainfall periods. The long rainy season occurred from March to May, while the shorter one was from October to December (Mombo et al., 2011). Flood peaks usually appeared during the long rainy season (Gabiri et al., 2018b). The main soil types were Fluvisols which dominated the sedimentary basin infillings of the seasonal alluvial floodplain, however fluvic Arenosols, Planosols and Gleysols were also found (Beck, 1964; Gabiri et al., 2018b). Agriculture represented the predominant economic activity of the region. The most prevalent food and cash crop was lowland rain-fed rice. Maize and peas were also cultivated in the valley, especially during the short rainy period (Gabiri et al., 2018b).

6.2.2. Study Design

The field experiment consisted of three rice cropping fields. The fields were located in three hydrological zones within the wetland. The fringe, mid-section and center position differed with respect to their topographic elevation and intensity of annual flooding (see also Figure 10 in chapter 5). Due to the flat topography of the Kilombero floodplain and in order to obtain the required topographic differences for the desired annual flooding periods, the three fields were selected with a linear distance of approximately 2-3 km between each field. The experimental plots (5 m x 6 m) were arranged in a randomized complete block design. In total, four agricultural treatments and one renaturation treatment within each hydrological zone were considered for the GHG measurements. GHG samples were collected in three replications ($n = 3$) of each treatment within each hydrological zone. Table 5 gives a summary of all applied agricultural treatments. T1 was characterized by renaturation. The plots were cleared in the beginning of the field experiment and left fallow to enable the regrowth of (semi-) natural vegetation. T2 represented the current rice cropping practise of local farmers without fertilization. T3 implemented bunding without fertilization to improve water retention and prevent nutrient leaching. T5 comprised intensive single crop management with application of mineral fertilizer. T6 involved the application of green manure (lablab) as a form of alternative nitrogen input (Ziegler et al., 2015). The applied amount was 48 kg/ha N in the fringe, 35 kg/ha N in the mid-section

Table 5: Agricultural treatments, Ifakara

Treatment	Description
T 1	renaturation with (semi-) natural vegetation
T 2	rice, no mineral N, no field bunds for rice
T 3	rice, no mineral N, plots bunded
T 5	rice, 120 kg urea-N/ha + 60 kg P/ha + 60 kg K/ha + supplementary irrigation, plots bunded
T 6	rice, no mineral N, pre-crop leguminous green manure (legume), plots bunded

and 39 kg/ha N in the center field during the first year, and 35 kg/ha N in the fringe, 51 kg/ha N in the mid-section and 42 kg/ha N in the center field during the second year, respectively.

6.2.3. GHG Sampling and Laboratory Analysis

The field trial was set up in January 2015. The sampling activities started simultaneously with the rice cropping season in March 2015. The collection of gas- and soil samples was conducted consistently every two weeks for two cropping periods and one fallow period until October 2016. Considerable fluctuations in GHG emissions were expected especially in the transition period between dry and rainy season. The GHG samples were collected with static chambers. The deployment time for each chamber was 60 min with sampling intervals of 20 min ($t_1 = 0\,min, t_2 = 20\,min, t_3 = 40\,min, t_4 = 60\,min$). The GHG samples were analysed in Germany at the Jülich Research Center with a gas chromatograph GC (European Greenhouse GC 8610C, SRI Instruments Europe GmbH, Bad Honnef, Germany) that was equipped with FID for CO_2 and CH_4 measurements and ECD for N_2O measurements. An autosampler was used to transfer each air sample from the sample vials to the injection port. A minimum overpressure of 0.16 bar within each vial was required to avoid an intrusion of ambient air into the transfer syringe and thus to prevent a contamination of the GHG samples. Subsequent to each measurement, every vial was checked with regard to its remaining overpressure.

6.2.4. Chamber Construction

Robust polypropylene boxes with a height of 0.4 m and matching polypropylene frames with a ground area of 0.6 m x 0.4 m were used for GHG sampling. Extension frames with 40 cm height were used additionally to extend the chamber headspace to a total height of 80 cm. This was necessary in the later stage of rice growth when the plants exceeded the initial chamber height. Because rice plants have an aerencyma which acts as preferential

escape paths for CH_4 emissions, the inclusion of the whole rice plant during measurement was essential to receive representative data (Wagatsuma et al., 1990; Jones, 2001). The chamber construction followed the example of chapter 5 and has been described in detail in section 5.2.3. Due to expected high flooding in the Kilombero floodplain, additional swimming rings made of styrodur (0.9 m x 0.7 m x 0,12 m) were provided and attached around frames so that they were able to float accordingly. The chambers could be attached on top of the floating frames and operated in the usual way. A schematic example is shown in picture A.8 (page 136). The floating chambers were only used in the center position where flooding height reached up to 2 m. Though, due to safety reasons gas sampling with floating chambers was only conducted until a flooding height maximum of 1 m. A temperature Probe (KHXL-IM30U-RSC-300 hand-held-probe, OMEGA, Deckenpfronn, Germany) and a datalogger (HH306A Type K Thermometer + Datalogger, OMEGA, Deckenpfronn, Germany) was mounted on one chamber to measure temperature variations in the chamber headspace during operation. Gas-pooling as described by Arias-Navarro et al. (2013) was applied for the GHG sampling in each treatment unit to minimize the number of required vials and to overcome spatial heterogenities. Thereby, the reliability of estimated GHG fluxes was maintained (Arias-Navarro et al., 2013). Because no vacuum pump for the evacuation of the sample containers was available, the vials were prepared according to a method described by Klein et al. (2003). Each vial was flushed with a sample air volume four times larger than the vial volume. Moreover, an additional quantity of 15 mL was inserted into each vial to create an overpressure of 0.75 bar which was essential for storage and GC analysis (compare with section 6.2.3).

6.2.5. Data Quality Management and Flux Calculation

All GHG results were checked with a newly developed DQMS. The DQMS controlled all collected GHG concentrations with respect to their applicability for linear flux calculation and removed outliers, if required. A detailed explanation of the DQMS can be found in chapter 4. Accredited concentration results were stored in the final data set and used for flux calculation. The final GHG fluxes were calculated according to the changes in concentrations within the chamber headspace over time. Thereby, air temperature and pressure correction were taken into consideration:

$$F = a\,h\frac{M}{V_0}\frac{p}{p_0}\frac{T_0}{T_C}\,0.6 \tag{23}$$

F is the final flux ($mg\,C\,m^{-2}\,h^{-1}$ or $mg\,N\,m^{-2}\,h^{-1}$, respectively), a is the concentration gradient ($ppm\,min^{-1}$), h is the chamber height (cm), M is the molar mass of C ($12\,g\,mol^{-1}$) or N ($28\,g\,mol^{-1}$), V_0 is the molar volume ($22.414\,L\,mol^{-1}$), p is the air pressure (hpa), p_0 is the normal pressure (1012 hpa), T_0 is the normal temperature ($273.15\,K$) and T_C is the chamber temperature (K), *0.6* is the residual correction factor which includes all unit

conversions ($\min h^{-1}$).

6.2.6. Soil Sampling and Laboratory Analysis

The collection of soil samples for the observation of mineral nitrogen dynamics was conducted every two weeks to receive long-term monitoring data. Soil sampling happened simultaneous with the collection of GHG samples . Considerable changes in nitrate (NO_3^-) and ammonium (NH_4^+) activities were expected to occur, especially during the transitions period from dry to rainy season. Soil sample collection and preparation was identical with the approach described in chapter 5 section 5.2.6. The soil samples were collected with a small soil auger (Ackerlandbohrer, ø13 mm, working length 25 cm, total length 58 cm by Eijkelkamp Soil & Water, EM Giesbeek, Netherlands) from 0 to 10 cm depth. Each soil sample consisted of five mixed samples from the same treatment plot and was cooled immediately after collection in an ice-packed cooling box in the field and later in a freezer at −16°C. After thawing at +7°C the samples were prepared for analysis. Dry weights were determined after 24 hours of drying at 105°C. The following preparation of soil extracts was derived from the presidedress soil nitrate test (PSNT) (Blume et al., 2015). An exact description of soil extract preparation was given in section 5.2.6. After extraction, a mobile photometer (pHotoFlex® STD, WTW, Weilheim, Germany) was used for the measurements of NH_4^+ and NO_3^- concentration. The applied reagents were WTW reagent NH4-3TC(HR) (measuring range 0.4- 50.0 mg/L NH_4-N, program no. 313), WTW reagent NO3-1 TC (measuring range 0.2 to 30.0 mg/L NO_3-N, program no. 314) and WTW reagent 14542 (measuring range 0.5 to 14.5 mg/L NO_3-N, program no. 17). NO_3^- was additionally measured with a Nitracheck Reflectometer (18.40 by Eijkelkamp Soil & Water, EM Giesbeek, Netherlands, measuring range 5 - 500 ppm (mg/L) NO_3^-). For the measurements of NH_4^+, an adjustment of the pH value of the soil extracts was required. 0.5 molar sodium hydroxide solution (NaOH) (VWR sodium hydroxide pellets extra pure) was added to the extracts until a pH value of ≈7 was reached. The dilution effect was corrected with a corresponding dilution factor. The final results were given in mg NO_3-N/kg and mg NH_4-N/kg.

6.2.7. Meteorological Data and Soil Moisture

The soil moisture and rainfall data were collected in collaboration with and according to Gabiri et al. (2018b). Soil moisture data at the fringe zone were recorded at 10 and 30 cm soil depth every 15 minutes with an automatic soil water station equipped with soil Hydra probe SDI-12 sensors that were connected to a data logger (Stevens Water Monitoring Systems Inc., 2007). Soil moisture in the mid-section and fringe position were monitored with 5TE soil moisture sensors connected to EM50 data loggers (Decagon Devices Inc., 2016a; Decagon Devices Inc., 2016b). The measurements were conducted at 10 and 30 cm depth once per hour. The soil moisture data from each hydrological position

were summarized to a daily time resolution. Soil moisture data gaps were inevitable due to sensor faults. The gaps in the fringe data set were closed by simulating soil moisture using an one-dimensional Hydrus-1D v4.09 software package from Simůnek et al. (2013), as described in Gabiri et al. (2018b). For the daily rainfall measurements, a tipping bucket rain gauge with a 0.2 mm resolution (Stevens Water Monitoring Systems Inc., 2007) was used. The daily temperature data were collected with an automatic weather station located at the Ifakara Health Institute (IHI) (Gabiri et al., 2018b). The described measurement methods are comparable with the methods applied in chapter 5.

6.2.8. Calculation of GWPI and GHGI

The GWP ($kg\,CO_2\,eq\,ha^{-1}yr^{-1}$) was calculated based on the approaches of Li et al. (2004), Nishimura et al. (2011) and Shi et al. (2013). The cumulative CH_4 and N_2O flux results ($kg\,C\,ha^{-1}yr^{-1}$ and $kg\,N\,ha^{-1}yr^{-1}$) were included into the calculation:

$$GWP = \frac{CH_4}{12} \cdot 16 \cdot 25 + \frac{N_2O}{28} \cdot 44 \cdot 298 \tag{24}$$

The calculation of GWPI ($kg\,CO_2\,eq\,kg^{-1}grain\,yield\,yr^{-1}$) was done according to Shang et al. (2010) and Shi et al. (2013). It included the division of GWP ($kg\,CO_2\,eq\,ha^{-1}yr^{-1}$) by grain yield ($kg\,C\,ha^{-1}$).

$$GWPI = \frac{GWP}{grain\,yield} \tag{25}$$

6.2.9. Data Processing and Statistical Analysis

The statistical analyses were implemented using R version 3.3.1 (2016-06-21). GHG results were given as weighted seasonal mean values. Because of missing flux results, some mean values were not based on three replications ($n=3$). Thus, mean values based on less than three replications were assumed to be less reliable and received a smaller weight to lower their impact and to prevent a bias on the total seasonal mean value. The cumulative fluxes for each cropping and fallow season were calculated by multiplying the respective number of days of each period with the corresponding weighted seasonal mean values. The results of the cropping and fallow season of the same year were summed up to annual cumulative flux results. The results were given plus or minus one standard deviation. Levene's test was applied to test the homogeneity of variance and normal distribution. Because the preconditions for an ANOVA were not given, the non-parametric Kruskal-Wallis test ($p < 0.05$) was applied to reveal significant differences between several groups. For the comparison of two groups, the independent Mann-Whitney U Test ($p < 0.05$) was applied. In addition, Spearman's rank correlation test was used to find relationships among N_2O fluxes and mineral nitrogen dynamics.

6.3. Results

Due to extremely high flooding in the center position, the rice yield of this field was lost in both years.

6.3.1. Methane

The CH_4 results showed increased emissions during the two rice-cropping periods of both sampling years from March to June 2015 and from February/March to June/July 2016 (Figure 19). In comparison, the emissions of the two fallow periods were considerably lower in all agricultural treatments and hydrological zones. The weighted seasonal mean values clearly demonstrated seasonal emission differences (Figure A.9 and A.10). The mean values ranged between 0.018 and 0.82 mg CH_4-C $m^{-2}h^{-1}$ during the first cropping period and between -0.43 and 0.83 mg CH_4-C $m^{-2}h^{-1}$ during the second cropping period. The results of the fallow periods ranged from -0.001 to 0.074 mg CH_4-C $m^{-2}h^{-1}$ in the first year and from -0.031 to 0.005 mg CH_4-C $m^{-2}h^{-1}$ in the second year. T1 in the mid-section field was the only treatment which showed a noticeable CH_4 uptake during the cropping period of the second year, all other treatments showed CH_4 emission during the cropping periods and no emissions or slight uptake rates during the fallow periods. In general, the emissions from the mid-section and center field of the second year were higher compared to the first year. Only the fringe field showed lower emissions in the second year than during the first year (Figure A.9, A.10 and 22). The cumulative CH_4 fluxes ranged from 0.22 to 26.33 kg CH_4-C $ha^{-1}yr^{-1}$ in the first year and from -13.55 to 23.46 kg CH_4-C $ha^{-1}yr^{-1}$ in the second year. The Kruskal-Wallis test ($p < 0.05$) revealed significant differences when regarding effects of treatment and position combinations. The emissions from T5 in the fringe position were significantly higher than the emissions from T2 in the mid-section position during the first year. However, no significant differences were observed within the second year (Figure 22). Moreover, the cumulative CH_4 fluxes did not show any significant treatment effects independent from the hydrological position during both years. However, significant hydrological zone effects were detected by the Kruskal-Wallis test for both years (Figure 23). During the first year, the emissions from the fringe position were significantly higher than the emissions from the mid-section position. During the second year, the emissions were again significantly different, but the emissions from the fringe position were lower than from the mid-section field.

6.3.2. Carbon Dioxide

Considerable seasonal variations were observed during both sampling years. The CO_2 emissions showed a clear increase during the cropping period and a decrease within the fallow period (Figure 20). Additional CO_2 emissions peaks were observed during the transition period from dry- to rainy season between December 2015 and January 2016. The weighted seasonal CO_2 mean values varied between 14.08 and 314.40 mg CO_2-C $m^{-2}h^{-1}$

during the first cropping period, while the results from the second cropping period ranged from 19.31 to 159.20 mg CO_2-C $m^{-2}h^{-1}$. In comparison, the emissions from the first and second fallow period ranged from 14.06 to 74.59 mg CO_2-C $m^{-2}h^{-1}$ and from 5.10 to 53.95 mg CO_2-C $m^{-2}h^{-1}$ (Figure A.9 and A.10). The cumulative flux results of the first year were higher compared to the emissions of the second year (Figure 22 and 23). The cumulative CO_2 emissions varied between 1389 and 12747 kg CO_2-C $ha^{-1}yr^{-1}$ during the first year and decreased to a range of 1009 to 7835 kg CO_2-C $ha^{-1}yr^{-1}$ during the second year. The Kruskal-Wallis test showed significantly increased emissions from the renaturation treatment T1 in the fringe position compared to T3 in the center position during the first year. Moreover, significant treatment effects were observed within the mid-section field between T1 and T3. No significant differences were observed during the second year (Figure 22). With regard to treatment effects (Figure 23), the cumulative CO_2 fluxes of both years showed that the highest CO_2 emissions originated from the renaturation treatment T1, which were significantly higher compared to the cropping treatment T3 in the first year. In the second year, the CO_2 emissions from T1 were significantly higher than all cropping treatments, except for the high productive treatment T5. Hydrological zone effects were also observed in both years. The center field showed significantly increased emissions compared to the fringe and mid-section field during the first year, while in the second year emissions from the fringe and mid-section position were significantly different.

6.3.3. Nitrous Oxide

Considerably increased N_2O emissions were observed in the beginning of the first rice-cropping period from March to April 2015 in all hydrological zones and again in the center position between June and July 2015. The emissions decreased to nearly zero during the fallow season 2015 and increased again in the beginning of the rainy season in December 2015, however only in the mid-section and center field. The increase in N_2O emissions from the fringe field occurred with a delay at the beginning of the second cropping period in February/March 2016. The center field showed additional N_2O peaks between June and September 2016 during the beginning of the dry season (Figure 21). In general, there was a clear difference in emission magnitude between cropping (rainy) season and fallow (dry) season. The N_2O emissions during the fallow seasons were much lower than during the cropping seasons (Figure A.9 and Figure A.10). However, the emission strength and duration varied considerably between fields. During the first cropping period the weighted seasonal mean values varied between 0.005 and 0.231 mg N_2O-N $m^{-2}h^{-1}$ and decreased slightly to -0.001 to 0.185 mg N_2O-N $m^{-2}h^{-1}$ during the second cropping period. The weighted seasonal mean values for the first fallow period ranged from -0.00005 to 0.032 mg N_2O-N $m^{-2}h^{-1}$ and from -0.002 to 0.113 mg N_2O-N $m^{-2}h^{-1}$ during the second fallow period. The cumulative N_2O fluxes of the first year varied from 0.377 to 6.215 kg N_2O-N $ha^{-1}yr^{-1}$, the values of the second year from -0.027 to

9.594 kg N_2O-N $ha^{-1}yr^{-1}$. The N_2O emissions of the first year were higher than during the second year at the fringe and mid-section position (see Figure 19, A.9, A.10 and 23), while the emissions from the center field were of the same order of magnitude in both years. The Kruskal Wallis test revealed clear hydrological zone effects with significantly increased N_2O emissions from the center position in both years (Figure 23). In contrast, no significant treatment differences were observed. However, an analysis of treatment and zone interactions showed significantly higher emissions from T3, T5 and T6 in the center field compared to the emissions from the same treatments in the mid-section field (Figure 22).

6.3.4. Nitrate and Ammonium

The NO_3^- and NH_4^+ concentrations in soil showed seasonal differences especially during the first observation year. The first cropping season from March to June 2016 showed clear concentration peaks, while the first fallow season showed a decrease in mineral nitrogen concentration between June and September 2015. The concentrations rose again with the beginning of the short rainy season in October 2015 and peaked again during the second cropping period in March/ April 2016. However, in contrast to the first year, no clear decrease in concentration could be observed during the second fallow period (Figure 25). The NH_4^+ values showed an increase in concentration from the first to the second year. The concentrations of the first cropping period varied from 0.00 to 81.97 mg NH_4^+-N kg^{-1}. and of the first fallow period from 0.00 to 67.84 mg NH_4^+-N kg^{-1}. In the second year, the NH_4^+ concentrations increased considerably. The range during the second cropping period was from 0.00 to 166.66 mg NH_4^+-N kg^{-1}, while the range during the second fallow period was from 0.00 to 201.84 mg NH_4^+-N kg^{-1}. In comparison, the NO_3^- concentrations decreased slightly between the first and the second year. The values of the first cropping period ranged between 0.00 and 115.91 mg NO_3^--N kg^{-1} and the results of the first fallow period ranged from 0.00 to 141.02 mg NO_3^--N kg^{-1}. In the second year, the results of the cropping season varied from 0.00 to 60.81 mg NO_3^--N kg^{-1} and the concentrations of the fallow period varied from 0.00 to 57.16 mg NO_3^--N kg^{-1}.

A reverse pattern regarding the increase and decrease of NO_3^- and NH_4^+ concentrations was observed during the second sampling year. Whenever NH_4^+ concentrations in soil rose, the NO_3^- concentrations decreased. The seasonal concentration dynamics of NO_3^- and NH_4^+ were comparable in all treatments and hydrological zones. The Kruskal Wallis test detected only a few significant differences, for example between the NO_3^- concentrations of T1 and T3 in the mid-section position during the first year. Other significant differences were observed between T1 and T5 as well as between T1 and T6 in the center position during the first year. However, no significant hydrological zone effects were detected during both years. Moreover, no relationships between the N_2O fluxes and the concentration dynamics of NO_3^- and NH_4^+ were found according to the Spearman correlation test (Table

A.4).

6.3.5. GWP and GWPI

The calculated GWP values were equal in magnitude for both years, as shown in Figure 24. It was striking that the center field showed significantly increased GWP values compared to the mid-section field during both years and to the fringe field during the second year. Treatment effects were not observed, except in the fringe position between T1 and T5 during the second year (Figure24, picture 2,3,4). The GWPI could not be calculated for the center position, because the yield of this field was destroyed by extreme flooding during both years. Moreover, no GWPI results were available for T1 because no rice was planted in this treatment. It represented the reestablishment of natural vegetation. In general, the GWPI results of both years showed no significant differences, neither for treatment nor for hydrological zone difference. The only striking feature was an increased GWPI for T2 in both years compared to the other treatments (T3, T5 and T6). However, these increases were not significantly different from each other.

6.4. Discussion

The present study contains a consistent GHG emission time series of one and a half years, which is a long measurement period compared to other studies conducted in SSA, e.g. Jones (2001), Krüger et al. (2013), and Nyamadzawo et al. (2014). Data gaps in the soil moisture measurements for the fringe position, that occurred due to sensor faults, were closed with simulated data (see section 6.2.7). No simulations were done for the mid-section and center position.

6.4.1. Methane

The seasonal increase of CH_4 emissions appeared simultaneous with the beginning of the cropping season. Delayed rice transplanting in the mid-section and center position during the second year resulted in delayed increase of CH_4 emissions from these two fields (figure 19). The findings indicated that rice cropping has a stronger measurable effect on CH_4 emissions than soil moisture dynamics as emissions began to rise with rice transplanting, not with the beginning of the rainy season around November/December. Similar observations were made in the study conducted in the Ugandan inland-valley wetland (see chapter 5). Rice plants are known to provide supporting conditions for CH_4 emissions by vascular transport through aerenchyma which results in increased flux rates (Wagatsuma et al., 1990; Schlesinger and Bernhardt, 2013). After harvest and simultaneous with the beginning of the dry season the CH_4 emissions decreased to nearly zero. The absence of water and availability of oxygen within the soil led to a termination of the CH_4 production because methanogens are obliged to strictly anoxic conditions

(Reddy and DeLaune, 2008). No significant treatment effects were observed in the CH_4 measurements. The renaturation treatment (T1) in the mid-section field was the only treatment that showed CH_4 uptake during the second year. It is supposed that low soil moisture levels in this unbunded treatment as well a the regrowth of natural vegetation led to CH_4 uptake. Observed hydrological zone differences were probably caused by the influence of different soil moisture levels. The CH_4 emissions from the fringe and mid-section field were significantly different in both years. However, the emissions from the fringe field were higher than from the mid-section field during the first year, while they were lower during the second year. Accordingly, the soil moisture levels, especially of the bunded treatments, were higher in the fringe position than in the mid-section field during the first year, while in the second year the soil moisture levels at 10 cm (T3/10) depth were reversed (Figure 26). This can explain the increased CH_4 emissions from the fringe position in the first year as well as the increased emissions from the mid-section field during the second year, as CH_4 production requires high soil moisture levels and anoxic soil conditions. However, the soil moisture measurements in the mid-section and center field showed considerable data gaps, thus this explanatory approach must be considered with a certain degree of uncertainty.

In comparison to other studies conducted in wetlands of SSA, the CH_4 emissions from this study were low. Other findings ranged between -0.07 and 8.22 mg CH_4-C $m^{-2}h^{-1}$ (Jones, 2001; Mitsch et al., 2012; Krüger et al., 2013; Nyamadzawo et al., 2014). Gondwe and Masamba (2013) even reported emissions up to 225 mg CH_4-C $m^{-2}h^{-1}$. Reported cumulative emissions amounted to 712.8 $\pm$ 262.8 kg CH_4-C $ha^{-1}yr^{-1}$ from wetlands (Kim et al., 2016) and 6.25 to 22.9 kg CH_4-C $ha^{-1}yr^{-1}$ from rice paddies (Cao et al., 1998; Kim et al., 2016). The observed annual CH_4 emissions of the study conducted in the Ugandan inland valley wetland (see chapter 5) ranged between -0.001 and 8.31 mg CH_4-C $m^{-2}h^{-1}$, while the cumulative emissions were in a range of -0.03 to 263.02 kg CH_4-C $ha^{-1}yr^{-1}$. In comparison, the cumulative emissions from the inland valley wetland exceeded the emissions from the Kilombero floodplain by a factor of up to 10. The differences in emission were probably caused by the differences in wetland types. Gondwe and Masamba (2013) reported lower mean CH_4 emission rates from floodplains compared to wetland river channels, which might explain the low CH_4 fluxes observed in this study. Moreover, the study conducted in Uganda also covered the investigation of land-use change effects, which is supposed to have a stimulating effect on CH_4 emissions (chapter 5). No land-use change effects were taken into account in the present study, thus an increase of CH_4 fluxes was not expected.

6.4.2. Carbon Dioxide

The CO_2 emissions showed seasonal dynamics that were aligned with the time course of the rainy season. The values increased with the beginning of the rainy season and decreased in the dry season. The influence of rice cropping on CO_2 efflux was found to be less

decisive compared to CH_4, as CO_2 fluxes already increased at the beginning of the rainy season around November/ December, while rice transplanting occurred in March/ April. No treatment effects on CO_2 emissions among the agricultural treatments were found. However, the natural revegetation treatment (T1) showed a considerable impact in form of increased CO_2 emissions in all hydrological zones. T1 simulated the reconversion to a natural wetland without water bunding and with regrowth of natural vegetation. Since soil respiration needs oxic soil conditions, the conditions in this treatment were supposed to be advantageous for CO_2 production. Field flooding in the bunded treatments is known to have an impeding effect on CO_2 emissions, thus net efflux of CO_2 is lower when the paddy field is inundated (Liu et al., 2013; MacCarthy et al., 2018). Differences between hydrological zones were observed in both years, but the findings differed considerably. However, in both years a tendency towards higher CO_2 emissions from the fringe field could be seen which is consistent with the assumption that the fringe field has the shortest flooding period and the most favourable conditions for soil respiration.

The values of this study were in the lower range compared to results of comparable studies which reported findings between 90 and 567.27 mg CO_2-C $m^{-2}h^{-1}$ (Nyamadzawo et al., 2014; MacCarthy et al., 2018). The cumulative CO_2 flux results of this study were lower than values from the literature. For example, Kim et al. (2016) reported cumulative CO_2 fluxes of 17727 kg CO_2-C $ha^{-1}yr^{-1}$ from rice fields in SSA. Moreover, the findings of this study were lower than the emissions reported in chapter 5. The findings presented in chapter 5 ranged from 51.32 to 185.69 mg CO_2-C $m^{-2}h^{-1}$ and from 5946 to 16001 kg CO_2-C $ha^{-1}yr^{-1}$, respectively. The cumulative CO_2 results of the Ugandan inland-valley wetland surpassed the emissions of the Kilombero floodplain on average by 68.4%. Wetland type and length of dry the season are supposed to be decisive for the magnitude of CO_2 emissions.

6.4.3. Nitrous Oxide

The seasonal dynamics of N_2O emissions were in phase with the dynamics of the rainy season. Due to increasing soil moisture levels and temporary flooding, the environmental conditions were advantageous for N_2O producing microorganisms, and N_2O emissions increased considerably during the rainy season (Reddy and DeLaune, 2008; Mitsch and Gosselink, 2015). Fertilizer application has been shown to have an increasing effect on N_2O efflux in many studies, such as Yan et al. (2000), Groffman et al. (2000), and Cooke et al. (2018). However, this effect could not be unambiguously confirmed in this study. The N_2O emissions from the agricultural treatments were not permanently higher than emissions from the revegetation treatment (T1), but only during the second observation year. Moreover, no significant differences among the fertilized (T5, T6) and non-fertilized (T2, T3) rice cropping treatments were observed. The results indicated that high amounts of fertilizer application do not necessarily result in increased N_2O emissions. However, it needs to be considered that the fields were fertilized for the first time in the course of this

study, thus it can be assumed that the nitrogen pools had to fill up first, before nitrogen loss can be observed. The findings confirmed the results of chapter 5. It is assumed that a major proportion of applied fertilizer was probably either immobilized, reduced to N_2 (Groffman et al., 2000), leached (Kimetu et al., 2006; Yeasmin et al., 2012) or was used for plant uptake (Snyder et al., 2009). Hydrological zone effects were observed in form of significantly higher N_2O emissions from the center position and correlated with extremely high and long flooding conditions in this zone during both years. It is assumed that N_2O producing microorganism found optimal anoxic soil conditions in the center field, which lead to the long periods of increased N_2O flux rates until August. As reported in other studies, wetlands with fluctuating water tables emitted more N_2O than permanently flooded wetlands (Hernandez and Mitsch, 2006; Lamers et al., 2007). As the alternating wetting and drying cycle was especially pronounced in the center position, it probably led to the increased N_2O emission compared to the other hydrological zones. Moreover, increased nitrogen concentrations in the floodplain during flooding and rapid NO_3^--N depletion through denitrification can also have an enhancing effect on N_2O emissions, as reported by Forshay and Stanley (2005).

In comparison to other studies, which reported average N_2O fluxes ranging from -0.02 to 3.70 mg N_2O-N m^{-2} h^{-1} (Krüger et al., 2013; Nyamadzawo et al., 2014), the N_2O emissions from the Ifakara floodplain were relatively low. However, the observed cumulative fluxes were higher than the results reported by Kim et al. (2016) with 1.27 $\pm$ 0.95 kg N_2O-N $ha^{-1}yr^{-1}$ from wetlands and floodplains and 0.12 kg N_2O-N $ha^{-1}yr^{-1}$ from rice fields. The results of the Ugandan inland valley wetland were higher by 47.2% compared to the findings of this study (see chapter 5). They ranged from 0.16 to 9.54 kg N_2O-N $ha^{-1}yr^{-1}$. It is assumed that GHG producing microorganism found more favourable environmental conditions in the Ugandan inland valley wetland due to more frequently high soil moisture levels and regular precipitation events, compared to the Tanzanian floodplain which was characterized by an annually appearing long dry period with a significant decrease of soil moisture levels.

6.4.4. Nitrate and Ammonium

The observed NH_4^+ and NO_3^- results neither showed a clear relationship between mineral nitrogen concentration in soil and fertilizer application, nor a correlation between mineral nitrogen concentrations and N_2O emissions. The nitrogen cycle involves different pathways and transformation steps which influence the relationships among nitrogen input, mineral nitrogen concentration in soil and gaseous output and impair possible correlations. For example, NO_3^- can be lost through runoff and leaching because NO_3^- is highly mobile (Kimetu et al., 2006; Yeasmin et al., 2012). NO_3^- is highly available for plants and easily taken up for plant growth (Snyder et al., 2009). Moreover, NH_4^+ concentration and N_2O fluxes are not necessarily interlinked, because nitrification activity depends on redox

potential rather than on NH_4^+ concentration (Yan et al., 2000). N_2O can be reduced to N_2 during denitrification if NO_3^- availability is limited (Groffman et al., 2000; Hernandez and Mitsch, 2006; Cooke et al., 2018). For instance, complete denitrification of NO_3^- to N_2 in constructed wetlands has been observed by Gale et al. (1993) and Picek et al. (2007). As the mineral nitrogen concentrations of this study were low in comparison to other studies (e.g. Nyamadzawo et al. (2014)), a reduction of N_2O to N_2 at the experimental site is probable.

6.4.5. GWP and GWPI

Effective management practices should aim to achieve a decent trade-off between food production and GHG emissions (Cassman, 1999; Snyder et al., 2009). Optimized exploitation of yield potentials can support the minimization of net GHG emissions from agricultural systems (Adviento-Borbe et al., 2007). The findings of this study demonstrated that intensification of food production does not necessarily have a negative influence on yield-based GHG emissions. The GWPs from fertilized treatments (T5 and T6) were not significantly increased compared to non-fertilized treatments (T2 and T3). The GWPI results even showed that the farmers practice (T2) has the poorest outcome due to disproportionately low rice yields and comparatively high GHG emissions. In contrast, the treatment with the highest productivity (T5) had the most advantageous GWPI. Accordingly, high yields have the ability to compensate increased GHG emissions. However, a net mitigation of GHG emissions is subject to the condition that natural wetland areas are spared from food production (Robertson, 2000; Snyder et al., 2009).

The comparison of hydrological zone effects clearly showed an increased GWP of the center field, which was mainly due to the high N_2O emissions. Thus, N_2O was a relevant contributor to GWP, as already reported by Shi et al. (2013). The range of observed GWP and GWPI values were comparable in magnitude with the findings of Shi et al. (2013) who reported results from 411.3 to 981.7 $kg\,CO_2\,eq\,ha^{-1}\,yr^{-1}$ and from 0.044 to 0.061 $kg\,CO_2\,eq\,kg^{-1}\,yield\,yr^{-1}$, respectively. The results of Ma et al. (2013) were higher with reported GWP results ranging from 4720 to 12150 $kg\,CO_2\,eq\,ha^{-1}\,yr^{-1}$ and GWPI results from 0.51 to 0.72 $kg\,CO_2\,eq\,kg^{-1}\,yield\,yr^{-1}$. The values reported of the Ugandan inland-valley wetland (chapter 5) were also higher. They ranged from 208 to 10095 $kg\,CO_2eq\,ha^{-1}\,yr^{-1}$ and 0.08 and 2.19 $kg\,CO_2eq\,kg^{-1}grain\,yield\,yr^{-1}$. More regularly occurring precipitation events and high soil moisture levels with only short periods of decreased soil moisture are possible reasons for the higher GHG emissions and correspondingly higher GWP and GWPI results from the Ugandan inland valley wetland, compared to the test site in Tanzania. Moreover, the Ugandan test site was influenced by land-use change which is known to have an increasing effect on GHG emission, for example because of drainage which can cause the exposure of great soil organic matter stocks to oxygen (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger

and Bernhardt, 2013), or due to additional nitrogen input through fertilization (Kroeze et al., 1999; Reddy and DeLaune, 2008).

In summary, the GWP and GWPI results proved to be valuable indicators for the mitigation potential of different cropping treatments regarding GHG emissions. This study indicated that the intensification of food production can be recommended for sustainable wetland management, provided that natural areas are spared from land-use change.

6.5. Summary

The GHG emissions from the Kilombero floodplain were relatively low compared to emissions from other wetlands in SSA. Especially, the CH_4 fluxes were significantly lower by a factor of up to 10 compared to the emissions from the Ugandan inland valley wetland reported in chapter 5. However, increased nitrogen concentrations in flooded floodplains and rapid NO_3^--N depletion by denitrification can result in high N_2O emissions, as observed in the hydrological center zone of this study (Forshay and Stanley, 2005). The regrowth of natural vegetation in unbunded wetland areas did not result in generally lower GWP results. In fact, the revegetation treatment (T1) indicated similarly high GWP results compared to the agricultural treatments during the first year, but the results decreased during the second year and even included CH_4 uptake. Revegetation of wetlands is considered to have a mitigating effect on GHG emission, though more long-term studies are required. Fertilizer application did not show an increasing effect on N_2O emissions. Thereby, the findings confirmed the results of chapter 5. As NH_4^+ and NO_3^- concentrations did not show any correlation with N_2O emissions, it is assumed that a major proportion of fertilizer was either immobilized in microbial biomass and organic soil substance, consumed by plant uptake, leached or reduced to N_2 (Groffman et al., 2000; Kimetu et al., 2006; Snyder et al., 2009; Yeasmin et al., 2012). Intensification of food production does not result in negative influences on yield-based GHG fluxes. High production treatments with nitrogen fertilizer application were associated with more advantageous GWPI results than the local farmer's practise without fertilizer application. In conclusion, sustainable wetland management can include the intensification of food production on condition that natural wetland areas are spared from land-use change. Therefore, the outcomes of this study confirmed the findings of chapter 5 and Snyder et al. (2009).

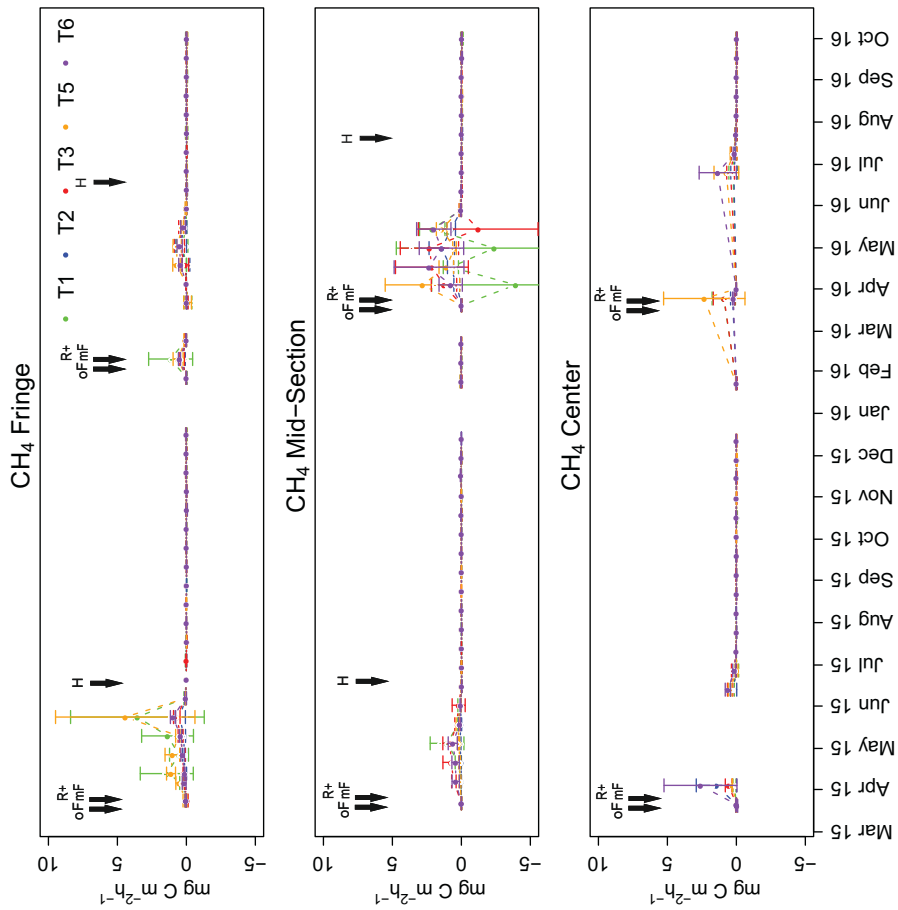

Figure 19: Seasonal variation of CH_4 fluxes, Ifakara, with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF) and harvest (H).

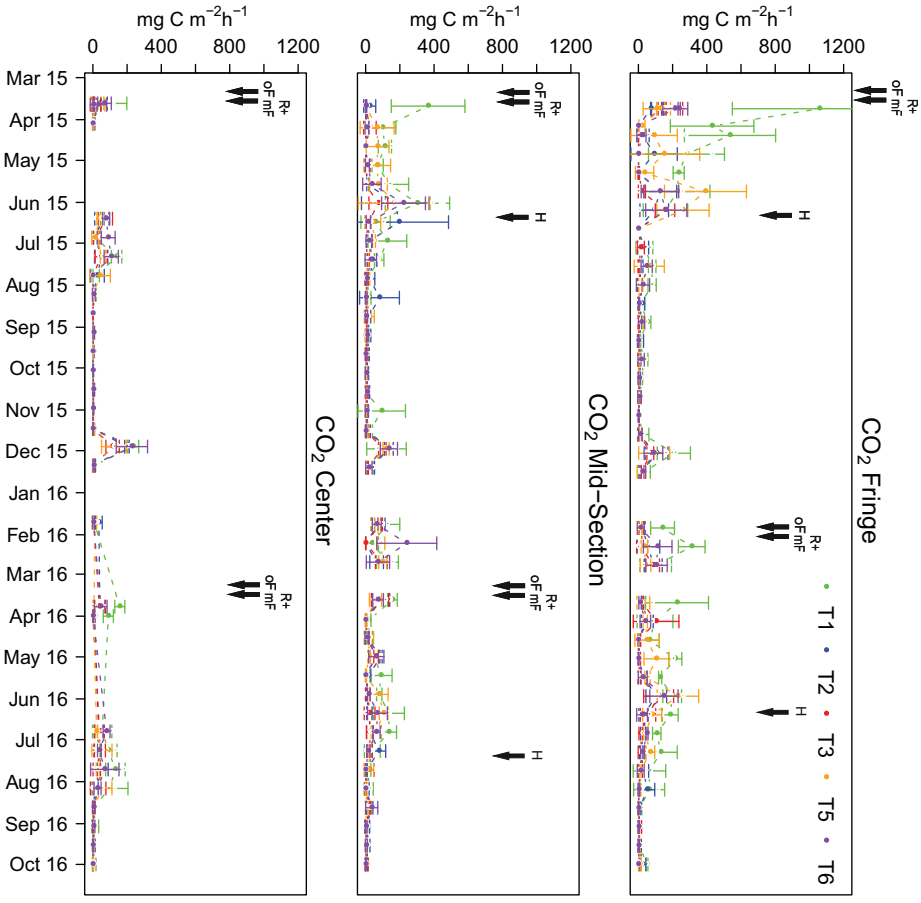

Figure 20: Seasonal variation of CO_2 fluxes, Ifakara, with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF) and harvest (H).

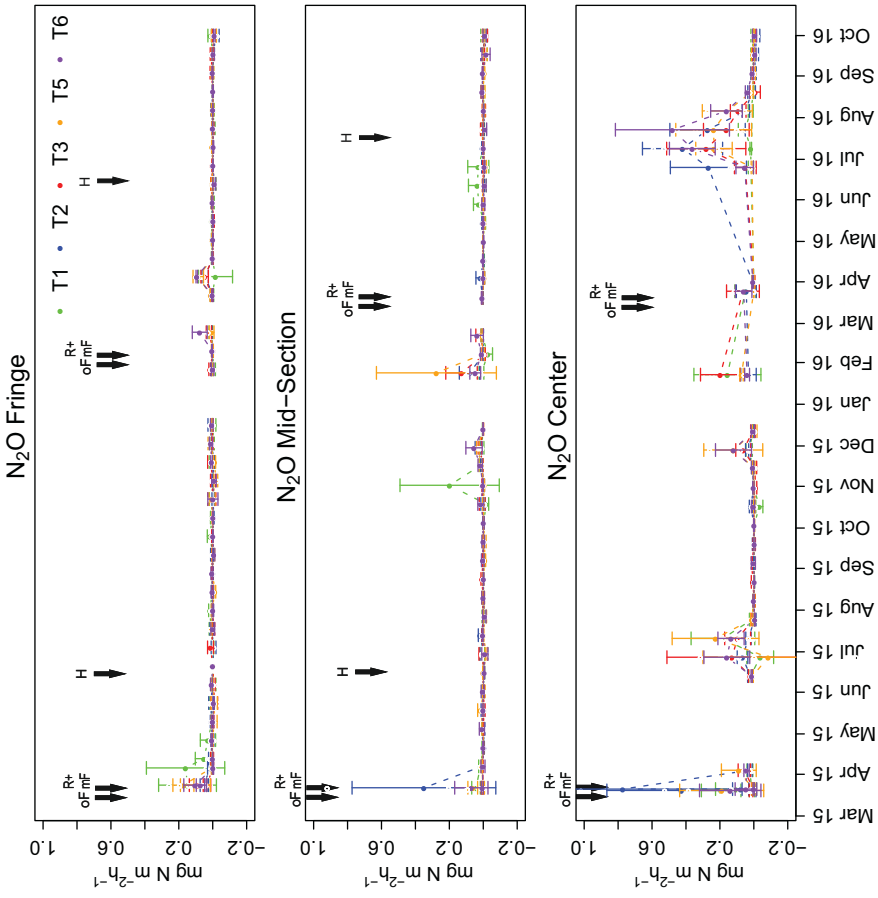

Figure 21: Seasonal variation of N_2O fluxes, Ifakara, with indication of rice planting (R), organic fertilizer application (oF), mineral fertilizer application (mF) and harvest (H).

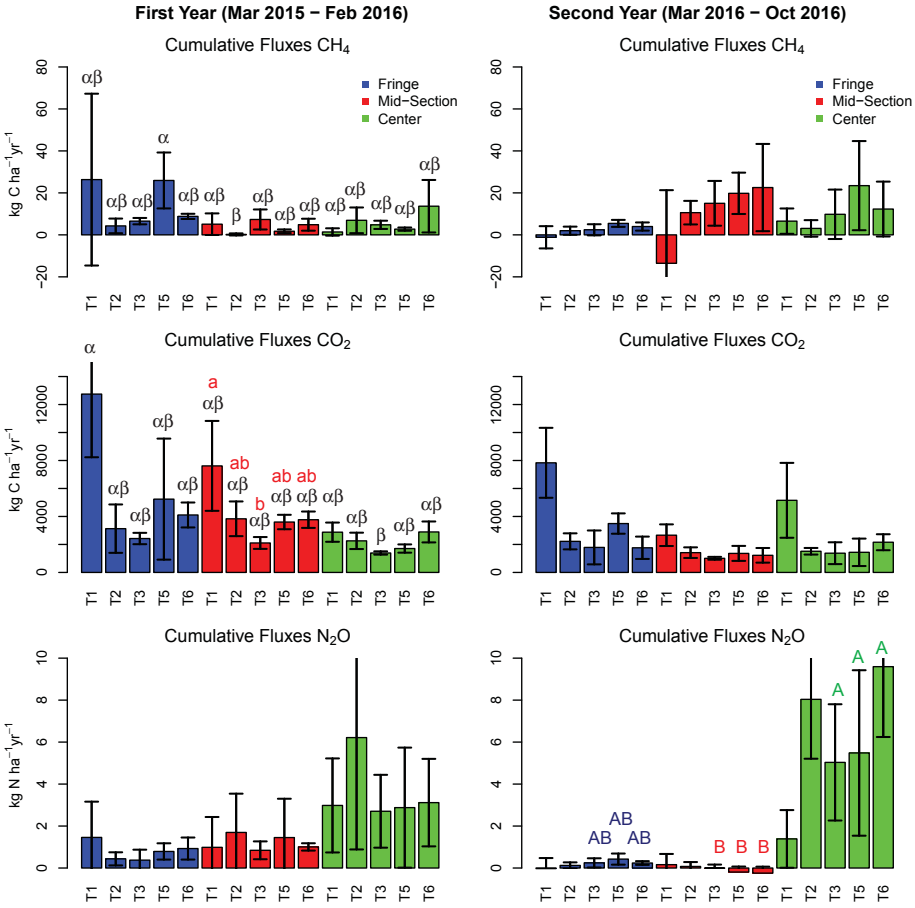

Figure 22: Cumulative GHG fluxes ± SD from the Ifakara field trial in Tanzania. The letters indicate significant differences at p = 0.05 by the Kruskal-Wallis Test:
- α, β: significant differences for treatment and hydrological zone interactions
- a, b: significant differences for treatment effects within one hydrological zone
- A, B: significant differences for hydrological zone effects within one treatment
T1: renaturation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application.

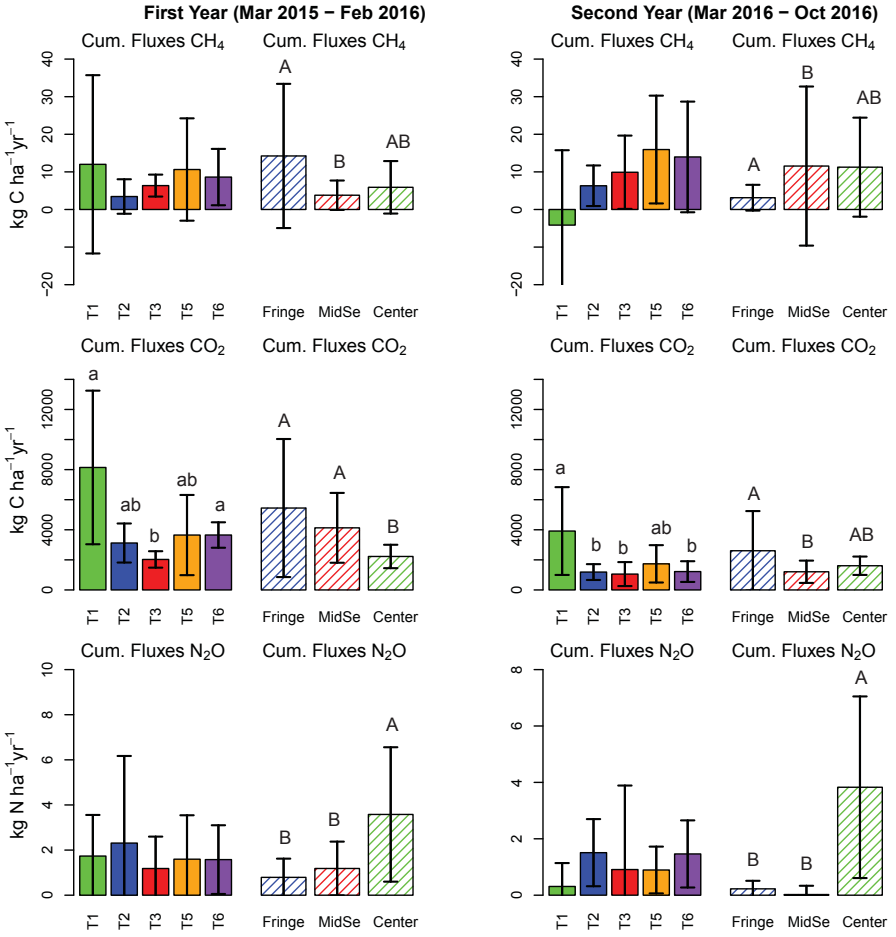

Figure 23: Treatment and hydrological zone summary for cumulative GHG fluxes ± SD from the Ifakara field trial in Tanzania.
The letters indicate significant differences at p <0.05 by the Kruskal-Wallis Test:
- a, b: significant differences for treatment effects
- A, B: significant differences for hydrological zone effects
T1: renaturation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application

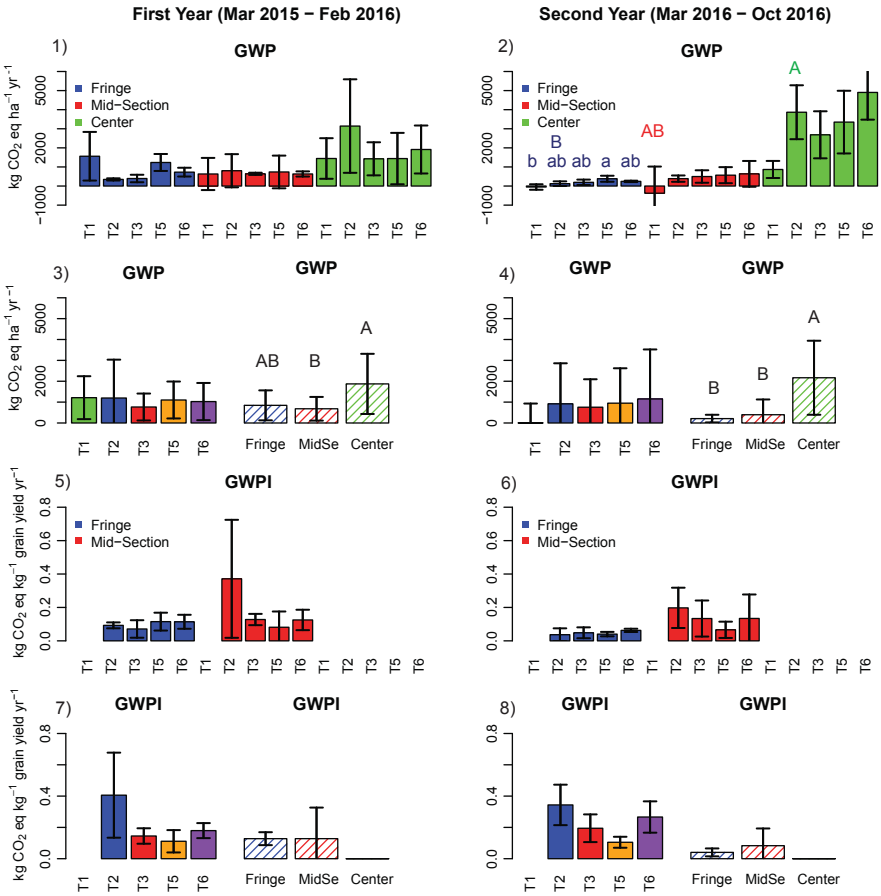

Figure 24: GWP and GWPI results from the Ifakara field trial in Tanzania.
Picture 1, 2, 5 and 6 show all individual treatment-zone combinations. Picture 3, 4, 7 and 8 show summaries for each treatment and zone.
Letters indicate significant differences at p = 0.05 by the Kruskal-Wallis Test:
- α, β: significant differences for treatment and hydrological zone interactions
- a, b: significant differences for treatment effects
- A, B: significant differences for hydrological zone effects
GWP: global warming potential, GWPI: global warming potential index, T1: renaturation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application

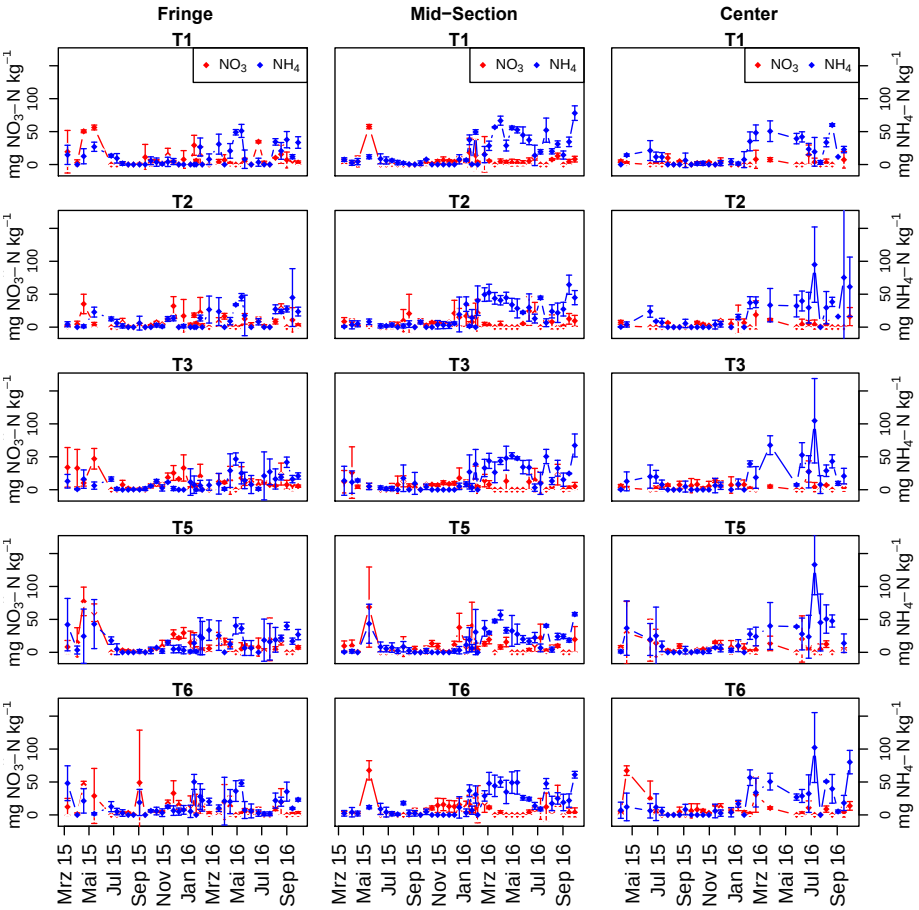

Figure 25: Nitrate and ammonium concentrations in soil, Ifakara.
T1: renaturation, T2: no mineral N fertilization or bunding, T3: bunding without N fertilization, T5: intensive crop management with 120 kg urea-N/ha + 60 kg PK, T6: green manure application

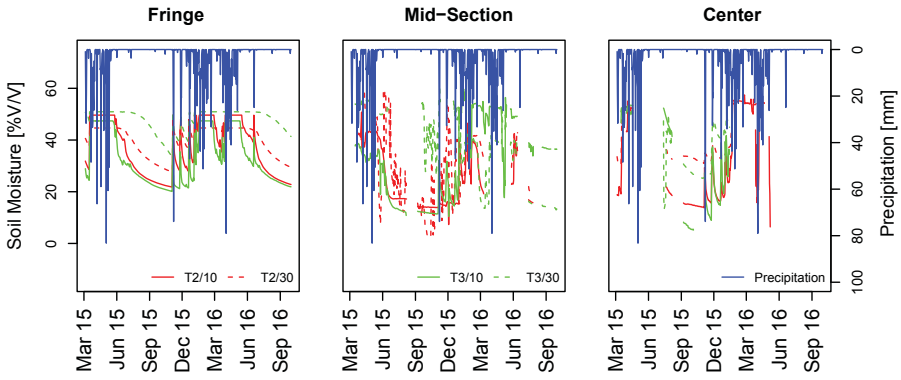

Figure 26: Rainfall and soil moisture data, Ifakara.
T2/10: Treatment 2, 10 cm depth. T2/30: Treatment 2, 30 cm depth. T3/10: Treatment 3, 10 cm depth. T3/30: Treatment 3, 30 cm depth.

7. Final Conclusions

The following section discusses the main research objectives of this study from chapter 3.

7.1. Development of Standardized Quality Criteria for GHG Data from Static Chamber Measurements

There is a lack of systematic quality assurance regarding GHG data sets collected with static chambers. A data quality management system (DQMS) with standardised specifications for data quality control is needed. The most widely used criterion for quality control is R^2 in linear flux estimation. However, the application of R^2 as only criterion can lead to the rejection of large parts of the data set and thus to a decline concerning its informative value. The developed DQMS aimed to provide a standardised system that improves data validity without accepting a considerable loss of data. The DQMS was based on eight control steps which enabled the automatic identification and rejection of single outliers, without rejecting total flux sets. The system is recommended for all kinds of GHG measurements with static chambers, especially for GHG data collected from areas with extreme physical conditions and taken under difficult sampling conditions. The DQMS improved the acceptance rate of the tested data sets by up to 37.2 pp for CH_4, 11.7 pp for CO_2 and 48.3 pp for N_2O. Moreover, it prevented the risk for under- or overestimation of the calculated GHG emissions values.

7.2. Assessment of GHG Emissions with Consideration of Contrasting Wetland Types

Two wetlands types were selected as test sites for the GHG measurements of this study. One test site was located in an inland valley wetland in Uganda ($00°31'30"N$, $32°36'54"E$), while the other one was situated in the lowland floodplain of the Kilombero river in Tanzania ($-8°07'59.99"S$, $36°40'59.99"E$). Both test sites were characterized by paddy cultivation. The GHG measurements were conducted for two cropping seasons, including the fallow- and transition periods, to achieve long-term monitoring data. Both wetland types differed in their environmental properties. The inland valley wetland showed an altitude of 1100 m above sea level with annual temperatures between 22.1 and 24.0°C. The climate was sub-humid with two dry- and two rainy seasons per year and an average precipitation of 1350 mm. The soils were characterized by Gleysols (Nsubuga et al., 2011; Leemhuis et al., 2016; Gabiri et al., 2018a). The floodplain had a low altitude of 250 to 300 m above sea level and a mean annual temperature of 23 to 25.0° C. The sub-humid tropical climate featured two rainfall periods with flood peaks usually occurring during the long rainy season between March and May. The annual rainfall was 1200 to 1400 mm. The soil types were dominated by Fluvisols, fluvic Arenosols, Planosols and Gleysols (Beck, 1964; Mombo et al., 2011; Koutsouris et al., 2015; Gabiri et al., 2018b). Due to the flat

topography of the Kilombero floodplain, the three hydrologically different test fields of the floodplain were distributed with a linear distance of 2-3 km between each field, whereas the fields in the inland valley wetland had a distance of only 30-50 m. The floodwater level in the Kilombero floodplain was much higher that in the inland valley wetland. According to information from residents of Ifakara, the water table in the hydrological center zone could reach up to 2 m.

The GHG measurements showed that the cumulative gas fluxes of all observed GHGs from the Kilombero floodplain were smaller than from the inland valley wetland (table 6, page 109). Especially the CH_4 fluxes showed significantly lower emissions of 86.0%. The cumulative CO_2 emissions were on average 68.4% lower, while the cumulative N_2O emissions showed a difference of 47.2% between the two wetland types. It is assumed that the differences in precipitation and soil moisture levels between the Ugandan and Tanzanian test site had a strong effect on the GHG production and emission. The climate in Uganda featured two dry and two rainy seasons per year, with a long rainy season occurring from March to May and a short rainy season occurring from September to November (Nsubuga et al., 2011). However, the observations of this study showed that some rainfall events also occurred during the dry seasons from June to August and from December to February. Consequently, the soil moisture observation showed fluctuating levels with only short periods of decreased soil moisture levels (see Figure 18, page 82). It is assumed that the regularly occurring rainfall events and the frequently high soil moisture levels provided favourable environmental conditions for GHG producing microorganism which resulted in increased GHG fluxes from the Ugandan inland valley wetland. In comparison, the climate in the Tanzanian floodplain showed a very long dry period without precipitation between June and December and a clear decrease of soil moisture levels, which was especially well observed in the fringe position (see Figure 26, page 106). As microorganism find optimal conditions for GHG production under dry soil conditions, the drop in soil moisture levels for a long time period explains the lower GHG fluxes from the Tanzanian floodplain.

Moreover, it needs to be considered that the inland valley wetland in Uganda was influenced by land-use change which clearly affected soil nutrient availability and GHG emission. Drainage is known to lead to an exposure of soil organic matter stocks to oxygen (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). Additional nitrogen input in form of fertilization can lead to an increase of N_2O emissions (Kroeze et al., 1999; Reddy and DeLaune, 2008). Therefore, it is assumed that land-use change led to increased GHG fluxes from the Ugandan test site. A more detailed description and discussion about observed land-use change effects on GHG emissions in the course of this study can be found in section 7.3.

In summary, the Ugandan inland valley wetland showed considerably higher emissions regarding all observed GHGs emissions compared to the Tanzanian floodplain, probably

Table 6: Comparison of cumulative GHG flux ranges from Uganda and Tanzania

GHG	Kilombero Floodplain (Tanzania)	Inland Valley Wetland (Uganda)
CH_4 [kg CH_4-C $ha^{-1}yr^{-1}$]	-13.55 to 26.33	-0.03 to 263.02
CO_2 [kg CO_2-C $ha^{-1}yr^{-1}$]	1009 to 12747	5946 to 16001
N_2O [kg N_2O-N $ha^{-1}yr^{-1}$]	-0.027 to 9.594	0.16 to 9.54

caused by frequent rainfall events and accordingly high soil moisture levels throughout the whole measuring period. The climate and soil conditions of the Tanzanian floodplain were less favourable for GHG producing microorganism, because of an annually occurring long dry season with clearly decreased soil moisture levels. Gondwe and Masamba (2013) reported that floodplains feature lower GHG emission rates than wetland river channels. The results of this study indicated that floodplains probably also emitted lower GHG flux rates than narrow inland valley wetlands. However, the findings of this study are only valid for the two observed wetlands in Uganda and Tanzania, and related to the specific climatic conditions. For a general conclusion concerning the influence of wetland types on GHG emissions, further research in different wetland types of East-Africa is required.

7.3. Assessment of GHG Emissions with Consideration of Different Types of Land Use

A main focus of this study was on the investigation of land-use change effects on GHG emissions. Therefore, an agronomic field trial was established in an Ugandan inland valley wetland which was converted from a (semi-) natural wetland to a paddy field. Land-use change is known to disturb the natural balance of carbon and nitrogen cycling. Drainage increases the exposure of organic matter stocks to oxygen and results in an increase of CO_2 emissions (Armentano and Menges, 1986; Reddy and DeLaune, 2008; Schlesinger and Bernhardt, 2013). Moreover, CH_4 fluxes are expected to decrease because of the change from anoxic to oxic soil conditions (Mitsch and Gosselink, 2015). Fertilization can lead to increased N_2O emissions. After soil N pools have filled up, the additional nitrogen added to the wetland ecosystem can be released in form of gaseous emissions (Kroeze et al., 1999; Reddy and DeLaune, 2008). However, the observed emission changes showed opposite effects. CO_2 emissions decreased, while CH_4 emissions increased. It is assumed that soil water levels increased after the conversion into a paddy field. The soil conditions were more favourable for CH_4 producing anaerobic micro-organism, rather than for soil respiration. Moreover, rice plants have an aerencyma which is known to provide an extra pathway for CH_4 efflux (Wagatsuma et al., 1990). The N_2O fluxes from the Ugandan field experiment increased as expected due to the additional nitrogen input. In general, emission differences between the (semi-) natural vegetation treatment and the

agricultural treatments were significant, in contrast to differences among the particular agricultural treatments. It was concluded that the impact of land-use change on GHG emissions is greater than the impacts of agricultural treatments (see also section 7.5). The field trial in the Kilombero floodplain in Tanzania took the regrowth of natural vegetation in unbunded wetland areas into consideration. However, as the results varied widely, no general conclusion was possible. The observations indicated a decreasing trend for CH_4 and N_2O during the second year, implying a decrease of GHG emissions after revegetation. More long-term studies are required to verify this observation. In summary, the increasing effect of land-use change on GHG emissions is significant, while the impact of renaturation is less distinct considering short-term periods.

7.4. Assessment of GHG Emissions with Consideration of Different Hydrological Positions in the Wetland

For the assessment of hydrological zone effects, the test sites in both wetland types were divided into three fields situated in different hydrological zones which differed in their lengths of annual flooding. The fields were named according to their topographic elevation in the wetland area with fringe, mid-section and center field (see figure 10, page 58). The observed GHG emissions from the hydrological zones in both countries showed high levels of heterogeneity for each measurement year. Although some significant zone differences were observed, no general emission patterns with regard to hydrological zone influences were found. The CO_2 fluxes from the fringe position in both wetland types were slightly increased, probably due to generally lower soil moisture levels, but statistical significances were not consistent. Moreover, a tendency towards increased N_2O emissions from the center position was observed, which can be explained by increased nitrogen concentrations in flooded areas close to the wetland river channel and high denitrification rates (Forshay and Stanley, 2005). The statistical differences were significant for the Kilombero floodplain in Tanzania, but not for the Namulonge inland-valley wetland in Uganda. The CH_4 results did not indicate noticeable emission differences from a specific hydrological zone. It is assumed that the influence of the environmental factors that cause hydrological zone effects, such as length of annual flooding and soil moisture levels, were not sufficiently differentiated among the zones to have a clear effect on the GHG emissions. The fringe, mid-section and center positions were selected with the assumption that the annual flooding periods vary from temporarily flooded in the fringe field to short term seasonally flooded in the mid-section field to long term seasonally flooded in the center field (see Figure 10, page 58). However, field observations showed that the lengths of flooding periods only differed by a few days or weeks. The associated soil moisture levels did not show distinct differences among the zones in both countries. The only significant difference was found in the fringe field of the Ugandan test site, which was significantly lower compared to the mid-section and center field (Gabiri et al., 2018a). The test-sites in Tanzania did not show clear soil moisture differences among the hydrological zones.

Correspondingly, the differences in soil oxygen availability between the zones is assumed to be too small to show a considerable zone effect on the production of GHGs. In summary, hydrological zone management has no substantial relevance for the development of wetland management recommendations with respect to GHG emission mitigation.

7.5. Evaluation of Several Agricultural Treatment Methods with Respect to Environmental Sustainability

The comparison of agricultural treatment differences among the individual cropping methods showed only few significant differences, but no general pattern of increased or decreased GHG emission from a particular treatment. The cropping treatments with fertilizer application did not feature continously high N_2O emissions. Increased N_2O emission were observed in fertilized as well as unfertilized treatments. It is assumed that the amount of applied fertilizers was either immobilized in form of soil organic matter or microbial mass, leached, consumed for plant uptake or reduced to N_2 (Groffman et al., 2000; Kimetu et al., 2006; Snyder et al., 2009; Yeasmin et al., 2012). Therefore, no correlation between NO_3^- and NH_4^+ concentration in soil and N_2O emissions was observed.

In general, soil-atmosphere gas fluxes can be utilized as sustainability indicators for the assessment of cropping methods. The findings of both test-sites showed that intensive cropping treatments with fertilizer application and high yields did not feature increased GWP and GWPI results compared to local farmer's practise without fertilization. In conclusion, the intensification of rice production does not affect yield-based GHG emissions in a negative way. Bunded high production treatments with mineral fertilizer or green manure application provide a possibility to increase food production and mitigate the total GWP. Nevertheless, land spared from land-use change and agricultural production provides the highest GHG mitigation potential (Snyder et al., 2009). Sustainable wetland management needs to consider a combination of high-yielding cropping treatments and protected natural wetland areas to achieve the best possible trade-off between food production and GHG emissions.

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A. Appendix

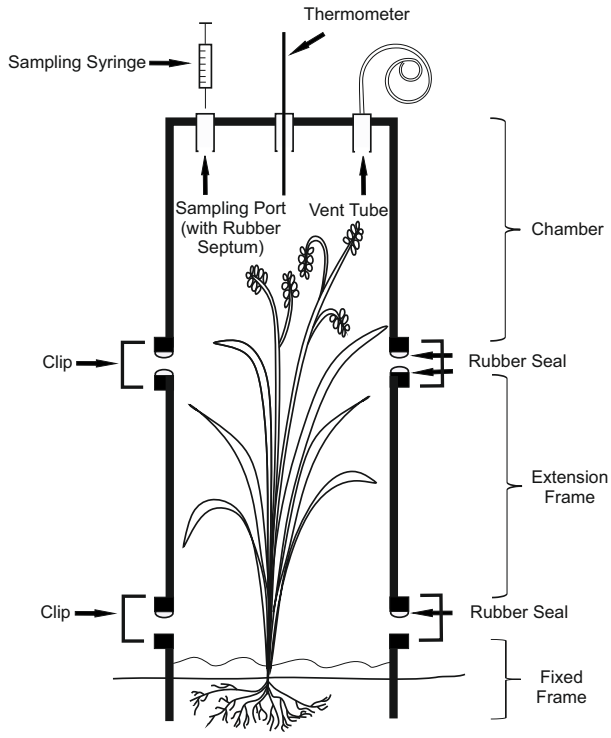

Figure A.1: Cross section of a static chamber construction with extension frame and rice plant included during the measurement.

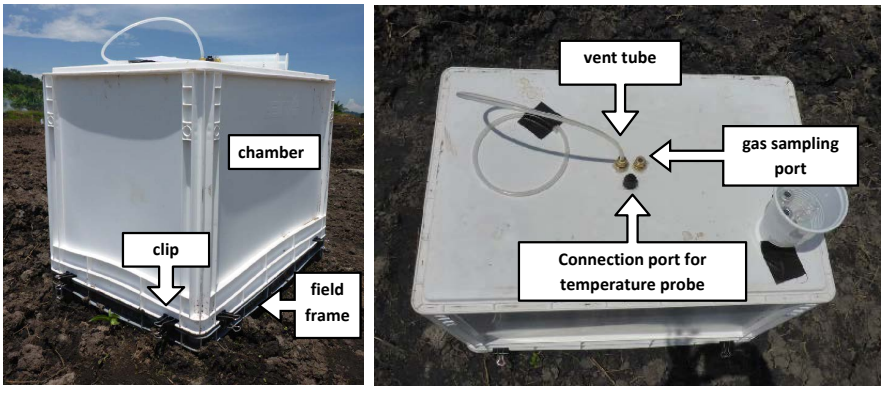

Figure A.2: GHG measuring chamber in the field (side view and top view)

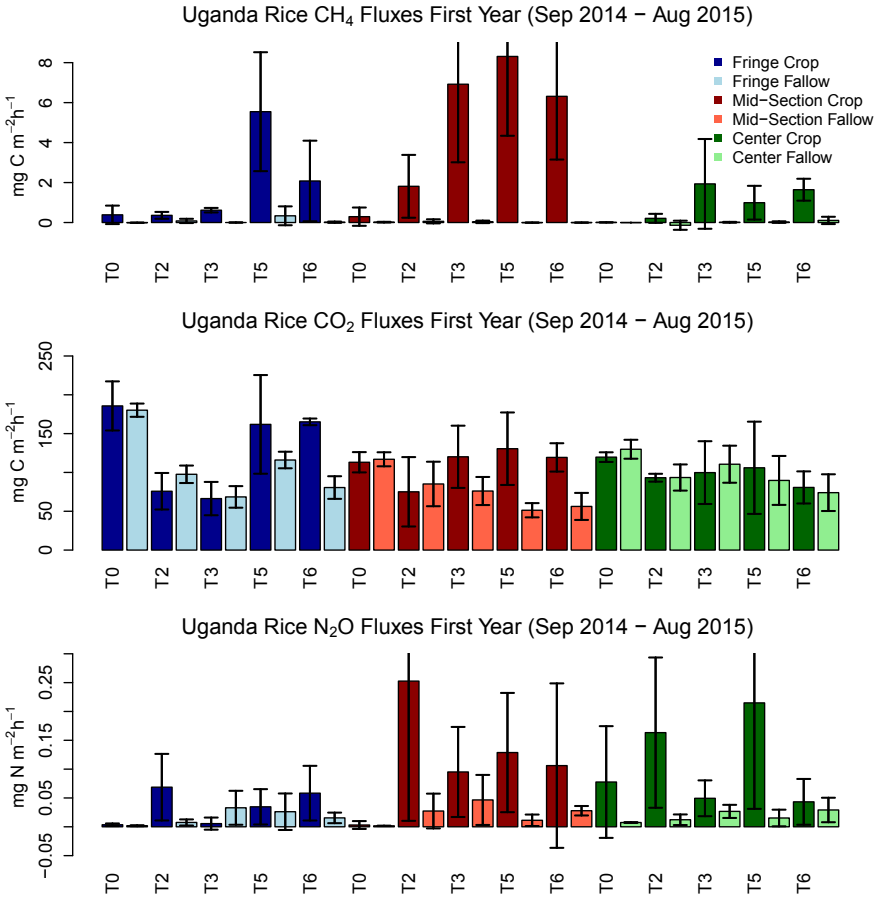

Figure A.3: Weighted seasonal mean values of the first year, Namulonge

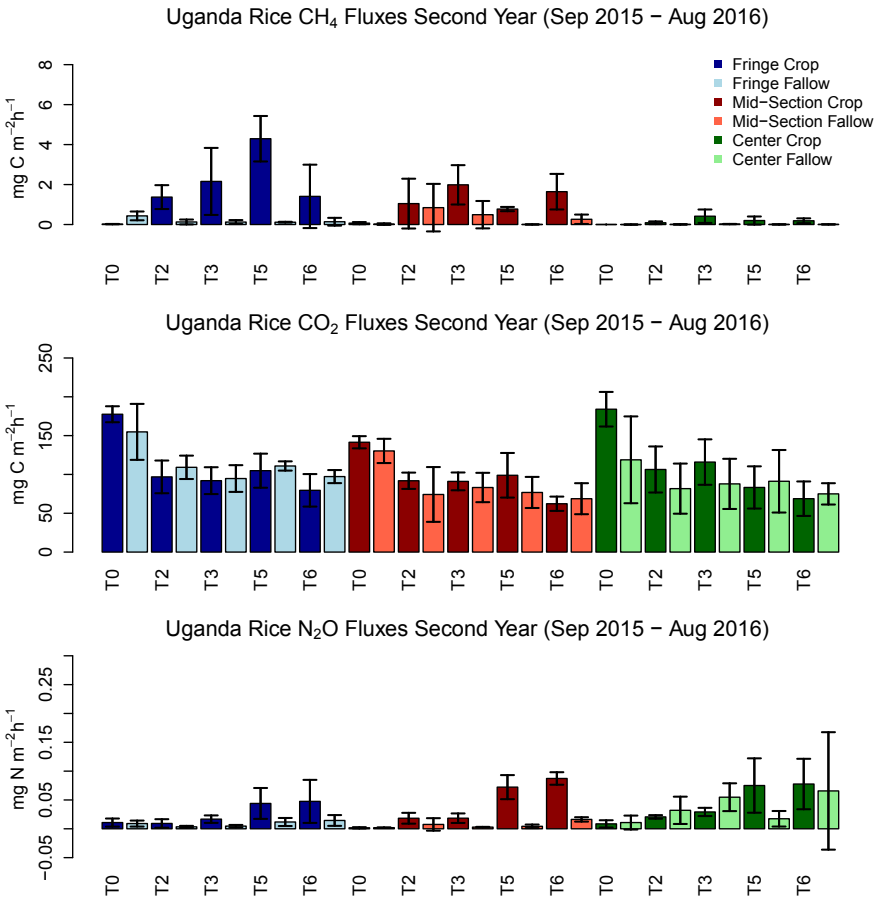

Figure A.4: Weighted seasonal mean values of the second year, Namulonge

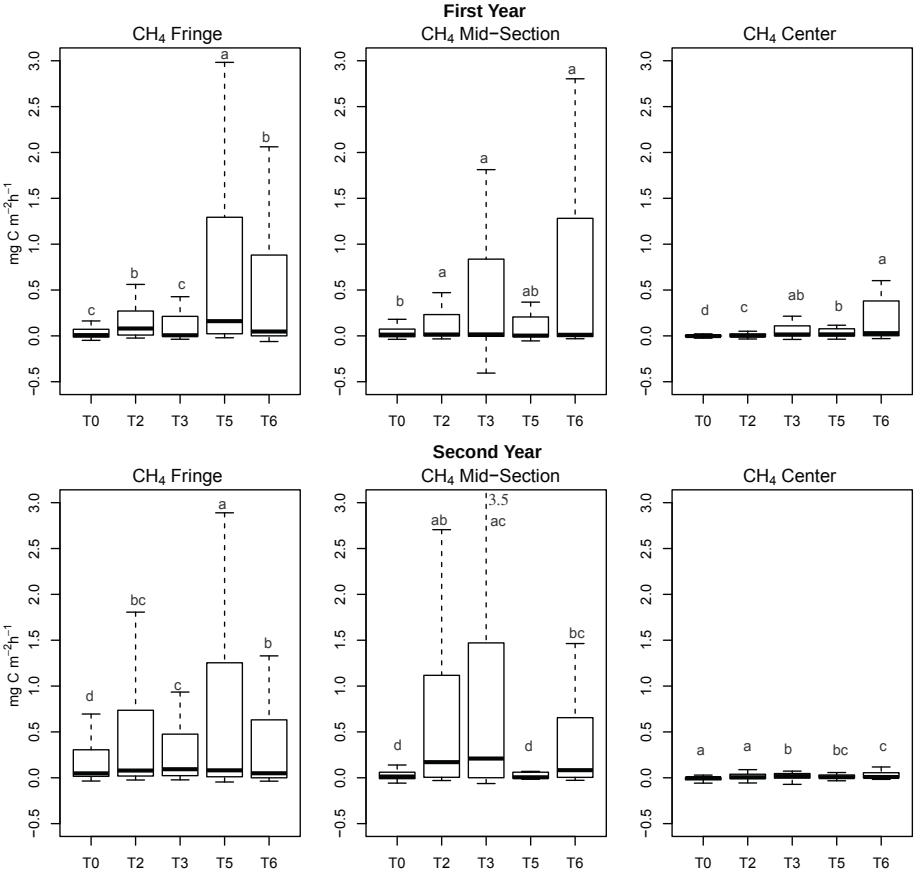

Figure A.5: Boxplots CH_4, Namulonge

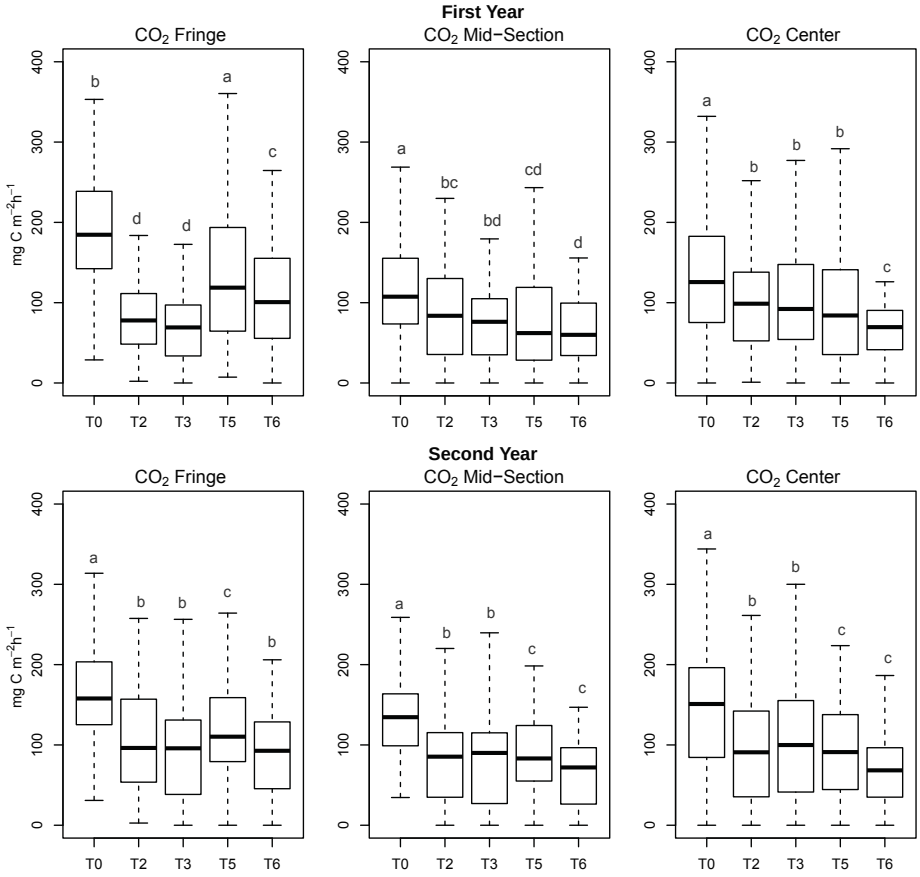

Figure A.6: Boxplots CO_2, Namulonge

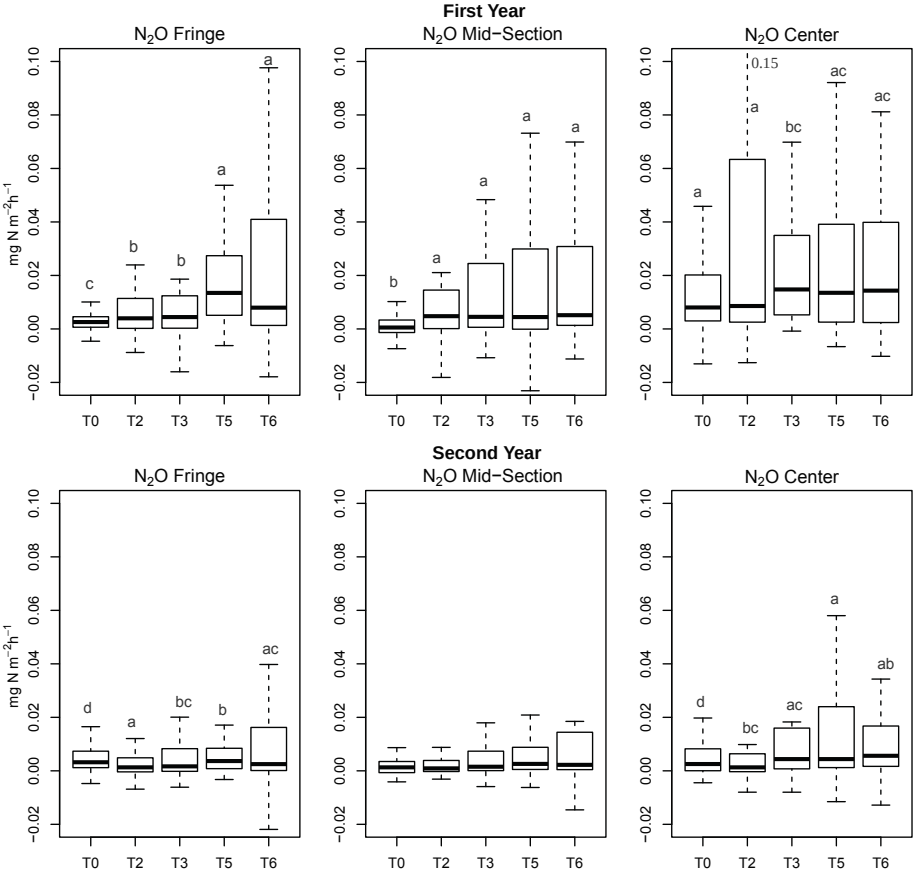

Figure A.7: Boxplots N_2O, Namulonge

Table A.1: Range of Uganda rice data set

	GHG	Zone	T0		T2		T3		T5		T6	
			min	max	min	max	min	max	min	max	min	max
1st Year	CH_4	Fringe	-0.05	5.23	-0.02	1.39	-0.04	3.71	-0.02	36.34	-0.06	18.01
	CH_4	Mid-Sec.	-0.03	3.46	-0.03	17.87	-0.41	46.86	-0.05	39.27	-0.03	74.72
	CH_4	Center	-0.05	0.06	-5.41	1.51	-0.79	20.15	-0.04	15.49	-0.03	10.32
	CO_2	Fringe	28.76	476	2.14	410	0.00	249	7.26	445	0.00	475
	CO_2	Mid-Sec.	0.00	345	0.00	475	0.00	747	0.00	613	0.00	604
	CO_2	Center	0.00	332	0.95	337	0.00	504	0.00	536	0.00	317
	N_2O	Fringe	-0.03	0.04	-0.03	0.80	-0.04	0.74	-0.01	0.56	-0.02	0.42
	N_2O	Mid-Sec.	-0.02	0.06	-0.02	3.30	-0.04	0.10	-0.02	1.21	-0.01	2.45
	N_2O	Center	-0.01	0.56	-0.01	1.15	0.00	0.47	0.00	0.91	-0.01	0.46
2nd Year	CH_4	Fringe	-0.04	4.21	-0.02	4.50	-0.02	7.52	-0.05	15.37	-0.04	7.79
	CH_4	Mid-Sec.	-0.06	0.69	-0.03	4.96	-0.06	5.96	-0.02	2.46	-0.03	4.98
	CH_4	Center	-0.08	0.43	-0.21	0.32	-0.07	2.73	-0.03	1.08	-0.02	1.13
	CO_2	Fringe	5.90	349	2.71	257	0.00	256	0.00	264	0.00	261
	CO_2	Mid-Sec.	0.00	283	0.00	269	0.00	280	0.00	198	0.00	146
	CO_2	Center	0.00	369	0.00	261	0.00	300	0.00	223	0.00	213
	N_2O	Fringe	-0.00	0.24	-0.01	0.05	-0.01	0.07	0.00	0.32	-0.02	0.52
	N_2O	Mid-Sec.	-0.02	0.02	0.00	0.27	-0.01	0.12	-0.01	0.46	-0.01	0.39
	N_2O	Center	-0.03	0.38	-0.01	0.69	-0.01	0.48	-0.04	0.59	-0.01	1.67

Table A.2: Correlation between mineral nitrogen concentration and N_2O flux in Namulonge

		total N		NO_3^-		NH_4^+	
Zone	Treatment	rho	p-value	rho	p-value	rho	p-value
Fringe	T0	0.14	0.16	-0.04	0.79	0.12	0.33
Fringe	T2	0.18	0.06	0.04	0.77	0.25	0.03
Fringe	T3	0.10	0.32	-0.16	0.24	0.12	0.33
Fringe	T5	0.18	0.05	0.08	0.57	0.42	0.00
Fringe	T6	0.03	0.77	-0.13	0.35	0.30	0.01
Mid-Section	T0	-0.06	0.50	0.06	0.68	-0.12	0.30
Mid-Section	T2	0.13	0.18	0.17	0.20	0.15	0.18
Mid-Section	T3	0.20	0.03	0.21	0.13	0.23	0.05
Mid-Section	T5	0.08	0.42	-0.08	0.52	0.27	0.03
Mid-Section	T6	0.13	0.18	0.46	0.00	0.23	0.05
Center	T0	0.18	0.06	0.16	0.21	0.22	0.04
Center	T2	0.10	0.27	0.16	0.21	0.14	0.25
Center	T3	0.07	0.42	0.07	0.56	0.15	0.16
Center	T5	0.06	0.50	0.06	0.64	0.39	0.00
Center	T6	0.10	0.31	0.11	0.40	0.28	0.02

Table A.3: Original values and units of GHG studies in SSA

GHG	study	original fluxes	converted fluxes
CH_4	Cao et al. (1998)	200 - 500 mg CH_4 $m^{-2}d^{-1}$	6.25 - 15.63 mg CH_4-C $m^{-2}h^{-1}$
	Gondwe and Masamba (2013)	0.2 - 300 mg CH_4 $m^{-2}h^{-1}$	0.15 - 225 mg CH_4-C $m^{-2}h^{-1}$
	Jones (2001)	128 mg CH_4 $m^{-2}d^{-1}$	4.0 mg CH_4-C $m^{-2}h^{-1}$
	Kim et al. (2016)	950.4 ± 350.4 kg CH_4 $ha^{-1}yr^{-1}$	712.8 ± 262.8 kg CH_4-C $ha^{-1}yr^{-1}$
	Kim et al. (2016)	30.5 kg CH_4 $ha^{-1}yr^{-1}$	22.9 kg CH_4-C $ha^{-1}yr^{-1}$
	Kim et al. (2016)	-4.8 - 3.5 kg CH_4 $ha^{-1}yr^{-1}$	-3.6 - 2.6 kg CH_4-C $ha^{-1}yr^{-1}$
	Krüger et al. (2013)	-2.14 - 56.20 mg CH_4 m^{-2} d^{-1}	-0.07 - 1.76 mg CH_4-C m^{-2} h^{-1}
	Mitsch et al. (2012)	72 ± 8 g C $m^{-2}yr^{-1}$	8.22 ± 1 mg CH_4-C $m^{-2}h^{-1}$
	Nyamadzawo et al. (2014)	0 - 1.8 mg CH_4 $m^{-2}h^{-1}$	0 - 1.35 mg CH_4-C $m^{-2}h^{-1}$
	Kim et al. (2016)	6.5 Mg CO_2 $ha^{-1}yr^{-1}$	17727 kg CO_2-C $ha^{-1}yr^{-1}$
	Kim et al. (2016)	3.3 - 57.0 Mg CO_2 $ha^{-1}yr^{-1}$	0.9 - 15.5 Mg CO_2-C $ha^{-1}yr^{-1}$
	MacCarthy et al. (2018)	330 mg CO_2 $m^{-2}h^{-1}$	90 mg CO_2-C $m^{-2}h^{-1}$
	Nyamadzawo et al. (2014)	500 - 2080 mg CO_2 $m^{-2}h^{-1}$	136.36 - 567.27 mg CO_2-C $m^{-2}h^{-1}$
N_2O	Kim et al. (2016)	2.0 ± 1.5 kg N_2O $ha^{-1}yr^{-1}$	1.27 ± 0.95 kg N_2O-N $ha^{-1}yr^{-1}$
	Kim et al. (2016)	0.19 kg N_2O $ha^{-1}yr^{-1}$	0.12 kg N_2O-N $ha^{-1}yr^{-1}$
	Kim et al. (2016)	-0.1 - 13.7 kg N_2O $ha^{-1}yr^{-1}$	-0.06 and 8.7 kg N_2O-N $ha^{-1}yr^{-1}$
	Krüger et al. (2013)	-0.64 to 139.84 mg N_2O m^{-2} d^{-1}	-0.02 - 3.70 mg N_2O-N m^{-2} h^{-1}
	Nyamadzawo et al. (2014)	0.1146 - 5.1858 mg N_2O $m^{-2}h^{-1}$	0.07 - 3.3 mg N_2O-N $m^{-2}h^{-1}$

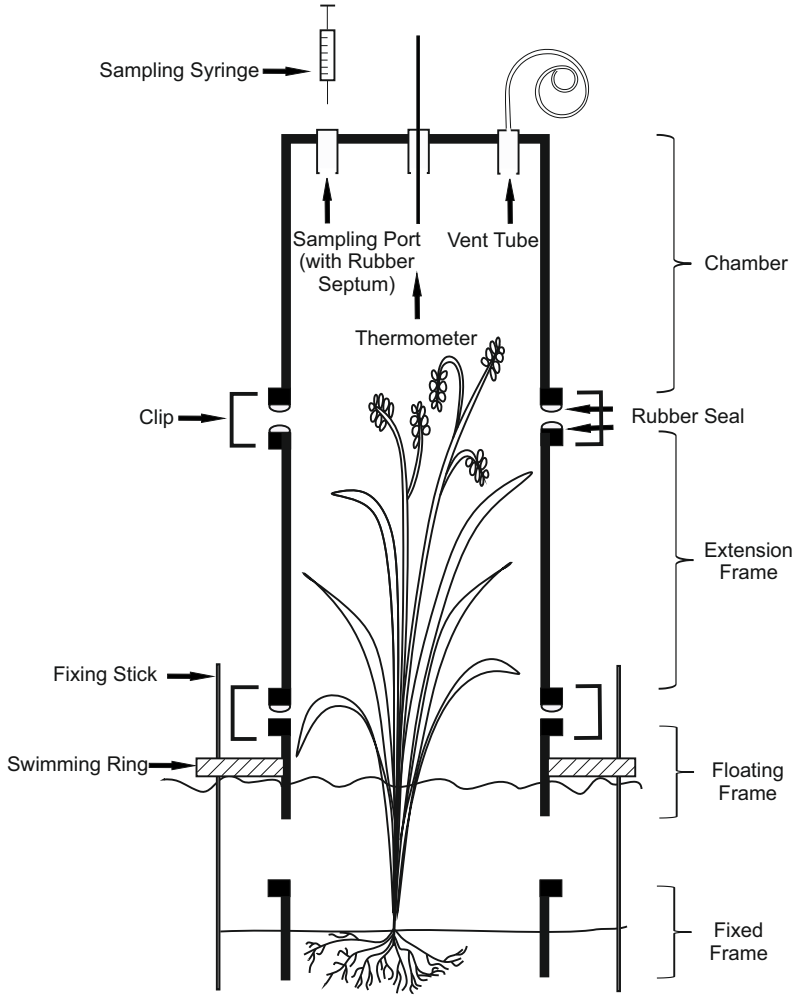

Figure A.8: Cross section of a floating chamber construction with swimming ring and extension frame.

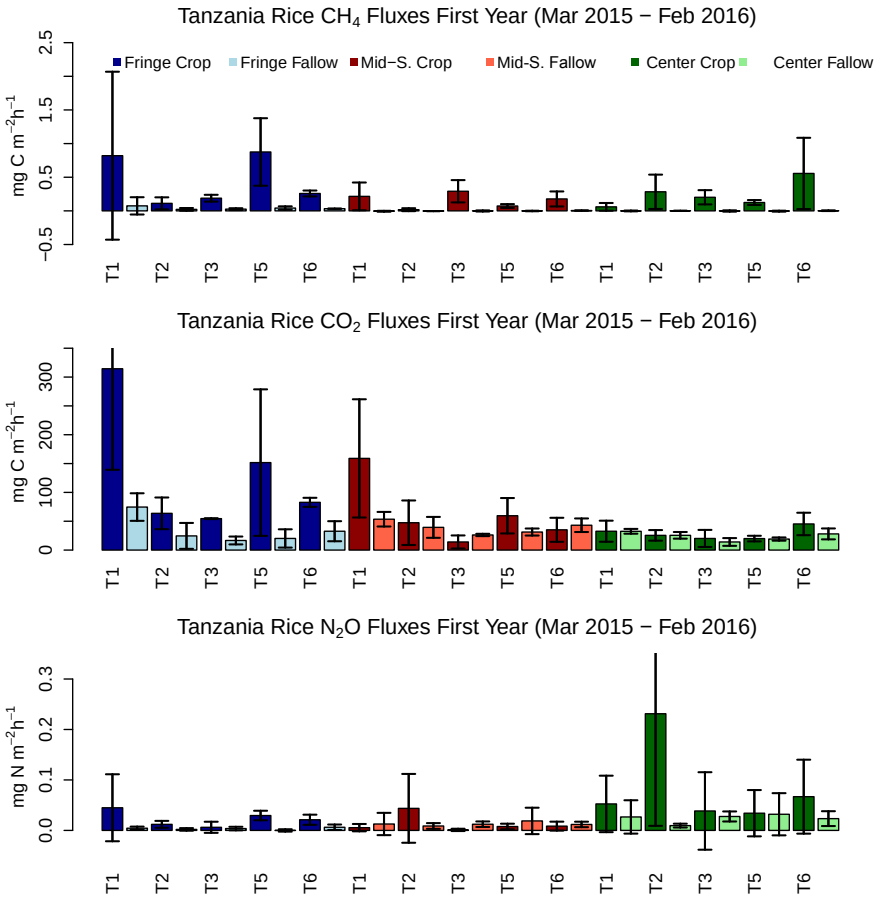

Figure A.9: Weighted seasonal mean values of the first year, Ifakara

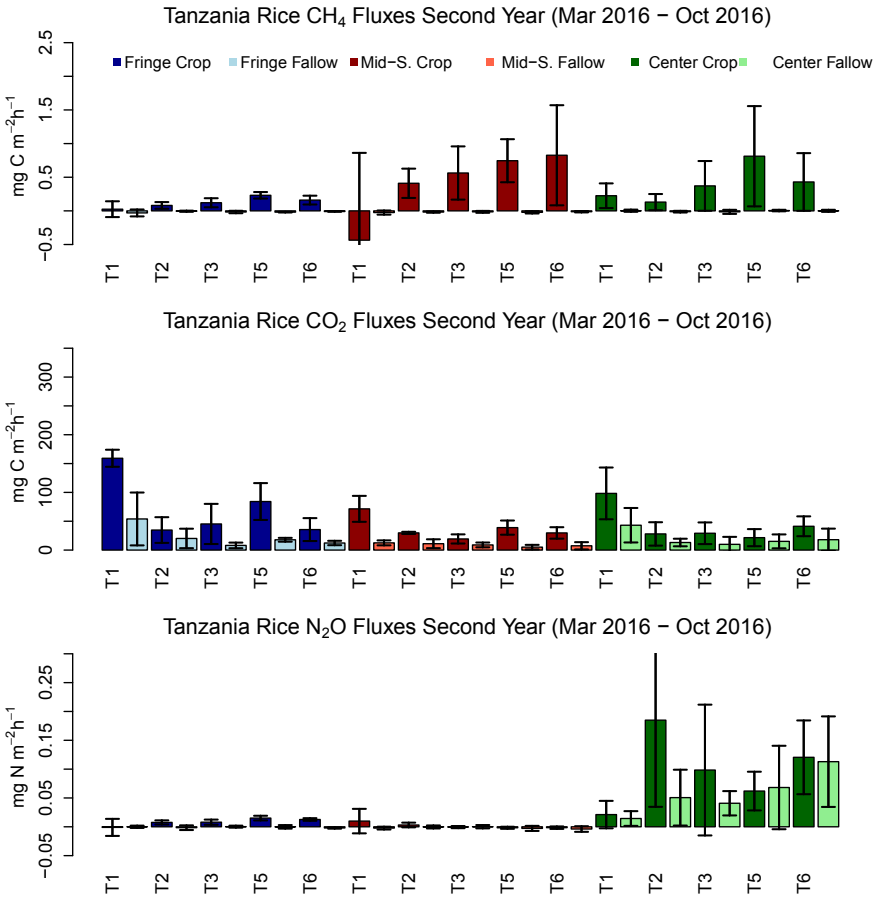

Figure A.10: Weighted seasonal mean values of the second year, Ifakara

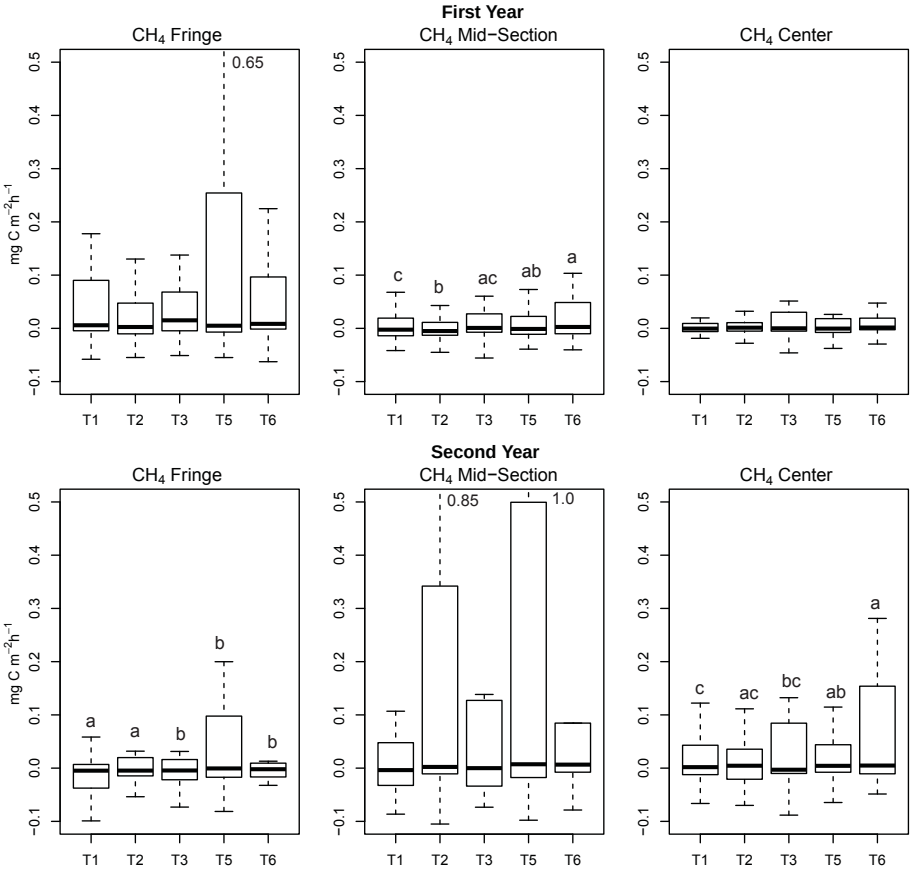

Figure A.11: Boxplots CH_4, Ifakara

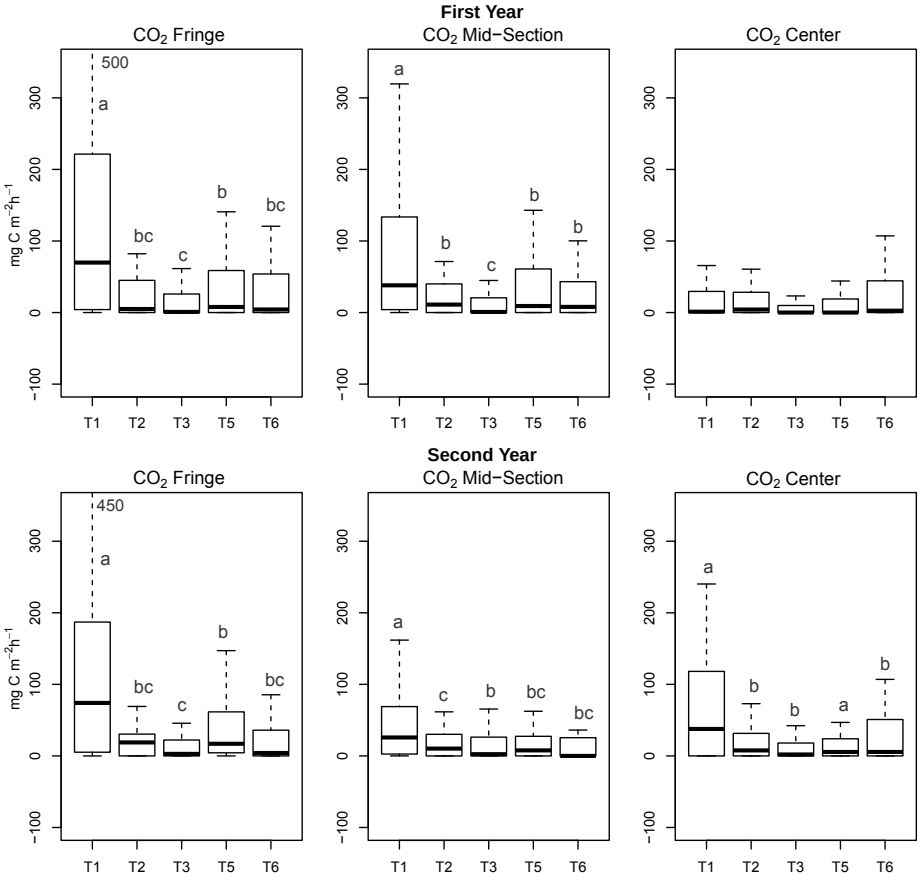

Figure A.12: Boxplots CO_2, Ifakara

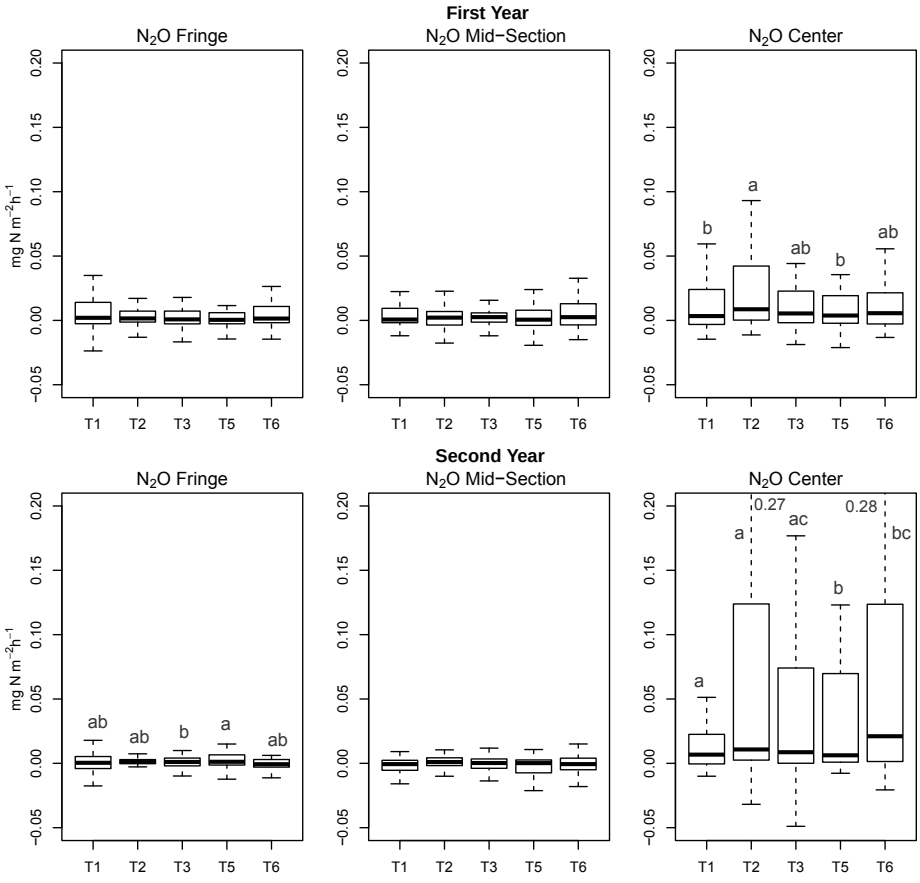

Figure A.13: Boxplots N_2O, Ifakara

Table A.4: Correlation between mineral nitrogen concentration and N_2O flux in Ifakara

		total N		NO_3^-		NH_4^+	
Zone	Treatment	rho	p-value	rho	p-value	rho	p-value
Zone	Treatment	rho	p-value	rho	p-value	rho	p-value
Fringe	T0	0.01	0.93	-0.13	0.25	0.08	0.46
Fringe	T2	0.10	0.36	0.19	0.09	0.00	0.97
Fringe	T3	-0.16	0.18	0.02	0.89	-0.18	0.13
Fringe	T5	0.11	0.38	0.19	0.12	0.03	0.79
Fringe	T6	0.00	0.98	0.16	0.20	-0.09	0.46
Mid-Section	T0	-0.18	0.11	-0.10	0.36	-0.20	0.08
Mid-Section	T2	-0.09	0.48	-0.04	0.75	-0.05	0.68
Mid-Section	T3	-0.08	0.49	0.15	0.21	-0.11	0.33
Mid-Section	T5	0.08	0.50	-0.02	0.89	0.10	0.39
Mid-Section	T6	-0.03	0.81	0.17	0.15	-0.15	0.20
Center	T0	0.03	0.84	0.02	0.92	0.05	0.73
Center	T2	0.13	0.36	-0.07	0.62	0.15	0.29
Center	T3	0.09	0.53	-0.13	0.38	0.17	0.24
Center	T5	0.30	0.04	-0.08	0.60	0.34	0.02
Center	T6	0.22	0.13	-0.30	0.04	0.32	0.03

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