University of Natural Resources and Life Sciences, Vienna

EGERTON UNIVERSITY

INFLUENCE OF LAND USE/COVER CHANGE ON GREENHOUSE GAS EMISSION AND CARBON STOCK IN A TROPICAL WETLAND IN KENYA

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By

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Affirmation

I confirm that this master thesis is my original work, written using tools and sources as quoted herein without use of any illegitimate support.

I further declare that I have not submitted this work to any other institution/university nationally or abroad for award of a degree.

Damaris Kinyua Vienna, 05/04/2015

Dedication to;

My princess, Delicia Claire Wanjiku

And

My parents;

Stanley K. Miano and Rachel W. Kinyua

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Table of Contents

Contents

List of figures

List of tables

List of abbreviations and acronyms

C	Carbon
CH_4	Methane
CO_2	Carbon dioxide
GC	Gas chromatograph
GHG	Greenhouse gases
GWP	Global warming potential
N	Nitrogen
N_2O	Nitrous Oxide
OC	Organic Carbon
OM	Organic matter
OM	Organic matter
P	Phosphorus
PCA	Principal Component analysis
SOC	Soil organic carbon

Abstract

Greenhouse gas (GHG) emissions have been well studied in the temperate wetlands, however, similar studies are almost absent in sub-Sahara Africa wetlands, which are currently under intense anthropogenic pressure. Wetlands are unique ecosystems because they play a major role in regulation of the global biogeochemical cycles. On the other hand, they are considered sources of the potent GHG, methane (CH_4) and nitrous oxide (N_2O). Conversion of wetlands to cropland alters the hydrology of the wetlands, resulting in less carbon (C) sequestration leading to increased emission of GHG. Concentration of GHG in the atmosphere has been rising over the years. Carbon sequestration in natural ecosystems such as wetlands, is one of the options proposed to reduce the GHG effect. Hence there is need to understand the emissions of the GHG from natural wetlands and whether conversion into farmlands influences their emissions. Therefore, this study aimed to assess the influence of wetland conversion into farmland on GHG (CH_4, CO_2 and N_2O) emissions and the nutrient stocks in a tropical wetland of East Africa. A two-month study was carried out on a weekly basis between December 2017 and January 2018. The static chamber method was used to compare CH_4, CO_2 and N_2O emissions between converted (unfertilized cocoyam farms) and unconverted areas of a tropical wetland. Furthermore, soil samples were analysed to compare for differences in soil C stock and nutrient (N and P) stocks between converted and unconverted areas of wetland. Converted area was a probable sink for CH_4 with flux ranging between -0.07 to 0.09 $mgm^{-2}h^{-1}$, and a source for CO_2 and N_2O (221.86± 17.86 $mgm^{-2}h^{-1}$, 187.06±25.41 $\mu gm^{-2}h^{-1}$ respectively). While unconverted area was a source for CH_4, flux ranged from 5.32 to 40.59 $mgm^{-2}h^{-1}$. Carbon and P stocks were higher in the unconverted areas (3.32±0.12%, 42.81±0.39 ppm) while all N species (TN, NO_3-N and NH_4-N) higher content was observed in the converted area. Conversion of wetland to farmland results to increased oxidation of OM, consequently, a reduction in CH_4 emission, while CO_2 and N_2O fluxes increases.

Key words: Carbon sequestration, greenhouse effect, Global warming potential, wetland drainage, Anyiko.

CHAPTER ONE

INTRODUCTION

1.1 Background information

Wetlands occupy 5-8 % of the earth's surface and encompass some of the most productive ecosystems (Bernal and Mitsch, 2013; Mitsch *et al.*, 2010). Wetlands have become increasingly important because of their unique role in regulating global biogeochemical cycles, a reason to promote sustainable use of these natural resources (Liu *et al.*, 2017; Zedler and Kercher, 2005). Wetlands' expansive root volume and canopy litter, coupled with the slow decomposition rate of organic matter due to the anaerobic conditions, makes them favourable climate regulators, through the sequestration of carbon (Batson *et al.*, 2014). They store approximately 538 pg of C in the soils, about 30% of global soil C, making them a major portion of the terrestrial C budget providing a balance in the radiation of the earth's atmosphere (Batson *et al.*, 2014; Bernal and Mitsch, 2012; Sjögersten *et al.*, 2014a).

Despite these natural ecosystems being valuable, they are under continuous threats. Population pressure and social economic changes have stimulated the need for more agriculturally productive land in quest to improve the food security (Dixon and Wood, 2003; Junk *et al.*, 2013; Mitchell, 2013). Globally about 64% of wetlands have been lost since 1900 because of human activities (Davidson, 2014). The main driver of wetland degradation and loss is land use change including, conversion to agriculture and pasture, reservoir building, urbanization and infrastructure development (Zedler and Kercher, 2005). Drainage for agriculture has been a prime cause of wetland loss to date, with an estimation of 26% of the global land area having been drained for intensive agriculture (56% to 65% in Europe and North America, 27% in Asia, 6% in South America and 2% in Africa) as of 1985 (Davidson, 2014). Continued drainage of wetlands without preserving their ecological integrity will eventually result in ecosystem disservices such as turning wetlands from C sinks into sources (Nath and Lal, 2017; Zedler and Kercher, 2005).

Besides wetlands ability to sequester C, they are considered sources of potent GHG (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) responsible for global warming (Kayranli *et al.*, 2010). Wetland drainage induce changes in the wetland hydrology, and consequently

the amount of organic C sequestered. Reduced C sequestration influences the CO_2, CH_4 and N_2O fluxes (Page and Dalal, 2011). Changes in the OM alters the equilibrium of the global C cycle resulting in increased release of GHG into the atmosphere (Craswell and Lefroy, 2001; Kindler *et al.*, 2011).

Wetlands are considered key sources of CH_4, a GHG having 25 times higher global warming potential (GWP) than CO_2 (Whiting and Chanton, 2001). Methane has been reported to be responsible for approximately 18% of the total greenhouse effect (Mitsch and Gosselink, 1993). Methane emissions occur through biological processes (Figure 1), comprising of (i) production in anaerobic conditions through methanogenesis (ii) consumption by the methanotrophs in the aerobic zones and (iii) direct transportation to the atmosphere e.g. through the aerenchyma of plants and in small quantities though diffusion and ebullition (Eusufzai *et al.*, 2010). These processes are dependent on water levels, temperature and soil composition, hence draining wetlands has seen a reduction in CH_4 emission.

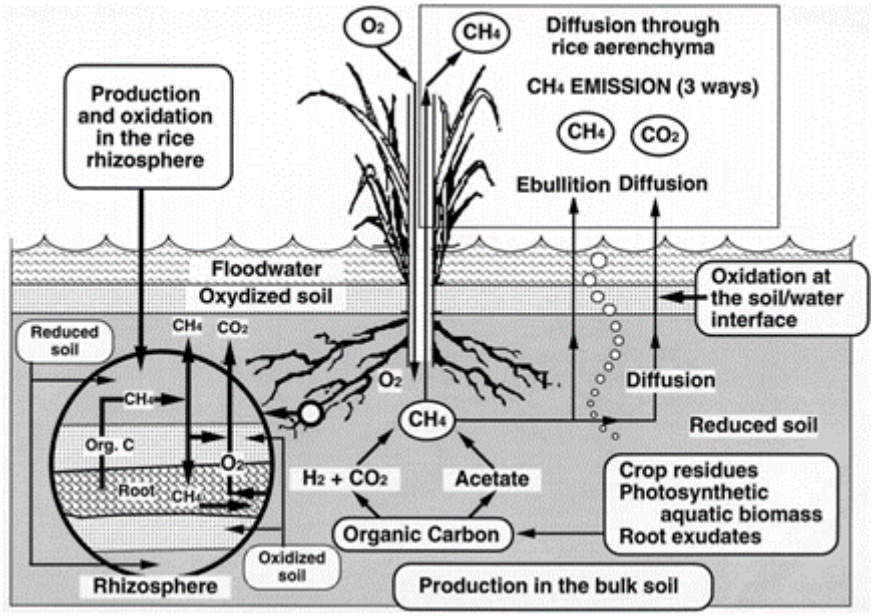

Figure 1: Schematic diagram of methane production, consumption and transfer pathways into atmosphere (Le Mer and Roger, 2001).

Carbon dioxide forms a major component of the C cycle (Figure 2). Its flux in the atmosphere is as a result of (i) uptake by plant via photosynthesis, (ii) emission through plant respiration and (iii) emission through microbial decomposition (Boone *et al.*, 2005). Drainage of the organic soils of the wetlands results in mineralization of the SOC and increased emissions of

C in to the atmosphere (Kasimir-Klemedtsson *et al.*, 1997). Wetlands have large pools of C, hence even a small increase in the rates of SOC oxidations through drainage results in high C outflows into the atmosphere and results into global warming and climate change(Ma *et al.*, 2016).

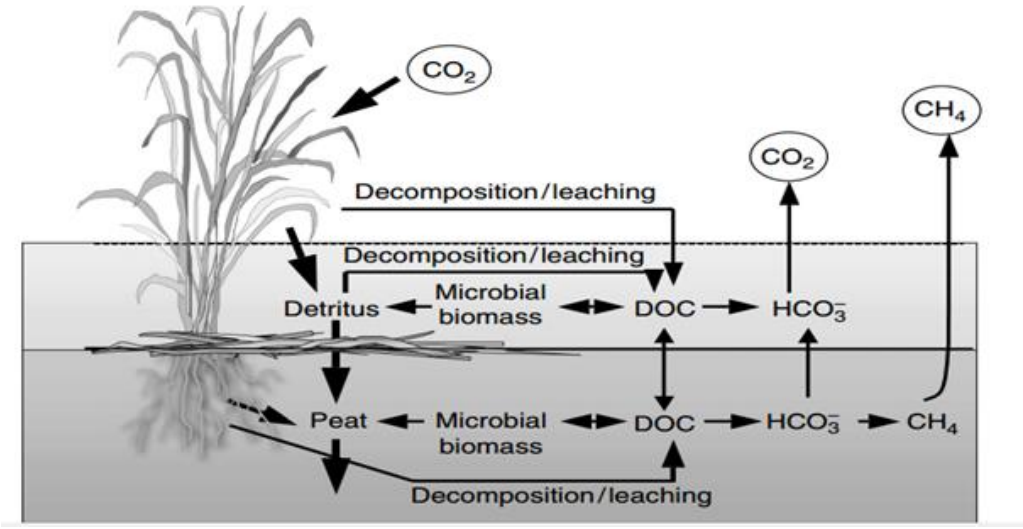

Figure 2: Schematic drawing highlighting carbon cycling processes and carbon storage (Reddy *et al.*, 2000).

Nitrous oxide is a component of the nitrogen (N) cycle (Figure 3), being released as a by-product of nitrification or an intermediate product of denitrification (Groffman *et al.*, 2006). Emission of N_2O is dependent on high temperatures, OM availability, inundated soils and availability of N nutrient components such as ammonium, nitrite and nitrates (Zhu *et al.*, 2013). Wetlands favour all these conditions hence considered a source of N_2O, besides intensified agriculture would also result in increased emission of this potent GHG with GWP being 298 times higher than that of CO_2 (Bernal and Mitsch, 2012).

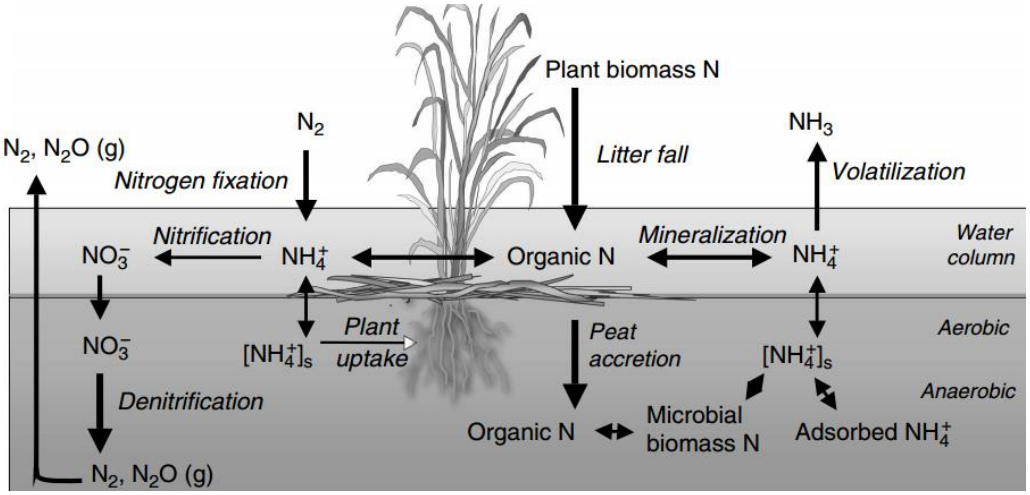

Figure 3: Schematic illustration of nitrogen transformation in wetlands (Reddy *et al.*, 2000)

Land use change result in wetland destruction and the largest effect is on the carbon fluxes. For instance, in North America, wetlands were highly influenced by land use (Bridgham *et al.*, 2006). This resulted in the reduction of their ability to sequester carbon, oxidation of soil carbon reserves upon drainage and reduction in methane emissions (Follett, 2001). Furthermore, drainage also affects the soil negatively by reducing SOM content and moisture levels. The soil is usually disturbed when crops are planted and the planted crops rarely bind soil like the natural wetland vegetation (Bridgham *et al.*, 2006). An average loss of 10.1 Mg ha of soil organic carbon on over 16 million ha of wetland, in North America was reported as a result of agricultural conversion (Euliss *et al.*, 2006). More studies in North America and the Prairies Pothole Region, have indicated that least disturbed wetlands catchments alongside native grasslands have relatively high C. On the other hand, soils in wetland catchments along agricultural settings had less C but could sequester more C if restored to similar natural states.

Although many studies on GHG have been carried out, there are still geographical regions and agricultural systems that have not been characterized. Most of these studies have been done in temperate regions whose climatic conditions, characteristics and environmental ranges differ widely with those of tropical regions such as Africa. Additionally, most studies in sub-Saharan wetlands have addressed hydrology, community structures and species diversity. This study did not find studies in Africa/sub-Sahara region conducted on conversion of the wetlands into other uses, and how these changes influence soil atmosphere GHG exchange. Recently, wetlands have gained attention as potential sink of the growing concentration of GHG, and therefore understanding the processes of C sequestration, and GHG emission in

tropical wetland is important. Furthermore, in October, 2016 Kenya, a sub-Saharan country, ratified the Paris agreement. The nations that ratified the agreement agreed to hold global warming to "well below" two degrees Celsius over pre-Industrial Revolution levels and on "pursuing efforts" to lower and keep it to 1.5°C. Accordingly the countries are required to take stocks of the emissions every five years starting 2018 (Rogelj et al., 2016). Therefore, to do proper accounting of the GHG inventory it's necessary to understand the emissions from the natural ecosystems as well as from the different land uses. A two-month study was carried out to assess how the conversion of wetlands to other uses affect the emissions of GHG and the C stocks. Information gained shall be used in consequent studies as a baseline, and can further be used to inform on importance of wetlands and the need to preserve these fast diminishing ecosystems.

1.2 Objectives and research questions

General Objective

To assess the effect of wetlands conversion into farmlands on SOC, N and P stocks and GHG emission.

Specific objectives, research questions and hypotheses

1. To quantify the standing stocks of C, N and P in converted and unconverted areas of a tropical wetland

Research question: How does the C, N and P stocks change in converted and unconverted areas of a tropical wetland?

i. H_1: SOC in the unconverted areas is higher than the converted areas due to increased accumulation of detrital organic matter and low decomposition rate.

ii. H_1: N is high in the unconverted areas compared to the converted ones due to N accumulation in the soil.

iii. H_1: P is high in the unconverted areas as compared to converted areas due to retention by the sediments.

2. To compare CO_2, CH_4 and N_2O emission and temporal patterns in converted and unconverted areas of a tropical wetland.

a) Research question: How does the CO_2, CH_4 and N_2O emission vary in converted and unconverted areas of a tropical wetland?

i. H_1: CO_2 emission in the converted areas of the wetland is higher than the unconverted areas because of increased oxidation of organic C and loss of biomass.

ii. H_1: CH_4 emission is higher in the unconverted areas of the wetland compared to the converted due to anaerobic oxidation of organic C.

iii. H_1: N_2O emission is higher in the unconverted areas of the wetland compared to converted areas due to nitrification and denitrification processes.

b) Research question: What are the temporal patterns of the CO_2, CH_4 and N_2O emission in converted and unconverted areas of a tropical wetland?

i. H_1: CO_2 emission expected to increase with a decrease in moisture content over time because of increased oxidation of organic C.

ii. H_1: CH_4 emission expected to decrease with a decrease in moisture content over time because of increased oxidation of organic C.

iii. H_1: N_2O emission will reduce with a decrease in moisture content over time because of reduced denitrification due to aerobic conditions.

3. To determine how the emission of GHG is affected by change in SOC, N and P stocks in converted and unconverted areas of a tropical wetland.

Research question: How does the gas flux vary with changes in C, N and P stocks in converted and unconverted areas of a tropical wetland?

H_1: CO_2 increases with decrease of C in the converted areas, CH_4 decreases with decrease in C in the converted area and there is little effect on N_2O with changes in C stocks.

$H_{1:}$ High availability of NO_3 results in high N_2O emissions because of enhanced denitrification in the unconverted area.

$H_{1:}$ Phosphorus availability results in increased CO_2 emission because of increased microbial heterotrophic and autotrophic respiration.

CHAPTER TWO

LITERATURE REVIEW

2.1 Global status of wetlands

Globally, wetlands can be found in all climates, from tropical to tundra, except in the Antarctica (Tiner, 2009). Wetlands occupy about 6% of the earth's surface covering about 10% of North America, 20% of South America, 10% of Russia, 7% of China, 3% of tropical and subtropical Asia, 3% of Australia, 7% of Africa and 5% of Europe (Junk *et al.*, 2013). Wetlands have been adversely affected especially in the densely populated regions such as Western Europe and China, and in countries with water shortages such as Australia, and in countries with powerful agro-industries such as the USA (Mitsch and Hernandez, 2013). Pressures facing global wetlands are mostly in the form of land reclamation, intense resource exploitation, changes in hydrology and pollution resulting to 30-90% of wetlands being destroyed or strongly modified depending on the region, mostly with no signs of abatement (Junk *et al.*, 2013).

Much of Africa lies in arid and semi-arid regions, estimated to occupy 1-16% of the total area of the continent (Dixon and Wood, 2003). However, their exact estimate cannot be quantified due to lack of scientific investigations and inconsistency in mapping policies (Bullock and Acreman, 2003). Although there are still several pristine wetlands in Africa, as compared to Europe and North America, many wetland areas are still facing immense pressure mostly from the demographic growth. Consequently, many wetlands have been drained for agriculture and settlement in addition to setting up of unplanned infrastructure (Schuyt, 2005).

Out of Kenya's 583,000 km², 14,000 km² (ca. 3 – 4 %) are occupied by wetlands ranging from deltas, estuaries, mudflats and mangroves, swamps, marshes and floodplains (Okeyo-Owuor *et al.*, 2016). Drive for economic growth, agricultural practices and development have been reported to be the major threats to papyrus wetlands and their biota, mainly through drainage, clearing and reclamation for subsistence crop production (Morrison et al., 2012). Mafabi (2000), in a comparative aerial survey of Lake Victoria basin wetlands between 1969 and 2000, reported 50, 47 and 34% loss in Dunga, Koguta and Kusa respectively.

In Kenya, drainage and conversion to arable land have been the key drivers to degradation of wetlands (Mironga, 2005). Kenya ratified the Ramsar convention in 1990, nevertheless, much of the wetlands had already been degraded through drainage and reclamation, overgrazing and pollution (Mironga, 2005). Communities living near the wetlands rely almost 100% on the wetlands for their livelihood especially for source of their water, food, and building materials such as clay, sand, wood and papyrus (Schuijt, 2002).

2.2 Land use cover change in wetlands

Globally, wetland loss has exceeded 50% of their original extent, however the rates of loss have decreased in Europe and North America (Mitsch and Hernandez, 2013). In other parts of the globe, rapid conversion of inland natural wetlands is still continuing (Davidson, 2014). In Costa Rica, wetland conversion was reported to be highly influenced by the topography. For instance, the wetlands that were easily accessible had the highest conversion rates (Daniels and Cumming, 2008). In general, people influence as well as depend on the ecosystem services provided by the wetlands (Clarkson *et al.*, 2013). Accordingly, in China a net loss of 50,360 km^2 due to human activities resulting from change in land use have been reported (Kirwan and Megonigal, 2013; Wang *et al.*, 2012). Furthermore, study by Song *et al.*, (2008), in the Sanjiang plain using remote sensing reported a change in the land use, where there was a decrease in the forests, and waterbodies in addition to an increase in the residential area and farmland. Their correlation analysis indicated that demographic growth was the main driving force for increased of farmland.

2.3 Carbon sequestration and greenhouse gases: CO_2, CH_4 and N_2O emission in wetlands

Carbon sequestration is the long-term removal of CO_2 from the atmosphere, through the plant biomass into SOM (Johnson *et al.*, 2007). Consequently, build-up of SOM results in increased soil quality and soil productivity, reduced risk of soil erosion, and decreased water contamination and eutrophication. Wetland soils play an important role in global climate alleviation (Nag *et al.*, 2017). The accumulation of C in wetlands soils is facilitated by the expansive root volume and canopy litter, and the slow decomposition of organic matter due to the anaerobic conditions (Batson *et al.*, 2014). While wetland ecosystems are considered to have a potential for C sequestration from the atmosphere, they could also be a source, since organic C is highly sensitive to environmental changes such as water content,

temperature, nutrient regimes and microbial activity (Kindler *et al.*, 2011; Veber *et al.*, 2017). Furthermore, human activities (including land use change) play a major role in the soil organic carbon (SOC) dynamics (Ma *et al.*, 2016).

Generally, GHG production and fluxes together with C sequestration and nutrients availability in wetland soils are controlled by abiotic factors that are highly variable among them, e.g. soil moisture and temperature, water depth, hydro period, water chemistry and redox conditions (Hernandez and Mitsch, 2006; Smith *et al.*, 2003). In addition, the vegetation and microbial community composition together with organic substrate availability are also important factors influencing the GHG flux and C (Liu and Greaver, 2009). All these biotic and abiotic factors are closely linked to climate, ground water interactions, and geomorphology, hence they are affected by human activities such as land use change (Tangen *et al.*, 2015).

A study conducted in the Zoige alpine wetland in China by Ma *et al.*, (2016) found that the CO_2 emission of the permanently saturated wetlands was lower (203 mg $m^{-2}h^{-1}$) than that of drained wetlands converted to grasslands (323 mg $m^{-2}h^{-1}$), providing evidence of C accumulation in the wetland. The amount of OC in soil influences the emission of the GHG, Cui *et al.*, (2015) and Li *et al.*, (2005) reported that increase in the SOC resulted to increase in N_2O production and a negligible effect on CH_4. They related this positive correlation between SOC and N_2O flux to the coupled biogeochemical cycles of C and N.

2.4. Influence of wetlands drainage and conversion (land use/cover change) on GHG emission

Wetlands are usually characterised by nutrient rich soils with high moisture content available all year round (Sakané *et al.*, 2013). Owing to this characteristic, wetlands are prone to conversion into agricultural production systems. In East Africa, wetlands have been reported to provide 10-40% of the rural population's annual food needs (Kamiri *et al.*, 2013). However, conversion of the wetlands has its negative impacts, such as loss of ecosystem services (ES) for instance C sequestration. Conversion of a pristine wetland in Ethiopia to farmland provides evidence of lost ES, where the drained wetland showed reduced water availability and reduction in crop yields (Dixon and Wood, 2003). Reduction in the crop yield was attributed to decrease in soil C and N, since drained is associated with aeration of soil consequently reducing C and N availability.

Land use change impacts the net emissions of the GHG. For example, if peatlands are drained for agriculture it results in increased emission of N_2O along with that of CO_2, but CH_4 emission is decreased (Smith *et al.*, 2004). In the Prairies Pothole Region, a study by Tangen *et al.*, (2015) on how land use affected the GHG fluxes reported that soil OC was lost when undisturbed wetlands were converted for agriculture. Veber *et al.* (2017) reported that human impacted peatlands had higher GHG emission, where agricultural activities, especially crop production together with intensive grazing in the mountain peatland pastures were the main factors studied. Furthermore, undisturbed vegetated wetlands in Australia were found to be net GHG sinks, but after drainage, the wetlands and the mangrove forest turned into carbon sources (Finlayson *et al.*, 2013).

2.5 Carbon, nitrogen and phosphorus stocks and land use change

Land use change may occur naturally or be influenced by human activities, resulting in land cover change and consequently a reduction in associated C stocks (Houghton et al., 2012). Soil OC is highly influenced by the vegetation cover, and any change in the land use may considerably modify the related characteristics of source or sink for the GHGs, because the plant species differ in root depth and spatial distribution in different land uses (Oertel *et al.*, 2016). Wetlands anaerobic condition nature allows them to inherently accumulate C in the soils. However, management such as drainage may cause return of the accumulated C into the atmosphere (Borges *et al.*, 2015).

Conversion of natural ecosystems to agriculture have been reported to decrease SOM and contribute significantly to the increase in CO_2 concentrations in the atmosphere (Puget and Lal, 2005). According to Guo and Gifford, (2002), a reduction of SOC was reported with soils losing 42 and 59% of SOC upon conversion from forest to croplands and from grassland to crop, respectively. Correspondingly, another study by Ma *et al.* (2016) using the Environmental Policy Integrated Climate (EPIC) model, reported drainage as one of the main driving factors for SOC loss in the wetlands, conversion of the wetlands via drainage to grassland since 1980 had resulted to a loss of approximately 4 t C ha-1 from the SOC stock. Qingshui *et al.*, (2014) using spatial analysis and statistics method reported a decline in the swamps and floodplain area in China, causing C loss from wetlands when the wetlands are converted to other land uses.

Phosphorous in ecosystems is mainly from mineralization and weathering process under natural conditions (Reddy et al., 2000). In this regard, wetlands are known to accumulate P in the sediments, vegetation and detrital materials. Therefore, conversion of wetland to agricultural land decreases the ability of wetlands to retain P.

2.6 Relationship between N, P, SOC and GHG emission

Greenhouse gases emission is dependent on microbial activities, chemical decay processes and heterotrophic respiration of soil fauna and fungi (Smith et al., 2003). Nutrient availability dominates these processes, coupled with soil moisture content, temperature, pH and land cover related parameters (Oertel et al., 2016). Naturally occurring N and C, together with atmospheric deposition, manure and fertilizer application play a critical role in the emissions of the GHGs (Chapuis-lardy *et al.*, 2007; Ludwig *et al.*, 2001; Oertel *et al.*, 2016).

Studies by Cobo *et al.*, (2010) and Song *et al.*, (2013), suggested that addition of N in wetland ecosystems altered soil physical characteristics, microbial communities and the vegetation communities which influence the GHG emissions. Oertel *et al.*, (2016) reported a negative correlation of N_2O emission with the C/N ratio, with the lowest emission being recorded at C/N ≥ 30 and highest at C/N values of 11. Further, the study reported a positive correlation of CO_2 and CH_4 emission with the C/N ratio. Increasing N content results to high soil respiration and high net ecosystem exchange, if C is not limiting. Intensive management of the peatlands was found to alter the soil C/N balance, leading to higher variability of GHG emission (Veber *et al.*, 2017).

In ecosystems that are N limited, addition of N causes higher N content in plant tissue, leaves and the litter fall, which in turn accelerates the assimilation and dissimilation processes of CO_2 and intensifies the substrate for N_2O emission from soil (Aronson and Helliker, 2010). The net atmosphere – biosphere exchange of GHG markedly depends on the coupled C-N cycles and the local conditions (Qingshui et al., 2014). According to Aronson and Helliker(2010), the effect of N on GHG emission is closely related to the form of N, duration and the timing of N fertilizer application, and its interaction with the abiotic factors.

Phoshoprus on the other hand is considered a key element of the microbial processes (Wang et al., 2017). Consequently, a positive correlation has been observed between CH_4 and CO_2

production with soil P content and microbial biomass (Wright and Reddy, 2001). Correspondingly, in Belize marsh sediments microbial activities were found to be positively affected by addition of P, and this led to the assumption that the system was P limited. In addition to this, methanogenesis was observed to increase in treatments with enriched P (Pivničková et al., 2010).

2.7 Greenhouse gas measurements in wetlands

Tropical wetlands play an important role in the global C cycle (Page and Dalal, 2011). On the other hand, they are under intense pressure from agriculture, resulting to increased CO_2 emissions into the atmosphere from these ecosystems (Houghton *et al.*, 2012). Increased GHG emission in tropical wetlands can be estimated and/or predicted using the existing wetland modelling tools (Sjögersten et al., 2014a). However, inclusion of these wetlands is hindered by lack of data to validate them (Farmer *et al.*, 2014). A review by Sjögersten *et al.*, (2014b) acknowledged a significant lack of data on carbon balance and GHG fluxes from the natural wetlands, hence limiting the ability of global climate change models to make accurate predictions on future climate. In this regard the study recommends exigent need for good quality data on carbon dynamics in natural wetlands in addition to, CO_2 and CH_4 flux data, accounting for spatial and temporal variation, to be used for evaluating model predictions. Correspondingly, a study Van Dam *et al.*, (2007) while creating a simulation model for papyrus wetland prompted need for more research on denitrification processes in natural wetlands. Besides, this will provide a comparison and robust understanding of how tropical wetlands differ from the well-studied temperate wetlands in addition to enabling incorporation of tropical wetlands into global climate change models.

Following the aforementioned gaps on lack of sufficient data on GHG emission, measuring the GHG fluxes exchange between soil and atmosphere in natural ecosystems such as wetlands and managed systems is of critical importance (Collier *et al.*, 2014). Furthermore, understanding the contribution of both natural and managed systems informs on development and evaluation of GHG mitigation strategies (Myhre *et al.*, 2013). There is a range of GHG measurements strategies varying from mass balance to micrometeorological approaches each characterised by pros and cons (Denmead, 2008). Chamber based (automated or manual) and micrometeorological measurements (gradient method or eddy

covariance) are the commonly used techniques to measure GHG flux exchange between terrestrial ecosystems and atmosphere (Butterbach-Bahl *et al.*, 2016).

Micrometeorological approaches are based on a real time direct measurement of vertical GHG fluxes. However, the assumption made by this approach is that fluxes are nearly constant with height and that concentrations change vertically but not horizontally (Denmead, 2008). This methods are considered to have advantage over chamber based method because the approach integrates fluxes over large areas (>ha), in addition to having high temporal resolution and no interference with the microenvironment (Butterbach-Bahl *et al.*, 2011; Hensen *et al.*, 2013). On the other hand, the method has its limitations among them, need for large homogenou**s** surfaces, requirement for fast response infrared sensors which are normally expensive (Collier *et al.*, 2014). Furthermore, atmospheric stability may affect the measurements during the night, causing constrain to the data captured (Hensen *et al.*, 2013).

Chamber based method, unlike the micrometeorological counterpart focus on gas concentration at the soil surface, where sampling is restricted to above ground headspace. Chamber method covers a finer scale usually small surface areas up to $< 1\ m^2$, hence are considered rather simple and therefore often used in most studies (Butterbach-Bahl *et al.*, 2016). Chamber based measurements have several advantages over the micrometeorological approach. To start with gas samples collected using chamber method can be stored for future analysis, secondly, its cheap in terms of capital cost considering the chambers do not require power supply at the site, also they do not require fast response sensors. Finally chamber method allows for process studies and experiments with many treatments (Denmead, 2008; Flechard *et al.*, 2007; Rosenstock *et al.*, 2016). Nonetheless, they are subject to high coefficients of variations due to spatial variation, moreover while installing the chambers into the sites, environmental conditions are disturbed which may influence the flux measured (Butterbach-Bahl *et al.*, 2016; Collier *et al.*, 2014). Due to the disturbance to the environmental conditions it is advised that the chamber bases are installed into the study site at least a week before collection of samples begin. Further recommendations on chamber methods are given by Parkin and Venterea, (2010). According to Hensen *et al.*, (2013), chamber based measurements are likely to miss peak events such as rainfall, because the experimentalist are not always at the site, in addition to the idea that chambers can only be

closed for a limited period per day. Therefore, Hensen *et al.*, (2013) recommended use of automatic chambers to address this limitation, although the automated chambers will require more capital cost, making them more expensive to use in the end covering smaller study area. This study used static chamber method, because it involves minimum capital cost, it's easy to follow in addition to having a laid out protocol.

Gas samples collected by the use of chamber based method are analysed commonly by either, gas chromatography (GC) or photoacoustic spectroscopy (PAS) method (Butterbach-Bahl *et al.*, 2016). The principle behind PAS is that the GHG absorbs light at specific wavelength, then the absorption is directly linked to the concentration. More specifically, PAS converts the absorption of light into acoustic signal which is then measured by a microphone (Leytem *et al.*, 2011). It's also possible to use PAS technique in the field, by making a closed loop connection between the chamber and the PAS instrument, in a way that air leaving the apparatus returns to the chamber avoiding dilution or under-pressure. This technique has gained popularity over the recent years, but according to Iqbal *et al.*, (2013), its precision and accuracy is still uncertain as compared to GC.

Gas chromatography technique follows principle of separating a compound into its molecular constituents (Hensen *et al.*, 2013). This analytical technique using GC is most commonly for determination of GHG concentration in gas samples from chambers (Butterbach-Bahl *et al.*, 2016). In general, 1 – 3 ml of gas sample is injected into the GC and the different compounds are separated in an analytical column. A ^{63}Ni Electron Capture Detector (ECD) is usually used for N_2O, operating at temperatures ranging from 330 to 350 °C, with highest sensitivity N_2O and lowest cross – sensitivity to CO_2 (Wang *et al.*, 2010). Whereas for CH_4, a flame ionisation detector (FID) is normally used, but in case a methanizer is introduced before the detector, CO_2 can also be measured with FID or use of thermal conductivity detector. Analysis of gases in this study followed GC technique.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Site description

Anyiko wetland is an inland permanent riverine wetland located in North East Ugenya location, Siaya County, Kenya. The wetland is in mid lower section of Nzoia River basin covering an area of 4 Km², with an average length of 6Km, situated within longitudes 0°16', 38°56''N, 0°14', 18°66''N and latitudes 34°16', 35°55''E, 34°18', 0°57''E (Figure 4). The wetland is fed by underground springs and six streams, drains into Nzoia River, which in turn drain into Lake Victoria. Anyiko is characterized by *Cyperus papyrus* as the dominant plant, in addition to *Phragmites sp*. The main activities in Anyiko include farming, mainly smallholder rice production in the lower sections and small-scale subsistence farming of cocoyam and vegetables. The community members also harvest papyrus for fibre and mats production as source of income.

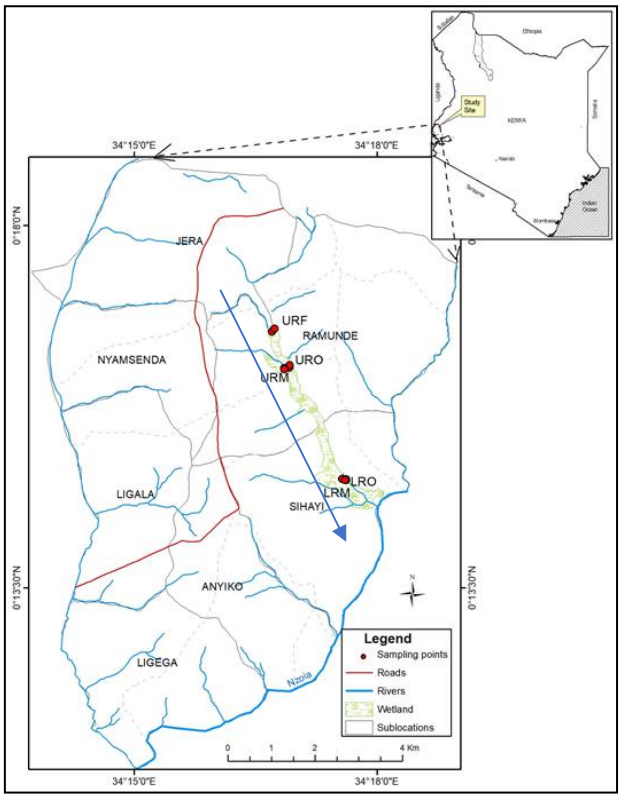

Figure 4: Map of the study site showing the hydrological connection of the wetland and the sampling sites shown by the red dots. The arrow indicates the direction of flow.

3.2 Study design

The study was conducted during the dry season between December 2017 and January 2018; no rainfall was recorded throughout the study period. The design included five sampling sites, selected in a manner that allowed comparison between (i) converted and unconverted, (ii) longitudinal (upper and lower reach) and (iii) transverse (seasonally and permanently flooded) variation within the wetland. Converted area of the wetland had one site (Farm), while the unconverted area had four sites, spread longitudinally along the upper reach (URO and URM) and lower reach (LRO and LRM), and transversely along the outer edge (URO and LRO) and middle section (URM and LRM) of the wetland respectively (Figure 5). In the unconverted areas, the papyrus crop was cleared to provide room for chamber base installation.

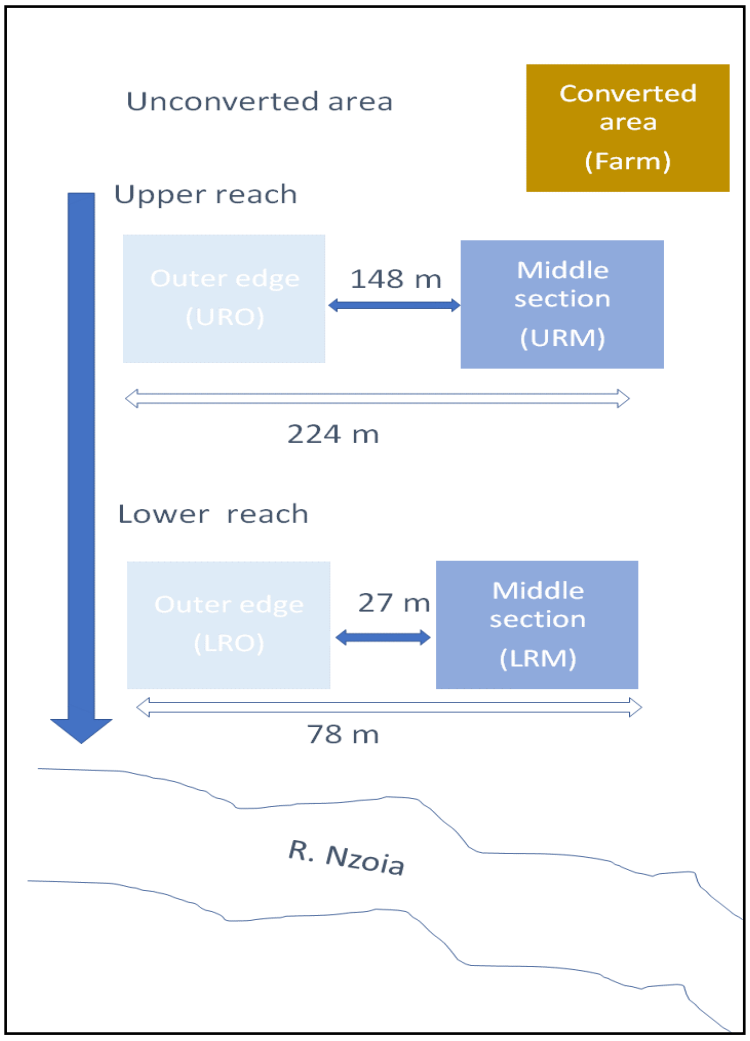

Figure 5: Schematic diagram showing the spatial study design, (not to scale).

3.2.1 Site characteristics

Converted area of the wetland (Farm) (URF 1 Lat: 0°16’ 31.24’’ N, Long: 34°16’ 44.76’’E, URF 2 Lat: 0°16’ 33.57’’ N, Long: 34°16’ 46.74’’E, URF 3 Lat: 0°16’ 33.36’’ N, Long: 34°16’ 46.71’’E): formerly part of the wetland, but the area was drained and converted into a farmland. The land was not under intensive management, i.e. no fertilizers were being applied to the farms. Cocoyam was the main crop produced in the chosen site (Figure 6), maturing for harvest in 6 months.

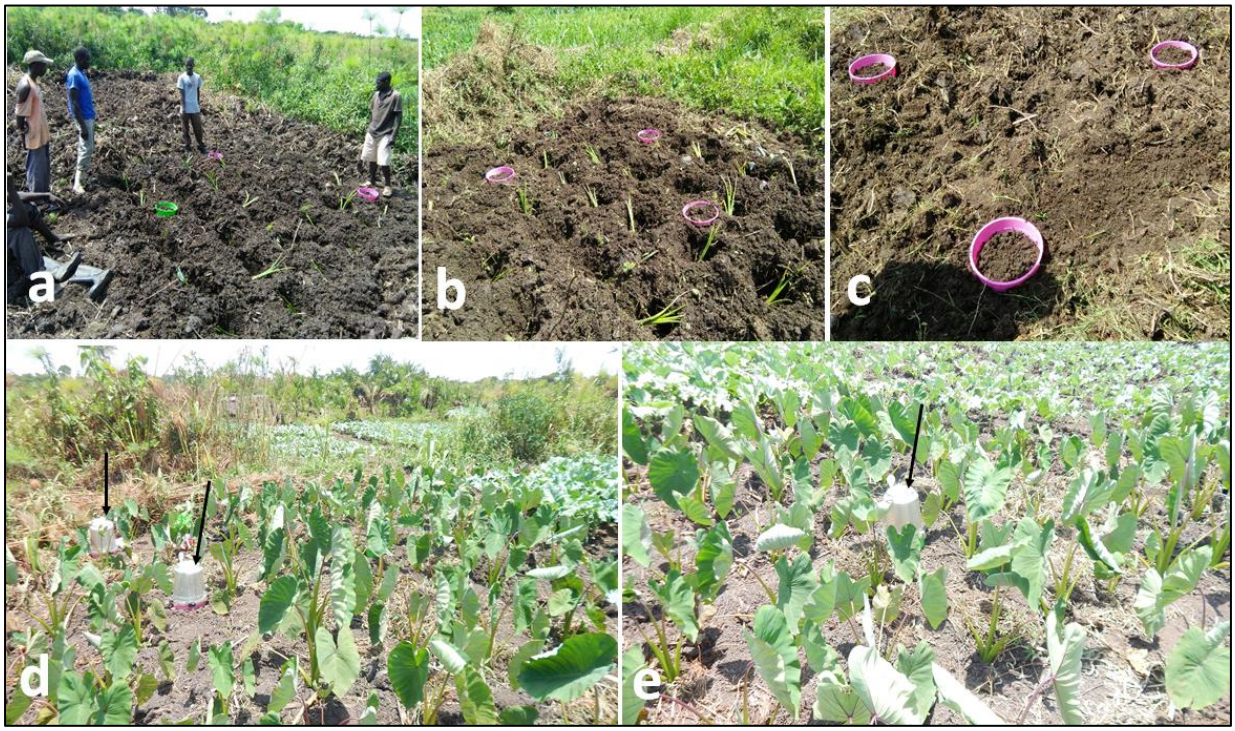

Figure 6: The converted area showing cocoyam as the main land use (a) URF 1, adjacent to the wetland, recently converted (<1Year) (b) URF 2, have been farmed for 6 years, (c) URF 3, adjacent to URF 2, farmed for 6 years, (d) and (e) shows the developed cocoyam plants , the arrows shows chambers during a sampling event.

Upper reach outer edge (URO) (URO 1 Lat: 0°14’ 4.0’’ N, Long: 34°16’ 54.3’’E, URO 2 Lat: 0°14’ 3.8’’ N, Long: 34°16’ 53.9’’E, URO 3 Lat: 0°16’ 3.9’’ N, Long: 34°16’ 53.8’’E): The site was in the unconverted area characterized by dense population of the papyrus (Figure 7 d & e). During the set-up of the chambers in the beginning of the study, the site had pools of water with an average water level of 39 cm (Figure 7 a, b & c), but by first sampling date site dried out having no water above the soil.

Figure 7: Characteristic of the URO site at the time of chamber installation (a) URO 1: average water level 36cm, Location; 11 m from the edge, (b) URO 2: average water level 40cm, Location; 13 m from the edge, (c) URO 3: average water level 30cm, Location; 9 m from the edge, (d) & (e) indicate the dense population of papyrus

Upper reach middle section (URM) (URM 1, Lat: 0°16’17.8’’ N, Long: 34°16’59.1’’E, URM 2 Lat: 0°14’ 6.2’’ N, Long: 34°16’ 58.0’’E, URM 3, Lat: 0°14’ 6.2’’ N, Long: 34°16’ 58.0’’E): The site was characterized by dense population of the papyrus (unconverted area). Deep pools of water having an average water level of 59 cm (Figure 8) described the site during the start of the study, however, site dried out with time having no water above the soil by second sampling date.

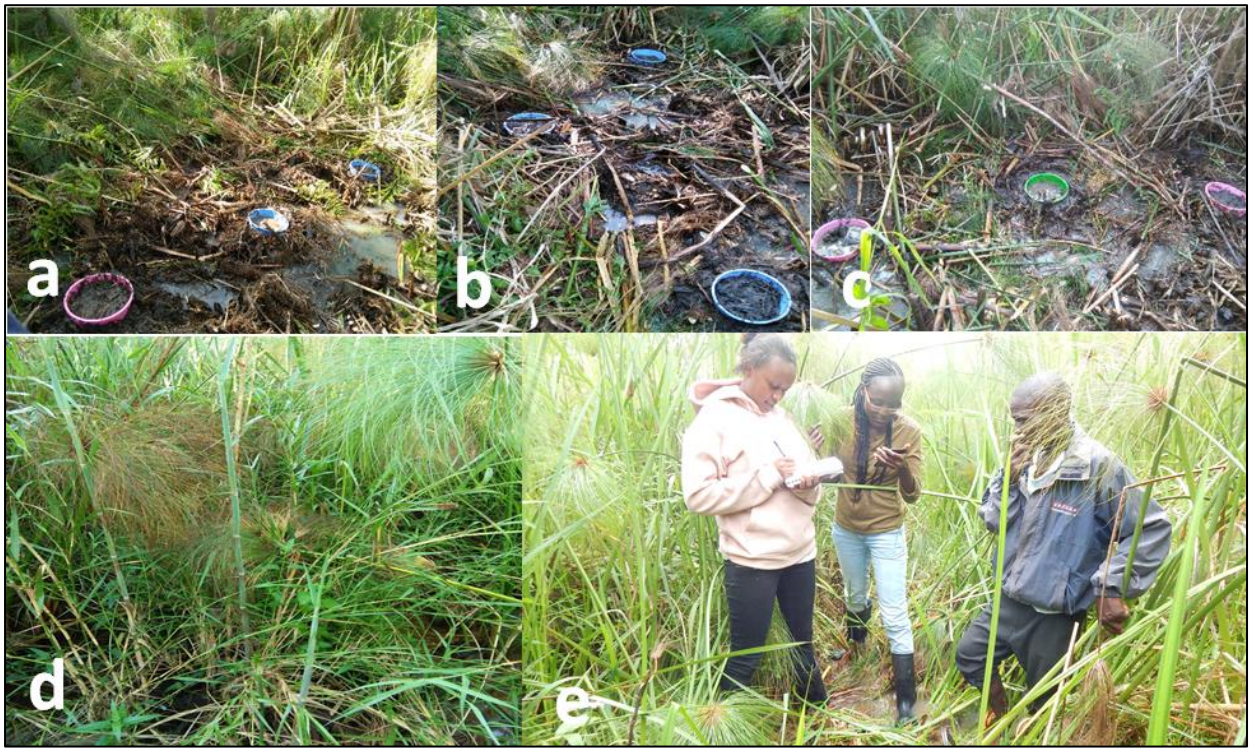

Figure 8: The status of URM at the time of chamber installation (a) URM 1: average water level 59cm, Location; 164 m from the edge, (b) URM 2: average water level 66cm, Location; 152m from edge, (c) URM 3: average water level 66cm, Location; 152m from edge, (d) papyrus stands densely populated, (e) site selection and mapping, also indicates the dense papyrus stands.

Lower reach outer edge (LRO) (LRO 1 Lat: 0°14’40.9’’ N, Long: 34°17’39.3’’E, LRO 2 Lat: 0°14’41.5’’ N, Long: 34°17’37.3’’E, LRO Lat: 0°14’40.8’’ N, Long: 34°17’39.2’’E): located in the lower sections of unconverted area, it was characterized by relatively dense population of the papyrus (Figure 9 d & e). At the start of the study, the site had shallow pools of water having an average water level of 24 cm (Figure 9 a, b & c), however, site dried out with time having no water above the soil by first sampling date. Throughout the study period, the site remained dry.

Figure 9: Characteristics of LRO at the time of chamber installation (a) LRO 1 Average water level, 24 cm, Location; 11 m from the outer edge, (b) LRO 2 Average water level; 27cm, Location; 13m from the outer edge, (c) LRO 3 Average water depth 22 cm, Location; 9 m from the edge, (d) and (e) show dense stands of papyrus, that were cleared for chambers to be installed.

Lower reach middle section (LRM) (LRM 1 Lat: 0°14’41.3’’ N, long:34°17’40.0’’E LRM Lat: 0°14’41.15’’ N, Long: 34°17’39.8’’E, LRM 3 Lat: 0°14’40.8’’ N, Long: 34°17’40.1’’E) : This site was in the unconverted area, and it was characterized by flowing waters at the time of chamber installation and thick, well developed pure stands of papyrus (Figure 10 d & e). The soil was fully inundated during the installation of the chambers (Figure 10 a, b & c), but the water levels went down rapidly and by the end of the experiment the site had no water levels above the soil, but soil remained moist.

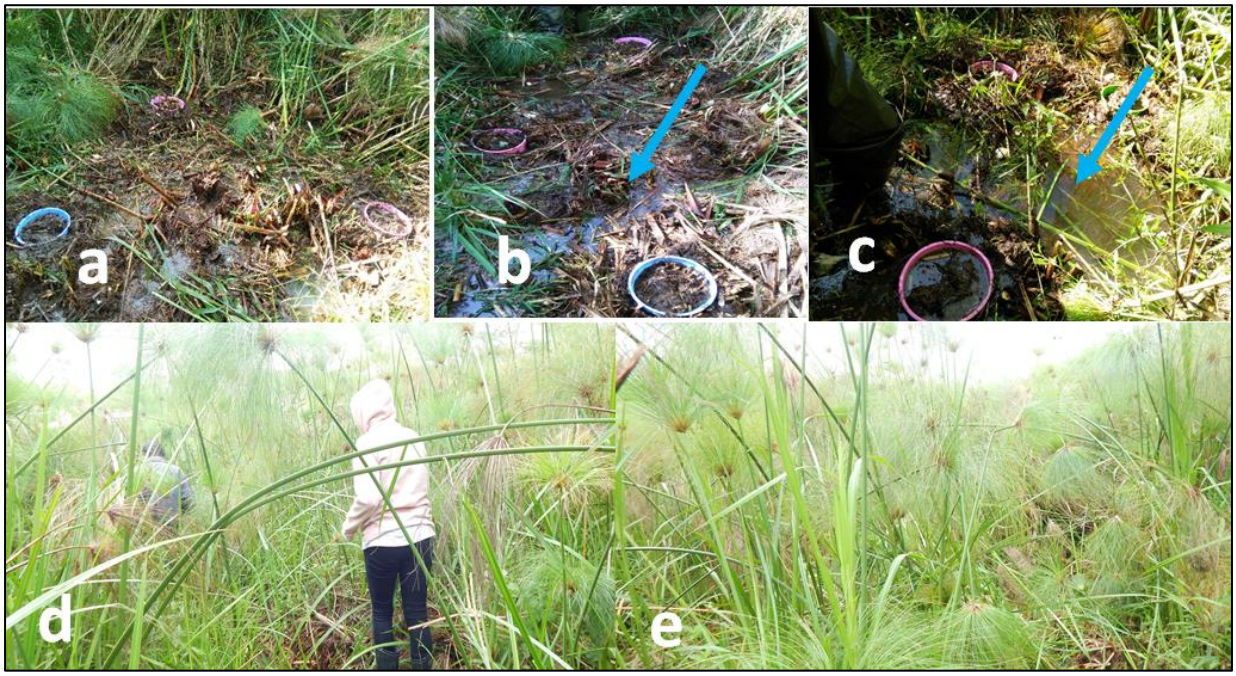

Figure 10: The characteristics of the sites (LRM) at the time of chamber installation(a) LRM 1 average water level 68cm, location; 37.5 m from the wetland edge, (b) LRM 2 average water level – 70cm, location; 40m from the outer edge, (c) LRM 3 average water level; 75 cm, location; 37.5m from the edge 2 m downstream from a, (d) and (e) indicate the dense population of papyrus that was cleared for the chambers to be installed.

3.3 Gas and soil sampling and analyses

Sampling took place for a period of two months (December 2017 to January 2018), covering the dry season in Kenya. The gases together with soil samples for analysis of NO_3-N, and NH_4-N were sampled weekly, whereas for OC, TN and TP soil samples were taken on three occasions, once in December and twice in January.

3.3.1 Soil sampling and analysis for TN, TP and OC

Soil samples were taken, at each sampling site using soil auger or a corer to a depth of 15 cm. Soon after, they were transferred into polythene bags, and placed in a cool box containing ice for transportation to the laboratory for further analysis. Standard procedures were followed to determine soil NO_3-N, NH_4-N, OC, TN, and TP. Nitrate-nitrogen and NH_4-N, were determined using 10 g of the fresh soil samples which were extracted with 100 ml of 1 M KCL (Rosenstock *et al.*, 2016). The samples were filtered and the supernatant was analysed

following the standard methods. Concentration was calculated from standard calibration curve determined from known concentrations (APHA, 2004).

Organic carbon content was determined by titration method. Soil samples were oven dried (70°C) to a constant weight. This was followed by complete oxidation of 0.3 g using 7.5 ml sulphuric acid and 10 ml aqueous potassium dichromate ($K_2Cr_2O_7$) mixture. The unused $K_2Cr_2O_7$, was titrated against ferrous ammonium sulphate to endpoint (colour change from greenish to brown). Difference between the added and residual $K_2Cr_2O_7$ gave the measure of OC content in soil (Okalebo *et al.*, 2002).

Total nitrogen was determined by acid digestion, followed by steam distillation then titration. From the oven dried (70°C) soil sample, 0.3 g were digested using 2.5 ml of digestion mixture (hydrogen peroxide + sulphuric acid + selenium + salicylic acid) at 360°C for two hours. Thereafter, an aliquot of 10 ml was transferred into a reaction chamber, 10 ml of 1% sodium hydroxide added then immediately steam distilled for two minutes into 5 ml of 1% boric acid. Finally, the distillate was titrated with N/140, hydrochloric acid until endpoint (colour change from green to definite pink). Total nitrogen was calculated following recommendations outlined by Okalebo *et al.*, (2002).

Total phosphorus was determined from the same digest solution as total nitrogen. Ascorbic acid method was used, where absorbance was read at 400 nm wavelength. Total P was determined using standard calibrated curve from known concentrations (Okalebo *et al.*, 2002).

3.3.2 Gas sampling and analysis for CO_2, CH_4 and N_2O

CO_2, CH_4 and N_2O fluxes were measured using vented, static chamber method, according to the minimum requirements for the GHG measurements as described by Parkin and Venterea, (2010) and Rosenstock *et al.*, (2016). The chambers were fabricated from 10 litre plastic buckets and they comprised of two parts (Figure 11 a, b & c): a base (diameter 25 cm height 15 cm) which was inserted 7-10 cm into the soil and a lid (diameter 25 cm height 25 cm) fitted with a gas sampling port, thermometer to measure chamber internal temperature, 50 cm long vent tube (2.5 mm diameter) and the top was lined with foam (Tully *et al.*, 2017). The lid was lined with an aluminium duct tape all round to minimize insulation and reduce the light going into the chamber since photosynthesis would affect the CO_2 fluxes.

Each site had nine gas chambers, where, three chambers were pooled into one sample vial, making three integrated replicates per site (Figure 11 e). The chamber bases were installed 5 to 7 cm into the ground, one week prior to the first measurement, and they remained in place until end of the experiment. Papyrus stands were cleared to provide ground for the base installation, hence there were rhizomes inside the bases. During sampling, the base and lid were clamped together using metallic clips to ensure they were airtight (Figure 11 d). In each sampling event, any growing shoots were cut to ground level before the measurements were taken. Gas sampling events took place weekly for a period of six weeks covering the dry season December 2017 and January2018.

Figure 11: (a) Fabricated chamber base, (b) fabrication of the chamber lid, (c) chamber base and lid, (d & e) gas sampling in the field showing the closed chambers, fitted with thermometers and vent tube.

In every sampling event, chambers were closed (Figure 11 d & e) for 30 minutes and gas samples were taken every 10 minutes giving a total of 4 samples per site (Table 1). Gas pooling technique from three chambers was applied to address the spatial heterogeneity, pooling (Arias-Navarro *et al.*, 2013). Gas was manually mixed before sample collection by pumping with a syringe several times to ensure homogeneity of the sample. Thereafter, gas sample

was collected using a 60 ml propylene syringe fitted with luer lock, where 20 ml sample was drawn from each of the three chambers per replica set and immediately transferred to a pre-labelled 10 ml glass vial with crimp seal. Flushing technique was used, where the first 30 – 40 ml was used to flush the vials, and the remaining filled the vial, over pressurizing it to minimize chances of leakage and contamination with ambient air. The samples were wrapped with parafilm over the crimp seal and transported to the International Livestock Research Institute (ILRI) laboratory for analysis within the next 12 hours after collection.

Table 1: Summary of the sampling design showing the sites and sampling frequency of the gas fluxes.

Transect	Site	Sampling frequency	No of samples	Total no. of samples
Converted area				
	Farm	6	(3*4) 12	72
Unconverted area				
Upper reach	URO	5	(3*4) 12	60
	URM	5	(3*4) 12	60
Lower reach	LRO	6	(3*4) 12	72
	LRM	6	(3*4) 12	72
	Total gas samples			**336**

Gas concentrations were analysed using the SRI GHG gas chromatograph (model 8610C; SRI) with a methanizer in combination with a flame ionization detector (FID) for CH_4 and CO_2, and a $_{63}$Ni electron capture detector (ECD) (Butterbach-Bahl *et al.*, 2011). Concentrations of the gases were calculated based on the peak areas measured by the GC relative to the peak areas measured from calibration gases. The ideal gas law, atmospheric pressure, internal chamber temperature and chamber volume measured during sampling were then used to convert the concentrations to mass per volume flux calculated following equation 1 (Butterbach-Bahl et al., 2011).

$$\text{Flux}_{GHG(mgm^{-2}h^{-1})} = \text{Ct} \times \left(\frac{\text{M}}{\text{Vm}}\right) \times \left(\frac{V_{ch}}{A_{ch}}\right) \times \left(\frac{273.15}{273.15+\text{t}}\right) \times \text{P} \times 60$$

[1]

Where: Ct = slope derived from the linear regression ($ppmmin^{-1}$) for CH_4 and CO_2 and ($ppbmin^{-1}$) for N_2O-N, M = molar weight ($gmol^{-1}$) (C=12 for CH_4 and CO_2, and N=28 for N_2O), Vm=molar gas volume ($m3mol^{-1}$), (22.41), V_{ch}= Volume of gas chamber, A_{ch}= Area of gas chamber, t= Chamber temperature (°C), P= Pressure at time of sampling (atm), 60= conversion factor of mins to hour.

3.4 Ancillary measurements

Environmental variables (soil temperatures, air temperature and bulk density), influencing GHG emission were measured, additionally the type of vegetation and weather conditions recorded. Air temperatures were measured using digital thermometer/hydrometers, while soil temperatures were taken using standard digital thermometers (Brannan) at depths 11 to 20 cm. Bulk density was determined on the upper 0 to 15 cm using the bulk density ring ($98cm^3$).

3.5 Statistical analysis

Data was tested for normality by Shapiro Wilk test and since it failed the normality test, even after log transformation non-parametric tests (Wilcoxon rank test and Kruskal Wallis) were conducted to test for differences between the converted and unconverted areas of the wetland and between the sites. Principal component analysis (PCA) was performed to determine the correlation of the variables and to show which components were contributing to the variations. Analysis was conducted using R studio (R version 3.4.3 Patched).

CHAPTER FOUR

RESULTS

4.1 General conditions of the sites

The mean air temperature was 36.76±1.39°C for the converted areas of the wetland and 33.61±1.06°C for the unconverted wetland areas. The soil temperature ranged from 22°C to 29.43°C in the converted area of the wetland, while in the unconverted area of the wetland ranged between 19.39°C to 28.04°C. Higher soil temperature was observed in the converted area of the wetland (25.39 ±1.51°C). Soil bulk density ranged from 0.63 to 1.33 gcm^{-3} in the converted area while in the unconverted area it ranged from 0.59 to 1.46 gcm^{-3}. Moisture content was higher in the unconverted area of the wetland (97.08±9.10%) than in the converted area (40.96±4.39%). Table 2 shows a summary (means ±standard error) of the ancillary variables measured by the study.

Table 2: Means ±SE for ancillary variables measured for the converted and unconverted areas of the wetland.

Site	Moisture content (%)	Soil temperature (°C)	Air temperature (°C)	Bulk density gm^{-3}
Farm	40.96± 4.39	23.78±0.45	36.76±1.39	1.05±0.03
URM	96.81±18.35	19.80±0.65	31.65±0.89	1.03±0.02
URO	100.84±22.96	20.94±0.51	34.35±1.49	1.04±0.01
LRM	106.36±17.95	20.35±1.27	32.93±1.68	1.05±0.03
LRO	82.38±16.07	20.08±1.23	35.09±1.13	0.99±0.03

4.2 Comparison of the soil C, N and P stocks in the converted and unconverted areas of the wetland

The unconverted areas of the wetland had a tendency of higher soil organic carbon and total phosphorus stocks (3.32±0.12% and 42.81±039 ppm) respectively, while the soil total nitrogen content (0.44±0.02%) was observed to be higher in the converted area of the wetland. Soil OC and TP in the converted and unconverted areas of the wetland showed no significant difference (Wilcoxon rank sum test, P > 0.05). Contrary, a significant difference was observed in soil TN stock between the converted and unconverted areas of the wetland (Wilcoxon rank sum test, W = 723, P <0.05) (Figure 12).

Soil nitrate-nitrogen was significantly higher in converted areas of the wetland (0.19±0.05 mg/l) compared to the unconverted area of the wetland (0.07±0.01 mg/l) (Wilcoxon rank sum test, P <0.05). A tendency of higher soil ammonia-nitrogen content was observed in the converted area of the wetland (2.97±0.61 mg/l), but no significant difference was observed (Wilcoxon rank sum test, P >0.05) (Figure 13).

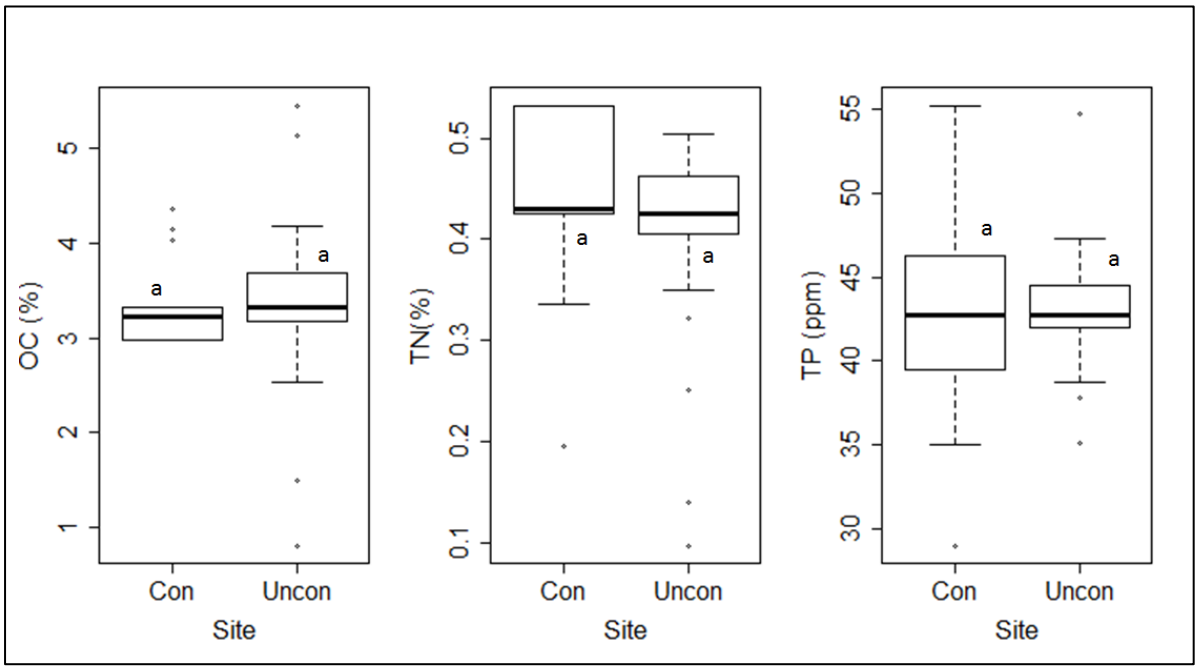

Figure 12: Comparison of the soil nutrient stocks (a) OC, (b) TN and (c) TP in the converted (Con) n=12 and unconverted n=42 (Uncon) areas of the wetland.

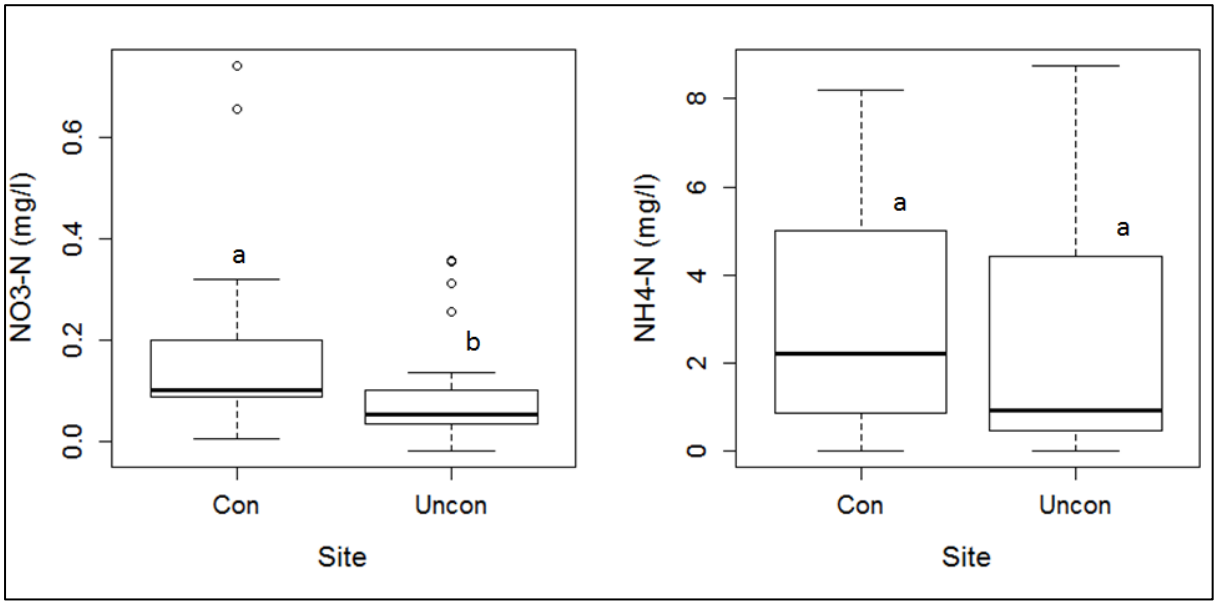

Figure 13: Comparison of the soil nutrient stocks (a) nitrate mg/l (b) ammonia mg/l in the converted (Con), n=15 and unconverted (Uncon) areas of the wetland, n=58. Letters indicate significance levels between the converted and unconverted area of the wetland; different letters denote significant difference, while similar letters indicate no significant difference.

4.3 Comparison of GHG (CH_4, CO_2 and N_2O) fluxes in the converted and unconverted areas of the wetland

Methane flux ranged between -0.07 to 0.09 $mgm^{-2}h^{-1}$ in the converted areas while in the unconverted areas it ranged from -5.32 to 40.59 $mgm^{-2}h^{-1}$. A significantly higher flux (5.34±0.008 $mgm^{-2}h^{-1}$) was observed in the unconverted areas of the wetland (Wilcoxon rank sum test, W = 134.5, P < 0.05). Carbon dioxide flux in converted and unconverted areas of the wetland ranged between 139.70 to 403.10 $mgm^{-2}h^{-1}$ and 74.65 to 608.80 $mgm^{-2}h^{-1}$ respectively. No significant difference in CO_2 flux was observed between the two areas (t-test, t = 0.448, P = 0.655). Nitrous oxide flux was however significantly higher in converted (187.06±25.41 $\mu gm^{-2}h^{-1}$) than in unconverted (24.28±5.00 $\mu gm^{-2}h^{-1}$) areas (Wilcoxon rank sum test, W = 1103, P <0.05) (Figure 14).

Pairwise comparison showed CH_4 fluxes in the converted area of the wetland (Farm) significantly differed from all the sites in the converted area of the wetland (URO, URM, LRM) (Wilcoxon rank sum test, p<0.05) except for LRO (Wilcoxon rank sum test, p>0.05). Along the longitudinal transect a significant difference in CH_4 flux was observed between LRO and upper reach sites (URO & URM) (Fig. 15). Nitrous oxide fluxes showed no significant difference along

the longitudinal gradient, while the converted area (Farm) significantly differed from all other sites in the converted area (URO, URM, and LRM) (Wilcoxon rank sum test, $p<0.05$) except LRO (Wilcoxon rank sum test, $p>0.05$) (Figure 15).

The converted area of the wetland acted as CH_4 sink, but a source for both N_2O and CO_2, while the unconverted area was a source for three gases. After the fluxes were summed up in their CO_2 equivalents, no significant difference was observed in overall GHG effect between converted and unconverted areas of the wetland (Wilcoxon rank sum test, W = 594, *p*-value = 0.8446) (Figure 16).

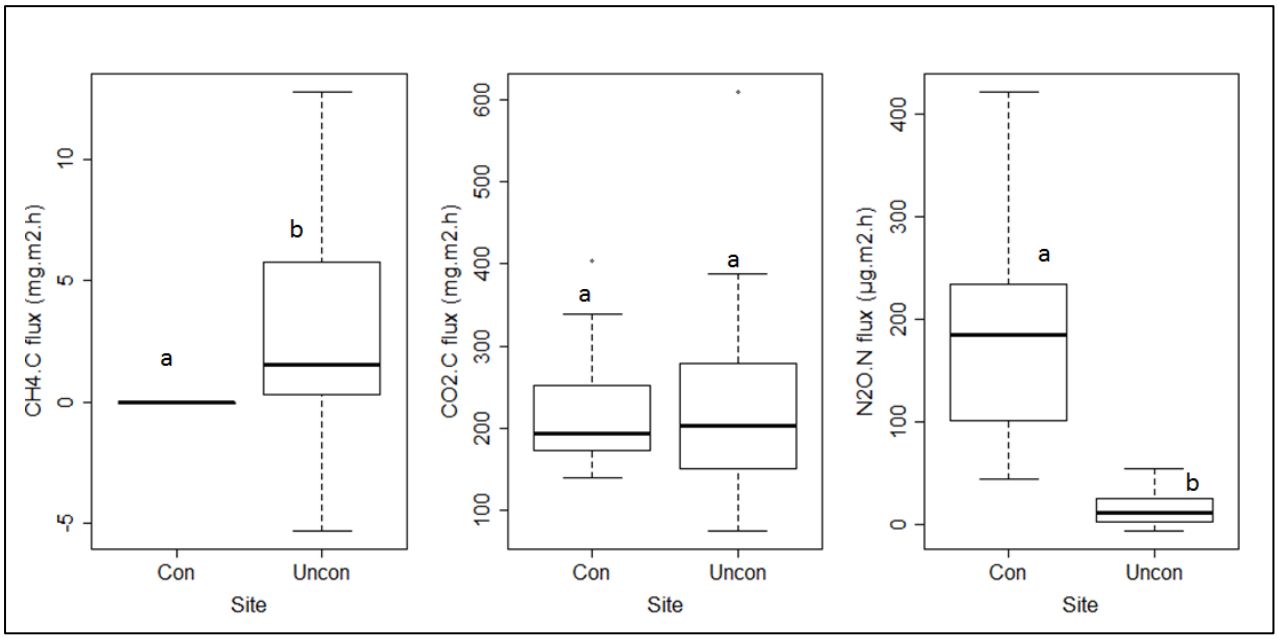

Figure 14:Comparison of the gas fluxes (a)CH4-C $mgm^{2-1-}h^{-1}$, (b)CO2-C $mgm^{2-1-}h^{-1}$ and (c)N2O-N $\mu gm^{2-1-}h^{-1}$ in the converted (Con) and unconverted (Uncon) areas of the wetland, n=6. Letters indicate significance levels between the converted and unconverted area of the wetland; different letters denote significant difference, while similar letters indicate no significant difference.

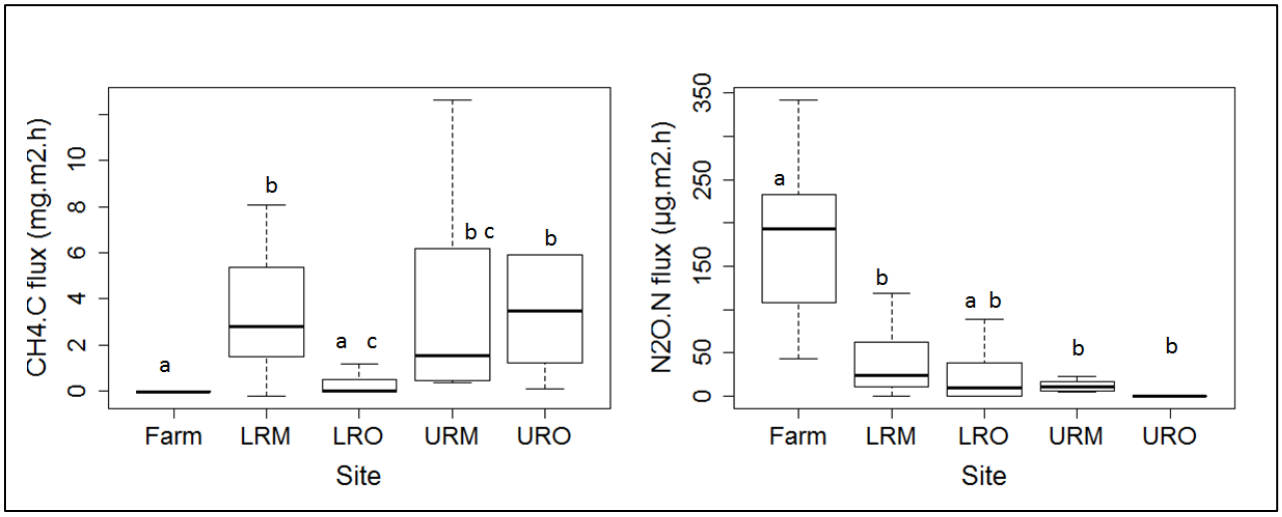

Figure 15: Pairwise comparison of the individual sites of the wetland, Letters indicate significance levels between the converted and unconverted area of the wetland; different letters denote significance difference, similar letters indicate no significance difference. Farm = converted area, LRO and LRM are sites along the longitudinal transect in the lower reach, URO and URM are sites along the longitudinal transect in the upper reach.

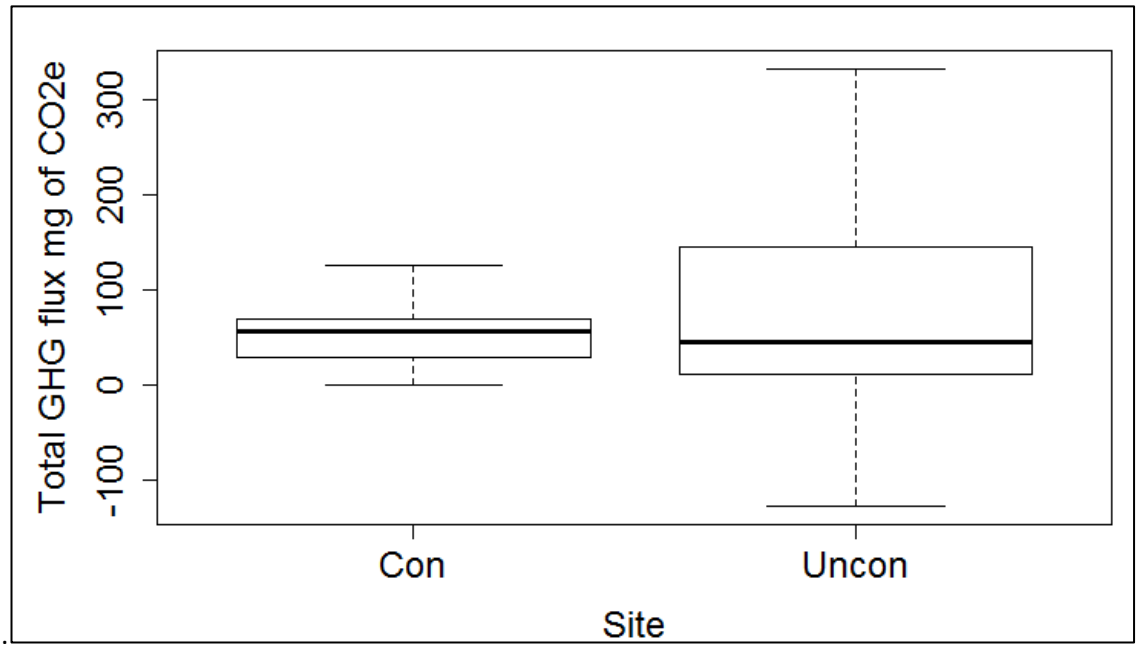

Figure 16: Total effect of the greenhouse gases summed up in mg CO_2e for the converted (Con) n=18 and unconverted (Uncon) areas of the wetland, n=63.

4.3.1 Temporal variations in the GHG gas fluxes

Significant variations were observed for the short-term temporal patterns of CH_4, N_2O and CO_2 (Kruskal Wallis, d.f = 5, P= 0.033, 0.0087, 0.0018, respectively) (Figure 17). A significant

difference was observed between the converted and unconverted area during the different sampling campaigns. Methane was significantly different for the second (S2) and fourth sampling (S4) (Wilcoxon rank sum test, P=0.0044, 0.0307 respectively. For CO_2 fluxes a significant difference was observed for the sixth sampling (S6) (Wilcoxson rank sum test, *P*= 0.049), while the N_2O fluxes were significantly different for sampling dates S2 to S6 (Table 2).

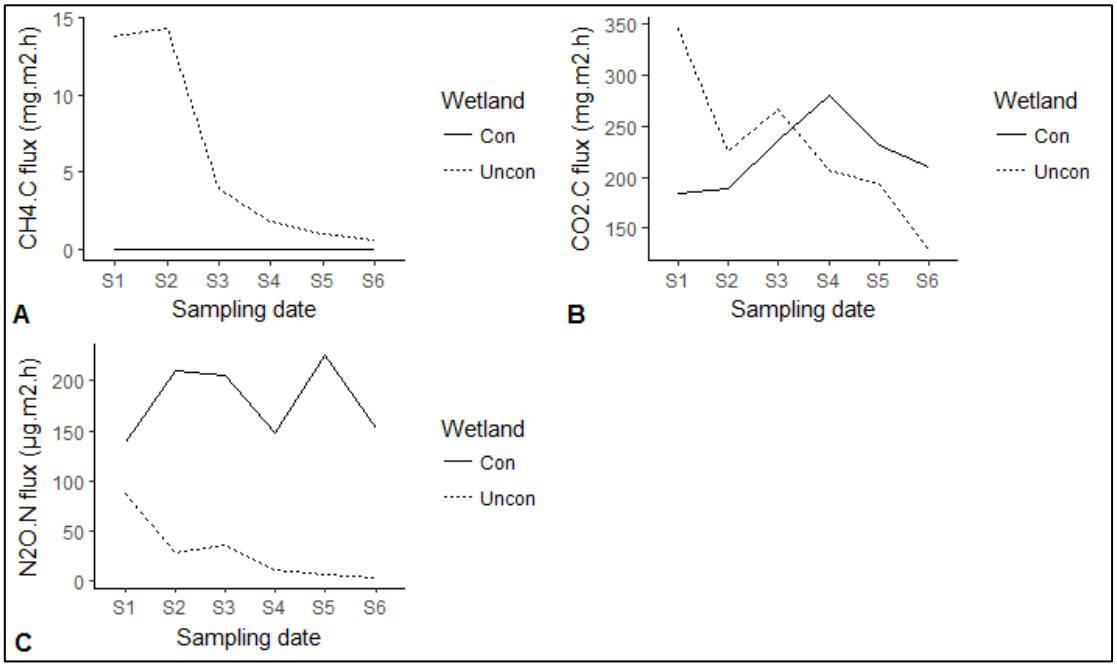

Figure 17: Temporal variations of the GHG fluxes in the converted (n=18) and unconverted (n=64) areas. S1 to S6 represent the sampling dates in a chronological order (S1 = 13/12/2017, S6 = 24/01/2018).

Table 3: Temporal variations in converted (n = 18) and unconverted (n = 64) areas of the wetland on different sampling dates.

Sampling date	CH4	CO2	N2O
S1	0.1667	0.1667	0.381
S2	0.0044 ×	0.8396	0.0043 ×
S3	0.014	0.6333	0.0171 ×
S4	0.0307 ×	0.3648	0.0108 ×
S5	0.2331	0.7341	0.0108 ×
S6	0.2043	0.049 ×	0.0137×

× Significant difference (Wilcoxon rank sum test)

4.4 Correlation between the soil nutrient stocks and the GHG fluxes in converted and unconverted areas

Principle component analysis, showed that the first two components explained more than 50% of the total variation (Dim 1: 36 and Dim 2: 16.9% respectively). The first component (Dim.1) described environmental conditions and reactions affecting nitrification and denitrification (N_2O, soil temperature, nitrate, ammonia, and air temperatures). The second component (Dim.2) was mainly explained by variables affecting organic matter build up and oxidation (OC, CO_2, CH_4, and soil moisture content) (Figure 18). The proportion of variance and the Eigen values for the PC are shown in Table 4 while contribution by each variable is shown in Table 5.

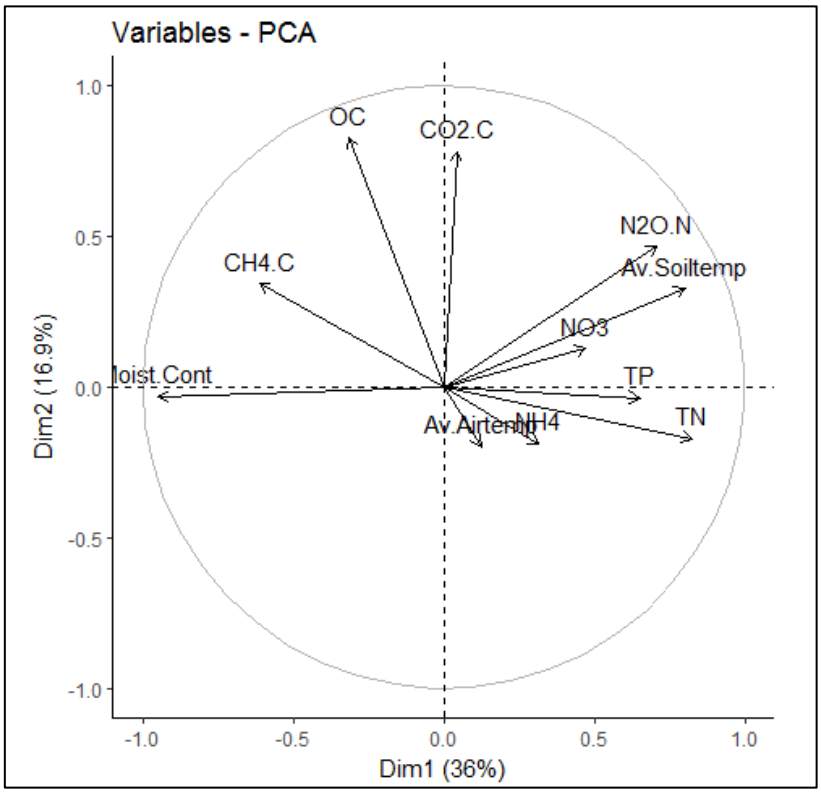

Figure 18: PCA correlation diagram of the environmental variables. Environmental variables: Moist.Cont = soil moisture content, CH_4.C= CH_4 ($mgm^{-2}h^{-1}$), OC= Organic carbon (%), CO_2.C =CO2 ($mgm^{-2}h^{-1}$), N_2O.N= N_2O ($\mu gm^{-2}h^{-1}$), Av.Soiltemp =average soil temperature (°C), NH_4 =ammonia (mg/l), Av.Airtemp =average air temperature (°C), NO_3= Nitrate (mg/l), TN = total nitrogen (%), TP= total phosphorus (ppm).

Table 4: Eigen values and the proportion variance explained by each principle component

	Eigenvalue	Variance percent	Cumulative variance percent
Dim.1	3.959898	35.99908	35.99908
Dim.2	1.859968	16.9088	52.90787

Table 5: Contribution of the variables to the Principal components.

	Dim.1	Dim.2
CH4.C	3.902042	1.487538
CO2.C	0.98709	1.495156
N2O.N	9.637206	6.058473
NO3	5.184839	20.61103
NH4	2.67039	17.12996
OC	1.208891	32.88221
TN	11.53854	8.169384
TP	15.68953	0.033644
Moisture content	18.70415	0.224905
Av. soil temp	18.44506	0.196118
Av. Air temp	12.03227	11.71158

CHAPTER FIVE

DISCUSSION

5.1 Nutrient stocks in the wetland soils

Wetlands are characterized by anoxic and water-logged conditions which allow for build-up of OM, because of the slow decomposition rate (Brown *et al.*, 2017). In this study, a decline in soil OC stocks (3.31±0.10%) was observed in the converted area of the wetland. Conversion of the wetland into farmlands, increase oxygen diffusion and hence increased OM oxidation, which could explain the lower values of OC in the converted areas of the wetland compared to unconverted areas (3.32±0.12%). Most studies on tropical wetlands have addressed, hydrology, community structures and species diversity (Bernal and Mitsch, 2013), limited studies have been conducted on the effects of conversion of the wetlands into other uses. For instance, Dixon and Wood, (2003), reported that wetland cultivation impacted negatively the hydrology of wetlands resulting to lower availability of water. Moreover, the study suggested drainage and cultivation of the wetland may lead to unsustainability and lead to ecosystem disservices such as reduced water storage capacity.

In temperate and boreal regions, several studies have reported a reduction in OC following wetland drainage (Bernal and Mitsch, 2012). For instance, a 25% loss in OC was reported in Australian wetlands by Page and Dalal, (2011). Other studies have reported lower SOC in drained wetlands in comparison to undrained wetlands, and loss of OC through cultivation (Euliss *et al.*, 2006; Kumar *et al.*, 2014; Streeter and Schilling, 2015; Ma *et al.*, 2016). Nag et al., (2017) in their study on a riparian marsh wetland in Ohio, observed measurements of OC close to this study, where C was highest in open water wetland, compared to vegetated and upland wetlands (5, 3.25, and 2.65%, respectively).

Higher measurements of TN, NO_3-N and NH_4-N content (0.44%±0.02, 0.19±0.48 and 2.97±0.61 mg/l respectively) were observed in the converted area of the wetland, which was characterized by dry soils, considered to be aerobic. Higher NO_3-N and NH_4-N content observed, in the converted area of the wetland could be associated to mineralization, which tends to occur in warmer and more aerobic conditions (Booth *et al.*, 2005). A study by Venterink as quoted by Brown *et al.,* (2017) found that drying of wetland soils increases

inorganic N and DON and triples the mineralization rates. Assimilation of nitrogen by microorganisms and mineralization of organic N to NH_3 are the two major biological processes responsible for nitrogen transformations in soil (Booth *et al.*, 2005). Balance between assimilation and mineralization is dependent on the C/N ratio of the soil. According to Signor and Cerri, (2013), a small C/N ratio (lower than 30:1), present a dominance of mineralization over assimilation and thence higher available N used for microbial processes. Similarly, a study by Zeshan *et al.*, (2012), while testing effect of C/N ratio in a thermophilic anaerobic digester reported that simulation with C/N ratio of 32 had 30% less ammonia in the digestate than the simulation with C/N ratio of 27. The C/N ratio in the converted area of the wetland ranged from 5:1 to 22:1, while in the unconverted area the C/N ratio ranged from 11.06:1 to 41.76:1. The lower C/N ratio in the converted area of the wetland could be explained by high mineralization and consequently the higher concentrations of NH_4-N and NO_3 -N in the converted area.

Soil TP content (42.81±0.39 ppm) was observed to be higher in the unconverted area of the wetland. The unconverted area of the wetland was characterised by dense stands of papyrus, with massive root volumes, thus this could be a reason for the higher content of TP observed. Vegetation and detritus accumulation are among the factors that influence P accumulation (Reddy *et al.*, 2000). Correspondingly, P has been reported to have a greater potential for retention in many ecosystems, mainly due to strong sorption of P by many soils in addition to the fact that P has fewer pathways through which it can be lost (McLauchlan, 2006). Erosion, uptake of P by leguminous plants and removal of plant biomass are the major mechanisms associated with P loss in agricultural soils (Johnston, 2003). In this regard, the lower TP content in the converted area can be associated to plant biomass loss, through the complete removal of above and below ground papyrus biomass to make the fields for cocoyam production.

5.2 Greenhouse gas fluxes in converted and unconverted areas of a wetland

Converted area of the wetland appeared to be a sink for CH_4, characterized by negative fluxes, while it was a source for both, CO_2 and N_2O fluxes. Methane is produced by methanogens under anaerobic conditions in water saturated soil, but converted areas of the wetland was characterised by dry soils and probably high in oxygen concentration. Drainage of wetlands

increases the oxygen diffusion and reduces the soil moisture content favouring aerobic conditions, therefore supporting methanotrophs. A study by Gondwe and Masamba, (2014) in a tropical delta wetland in Botswana, reported higher CH_4 emissions in the flood plains and lagoon, which is consistent with this study. Unconverted area had higher measurement of the methane fluxes (Figure14), in addition to higher soil moisture content and OC (96.25%±9.13 and 3.32%±0.12 respectively). Other studies have reported higher fluxes of CH_4 from natural wetlands in boreal and temperate regions (Smith *et al.*, 2004; Tangen *et al.*, 2015; Veber *et al.*, 2017). Higher fluxes in the unconverted area could be attributed to the unconverted area of the wetland having higher soil moisture content and OC content, conditions that are favourable for proliferation of the methanogens and consequently methanogenesis (Veber *et al.*, 2017).

A significant difference in CH_4 flux was observed along the transverse wetness gradient of intermittent wet LRO, (2.27±1.20 $mgm^{-2}h^{-1}$) to permanent wet LRM (5.13±1.91 $mgm^{-2}h^{-1}$) (Figure 15). Altor and Mitsch, (2006) in their study at Schiermeier Olentangy River Wetland Research Park at the Ohio State University, USA, reported similar results of higher methane fluxes in permanenlty inundated areas than in the intermitently flooded areas. Correspondingly, a study by Mishra *et al.*, (1997) observed that earthenware pots planted with rice that were permanently flooded with water emmitted higher CH_4 than pots that were intermitentely inundated. Higher CH_4 emission in inundated areas could be attributed to high productivity from vegetation which in turn provide substrate for methanogens, where substrate availability is considered the limiting factor of methanogenesis at anoxic conditions (Smith *et al.*, 2000; Whalen, 2005).

A negative correlation between methane and soil temperature (Figure 18) was a noteworthy observation in this study, which contrast other studies. Generally, methanogenesis and methane consumption are temperature dependent processes (Dobbie and Smith, 2001). Positive correlation between CH_4 emission and soil temperature was reported in previous studies (Dobbie and Smith, 2001; Gondwe and Masamba, 2014). Furthermore, temperature influences mineralization of organic matter, consequenltly resulting in faster depletion of alternative electron acceptors, in addition to enhanced archeal CH_4 production in anoxic soils and bacterial oxidation at the oxic-anoxic interface (Marinho *et al.*, 2009). The negative

correlation observed in this study could probably be explained by methanogenesis being influenced more by moisture content rather than temperature.

Unlike methane, CO_2 fluxes in the converted and unconverted areas of the wetland were not significantly different. Both converted and unconverted areas of the wetland were dry, which may explain the observed results. Drainage of saturated wetland soils in addition to its natural dryness result in increased oxygen diffusion, translating to higher rates of decomposition of OC, consequently an increase in CO_2 emissions (Heikkinen *et al.*, 2002; Mitsch *et al.*, 2013). Consistent with findings of this study, Gao *et al.*, (2009) observed an increase in CO_2 emission when water saturation declined in Zoige Alpine wetland soils. A study by Veber *et al.*, (2017), in natural and managed peatlands in America reported higher CO_2 emissions median (572.4 $mgm^{-2}h^{-1}$) in the managed site compared to the natural site (26.8 $mgm^{-2}h^{-1}$). These results are consistent to this study where higher CO_2 flux measurements were higher in the converted area.

Phosphorus is a key element for growth of microorganisms (Reddy et al., 2000). In the study of Wright and Reddy (2001), P was positively correlated to CO_2 flux, which was linked to increased soil microbial heterotrophic and autotrophic respiration. However, in this study, CO_2 was negatively correlated to P (Figure 18). The negative correlation observed in this study would lead to assumption that P is not the limiting nutrient in the study area. Accordingly, a study by Lukito *et al.*, (1998) defines existence of critical P concentration of microbial biomass, where above this critical P concentration, microbial activity is normally decreased.

Significantly higher N_2O flux was observed in the converted area of the wetland (187.06±25.41 $\mu gm^{-2}h^{-1}$), as opposed to the hypothesized higher emission in the unconverted area of the wetland. The availability of N in soil (NH_4 and NO_3) influences emission of N_2O, because they are the key nutrients required for the microbial processes of nitrification and denitrification through which N_2O can be produced and released into the atmosphere (Cowan *et al.*, 2015; Davidson and Verchot, 2000). In general, N_2O is produced by nitrifying microorganism under aerobic conditions, while denitrifying microorganisms are responsible for N_2O production in anaerobic conditions (Chapuis-lardy et al., 2007). Although denitrification is considered responsible for most of the N_2O emission from soils, under low oxygen conditions nitrifying microorganisms can contribute considerably to N_2O emissions (Signor and Cerri, 2013).

Denitrification involves reduction of NO_3 to the gaseous forms of N, while nitrification involve oxidation of NH_4 to NO_3, with gaseous form of N as by products representing the loss of N from soils into atmosphere (Sirivedhin and Gray, 2006).

According to Signor and Cerri, (2013), soil OC content increases the production of N_2O, relating this to increased decomposition where microbial growth and activity is stimulated. The converted area of the wetland may have increased decomposition rates because of the drained, well aerated soils, consequently higher N_2O flux. Furthermore, denitrification and nitrification processes are influenced by the available C in soil (Bremner, 1997). This is due to increased microbial activity which results in increased consumption of oxygen creating anaerobic conditions required for denitrification. Carmo *et al.*, (2005) in their study in Amazonian soils, observed a peak of N_2O emission after inputs of labile C and plant residues to soil.

Based on the PCA analysis (Figure 18), a strong relationship was observed between N_2O and soil temperatures. Both nitrification and denitrification are dependent on soil temperatures, moisture content and oxygen availability (Booth et al., 2005). Nitrification is the dominant process in well aerated, dry soils whereas denitrification is an anaerobic process, in saturated soils. Studies by Booth *et al.*, (2005) and Bedard-Haughn *et al.*, (2006) reported similar observations where, mineralization and nitrification in cultivated wetland soils were positively correlated to soil temperatures. Besides influencing nitrification and denitrification processes, soil temperatures alongside moisture content have also been reported to influence N_2O production and diffusion into the atmosphere (Signor and Cerri, 2013).

The global warming potential differs for the various GHG, and for that reason their effect cannot be compared on a mole to mole basis or volume to volume (Forster *et al.*, 2007). Carbon dioxide equivalents are used as the units of measure for the GWP effect for the gases, where over a 100 year span the GWP for CO_2, CH_4 and N_2O are 1, 25,and 298 respectively (Solomon *et al.*, 2007). The GWPs were applied to the fluxes measured in this study, to evaluate the overall effect of GHG production on climate change (Figure 16). The two areas showed no significant difference, probably because of the low number of observations in this study. Another probable explanation to this would be the fact that this study was conducted during the dry season. During study period, the unconverted area hydrological conditions

tended towards the converted area condition. Moisture content reduced significantly (Kruskal Wallis, $p < 0.05$) from start of experiment progressively towards the end. Reduced moisture content results to increased diffusion of atmospheric oxygen and consequently hindering methanogenesis (McLain and Ahmann, 2008).

Gas fluxes varied significantly over the short study period during dry season. This may be associated to the disturbance effect on the unconverted areas of the wetland, while installing the chambers, since papyrus stands were cleared to allow room for the chamber base installation. Moreover, unconverted areas of the wetland dried out gradually with time showing significant reduction in moisture content, creating more aerobic conditions. Methane showed a distinct pattern of reduction with time (Figure 17a), but CO_2 and N_2O patterns were not distinct (Figure17 b &c).

Different agriculture management practices influence the GHG emission in different ways (Tangen et al., 2015; Wang et al., 2017). This study focus was on land use without intensive agricultural management, meaning no fertilizer was added to the agriculture fields. This is evident in the limited effect observed in the converted area of the wetland, confirmed by lack of statistical differences between converted and unconverted areas, in total greenhouse gas effect as well as OC and TP stocks. Addition of N fertilizer has been observed to increase N_2O emissions. But varying effects on CH_4 emissions have been reported, if the fertilizer applied contained urea. Increase in N_2O and decrease in CH_4 was reported when urea fertilizer was added to rice paddies (Bosse and Frenzel, 1998; Cai et al., 2007; Zou et al., 2005), while an increase in both N_2O and CH_4 was reported by Cai et al., (2007) and Das and Adhya, (2014). Mean CH_4 flux observed in the unconverted area (5.34±0.008 $mgm^{-2}h^{-1}$), whose characteristics are similar to the rice paddies, was lower compared to the observed mean flux under intensively managed rice paddies 25.6 $mgm^{-2}h^{-1}$. This could be an indication that under different management practices the effect on GHG emission would be different, may be higher or lower depending on the intensity of the management and the water regimes.

Under different nitrogen fertilization loads, where 300 kg/ ha, was considered as a high load was reported to increase N_2O and CH_4 emissions, whereas low loads 150 kg/ ha was observed to reduce N_2O and CH_4 emissions in Chongming Island, Shanghai (Zhang et al., 2014). Furthermore, addition of P and N fertilizer were found to influence gross primary productivity

as well as ecosystem respiration in a Swedish peatland having different N deposition rates (Lund et al., 2009). Review by (Snyder et al., 2009) indicated that (i) best management practises for N fertilizer play a critical role in minimizing soil nitrate residual which aid in lowering the risk of increased N2O emission; (ii) influence of N fertilization on N2O emission is dependent upon site and weather conditions; (iii) appropriate use of N fertilizer supports increase in biomass production and consequently restoration and maintenance of SOC levels; (iv) soil tillage practices of that do not disturb the soil, maintains crop residue on the surface subsequently increasing SOC levels, and finally (v) intensively managing crop systems may not necessarily lead to increased GHG emission per unit of crop or production; but they may aid in conserving natural ecosystems from being converted to cropland and allow restoration of converted natural ecosystems into their original form for mitigation of GHG.

6. Conclusion

Higher carbon and phosphorus stocks were observed in the unconverted area in line with the hypothesis of this study. However, nitrogen stocks for all the N species (TN, NO_3-N, and NH_4-N) were all higher in the converted area of the wetland contradictory to the hypothesis of higher N in the unconverted area of the wetland.

The converted area of the wetland is a probable sink of CH_4, but on the other hand a source for CO_2 and N_2O. Drainage of wetlands alters its hydrological connectivity, as a result increases oxidation of SOM, resulting in increased emissions of CO_2. Increased availability of OC result in increased mineralization of N, subsequently increased N_2O emissions.

Carbon and nutrient influence the GHG gas emissions. Organic carbon is strongly correlated to CH_4 and CO_2, while N_2O was strongly correlated to nitrate, ammonia and total nitrogen. Other abiotic factors also showed strong correlation with the GHG fluxes, a high correlation was found for moisture content and CH_4 and CO_2, while N_2O showed a strong correlation with soil temperatures.

7. Limitations

This study was carried out within a short period which coincides with the dry season only, therefore seasonal variations could not be studied.

The study was based on a comparison between the converted and unconverted areas of the wetland, hence mainly focused on quantification of the stocks and the gas fluxes. Nutrient stocks and GHG fluxes are dependent on other variables both biotic and abiotic. An assessment of microorganisms' composition, oxygen levels in the soil and soil pH, in addition to the ancillary measurements addressed by this study (soil bulk density, air temperature, and soil temperature), may have added information to explain the results. Time and availability of funds, in addition to lack of equipment such as soil dissolved oxygen meter were some of the factors that hindered analysis of the additional variables.

Only one land use type was studied, i.e. production of cocoyam without intensive management. A comparison of different land uses could give a broader picture of how the change in land use/cover influence the change in nutrient stock and GHG emissions.

Furthermore, future studies should consider different types of management, since addition of organic and inorganic fertilizers has different levels of impacts on GHG emission as well on nutrient stocks.

During the study, it was brought to attention that the conversion of the wetland to cropland happened at different periods, meaning a spatial and temporal variation in the converted area, should be incorporated in the design to show the effects over time.

8. Recommendations

Seasonal changes influence the wetlands hydrology, therefore the study on GHG fluxes should be carried out for at least one year to capture the seasonal and temporal variations.

Despite costs being a boundary condition for greenhouse gases study, other inexpensive methods are recommended such as modelling. However, to be able to effectively use the prediction models, direct measurements of the fluxes are necessary to provide data for model verifications.

References

Altor, A.E., Mitsch, W.J., 2006. Methane flux from created riparian marshes: Relationship to intermittent versus continuous inundation and emergent macrophytes. Ecol. Eng. 28, 224–234. https://doi.org/10.1016/j.ecoleng.2006.06.006

APHA, 2004. APHA: Standard Methods for the Examination of Water and Wastewater. Am. Public Heal. Assoc. Water Work. Assoc. Environ. Fed. 552.

Arias-Navarro, C., Díaz-Pinés, E., Kiese, R., Rosenstock, T.S., Rufino, M.C., Stern, D., Neufeldt, H., Verchot, L. V., Butterbach-Bahl, K., 2013. Gas pooling: A sampling technique to overcome spatial heterogeneity of soil carbon dioxide and nitrous oxide fluxes. Soil Biol. Biochem. 67, 20–23. https://doi.org/10.1016/j.soilbio.2013.08.011

Batson, J., Noe, G.B., Hupp, C.R., Krauss, K.W., Rybicki, N.B., Schenk, E.R., 2014. Journal of Geophysical Research: Biogeosciences 77–95. https://doi.org/10.1002/2014JG002817.Received

Bedard-Haughn, A., Matson, A.L., Pennock, D.J., 2006. Land use effects on gross nitrogen mineralization, nitrification, and N2O emissions in ephemeral wetlands. Soil Biol. Biochem. 38, 3398–3406. https://doi.org/10.1016/j.soilbio.2006.05.010

Bernal, B., Mitsch, W.J., 2013. Carbon sequestration in freshwater wetlands in Costa Rica and Botswana. Biogeochemistry 115, 77–93. https://doi.org/10.1007/s10533-012-9819-8

Bernal, B., Mitsch, W.J., 2012. Comparing carbon sequestration in temperate freshwater wetland communities. Glob. Chang. Biol. 18, 1636–1647. https://doi.org/10.1111/j.1365-2486.2011.02619.x

Boone, R.D., Grigal, D.F., Sollins, P., Ahrens, R.J., Armstrong, D.E., Fraterrigo, J., Turner, M., Pearson, S.M., Dixon, P., Guo, L.B., Gifford, R.M., Harrison, R.B., Footen, P.W., Strahm, B.D., Rowe, R., Sohi, S., Vanguelova, E., Mcnamara, N.P., Klironomos, J.N., Rillig, M.C., Allen, M.F., Liebig, M.A., Johnson, H.A., Hanson, J.D., Frank, A.B., Pennock, D.J., Wei, J., Xiao, D., Zhang, X., Li, X., 2005. Designing field studies in soil science. Ecol. Monogr. https://doi.org/10.4141/S03-039

Booth, M.S., Stark, J.M., Rastetter, E., 2005. Controls on nitrogen cycling in terrestrial ecosystems: A synthetic analysis of literature data. Ecol. Monogr. 75, 139–157. https://doi.org/10.1890/04-0988

Borges, A. V., Darchambeau, F., Teodoru, C.R., Marwick, T.R., Tamooh, F., Geeraert, N., Omengo, F.O., Guérin, F., Lambert, T., Morana, C., Okuku, E., Bouillon, S., 2015. Globally significant greenhouse-gas emissions from African inland waters. Nat. Geosci. 8, 637–642. https://doi.org/10.1038/ngeo2486

Bosse, U., Frenzel, P., 1998. Methane emissions from rice microcosms: The balance of production, accumulation and oxidation. Biogeochemistry 41, 199–214. https://doi.org/10.1023/A:1005909313026

Bremner, J.M., 1997. Sources of nitrous oxide in soils. Nutr. Cycl. Agroecosystems 49, 7–16. https://doi.org/10.1023/A:1009798022569

Bridgham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C., 2006. The carbon balance of North American wetlands. Wetlands 26, 889–916. https://doi.org/10.1672/0277-5212(2006)26[889:TCBONA]2.0.CO;2

Brown, R., Zhang, Z., Comeau, L.P., Bedard-Haughn, A., 2017. Effects of drainage duration on mineral wetland soils in a Prairie Pothole agroecosystem. Soil Tillage Res. 168, 187–197. https://doi.org/10.1016/j.still.2016.12.015

Bullock, A., Acreman, M., 2003. The role of wetlands in the hydrological cycle. Hydrol. Earth Syst. Sci. 7, 358–389. https://doi.org/10.5194/hess-7-358-2003

Butterbach-Bahl, K., Kiese, R., Liu, C., 2011. Measurements of biosphere atmosphere exchange of CH4in terrestrial ecosystems, 1st ed, Methods in Enzymology. Elsevier Inc. https://doi.org/10.1016/B978-0-12-386905-0.00018-8

Butterbach-Bahl, K., Sander, B.O., Pelster, D., Díaz-Pinés, E., 2016. Quantifying greenhouse gas emissions from managed and natural soils, in: Methods for Measuring Greenhouse Gas Balances and Evaluating Mitigation Options in Smallholder Agriculture. pp. 71–96. https://doi.org/10.1007/978-3-319-29794-1_4

Cai, Z., Shan, Y., Xu, H., 2007. Effects of nitrogen fertilization on CH4 emissions from rice fields. Soil Sci. Plant Nutr. 53, 353–361. https://doi.org/10.1111/j.1747-0765.2007.00153.x

Chapuis-lardy, L., Wrage, N., Metay, A., Chotte, J.L., Bernoux, M., 2007. Soils, a sink for N2O? A review. Glob. Chang. Biol. https://doi.org/10.1111/j.1365-2486.2006.01280.x

Clarkson, B.R., Ausseil, A.E., Gerbeaux, P., 2013. Wetland Ecosystem Services, Ecosystem services in New Zealand - conditions and trends.

Cobo, J.G., Dercon, G., Cadisch, G., 2010. Nutrient balances in African land use systems across different spatial scales: A review of approaches, challenges and progress. Agric. Ecosyst. Environ. 136, 1–15. https://doi.org/10.1016/j.agee.2009.11.006

Collier, S.M., Ruark, M.D., Oates, L.G., Jokela, W.E., Dell, C.J., 2014. Measurement of Greenhouse Gas Flux from Agricultural Soils Using Static Chambers. J. Vis. Exp. 1–8. https://doi.org/10.3791/52110

Cowan, N.J., Norman, P., Famulari, D., Levy, P.E., Reay, D.S., Skiba, U.M., 2015. Spatial variability and hotspots of soil N2O fluxes from intensively grazed grassland. Biogeosciences 12, 1585–1596. https://doi.org/10.5194/bg-12-1585-2015

Craswell, E., Lefroy, R., 2001. The role and function of organic matter in tropical soils. Nutr. Cycl. Agroecosystems 61, 7–18. https://doi.org/10.1023/A

Cui, Q., Wang, X., Li, C., Cai, Y., Liu, Q., Li, R., 2015. Ecosystem service value analysis of CO2management based on land use change of Zoige alpine peat wetland, Tibetan Plateau. Ecol. Eng. 76, 158–165. https://doi.org/10.1016/j.ecoleng.2014.03.035

Das, S., Adhya, T.K., 2014. Effect of combine application of organic manure and inorganic fertilizer on methane and nitrous oxide emissions from a tropical flooded soil planted to rice. Geoderma 213, 185–192. https://doi.org/10.1016/j.geoderma.2013.08.011

Davidson, E.A., Verchot, L. V., 2000. Testing the hole-in-the-pipe model of nitric and nitrous oxide emissions from soils using the TRAGNET database. Global Biogeochem. Cycles 14, 1035–1043. https://doi.org/10.1029/1999GB001223

Davidson, N.C., 2014. How much wetland has the world lost? Long-term and recent trends in

global wetland area. Mar. Freshw. Res. 65, 934–941. https://doi.org/10.1071/MF14173

Denmead, O.T., 2008. Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere. Plant Soil 309, 5–24. https://doi.org/10.1007/s11104-008-9599-z

Dixon, A.B., Wood, A.P., 2003. Wetland cultivation and hydrological management in eastern Africa: Matching community and hydrological needs through sustainable wetland use. Nat. Resour. Forum 27, 117–129. https://doi.org/10.1111/1477-8947.00047

Do Carmo, J.B., de Andrade, C.A., Cerri, C.C., Piccolo, M.D., 2005. Nitrogen availability and N2O fluxes frompasture soil after herbicide application. Rev. Bras. Cienc. DO SOLO 29, 735–746.

Dobbie, K. E. & Smith, A., 2001. The effects of temperature, water fuilled pore space and land use on N2O emissions from an imperfectly drained gleysol. Eur. J. Soil Sci. 667–673.

Euliss, N.H., Gleason, R.A., Olness, A., McDougal, R.L., Murkin, H.R., Robarts, R.D., Bourbonniere, R.A., Warner, B.G., 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. Sci. Total Environ. 361, 179–188. https://doi.org/10.1016/j.scitotenv.2005.06.007

Eusufzai, M.K., Tokida, T., Okada, M., Sugiyama, S.I., Liu, G.C., Nakajima, M., Sameshima, R., 2010. Methane emission from rice fields as affected by land use change. Agric. Ecosyst. Environ. 139, 742–748. https://doi.org/10.1016/j.agee.2010.11.003

Farmer, J., Matthews, R., Smith, P., Smith, J.U., 2014. The Tropical Peatland Plantation-Carbon Assessment Tool: estimating CO2 emissions from tropical peat soils under plantations. Mitig. Adapt. Strateg. Glob. Chang. 19, 863–885. https://doi.org/10.1007/s11027-013-9517-4

Finlayson, C.M., Davis, J.A., Gell, P.A., Kingsford, R.T., Parton, K.A., 2013. The status of wetlands and the predicted effects of global climate change: The situation in Australia. Aquat. Sci. 75, 73–93. https://doi.org/10.1007/s00027-011-0232-5

Flechard, C.R., Ambus, P., Skiba, U., Rees, R.M., Hensen, A., van Amstel, A., Dasselaar, A. van

den P. van, Soussana, J.F., Jones, M., Clifton-Brown, J., Raschi, A., Horvath, L., Neftel, A., Jocher, M., Ammann, C., Leifeld, J., Fuhrer, J., Calanca, P., Thalman, E., Pilegaard, K., Di Marco, C., Campbell, C., Nemitz, E., Hargreaves, K.J., Levy, P.E., Ball, B.C., Jones, S.K., van de Bulk, W.C.M., Groot, T., Blom, M., Domingues, R., Kasper, G., Allard, V., Ceschia, E., Cellier, P., Laville, P., Henault, C., Bizouard, F., Abdalla, M., Williams, M., Baronti, S., Berretti, F., Grosz, B., 2007. Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe. Agric. Ecosyst. Environ. 121, 135–152. https://doi.org/10.1016/j.agee.2006.12.024

Follett, R.., 2001. Soil management concepts and carbon sequestration in cropland soils. Soil Tillage Res. 61, 77–92. https://doi.org/10.1016/S0167-1987(01)00180-5

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., A, M.S., Dorland, R. Van, 2007. Changes in Atmospheric Constituents and in Radiative Forcing. In: Climate Change 2007: The Physical Science Basis, in: Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. https://doi.org/10.1103/PhysRevB.77.220407

GAO, J.-Q., OUYANG, H., XU, X.-L., ZHOU, C.-P., ZHANG, F., 2009. Effects of Temperature and Water Saturation on CO2 Production and Nitrogen Mineralization in Alpine Wetland Soils. Pedosphere 19, 71–77. https://doi.org/10.1016/S1002-0160(08)60085-5

Gondwe, M.J., Masamba, W.R.L., 2014. Spatial and temporal dynamics of diffusive methane emissions in the Okavango Delta, northern Botswana, Africa. Wetl. Ecol. Manag. 22, 63–78. https://doi.org/10.1007/s11273-013-9323-5

Groffman, P.M., Altabet, M.A., Böhlke, J.K., Butterbach-Bahl, K., David, M.B., Firestone, M.K., Giblin, A.E., Kana, T.M., Nielsen, L.P., Voytek, M.A., 2006. Methods for measuring denitrification: Diverse approaches to a difficult problem. Ecol. Appl. https://doi.org/10.1890/1051-0761(2006)016[2091:MFMDDA]2.0.CO;2

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: A meta analysis. Glob. Chang. Biol. 8, 345–360. https://doi.org/10.1046/j.1354-1013.2002.00486.x

Heikkinen, J.E.P., Elsakov, V., Martikainen, P.J., 2002. Carbon dioxide and methane dynamics and annual carbon balance in tundra wetland in NE Europe, Russia. Global Biogeochem. Cycles 16, 16. https://doi.org/10.1029/2002gb001930

Hensen, A., Skiba, U., Famulari, D., 2013. Low cost and state of the art methods to measure nitrous oxide emissions. Environ. Res. Lett. 8. https://doi.org/10.1088/1748-9326/8/2/025022

Hernandez, M.E., Mitsch, W.J., 2006. Influence of hydrologic pulses, flooding frequency, and vegetation on nitrous oxide emissions from created riparian marshes. Wetlands 26, 862–877. https://doi.org/10.1672/0277-5212(2006)26[862:IOHPFF]2.0.CO;2

Houghton, R.A., House, J.I., Pongratz, J., Van Der Werf, G.R., Defries, R.S., Hansen, M.C., Le Quéré, C., Ramankutty, N., 2012. Carbon emissions from land use and land-cover change. Biogeosciences 9, 5125–5142. https://doi.org/10.5194/bg-9-5125-2012

Hu, M., Ren, H., Ren, P., Li, J., Wilson, B.J., Tong, C., 2017. Response of gaseous carbon emissions to low-level salinity increase in tidal marsh ecosystem of the Min River estuary, southeastern China. J. Environ. Sci. (China) 52, 210–222. https://doi.org/10.1016/j.jes.2016.05.009

Iqbal, J., Castellano, M.J., Parkin, T.B., 2013. Evaluation of photoacoustic infrared spectroscopy for simultaneous measurement of N2O and CO2 gas concentrations and fluxes at the soil surface. Glob. Chang. Biol. 19, 327–336. https://doi.org/10.1111/gcb.12021

Johnson, J.M.F., Franzluebbers, A.J., Weyers, S.L., Reicosky, D.C., 2007. Agricultural opportunities to mitigate greenhouse gas emissions. Environ. Pollut. 150, 107–124. https://doi.org/10.1016/j.envpol.2007.06.030

Johnston, K.J., 2003. The intensification of pre-industrial cereal agriculture in the tropics: Boserup, cultivation lengthening, and the classic Maya. J. Anthropol. Archaeol. https://doi.org/10.1016/S0278-4165(03)00013-8

Junk, W.J., An, S., Finlayson, C.M., Gopal, B., Květ, J., Mitchell, S.A., Mitsch, W.J., Robarts, R.D., 2013. Current state of knowledge regarding the world's wetlands and their future under

global climate change: A synthesis. Aquat. Sci. 75, 151–167. https://doi.org/10.1007/s00027-012-0278-z

Kamiri, H., Kreye, C., Becker, M., 2013. Dynamics of agricultural use differentially affect soil properties and crop response in East African wetlands. Wetl. Ecol. Manag. 21, 417–431. https://doi.org/10.1007/s11273-013-9315-5

Kasimir-Klemedtsson, A., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J., Oenema, O., 1997. Greenhouse gas emissions from farmed organic soils: a review. Soil Use Manag. 13, 245–250. https://doi.org/10.1111/j.1475-2743.1997.tb00595.x

Kayranli, B., Scholz, M., Mustafa, A., Hedmark, Å., 2010. Carbon storage and fluxes within freshwater wetlands: A critical review. Wetlands 30, 111–124. https://doi.org/10.1007/s13157-009-0003-4

Kindler, R., Siemens, J., Kaiser, K., Walmsley, D.C., Bernhofer, C., Buchmann, N., Cellier, P., Eugster, W., Gleixner, G., Grunwald, T., Heim, A., Ibrom, A., Jones, S.K., Jones, M., Klumpp, K., Kutsch, W., Larsen, K.S., Lehuger, S., Loubet, B., Mckenzie, R., Moors, E., Osborne, B., Pilegaard, K., Rebmann, C., Saunders, M., Schmidt, M.W.I., Schrumpf, M., Seyfferth, J., Skiba, U., Soussana, J.F., Sutton, M.A., Tefs, C., Vowinckel, B., Zeeman, M.J., Kaupenjohann, M., 2011. Dissolved carbon leaching from soil is a crucial component of the net ecosystem carbon balance. Glob. Chang. Biol. 17, 1167–1185. https://doi.org/10.1111/j.1365-2486.2010.02282.x

Kirwan, M.L., Megonigal, J.P., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature. https://doi.org/10.1038/nature12856

Kumar, S., Nakajima, T., Mbonimpa, E.G., Gautam, S., Somireddy, U.R., Kadono, A., Lal, R., Chintala, R., Rafique, R., Fausey, N., 2014. Long-term tillage and drainage influences on soil organic carbon dynamics, aggregate stability and corn yield. Soil Sci. Plant Nutr. 60, 108–118. https://doi.org/10.1080/00380768.2013.878643

Le Mer, J., Roger, P., 2001. Production, oxidation, emission and consumption of methane by soils: A review. Eur. J. Soil Biol. 37, 25–50. https://doi.org/10.1016/S1164-5563(01)01067-6

Leytem, A.B., Dungan, R.S., Bjorneberg, D.L., Koehn, A.C., 2011. Emissions of Ammonia, Methane, Carbon Dioxide, and Nitrous Oxide from Dairy Cattle Housing and Manure Management Systems. J. Environ. Qual. 40, 1383. https://doi.org/10.2134/jeq2009.0515

Li, C., Frolking, S., Butterbach-Bahl, K., 2005. Carbon sequestration in arable soils is likely to increase nitrous oxide emissions, offsetting reductions in climate radiative forcing. Clim. Change 72, 321–338. https://doi.org/10.1007/s10584-005-6791-5

Liu, L., Greaver, T.L., 2009. A review of nitrogen enrichment effects on three biogenic GHGs: The CO 2 sink may be largely offset by stimulated N2O and CH 4 emission. Ecol. Lett. 12, 1103–1117. https://doi.org/10.1111/j.1461-0248.2009.01351.x

Liu, X., Jiang, M., Dong, G., Zhang, Z., Wang, X., 2017. Ecosystem Service Comparison before and after Marshland Conversion to Paddy Field in the Sanjiang Plain, Northeast China. Wetlands 37, 593–600. https://doi.org/10.1007/s13157-017-0894-4

Ludwig, J., Meixner, F.X., Vogel, B., Forstner, J., 2001. Soil-air exchange of nitric oxide: An overview of processes, dnvironmental vactors, and modeling studies. Biogeochemistry 52, 225–257. https://doi.org/10.1023/a:1006424330555

Lukito, H.P., Kouno, K., Ando, T., 1998. Phosphorus requirements of microbial biomass in a regosol and an andosol. Soil Biol. Biochem. 30, 865–872. https://doi.org/10.1016/S0038-0717(98)00011-X

Lund, M., Christensen, T.R., Mastepanov, M., Lindroth, A., Ström, L., 2009. Effects of N and P fertilization on the greenhouse gas exchange in two northern peatlands with contrasting N deposition rates. Biogeosciences 6, 2135–2144. https://doi.org/10.5194/bg-6-2135-2009

Ma, K., Liu, J., Balkovič, J., Skalský, R., Azevedo, L.B., Kraxner, F., 2016. Changes in soil organic carbon stocks of wetlands on China's Zoige plateau from 1980 to 2010. Ecol. Modell. 327, 18–28. https://doi.org/10.1016/j.ecolmodel.2016.01.009

Mafabi, P., 2000. The role of wetland policies in the conservation of waterbirds: The case of Uganda. Ostrich 71, 96–98. https://doi.org/10.1080/00306525.2000.9639880

Marinho, C.C., Palma-Silva, C., Albertoni, E.F., Trindade, C.R., Esteves, F.D.A., 2009. Seasonal dynamics of methane in the water column of two subtropical lakes differing in trophic status. Braz. J. Biol. 69, 281–287. https://doi.org/10.1590/S1519-69842009000200007

McLain, J.E.T., Ahmann, D.M., 2008. Increased moisture and methanogenesis contribute to reduced methane oxidation in elevated CO2soils. Biol. Fertil. Soils 44, 623–631. https://doi.org/10.1007/s00374-007-0246-2

McLauchlan, K., 2006. The nature and longevity of agricultural impacts on soil carbon and nutrients: A review. Ecosystems. https://doi.org/10.1007/s10021-005-0135-1

Mironga, J.M., 2005. Effect of farming practices on wetlands of Kisii District, Kenya. Appl. Ecol. Environ. Res. 3, 81–91. https://doi.org/10.15666/aeer/0302_081091

Mishra, S., Rath, A.K., Adhya, T.K., Rao, V.R., Sethunathan, N., 1997. Effect of continuous and alternate water regimes on methane efflux from rice under greenhouse conditions. Biol. Fertil. Soils 24, 399–405. https://doi.org/10.1007/s003740050264

Mitchell, S.A., 2013. The status of wetlands, threats and the predicted effect of global climate change: The situation in Sub-Saharan Africa. Aquat. Sci. 75, 95–112. https://doi.org/10.1007/s00027-012-0259-2

Mitsch, W.J., Bernal, B., Nahlik, A.M., Mander, Ü., Zhang, L., Anderson, C.J., Jørgensen, S.E., Brix, H., 2013. Wetlands, carbon, and climate change. Landsc. Ecol. 28, 583–597. https://doi.org/10.1007/s10980-012-9758-8

Mitsch, W.J., Gosselink, J.G., 1993. Wetlands, Second edition, Wetlands, Second edition.

Mitsch, W.J., Hernandez, M.E., 2013. Landscape and climate change threats to wetlands of North and Central America. Aquat. Sci. 75, 133–149. https://doi.org/10.1007/s00027-012-0262-7

Mitsch, W.J., Nahlik, A., Wolski, P., Bernal, B., Zhang, L., Ramberg, L., 2010. Tropical wetlands: Seasonal hydrologic pulsing, carbon sequestration, and methane emissions. Wetl. Ecol. Manag. 18, 573–586. https://doi.org/10.1007/s11273-009-9164-4

Morrison, E.H.J., Upton, C., Odhiambo-K'oyooh, K., Harper, D.M., 2012. Managing the natural

capital of papyrus within riparian zones of Lake Victoria, Kenya. Hydrobiologia 692, 5–17. https://doi.org/10.1007/s10750-011-0839-5

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and Natural Radiative Forcing. Clim. Chang. 2013 Phys. Sci. Basis. Contrib. Work. Gr. I to Fifth Assess. Rep. Intergov. Panel Clim. Chang. 659–740. https://doi.org/10.1017/ CBO9781107415324.018

Nag, S.K., Liu, R., Lal, R., 2017. Emission of greenhouse gases and soil carbon sequestration in a riparian marsh wetland in central Ohio. Environ. Monit. Assess. 189. https://doi.org/10.1007/s10661-017-6276-9

Nath, A.J., Lal, R., 2017. Managing tropical wetlands for advancing global rice production: Implications for land-use management. Land use policy 68, 681–685. https://doi.org/10.1016/j.landusepol.2017.08.026

Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F., Erasmi, S., 2016. Greenhouse gas emissions from soils—A review. Chemie der Erde - Geochemistry 76, 327–352. https://doi.org/10.1016/j.chemer.2016.04.002

Okalebo, J.R., Gathua, K.W., Woomer, P.L.., 2002. Laboratory Methods of Soil and Plant Analysis: A Working Manual.

Okeyo-Owuor J.B, Raburu P.O, M.F.. & O.S.., 2016. Wetlands of Lake Victoria Basin, Kenya: distribution, current status and conservation challenges. Community Based Approach to Manag. Nyando Wetl. Lake Victoria Basin, Kenya 2–3.

Page, K.L., Dalal, R.C., 2011. Contribution of natural and drained wetland systems to carbon stocks, CO2, N2O, and CH4 fluxes: an Australian perspective. Soil Res. 49, 377–388. https://doi.org/10.1071/SR11024

Pivničková, B., Rejmánková, E., Snyder, J.M., Šantrůčková, H., 2010. Heterotrophic microbial activities and nutritional status of microbial communities in tropical marsh sediments of different salinities: The effects of phosphorus addition and plant species. Plant Soil 336, 49–63. https://doi.org/10.1007/s11104-010-0439-6

Puget, P., Lal, R., 2005. Soil organic carbon and nitrogen in a Mollisol in central Ohio as affected by tillage and land use. Soil Tillage Res. 80, 201–213. https://doi.org/10.1016/j.still.2004.03.018

Qingshui, L., Zhiqiang, G., Zhiping, Z., Jicai, N., Xiaoli, B., 2014. Dynamics of wetlands and their effects on carbon emissions in china coastal region - case study in bohai economic rim. Ocean Coast. Manag. 87, 61–67. https://doi.org/10.1016/j.ocecoaman.2013.11.004

Reddy, K.R., D'Angelo, E.M., Harris, W.G., 2000. Biogeochemistry of wetlands. Handb. soil Sci. G89-119. https://doi.org/10.1016/j.ecoleng.2010.01.001

Rogelj, J., Den Elzen, M., Höhne, N., Fransen, T., Fekete, H., Winkler, H., Schaeffer, R., Sha, F., Riahi, K., Meinshausen, M., 2016. Paris Agreement climate proposals need a boost to keep warming well below 2 °c. Nature 534, 631–639. https://doi.org/10.1038/nature18307

Rosenstock, T.S., Rufino, M.C., Butterbach-Bahl, K., Wollenberg, E., Richards, M., 2016. Methods for Measuring Greenhouse Gas Balances and Evaluating Mitigation Options in Smallholder Agriculture. Methods Meas. Greenh. Gas Balanc. Eval. Mitig. Options Smallhold. Agric. 1–203. https://doi.org/10.1007/978-3-319-29794-1

Sakané, N., Becker, M., Langensiepen, M., Van Wijk, M.T., 2013. Typology of smallholder production systems in small east-African wetlands. Wetlands 33, 101–116. https://doi.org/10.1007/s13157-012-0355-z

Schuijt, K., 2002. Land and water use of wetlands in Africa: Economic values of African wetlands 41.

Schuyt, K.D., 2005. Economic consequences of wetland degradation for local populations in Africa. Ecol. Econ. 53, 177–190. https://doi.org/10.1016/j.ecolecon.2004.08.003

Signor, D., Cerri, C.E.P., 2013. Nitrous oxide emissions in agricultural soils: a review. Pesqui. Agropecuária Trop. 43, 322–338. https://doi.org/10.1590/S1983-40632013000300014

Sirivedhin, T., Gray, K.A., 2006. Factors affecting denitrification rates in experimental wetlands: Field and laboratory studies. Ecol. Eng. 26, 167–181.

https://doi.org/10.1016/j.ecoleng.2005.09.001

Sjögersten, S., Black, C.R., Evers, S., Hoyos-santillan, J., Wright, E.L., Turner, B.L., 2014a. Global Biogeochemical Cycles. Global Biogeochem. Cycles 28, 1371–1386. https://doi.org/10.1002/2014GB004844.Received

Sjögersten, S., Black, C.R., Evers, S., Hoyos-Santillan, J., Wright, E.L., Turner, B.L., 2014b. Tropical wetlands: A missing link in the global carbon cycle? Global Biogeochem. Cycles 28, 1371–1386. https://doi.org/10.1002/2014GB004844

Smith, K.A., Smith, K.A., Conen, F., 2004. Impacts of land management on fluxes of trace greenhouse gases. Soil Use Manag. 20, 255–263. https://doi.org/10.1079/SUM2004238

Smith, K. a., Ball, T., Conen, F., Dobbie, K.E., Massheder, J., Rey, a., 2003. Exchange of greenhousegases between soil and atmosphere: interactions of soil physical factors and biological processes. Eur. J. Soil Sci. 54, 779–791. https://doi.org/10.1046/j.1365-2389.2003.00567.x

Smith, L.K., Lewis, W.M., Chanton, J.P., Cronin, G., Hamilton, S.K., 2000. Methane emissions from the Orinoco River floodplain, Venezuela. Biogeochemistry 51, 113–140. https://doi.org/10.1023/a:1006443429909

Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric. Ecosyst. Environ. https://doi.org/10.1016/j.agee.2009.04.021

Solomon, S., Qin, D., Manning, M., Alley, R.B., Berntsen, T., Bindoff, N.L., Chen, Z., Chidthaisong, A., Gregory, J.M., Hegerl, G.C., Heimann, M., Hewitson, B., Hoskins, B.J., Joos, F., Jouzel, J., Kattsov, V., Lohmann, U., Matsuno, T., Molina, M., Nicholls, N., Overpeck, J., Raga, G., Ramaswamy, V., Ren, J., Rusticucci, M., Somerville, R., Stocker, T.F., Whetton, P., Wood, D., Wratt, R. a, 2007. Climate Change 2007 The Physical Science Basis Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Ipcc Wg1 23–78. https://doi.org/10.1007/s10894-008-9162-1

Song, C., Wang, L., Tian, H., Liu, D., Lu, C., Xiaofeng Xu, Zhang, L., Yang, G., Wan, Z., 2013.

Effect of continued nitrogen enrichment on greenhouse gas emissions from a wetland ecosystem in the Sanjiang Plain, Northeast China: A 5 year nitrogen addition experiment. J. Geophys. Res. Biogeosciences 118, 741–751. https://doi.org/10.1002/jgrg.20063

Streeter, M.T., Schilling, K.E., 2015. A comparison of soil properties observed in farmed, restored and natural closed depressions on the Des Moines Lobe of Iowa. Catena 129, 39–45. https://doi.org/10.1016/j.catena.2015.02.021

Tangen, B.A., Finocchiaro, R.G., Gleason, R.A., 2015. Effects of land use on greenhouse gas fluxes and soil properties of wetland catchments in the Prairie Pothole Region of North America. Sci. Total Environ. 533, 391–409. https://doi.org/10.1016/j.scitotenv.2015.06.148

Timothy B.Parkin and Rodney T. Venterea, 2010. USDA-ARS GRACEnet Project Protocols Chapter 3. Chamber-Based Trace Gas Flux Measurements. Flux 2010, 1–39.

Tiner, R.W., 2009. Global Distribution of Wetlands, in: Encyclopedia of Inland Waters. pp. 526–530. https://doi.org/10.1016/B978-012370626-3.00068-5

Tully, K.L., Abwanda, S., Thiong'o, M., Mutuo, P.M., Rosenstock, T.S., 2017. Nitrous Oxide and Methane Fluxes from Urine and Dung Deposited on Kenyan Pastures. J. Environ. Qual. 46, 921. https://doi.org/10.2134/jeq2017.01.0040

Van Dam, A.A., Dardona, A., Kelderman, P., Kansiime, F., 2007. A simulation model for nitrogen retention in a papyrus wetland near Lake Victoria, Uganda (East Africa). Wetl. Ecol. Manag. 15, 469–480. https://doi.org/10.1007/s11273-007-9047-5

Veber, G., Kull, A., Villa, J.A., Maddison, M., Paal, J., Oja, T., Iturraspe, R., Pärn, J., Teemusk, A., Mander, Ü., 2017. Greenhouse gas emissions in natural and managed peatlands of America: Case studies along a latitudinal gradient. Ecol. Eng. https://doi.org/10.1016/j.ecoleng.2017.06.068

Wang, J., Chen, Y., Shao, X., Zhang, Y., Cao, Y., 2012. Land-use changes and policy dimension driving forces in China: Present, trend and future. Land use policy 29, 737–749. https://doi.org/10.1016/j.landusepol.2011.11.010

Wang, W., Sardans, J., Wang, C., Zeng, C., Tong, C., Asensio, D., Peñuelas, J., 2017. Relationships between the potential production of the greenhouse gases CO2, CH4and N2O and soil concentrations of C, N and P across 26 paddy fields in southeastern China. Atmos. Environ. 164, 458–467. https://doi.org/10.1016/j.atmosenv.2017.06.023

Wang, Y., Wang, Y., Ling, H., 2010. A new carrier gas type for accurate measurement of N2O by GC-ECD. Adv. Atmos. Sci. 27, 1322–1330. https://doi.org/10.1007/s00376-010-9212-2

Whalen, S.C., 2005. Natural Wetlands and the Atmosphere. Environ. Eng. Sci. 22, 73–94. https://doi.org/10.1089/ees.2005.22.73

Whiting, G.J., Chanton, J.P., 2001. Greenhouse carbon balance of wetlands: Methane emission versus carbon sequestration. Tellus, Ser. B Chem. Phys. Meteorol. 53, 521–528. https://doi.org/10.3402/tellusb.v53i5.16628

Wright, a. L., Reddy, K.R., 2001. Heterotrophic microbial activity in northern Everglades wetland soils. Soil Sci. Soc. Am. J. 65, 1856–1864. https://doi.org/10.2136/sssaj2001.1856

Zedler, J.B., Kercher, S., 2005. WETLAND RESOURCES: Status, Trends, Ecosystem Services, and Restorability. Annu. Rev. Environ. Resour. 30, 39–74. https://doi.org/10.1146/annurev.energy.30.050504.144248

Zeshan, Karthikeyan, O.P., Visvanathan, C., 2012. Effect of C/N ratio and ammonia-N accumulation in a pilot-scale thermophilic dry anaerobic digester. Bioresour. Technol. 113, 294–302. https://doi.org/10.1016/j.biortech.2012.02.028

Zhang, X., Yin, S., Li, Y., Zhuang, H., Li, C., Liu, C., 2014. Comparison of greenhouse gas emissions from rice paddy fields under different nitrogen fertilization loads in Chongming Island, Eastern China. Sci. Total Environ. 472, 381–388. https://doi.org/10.1016/j.scitotenv.2013.11.014

Zhu, X., Burger, M., Doane, T.A., Horwath, W.R., 2013. Ammonia oxidation pathways and nitrifier denitrification are significant sources of N2O and NO under low oxygen availability. Proc. Natl. Acad. Sci. 110, 6328–6333.

https://doi.org/10.1073/pnas.1219993110

Zou, J., Huang, Y., Jiang, J., Zheng, X., Sass, R.L., 2005. A 3-year field measurement of methane and nitrous oxide emissions from rice paddies in China: Effects of water regime, crop residue, and fertilizer application. Global Biogeochem. Cycles 19, 1–9. https://doi.org/10.1029/2004GB002401

Popular scientific summary

Convert or conserve wetlands

In Kenya, wetlands are under intense anthropogenic pressure arising from the need of productive agricultural land and economic growth. It is estimated that 50% of the ca.3-4% area covered by wetlands has been lost due to conversion to agriculture. This study aimed at assessing the influence of wetland conversion to farmland on greenhouse gas emission and carbon stocks. Generally, wetlands are characterised by water logged soils facilitating carbon storage, through reduced decomposition rate. Conversion to farmland increases oxygen diffusion, consequently releasing stored carbon in form of carbon dioxide. Methane emission is reduced in farmlands while nitrous oxide increases due to availability of organic carbon, favouring microbial activity and growth.