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Water Education**
under the auspices of UNESCO

**IMPACT OF WETLAND CONVERSION TO CROPLAND ON SOIL ORGANIC
CARBON AND GREENHOUSE GAS EMISSIONS IN YALA WETLAND, KENYA**

Master of Science Thesis

By

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DECLARATION AND RECOMMENDATION

Declaration

This thesis is my original work and has not been presented in this University or any other for the award of a degree.

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Recommendation

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DEDICATION

I dedicate this work to my late brother Kizza George and my grand father, Mr. Perez Bukumunhe, for believing in me and giving me an opportunity to access education, without you, this scholarship would not have come through. Thanks a lot.

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ABSTRACT

Climate change is one of the major global concerns, with scientific evidence linking it to the rising emissions of greenhouse gases (GHGs), particularly carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) in the atmosphere. Wetlands provide an optimum natural environment for mitigation of climate change through carbon sequestration. Converting wetlands to cropland is well known to reduce soil carbon storage and increase emissions of GHGs. However, there is limited data on how different wetland plants and conversion of wetlands to cropland influence carbon and GHG fluxes. This study aimed to fill the gap in understanding how different wetland plant communities and land conversion affect carbon and nitrogen fluxes. This study was conducted during a dry season i.e. from December 2024 to March 2025 in Kadenge and Usonga sections of Yala wetland. The sampling design was stratified and included four sampling sites that is, wetland areas converted to sugarcane and those dominated by *Cyperus alopecuroides*, *Typha latifolia*, and *Cyperus papyrus*. Each site had three sub-sampling sites (replicates), each with three chambers to cater for spatial heterogeneity. Sampling of gases was carried out twice a month using the closed static chamber method. The gas samples were analysed using the SRI GHG gas chromatograph. Soil samples were collected during each sampling session and analysed for NO₃-N, NH₄-N, soil organic carbon (SOC), total phosphorus, and total nitrogen.

The Kruskal–Walli’s test was used to examine the differences in the medians of SOC and GHGs under the different wetland plant dominance and cropland. The results showed that SOC was significantly lower in areas converted to sugarcane (1.73%) than areas dominated by either *Cyperus alopecuroides* (2.98 %), *Typha latifolia* (3.50%) or *Cyperus papyrus* (3.09 %) (adj. p < 0.05). There were no significant differences in the proportion of SOC among the plant communities (adj. p > 0.05). Both CO₂ and N₂O did not show significant differences in their emissions among the sites (adj p > 0.05). Significantly higher CH₄ emissions were observed in *Cyperus alopecuroides* (27.19 mg m⁻² h⁻¹) compared to *Cyperus papyrus* (0.09 mg m⁻² h⁻¹) and sugarcane areas (0.31 mg m⁻² h⁻¹). In this study, conversion of a wetland area to a sugarcane farm significantly reduced the SOC but had no significant effect on GHG emissions except for CH₄ which was significantly higher in *Cyperus alopecuroides*. Therefore, this necessitates sustainable use of wetlands to minimize SOC loss and consequently GHG emissions.

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LIST OF ABBREVIATIONS AND ACRONYMS

C	Carbon
ES	Ecosystem Services
GHGs	Greenhouse Gases
MEA	Millennium Ecosystem Assessment
N	Nitrogen
P	Phosphorus
SM	Soil Moisture
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
SPSS	Statistical Package for the Social Sciences
TEMP.	Temperature
TN	Total Nitrogen
TP	Total Phosphorus

CHAPTER ONE

INTRODUCTION

1.1 Background information

Climate change is among the most pressing global environmental issues, with empirical evidence linking it to the increasing emissions of potent greenhouse gases (GHGs) in the atmosphere (WMO, 2019). In fact, the Intergovernmental Panel on Climate Change, for example, estimates that if the current emission rates are maintained, global warming will exceed 2.0 °C somewhere between the years 2040 and 2070, a level it termed as "dangerous" (Cubasch *et al.*, 2001). Greenhouse gases, particularly, CO₂, CH₄ and N₂O are the main contributors of global warming, accounting for 66%, 17% and 6%, respectively of the radiative long-lived GHGs (WMO, 2019). The concentrations of CO₂, CH₄ and N₂O in the atmosphere have steadily risen over the years reaching 407.8 ppm, 1869 ppb and 270 ppb in 2018, which is 147%, 259%, 123% increase respectively, compared to pre-industrial levels which were 278 ppm for CO₂, 722 ppb for CH₄ and 270 ppb for N₂O (WMO, 2019). Reducing emissions of these GHGs into the atmosphere while increasing their long-term storage in various ecosystems are key in climate change mitigation.

Wetlands provide an optimum natural environment for climate change mitigation through carbon sequestration (Mitsch *et al.*, 2012). These natural resources are highly productive ecosystems where plants convert atmospheric CO₂ through photosynthesis (Reddy *et al.*, 2008). As wetland plants senesce, their biomass accumulate into the soil as organic matter, contributing to the carbon pool in wetland soils (Villa & Bernal, 2018 ; Mitsch & Gosselink, 2000). In addition, the waterlogged conditions in wetlands create anaerobic environment, slowing down organic matter decomposition. This leads to the accumulation of partially decomposed plant material, known as peat, which stores carbon for thousands of years. The previous studies estimate that wetlands store 20-33% of the carbon pool of the Earth (Mitsch & Gosselink, 2015). The stored organic carbon, however, undergoes decomposition leading to production and emissions of potent GHGs to the atmosphere.

Wetlands emit three potent GHGs; CH₄, CO₂ and N₂O (Reddy *et al.*, 2008). Methane is a powerful greenhouse gas with a global warming potential roughly 25 times that of CO₂ over a century (Woodward *et al.*, 2014). It is generated in anaerobic conditions by methanogenic archaea during the breakdown of organic materials (Reddy *et al.*, 2008). Carbon dioxide (CO₂) in wetland soils is predominantly generated via microbial respiration (Bridgham *et al.*, 2013). Soil microorganisms degrade organic materials, converting them into simpler compounds and releasing CO₂ as a byproduct. In addition, CO₂ is also emitted from wetlands

when CH₄ is oxidised through a process called methanotrophy, carried out by methanotrophic bacteria (Bridgham *et al.*, 2013). This oxidation typically occurs in the aerobic zones of the wetland soil, where oxygen is available. Nitrous oxide is another powerful greenhouse gas, possessing a global warming potential approximately 298 times more than that of CO₂ over a century (Woodward *et al.*, 2014). N₂O is generated in wetlands mainly via nitrification and denitrification processes. The emissions of these GHGs from wetlands, however, are largely part of the natural carbon and nitrogen cycling, contributing to maintaining the natural greenhouse effect (Bridgham *et al.*, 2014; Saarnio *et al.*, 2009).

Different factors control the amount of organic carbon stored in wetland soil and GHG emitted (Wilson *et al.*, 2016). These factors include wetland plant community and biomass, hydrology (hydroperiod), soil type and texture, temperature and climate, nutrient availability and land use and management practices (Mitsch & Gosselink, 2015). Among these factors, wetland plant community plays a crucial role in influencing SOC storage and GHG emissions (Marín-Muñiz *et al.*, 2015; Zhang *et al.*, 2017). This is mainly because organic matter produced by different plant species have different levels of recalcitrance and hence differential rates of decomposition (Sjögersten *et al.*, 2014). This implies that even within the same wetland, differences in plant communities is likely to affect the amount of SOC stored as well as the subsequent GHGs emitted from the wetland.

Drainage and conversion of wetlands to cropland leads to decline in SOC and an increase in emissions of CO₂ and N₂O but a decline in CH₄ emission (Lal, 2004; Murdiyarso *et al.*, 2010). Draining a wetland, removal of vegetation, and cultivation creates aerobic conditions, resulting into faster decomposition of soil organic matter thus increasing CO₂ emissions into the atmosphere (Saunders *et al.*, 2007). The aerobic condition, however, inhibit the CH₄ producing methanogenic process (Reddy *et al.*, 2008). The rise in emissions of N₂O is as a result of the oxic conditions enhancing mineralization of organic nitrogen to ammonium-nitrogen before being oxidized to nitrate-nitrogen through nitrification. The formed nitrate-nitrogen diffuses to anaerobic zones where it is reduced to N₂O through denitrification process (Butterbach-Bahl *et al.*, 2013). Therefore, based on the global warming potential of the respective GHGs, the conversion of natural wetlands into cropland could contribute to mitigating CH₄ emissions but may pose more environmental threat because of increased N₂O and CO₂ emissions as well as decline in SOC stocks.

Studies on influence of wetland plant communities and land use/cover change to cropland on SOC and GHGs have been carried out, but most of these studies have been done in temperate regions (Hedman, 2019; Lin *et al.*, 2022; Sharma & Ravindranath, 2019; Tan *et*

et al., 2023). Very few studies have been done in Africa's tropical region to understand the effect of land-use/land-cover conversion to cropland on wetland SOC and GHG emissions (Ondiek *et al.*, 2021; Were *et al.*, 2021; Were *et al.*, 2020a; Were *et al.*, 2020b). However, these studies either focused on conversion of *Cyperus papyrus* dominated areas to cocoyam farm (Ondiek *et al.*, 2021) or conversion of wetlands areas dominated by various plant communities to paddy rice fields (Were *et al.*, 2020a; Were *et al.*, 2020b) unlike the current study where some wetland areas are converted to sugarcane. Hence, this study aimed at enhancing understanding on how wetland plant communities and land use/cover change to cropland influence SOC and GHG emissions from a tropical freshwater wetland. The findings of this study inform sustainable agricultural practices in the wetlands by examining how plant community composition and land use changes affect SOC and emission of GHGs.

1.2 Statement of the problem

Within the Yala wetland, approximately 64% of the area is covered by *Cyperus papyrus* and other macrophytes such as *Cyperus alopecuroides*, *Cyperus immensus*, *Typha latifolia*, *Phragmites mauritianus* among others, while 10% is open water (Muoria *et al.*, 2015). The conversion of wetland areas to cropland leads to the release of stored SOC and increased GHGs emissions. The Yala wetland has lost approximately 33% of its area over the past 25 years (Odero & Odenyo, 2021). Local communities currently cultivate about 11.5% of the wetland, and another 9.4% is under sugarcane production by a private investor. This implies that if current trends continue, wetland conversion is projected to rise from 20.9% to 30.5% over the next 25 years (Odero & Odenyo, 2021) undermining the climate regulating function of the wetland. However, the impact of converting the wetland to cropland on SOC and GHG emissions remains unknown. This study's aim, therefore, was to assess the influence of wetland plant communities and land use/cover change to cropland on climate change mitigation functions of the Yala wetland, thereby bridging this knowledge gap.

1.3 Objectives

1.3.1 General objective

To contribute to mitigation of climate change through assessment of the effect of converting wetland areas dominated by different plant communities to cropland on SOC and GHGs emissions.

1.3.2 Specific objectives

- i. To determine the amount of soil organic carbon (SOC) in areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides*, *Typha latifolia*, and in areas under sugarcane cultivation in the Yala wetland.
- ii. To quantify CO₂, CH₄ and N₂O emissions from areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides*, *Typha latifolia*, and from sugarcane-cultivated areas in the Yala wetland.
- iii. To assess the relationships between SOC and soil physico-chemical properties, as well as between GHG emissions (CO₂, CH₄, N₂O) and soil physico-chemical characteristics in the above land cover types.

1.4 Hypotheses

- i. There is no significant difference between the amount of SOC in areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides*, and *Typha latifolia* and areas under crop (sugarcane) cultivation in Yala wetland.
- ii. There is no significant difference between GHGs emitted from areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides*, and *Typha latifolia* and from sugarcane-cultivated areas in the Yala wetland.
- iii. There is no significant difference between SOC and soil physico-chemical properties, as well as between GHG emissions and soil physico-chemical characteristics in the above land cover types.

1.5 Justification

Wetlands are one of the most productive ecosystems on Earth, offering critical ecosystem services such as climate regulation. Yala wetland, despite having a land use plan is under threat from agricultural expansion. This scientific report on Yala wetland contributes to the required scientific evidence for the implementation of the plan. Also, it contributes to the national greenhouse gas inventory, a key instrument in the tracking of the country's progress in reducing emissions. The research aligns with the Kenya National Development Plan 2030, which highlights environmental integrity as a key factor of development as well as the attainment of SDG 13 on taking immediate action to combat climate change. This investigation into the impact of converting wetlands to cropland on SOC and GHG emissions informs strategies to maximize carbon storage and minimize emissions from wetlands. The findings of this study provide baseline data that aids policymakers, wetland users, land managers, and

conservationists in advocating for the conservation and sustainable use of wetlands, thus contributing to global wetland conservation and climate change mitigation efforts.

CHAPTER TWO

LITERATURE REVIEW

2.1 Global extent and distribution of wetlands

Wetlands are areas of marsh, fen or water (natural or artificial, permanent or temporary) where the water is static or flowing, fresh, brackish or salt, and includes marine areas where the water depth at low tide does not exceed six metres (Convention on Wetlands, 2018). To cater for all the wetland habitats for the migrating birds, the Ramsar Convention definition considers all lakes and rivers regardless of their depth to be wetlands. These unique ecosystems are found all over the world except in the Antarctica (Tiner, 2016). The Convention on Wetlands categorizes wetlands into three broad types: inland, marine and coastal and human-made, and classifies them into 42 categories (Convention on Wetlands, 2008).

Global wetland area is estimated to be more than 12.1 million Km² (Davidson, 2018). This estimate consists of inland and coastal wetlands, of which 54% are inundated permanently and 46% inundated seasonally (Davidson, 2018; Convention on Wetlands, 2018). The Asian continent is having the biggest share of wetlands globally, occupying approximately 32% of the global area then follows North America at 27% and Latin America and Caribbean at 16%. Wetlands in Europe, Africa and Oceania are relatively smaller, occupying global area of 13%, 10%, and 3%, respectively (Davidson, 2018). In densely populated regions such as Europe and China and in areas with water shortages such as Australia and in countries with extensive agro-industries such as the USA, wetlands loss has exceeded 50% of their original extent (Mitsch & Hernandez, 2012).

In Africa, where data is available, wetlands cover 1 -16% of the total continent area (Dixon & Wood, 2003). This is mainly because much of the continent lies in arid areas and semi arid regions. Although there are still several near pristine wetlands in Africa (Mitchell *et al.*, 2013), as compared to western Europe and North America, many African wetland areas are currently facing immense pressure mostly from the demographic growth (Junk *et al.*, 2012). In the East African countries of Tanzania, Kenya, Uganda and Rwanda, wetlands cover approximately 18 Mio ha of the total landcover (Leemhuis *et al.*, 2016). For Kenya, wetlands cover 3 – 4 % of its total land area (MEMR, 2012). These Kenyan wetlands range from deltas, estuaries, mudflats and mangroves, swamps, marshes, and floodplains (MEMR, 2012). Out of forty-two types of wetlands categorized by the Convention on Wetlands, six including riverine, lacustrine, marine, palustrine, estuarine and human-made ones can be found in Kenya (MEMR, 2012).

2.2 Wetland loss and agriculture

Worldwide, though the rates at which wetlands are lost have decreased in Europe and North America, wetlands are still declining at an alarming rate in other parts of the world, currently at about 50% of their original extent (Mitsch & Hernandez, 2012). Conversion of into agricultural land is a major driver to the degradation and loss wetland ecosystems (Davidson, 2014; Junk *et al.*, 2012; Mitsch & Gosselink, 2015). Agriculture impacts wetlands on two levels that is; directly by draining the wetlands and converting them to crops or indirectly by having agricultural activities upstream of wetlands which results in the alteration flows of water, alien species and nutrients to wetlands (Wood & van Halsema, 2008). Such activities range from the producing of food to its consumption including its processing, packaging, distribution, and retailing activities. With both types of impact, the roles of these wetlands are affected, with consequences on carbon sequestration and other ecosystem services (Muoria *et al.*, 2015). In general, people both depend on, and influence wetland ecosystem services. For example, in China, due to human activities resulting from change in land use, a net loss of 50,360 km² of wetland have been reported (Wang *et al.*, 2010). Furthermore, a study using remote sensing by Song *et al.* (2014) in the Sanjiang plain reported a significant change in the land use, where there was a decrease in the wetlands, forests, and waterbodies in addition to an increase in the residential area and farmlands. In Kenya, wetlands are being lost three times faster than forests and at least 35% of wetlands have been lost since the year 1970 (NEMA, 2023).

2.3 Carbon sequestration in wetlands

Carbon sequestration refers to the process of capturing and storing atmospheric carbon dioxide (Villa & Bernal, 2018). Wetlands essentially sequester carbon in three primary ways; (i) primary production and organic matter accumulation, when wetland plants, algae, and other organisms utilize atmospheric CO₂ and convert to organic matter via photosynthesis, and (ii) anaerobic conditions and peat formation, where waterlogged conditions in a wetland environment create anaerobic conditions to slow down decomposition of organic matter and convert to peat material that can sequester carbon for millennia, and finally (iii) sediment trapping, wetlands trap sediments (often containing organic carbon) carried by the water (Kayranli *et al.*, 2009 & Mitsch *et al.*, 2012). The three processes contribute to the burial of wetland soil carbon and sediments, further enhancing their role as sinks of carbon.

Carbon sequestration in wetlands is influenced by several factors. To start with, the hydrological regime of wetlands, including flow patterns, water depth and hydroperiod have a significant influence on the carbon sequestration process (Panchal *et al.*, 2022; Wang *et al.*,

2010). Stable water levels and prolonged inundation promote anaerobic conditions, enhancing soil carbon storage (Reddy *et al.*, 2008). The type and productivity of vegetation in wetlands affect the amount of the sequestered carbon (Panchal *et al.*, 2022). Plants with high biomass production such as *Cyperus papyrus*, sedges and mosses, due to their slow decomposition rates, contribute more to carbon sequestration (Chimner & Ewel, 2005). The slow decomposition rates are primarily attributed to their chemical composition particularly high lignin content and the presence of secondary metabolites (Keller, 2000). Similarly, soil properties including nutrient content, texture and microbial activity, play a critical role in accumulation. Soils with high organic matter content and low decomposition rates are more effective in storing carbon (Bridgham *et al.*, 2014). In addition, climatic parameters such as precipitation patterns and temperature influence the carbon dynamics in wetlands. Warmer temperatures can increase plant productivity but accelerate decomposition rates, potentially reducing net carbon sequestration. Conversely, higher precipitation can enhance waterlogging and carbon storage. Globally, the potential of wetlands for carbon sequestration is unmatched. For example, peatlands cover only about 3% of the Earth's land surface but store approximately 30% of the world's soil carbon, which makes them one of the most significant carbon sinks (Yu *et al.*, 2010). Also, mangrove forests in coastal regions store almost four times more carbon per unit area than our terrestrial forests (Ma *et al.*, 2016). While their carbon storage capacity per unit area may be lower than that of peatlands and mangroves, freshwater wetlands are crucial in carbon sequestration. Their widespread distribution makes them an important contributor to global carbon storage (Mitsch *et al.*, 2012).

2.4 Greenhouse gases production and emissions from wetlands

The three main GHGs from wetlands are CO₂, CH₄ and N₂O. CO₂ is one of the major components of the carbon cycle (Figure 2.1). During respiration, plants consume oxygen and release CO₂ as they convert stored carbohydrates into energy (Li *et al.*, 2013). Also, when dead plant matter and SOC are broken down by bacteria and fungi, CO₂ is released as a byproduct of microbial respiration (Song *et al.*, 2014). This CO₂ is then emitted into the atmosphere.

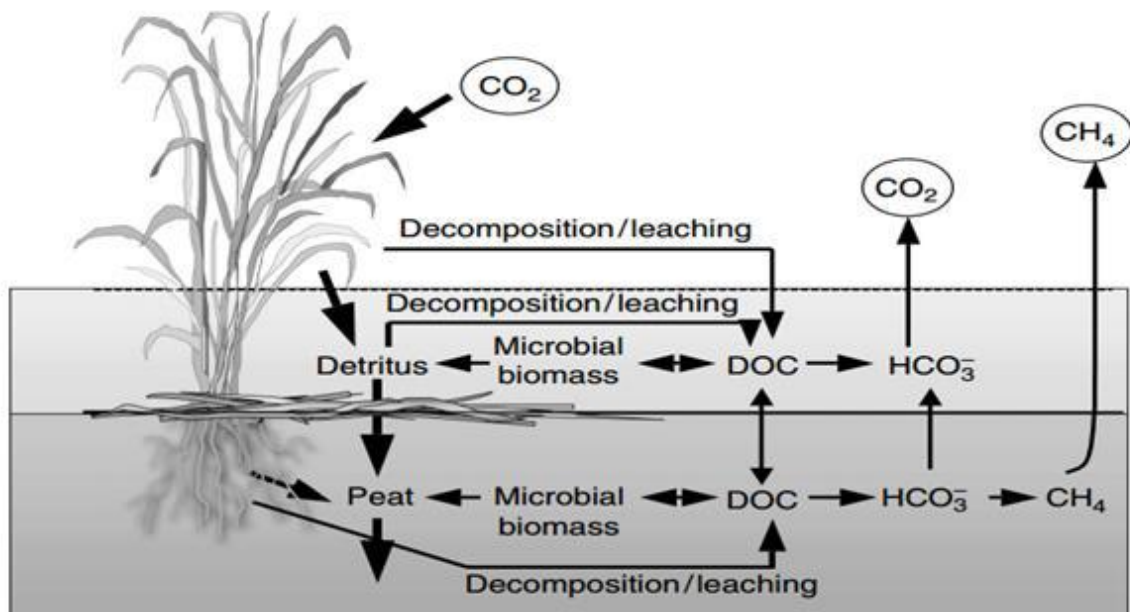


Figure 2.1: Schematic illustration highlighting carbon cycling processes and CO₂ emission (Reddy *et al.*, 2008)

Wetlands globally are the largest natural source of CH₄, which is a potent greenhouse gas (Reddy *et al.*, 2015). Recent studies indicate that wetlands' CH₄ emissions have been rising significantly, with estimates suggesting an increase of 1.2-1.4 million tonnes per year over the past two decades, mainly because of climate change impacts on wetland hydrology and temperature conditions (Zhang *et al.*, 2017). CH₄ was reported as being responsible for approximately 18% of the total greenhouse effect (Mitsch & Gosselink, 2015). Its emissions occur through biological processes (Figure 2.2), comprising of; production in anaerobic conditions through methanogenesis, consumption by the aerobic zones methanotrophs and direct transportation to the atmosphere e.g. through the aerenchyma of plants and in small quantities through diffusion and ebullition (Reddy *et al.*, 2015). All these processes are dependent on temperature, water levels and SOC and were important in this study (Tully *et al.*, 2017; Were *et al.*, 2021; Zhang *et al.*, 2017).

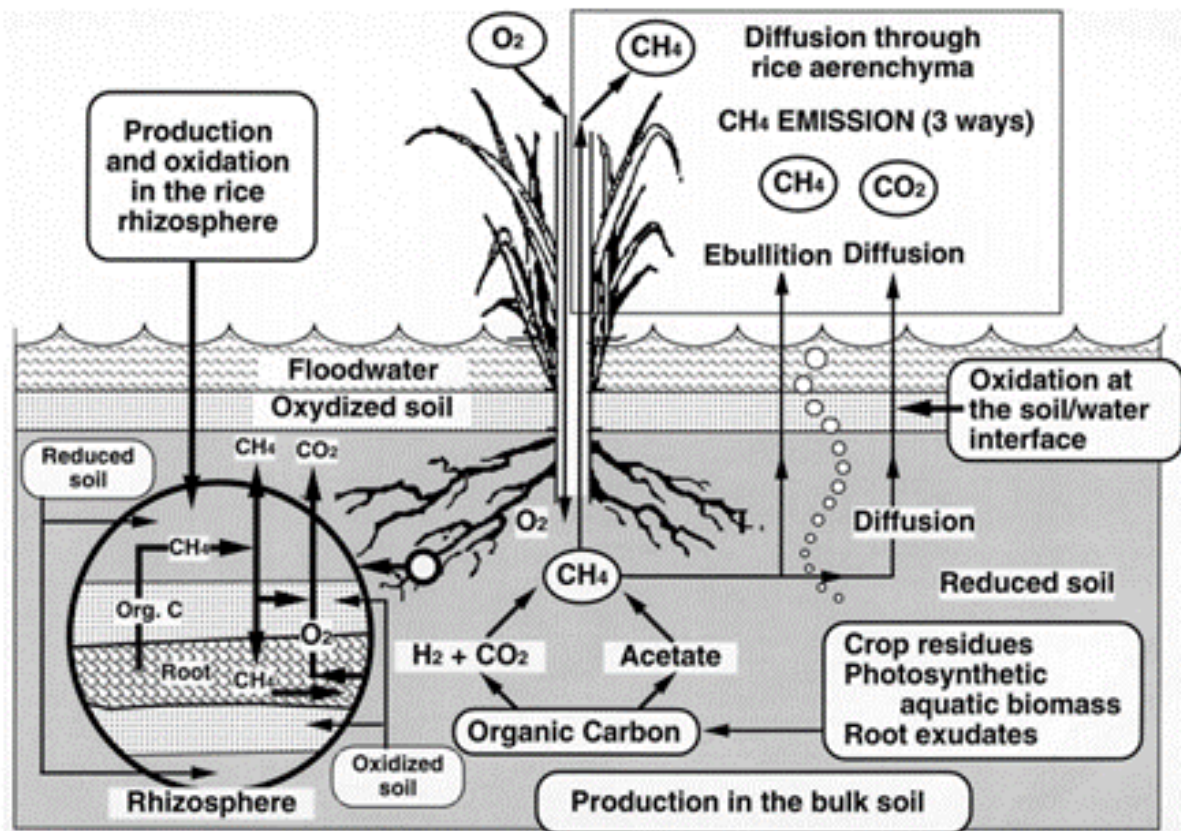


Figure 2.2: Schematic diagram highlighting production, transfer and consumption of methane into the atmosphere (Rogers *et al.*, 2017)

Nitrous oxide is a component of the nitrogen (N) cycle (Figure 2.3) (Bernal & Mitsch, 2012). Its emission to the atmosphere involves several key steps such as nitrogen fixing, nitrification, assimilation, ammonification, and denitrification (Galloway *et al.*, 2004). In wetlands, the production of N_2O takes place by two processes; (i) nitrification, where ammonia is oxidized to NO_2 and then to NO_3 by nitrifying bacteria (Prosser, 1990) and (ii) denitrification, where under anaerobic conditions, denitrifying bacteria convert nitrate (NO_3) back into either nitrogen gas (N_2) or N_2O (Knowles, 1982). N_2O emission is dependent on organic matter availability, high temperatures, inundated soils and availability of N nutrient components such as ammonium, nitrite and nitrates, all of which are present in wetlands and hence they considered a source of N_2O (Zhang *et al.*, 2017).

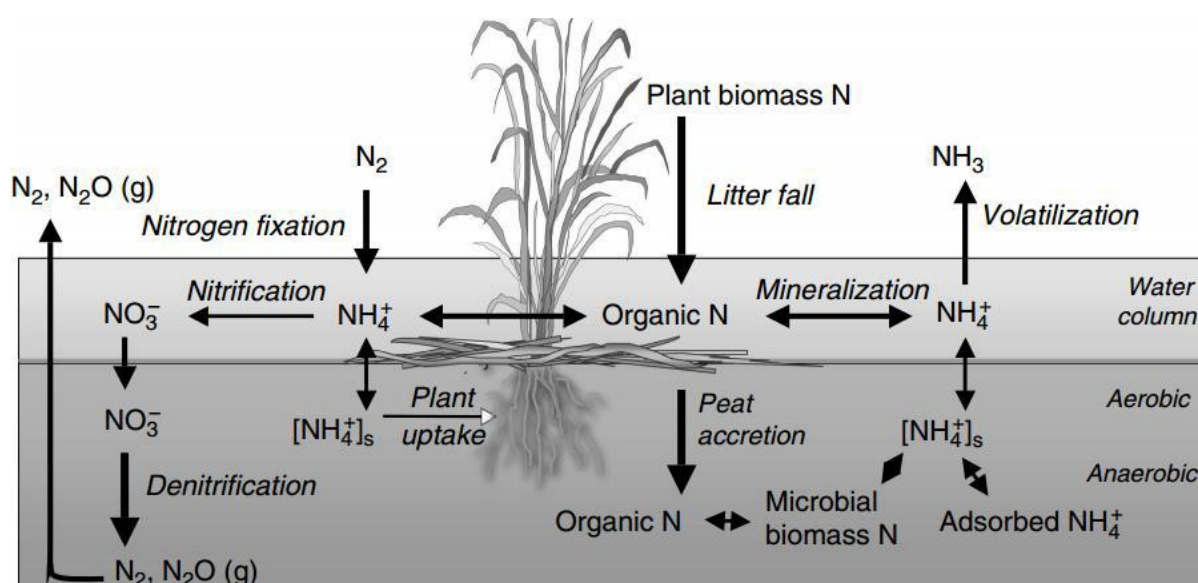


Figure 2.3: Schematic diagram of nitrogen transformations in a wetland (Reddy *et al.*, 2008)

2.5 Influence of plant communities on SOC and GHGs

Wetland plant communities significantly and in several ways influence SOC levels and GHG emissions. Different plant species and communities can affect SOC levels through various mechanisms, including biomass production, root exudation, litter quality, and interactions with soil microorganisms (Yan *et al.*, 2018). Plant communities with high biomass production contribute more organic matter to the soil, which can enhance SOC levels. The quantity and quality of plant litter (leaves, stems, roots) are crucial for SOC accumulation (Panchal *et al.*, 2022). Also, plants release organic compounds through their roots, known as root exudates, which can stimulate microbial activity and contribute to SOC formation (Panchal *et al.*, 2022). In the same way, the amount of organic matter added to wetland soils varies with different plant species (Vitousek *et al.*, 2002). The key point is that SOC accumulation and GHG flux patterns may vary spatially with plant type, even in the same wetland site.

Different plant species contribute varying quantities of organic matter to the soils through their biomass. The chemical composition of the organic matter, for instance, lignin, cellulose, and nutrient content, affects decomposition rates and SOC turnover and the subsequent GHGs (Gleixner *et al.*, 2001). Wetland plant communities are adapted to specific environmental gradients, such as water availability and soil saturation levels (Brinson, 1993). These adaptations not only affect the types of plants thriving in those conditions but also influence the overall SOC dynamics. For instance, plants that can tolerate flooding contribute organic materials that enrich the soil, while their root systems help stabilize sediments, further promoting SOC accumulation. Wetland plant communities additionally influence the GHG fluxes in different ways. For example, Moore and Dalva (1993) documented a sixfold variation

in CH₄ emissions among various plant species in a Swiss wetland. Marín-Muñiz *et al.* (2015) documented that CO₂ emissions from wetlands dominated by *Arundo donax*, *Miscanthus giganteus*, and *Phragmites australis* were more than double those from wetlands with *Chrysopogon zizanioides* and those that were unvegetated. In the same study, CH₄ emissions from planted wetlands were substantially greater than those from the unvegetated ones.

2.6 Influence of wetland conversion to cropland on SOC and GHG emissions

In East Africa, wetlands provide 10-40% of the rural population's yearly food needs (Kamiri *et al.*, 2013). This is because wetlands are very fertile and have a high-water content all year round. However, conversion of the wetlands has its negative impacts, such as loss of ecosystem services (ES) for instance C sequestration. According to a study from Ethiopia, conversion of a pristine wetland to farmland provided evidence of lost ES, as the drained wetland showed remarkable reduction in water availability and consequently reduction in crop yields (Dixon & Wood, 2003). The reduced crop yields were attributed to the general reduction in soil carbon and nitrogen occasioned by aeration of the drained soil. Converting wetlands to cropland has far reaching impacts on SOC due to changes in land use and overall soil management practices.

Conversion of wetlands to cropland drains the soils as well as aerating it, leading to intensified microbial activity and faster decomposition of organic matter (Saunders *et al.*, 2016). A study on soil carbon sequestration by Lal (2004) estimated that such conversions can result in a loss of 20-50% of the original SOC within a few decades. Further, the process of converting wetlands to cropland involves significant soil disturbance through ploughing and tilling. This disturbance exposes SOC to oxidation, further accelerating its loss. Wetlands do support a big number of plant species that contribute to high levels of organic matter. Cropland, on the other hand, typically supports monocultures or a limited variety of crops, which contribute minimal amount of organic matter to the soil (Mitsch *et al.*, 2012). The impact on SOC is more pronounced in newly converted cropland and this is because the initial conversion process involves significant soil disturbance and drainage, leading to a rapid loss of SOC. Studies have shown that the first few years after conversion have the highest rates of SOC loss (Smith *et al.*, 2007). Over time, the rate of SOC loss tends to stabilize, but the SOC levels in old cropland remain significantly lower than in natural wetlands (Ji *et al.*, 2020). Long-term agricultural practices, such as crop rotation, cover cropping, and reduced tillage, can help mitigate SOC loss but typically do not restore SOC to pre-conversion levels (Kuleshov *et al.*, 2019).

The transformation of wetlands into agricultural land substantially affects greenhouse gas emissions as a result of alterations in land use and management strategies (Lin *et al.*, 2022; Smith *et al.*, 2007; Tan *et al.*, 2023). Wetlands are carbon sinks, with their soils and vegetation sequestering substantial quantities of carbon. The conversion of wetlands to agriculture exposes soil organic matter to oxygen, resulting in heightened microbial activity and the oxidation of organic carbon. This process emits CO₂ into the atmosphere (Mitsch *et al.*, 2012). Wetlands are known sources of CH₄ due to anaerobic decomposition of organic matter. However, when wetlands are drained for agriculture, CH₄ emissions typically decrease because the anaerobic conditions necessary for CH₄ production are reduced (Mitsch *et al.*, 2012). Despite this, the overall GHG concentrations may still be high because even the CH₄ produced in the deeper soil layers is oxidized to CO₂ in the aerobic layers leading to a considerable increase in CO₂ emissions. Based on a study by Ma *et al.* (2016), due to wetland conversion, there has been approximately 4 tons of C/ha loss of SOC since 1980. This is because, draining of wetlands supply oxygen into the soil allowing rapid decomposition of SOM; hence little or no accumulation of carbon in the soil as most of it is released as CO₂. Changing a wetland to cropland does not only impact CO₂ emissions but also other potent GHGs like CH₄ and N₂O. Smith and Cohen (2004) observed that when peatlands are drained for agriculture, N₂O and CO₂ emissions rise, though, CH₄ emission declines. Type of plant growing in the wetland also influences carbon storage and release of GHGs (Tongwane *et al.*, 2016). Sugarcane and maize for example requires complete drainage and cultivation which alters the wetland's hydrology and accelerates decomposition of SOM into GHGs (Uwimana, 2019). On the other hand, rice thrives in wetland conditions (water logging), thus rice paddies could be net sinks or sources of GHGs depending on activities taking place for example, irrigation schedule and fertilizer application (Uwimana, 2019).

In wetlands, where agriculture remains non-intensive for about ten years, the N₂O emissions decline due to the exhaustion of substrates for the nitrification process (Van Lent *et al.*, 2015). However, when agriculture becomes intensive and with increased input of nitrogen from application of fertilizer, N₂O emissions increase due to increased nitrogen input from fertilizer application and the oxic soil conditions (Reddy *et al.*, 2008; Reddy & DeLaune, 2008). This is observation agrees with other studies such as Owino *et al.* 2020 and Wang *et al.* (2017) who also noted that nitrogen fertilizer application led to increased N₂O emissions. In particular, Owino *et al.* (2020) observed that N₂O emission was higher in paddy rice fields that were fertilized rice fields than those that were not fertilized. This is because the supply of nitrogen and oxygen levels in wetlands enhances the nitrification and denitrification processes

leading to increased emission of N₂O (Mitsch & Gosselink, 2015). Rogers *et al.*, (2017) reported that human impacted peatlands had higher GHG emissions, where agricultural activities, especially crop production was the main activity involved. Tangen *et al.* (2015) study how land use affected the GHG fluxes in the Prairies pothole region reported that SOC declined significantly when undisturbed wetlands were converted for agriculture.

Tropical natural wetlands have high productivity and therefore, the rate of conversion to crop land is very high. However, there are only a few studies investigating the effect of converting wetland to cropland in Kenya (Ondiek *et al.*, 2021). Conversion of wetlands to cropland is well known to cause large carbon losses via several pathways like GHG emissions, but critical data is lacking to evaluate the magnitude of carbon loss and therefore, this study will provide information to understand Yala wetland's SOC and GHGs fluxes.

2.7 Greenhouse gas measurements in wetlands

GHGs can be measured using different methods and approaches. These include indirect and direct methods, remote sensing and modelling approaches. However, 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 follow a real time and direct measurement of vertical GHG fluxes. This approach assumes that fluxes are always nearly constant with height, and that concentrations change only upwards but not horizontally. The method has an advantage over chamber-based in that, it integrates fluxes over large areas (> ha), has high temporal resolution and no interference with the microenvironment (Butterbach-Bahl *et al.*, 2016). However, the method has its limitations; need for large homogenous 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 methods involve placing a sealed chamber over the soil or water surface and measure gas concentration changes within the chamber over time. This focuses on soil surface gas concentration and thus sampling is restricted to above ground headspace. Chamber method covers a finer scale usually small surface areas up to < 1 m², hence are simple and therefore often used in most studies (Butterbach-Bahl *et al.*, 2016). Chamber based measurements have several advantages. It is cheap in terms of capital cost considering the chambers do not require power supply or fast response sensors at the site. Additionally, collected gas samples can be stored and analysed in the future and allows studies and experiments with many treatments

(Rosenstock *et al.*, 2018). While employing the static chamber gas measurement methodology, a few precautions need to be observed to ensure accuracy. For instance, due to the disturbance to the environmental conditions during installation of the chambers at the sites, all chamber bases should be installed at least a week before collection of samples begins (Butterbach-Bahl *et al.*, 2016; Collier *et al.*, 2014) and in order to eliminate high spatial variation coefficients, several chambers (three replicates) are installed to account for the spatial heterogeneity of the soil. Also, potential high emissions during sampling especially for methane are minimized by installing boardwalks in the wetland/study sites. However, according to Henson *et al.* (2013), chamber-based measurements are most likely to miss peak events such as rainfall, because the researcher may not always be at the site, in addition to the fact that chambers can only be used for a limited period per day. Consequently, Hensen *et al.* (2013) recommended use of automatic chambers to address this limitation. However, the automated chambers require more capital, making them more expensive to use and yet they too end up covering a small study area. This study employed the static chamber method, because of low capital funding, collected samples were stored for possible future analysis and, the method allowed for various experiments with many different replicates.

Gas samples collected using chamber-based method are usually analysed by either, photoacoustic spectroscopy (PAS) method or gas chromatography (GC) (Butterbach-Bahl *et al.*, 2016). In PAS, the GHG absorbs light at specific wavelength and absorption is linked to the concentration, that is; PAS usually translates the absorbed light into acoustic signal which is measured by a microphone. Gas chromatography technique employs the principle of separating a compound into its molecular constituents (Hensen *et al.*, 2013). This is the predominant method for assessing GHG concentration in gas samples from chambers (Butterbach-Bahl *et al.*, 2016). In general, a gas sample of 1–3 ml is injected into the GC, where various chemicals are separated within an analytical column. A ^{63}Ni Electron Capture Detector is typically employed for N_2O detection, functioning at temperatures between 330 and 350 °C, exhibiting maximal sensitivity to N_2O and minimal sensitivity to CO_2 (Wang *et al.*, 2010). While a flame ionisation detector (FID) is employed for CH_4 , the introduction of a mechaniser before to the detector allows for the measurement of CO_2 using either FID or a thermal conductivity detector.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Description of the study area

Yala wetland, covering 20,756 ha, is Kenya's largest freshwater wetland (Muoria *et al.*, 2015; Odero & Odenyo, 2021). It's a trans-county wetland located in Busia and Siaya counties on the north-eastern shoreline of Lake Victoria. The wetland is located between latitudes 0°6'0" S and 0°6'0" N and longitudes 33°54'0" E and 34°15'0" E (Figure 3.1). The altitude ranges between 1,150 m and 1,135 m. Yala wetland was formed because of the backflow of water from Lake Victoria and flooding from Nzoia and Yala rivers. The wetland contains three freshwater lakes: Kanyaboli, Nyamboyo and Sare, in addition to one man-made lake, Lake Bob. The wetland is characterized by *Cyperus papyrus* as the dominant plant with *Cyperus alopecuroices*, *Cyperus immensus*, *Typha latifolia*, *Phragmites mauritianus* among others as the other vegetation cover. The wetland's soils are largely fertile alluvial clays, derived from deposits from the lakes and rivers.

Yala wetland experiences a bimodal rainfall pattern with 'long rains' from March to June and 'short rains' of September to December. Yala and Nzoia catchments have high precipitation in Northern highland (1,800-2,000 mm per annum) and low precipitation in southwestern lowland (800-1,600 mm per annum), with the average rainfall of 760 mm around the wetland (Muoria *et al.*, 2015). The humidity is relatively high and the mean evaporation level is recorded as varying between 1800 mm to 2,200 mm per annum (Odero & Odenyo, 2021). The mean annual temperature is recorded as 22 °C which varies in the months between 15 °C in July to 30 °C of February and March (Muoria *et al.*, 2015).

Yala wetland is among some of the most floodplain wetlands and an important riparian habitat around the lake. It is characterized by high biodiversity richness and provides habitat for some fish species which have otherwise been lost from Lake Victoria (Muoria *et al.*, 2015). The wetland is considered an Important Bird Area (IBA) and over 172 bird species have been recorded at the site (Kratter, 2015). Some of the bird species found in the wetland are near-threatened and vulnerable, as categorized by the International Union for Conservation of Nature. They include Papyrus Yellow Warbler and Papyrus Gonolek. Mammals such as vervet monkeys, wild pigs (*Cercopithecus aethiops*) and Sitatunga (*Tragecephalus spekeii*) have also been recorded in the wetland.

Yala wetland is an important part of the economy of the rural communities through provision of various provisioning ecosystem services. The local communities neighbouring the wetland collect water for use at homes, medicinal plants, fish, papyrus for handicraft,

transportation and building materials for their livelihoods from the wetland. The wetlands also support small-scale agriculture by local communities. The communities grow crops such as maize, coco yams, vegetables, cassava, and potatoes among others which are not only for subsistence use but also for sale in the local markets. The wetland also supports large-scale commercial sugarcane production by a private investor, Lake Agro Limited.

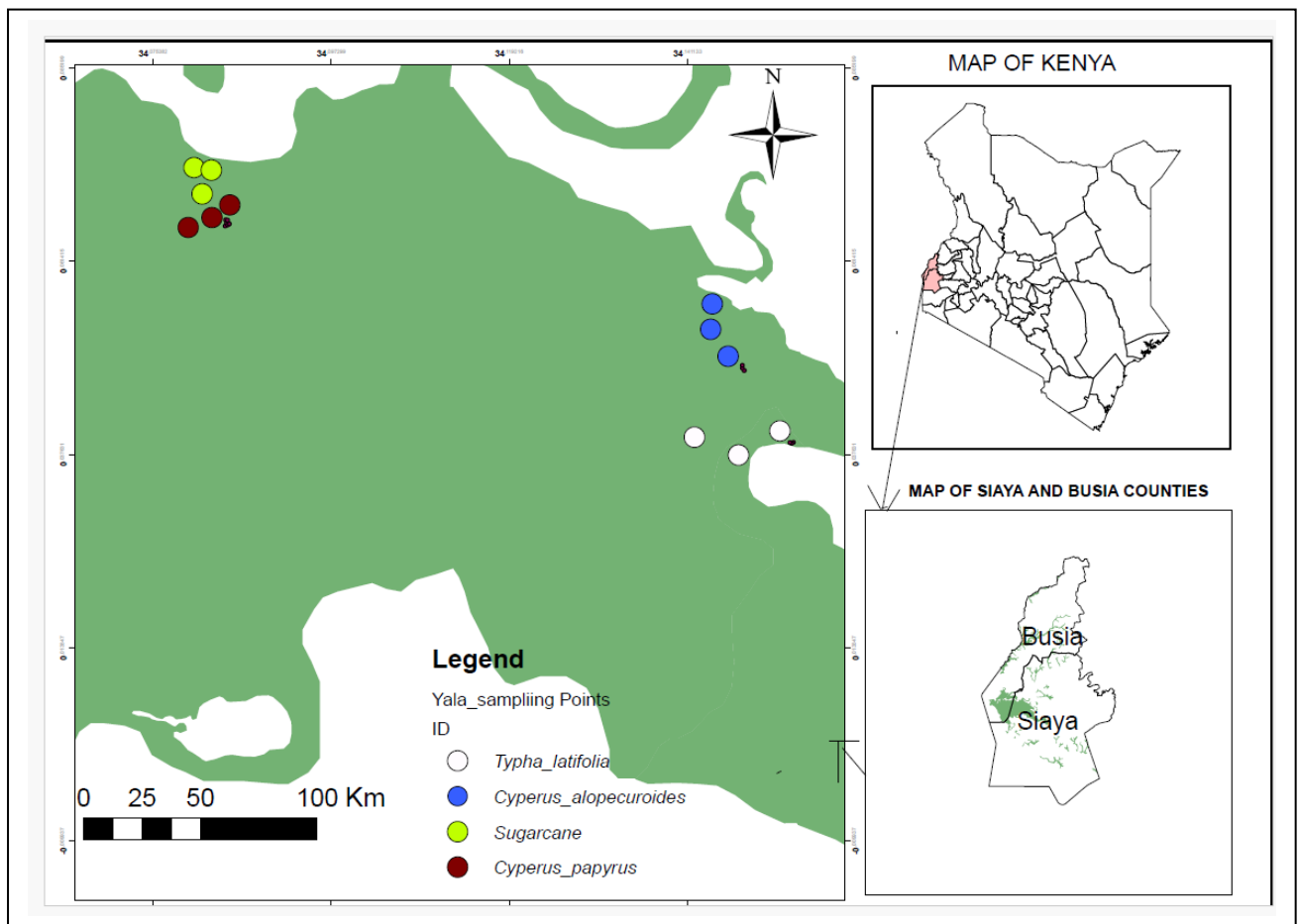


Figure 3.1: Map of Yala wetland showing selected sampling points.

Source (Modified from Topographical map of Kenya, scale 1:50,000)

3.2 Study design

The study was conducted for 3 months during the dry season between December 2024 and March 2025. The sampling design was stratified and included four sampling sites, selected in a manner that allowed for comparison among areas dominated by (i) *Cyperus papyrus* (ii) *Cyperus alopecuroides* (iii) *Typha latifolia* and (iv) areas converted to cropland. Each site had three sub-sampling sites (replicates) (Figure 3.2), each with three chambers to cater for spatial heterogeneity (Arias-Navarro *et al.*, 2013). Thus, a total of nine chambers were installed within each site. GHGs were sampled twice a month using closed-static chamber method. Soil samples

were collected in triplicates twice a month for the 3 months from each of the sampling sites up to a depth of 15cm. The soil samples for analysis of NO₃-N, and NH₄-N were collected twice a month, whereas for OC, TN, TP and bulky density, once in a month.

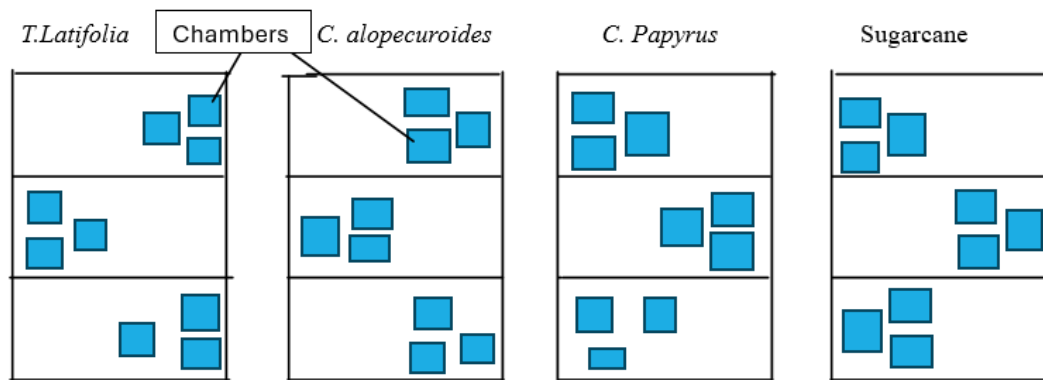


Figure 3.2: Illustration of the sampling design with each site containing a total of nine chambers.

3.3 Soil sampling

During each gas-sampling event, in each of the three dominated plant communities i.e. *C. Cyperus*, *T. latifolia* and *C. alopecuroides* and in areas converted to cropland, three 2 m² sampling plots were randomly selected along transects. Due to lack of specialized equipment (Russian Peat Borer) for soil sampling at different depths, samples for nutrients and SOC analysis randomly collected from each sampling site over a soil - depth profile of 0-15 cm using a soil auger. To cater for SOC differences due to variation in decomposition, the soil samples were collected from points in a triangular pattern. Soil samples for bulk density determination were collected once using density cups with a diameter of 5 cm and a height of 5 cm. The density cups were banged into the soil at each site in its natural state and samples obtained. The soil samples were later dried while still in the cup. The soil samples were then packed in labelled Ziploc polyethylene bags and transported to the Soil Science Laboratory at Egerton University for analysis.

3.4 Auxiliary measurements

Environmental variables including soil temperature, soil moisture and water level driving the emissions of the GHGs and SOC stock were measured in all the sampling sites. Soil temperature was measured using standard digital thermometer (Brannan) from the soil surface up to 10 cm. Water level was determined using a graduated ruler. Soil moisture content was measured using gravimetric method by oven drying 50g of soil at 105° C for 48 hours to

constant weight. This was followed by reweighing the dried soil and calculating soil moisture as described by Equation 3.2 (Okalebo *et al.*, 2002)

$$\text{Soil moisture content (\%)} = \left(\frac{\text{Weight of wet soil} - \text{weight of the dry soil}}{\text{Weight of the dry soil}} \right) * 100 \quad 3.1$$

Bulk density was calculated by filling a container of known volume (V) and weight (W1) with soil. Soil was oven dried at 105°C for 48 hours to obtain a constant weight (W2). The bulk density was then calculated as described by equation (3.2) (Okalebo *et al.*, 2002)

$$\text{Bulk density } \left(\frac{\text{g}}{\text{cm}^3} \right) = \frac{W2 - W1}{V} \quad 3.2$$

3.5 Nutrient and organic carbon analyses

In the laboratory, standard procedures as elaborated by Okalebo *et al.* (2002) were followed to determine the amount of NO₃-N, NH₄-N, SOC, TP, and TN in the soil. NO₃-N and NH₄-N, were determined using colorimetric method, where 10 g of the fresh soil samples were extracted with 100 ml of 0.5M K₂SO₄ (Okalebo., *et al.*, 2002). These samples were then filtered using membrane filter paper 47 mm diameter with pore size of 0.45µm, and the supernatant analysed for NO₃-N and NH₄-N using standard procedures for colorimetry and concentrations determined according to equation (3.3) (Okalebo *et al.*, 2002).

$$\text{NH}_4^+ (\mu\text{g kg}^{-1}) / \text{NO}_3^- (\mu\text{g kg}^{-1}) = \frac{(a-b) * V * \text{MCF} * f * 1000}{w} \quad 3.3$$

Where a = concentration of N in the solution, b = concentration of N in the blank, v = volume of the extract; w = weight of the fresh soil; MCF = moisture correction factor; f = dilution factor.

Total nitrogen (TN) was quantified by colorimetric technique. From the soil samples that were dried at 70°C in the oven, 0.3 g was subjected to digestion with 2.5 ml of a mixture of hydrogen peroxide, sulphuric acid, selenium and salicylic acid at 360°C for two hours in a block digester. Subsequently, the usual approach for the determination of NH₄-N was adhered to. The concentration of TN was determined using equation (3.4) as per Okalebo *et al.* (2002).

$$\text{TN in soil sample (\%)} = \frac{(a-b) * v * 100}{1000 * w * al * 1000} \quad 3.4$$

Where a is the concentration of N in the solution, b is the concentration of N in the blank, v is the total volume at the end of analysis, w is the weight of dried sample and al is the aliquot of the solution taken.

Total phosphorus (TP) was determined using an ascorbic method following acid digestion and using a digestion mixture at 360° C for two hours. TP concentration was then calculated as per Okalebo *et al.* (2002) using the equation (3.5) below.

$$P \text{ in the sample (\%)} = \frac{c * v * f}{w} \quad 3.5$$

Where c is the the corrected concentration of P in the sample; v is the volume of the digest; f is the dilution factor; w is the weight of the sample.

Organic carbon in the soil was quantified with the Walkley-Black method, which involves digestion with a mixture of sulfuric acid (H₂SO₄) and aqueous potassium dichromate. The soil samples were oven dried at 70°C until a constant weight was obtained. Subsequently, 0.3 g of soil underwent total oxidation utilizing a mixture of 7.5 ml concentrated H₂SO₄ and 5 ml. The principle states that Dichromate usage correlates directly with the quantity of reactive carbon. The excess K₂Cr₂O₇ was titrated with ferrous ammonium sulphate until endpoint was reached. The amount of SOC was then calculated according to equation (3.6) by Okalebo *et al.* (2002).

$$\text{Organic Carbon (\%)} = \frac{(0.003 * 0.2 (V_b - V_s) * 100)}{w} \quad 3.6$$

Where V_b is the volume of 0.2 M ferrous ammonium sulphate utilized to titrate the reagent blank solution, V_s is the volume of 0.2 M ferrous ammonium sulphate used to titrate sample solution and 12/4000 is the Mili-equivalent weight of C in grams.

SOC content and density for the different wetland plant communities and cropland, as well as their respective SOC storage potentials were determined as per the equations 3.7 (Bernal and Mitsch, 2008), 3.8 (Huo *et al.* 2013) and 3.9 (Huang *et al.* 2012)

$$SOC \text{ content} = 10 * SOC_i\% \quad 3.7$$

$$SOC \text{ density} = SOC_i\% * BD * \frac{h}{10} \quad 3.8$$

$$SOC \text{ storage potential} = \sum_{i=1}^n \left(SOC\% * BD_i * \frac{h_i}{10} \right) * 1000 \quad 3.9$$

where: SOC_i (%) is the SOC% of a wetland plant community or cropland, BD_i is the bulk density of a wetland plant community or cropland (g cm⁻³), and h_i is the soil depth of a wetland plant community or cropland (cm).

3.6 Gas sampling and analysis for CO₂, CH₄ and N₂O

The emissions of CO₂, CH₄ and N₂O from areas of the wetland dominated by *Cyperus*, *Typha latifolia* and *Cyperus alopecuroides* and in areas converted to cropland were measured using vented static chamber method, consistent with the minimum criteria for GHG measurements outlined by Rosenstock *et al.* (2016). The lid included ports for gas sampling and thermometers to measure internal temperature of the chamber, one vent tube with a length of 50 cm that has a diameter of 2.5 mm, and the top lined with foam (Butterbach-Bahl & Dannenmann, 2011). To minimize insulation and reduce the light going into the chamber, the lid was lined with an aluminium duct tape all round so that photosynthesis does not affect the CO₂ fluxes. Each site had three gas chambers, the three chambers were pooled into one sample vial, making one integrated sample per site. The chamber bases were installed 5 to 7 cm into the ground, a week before the first GHG sampling, and they were left until the end of the sampling period.

In the sampling sites, *Cyperus papyrus*, *Cyperus alopecuroides* and *Typha latifolia* stands and crops were cleared to provide ground for the base installation and thus there were rhizomes or roots inside the bases. Wooden walking platforms were also installed a month before chamber installation to limit emissions especially CH₄ through artificial gas ebullition as result of disturbing/stepping on the soil during GHG sampling. The wooden walking boards made sampling easier while standing on the boards when the ground was wet enough during measurement. In addition, floating chambers were also used in areas that flooded more, but especially in the *Cyperus papyrus* site which remained flooded throughout the entire sampling period. During the sampling, the lid and base were be clamped together with metallic clips, making the chambers airtight. In addition, during each sampling, any growing shoots were trimmed to ground level. Gas sampling occurred twice a month for a period of twelve weeks throughout the dry season from mid-December 2024 to mid-March 2025.

During each sampling, the chambers were sealed for a thirty-minute duration, after which gas samples were collected every ten minutes amounting to a total of four gas samples per site for the thirty-minute duration. To reduce the spatial heterogeneity, gas pooling technique as described by Bernal & Mitsch (2012) was employed to collect the gases from the three chambers at each site. The gas within a chamber was manually mixed before sample collection by pumping with a syringe several times to ensure that the sample is homogeneous. Thereafter, gas sample was collected using a 60 ml propylene syringe fitted with a luer lock, where 20 ml sample were drawn from each of the three chambers per replica set and

immediately transferred to a pre- labelled 10 ml glass vial with crimp seal. Using the flushing technique, the first 40 ml were used to flush the vials, and the remaining 20ml fills the vial, pressurizing it to minimize chances of leakage and contamination with ambient air. Within 12 hours after collection, the wrapped samples were transported to International Livestock Research Institute laboratory in Nairobi for CO₂, CH₄, and N₂O analyses.

Analysis of gases in this study followed the gas chromatography technique (GC) (Bucha *et al.*, 2025). Gas concentrations were measured using the SRI GHG gas chromatograph (model 8610C; SRI) with a mechaniser in combination with a flame ionization detector for CH₄, CO₂ and a 63Ni electron capture detector (Butterbach-Bahl & Dannenmann, 2011). Concentrations of the gases were then calculated based on the peak areas measured by the GC relative to the measured peak areas from calibration gases. Concentrations were converted to mass per volume flux using the ideal gas law, atmospheric pressure, internal chamber temperature, and chamber volume at the time of sampling (Butterbach-Bahl & Dannenmann, 2011).

$$\text{Flux}_{\text{GHG}}(\text{mgm}^{-2}\text{h}^{-1}) = \text{Ct} * \left(\frac{\text{M}}{\text{Vm}}\right) * \left(\frac{\text{V}_{\text{ch}}}{\text{A}_{\text{ch}}}\right) * \left(\frac{273.15}{273.15+t}\right) * \text{P} * 60 \quad 3.10$$

Where, Ct is the slope derived from the linear regression (ppmmin⁻¹) for CH₄ and CO₂ and (ppbmin⁻¹) for N₂O, M is the molar weight (gmol⁻¹) (C=12 for CH₄ and CO₂, and N=28 for N₂O), Vm is the molar gas volume (m³mol⁻¹), (22.41), Vch is the Volume of gas chamber, Ach is the area of gas chamber, t is the chamber temperature (°C), P is the pressure at time of sampling in atm and 60 is conversion factor of mins to hour.

Chamber fluxes were calculated when R squared was ≥ 0.70 (Rochette and Bertrand, 2008). The dynamics of CO₂ concentration were analysed during a 30-minute interval to corroborate the data for each chamber measurement. Chambers exhibiting a CO₂ reduction over 10% between any time intervals were presumed to be leaking. In this instance, all gas fluxes were excluded from subsequent study. The sole exception was when the reduction transpired in the last measurement, as the flux rate could still be computed using the initial three measurement points (Pelster *et al.*, 2017). Nonetheless, in the absence of leakage, negative CH₄ and N₂O readings were identified as uptake of the corresponding GHGs by soil.

3.7 Statistical data analysis

Data on CO₂, CH₄, N₂O, NH₄-N, NO₃-N, TP, TN, SOC, soil temperature, soil moisture, bulk density, and water level were entered into Microsoft Excel (2013) and analysed statistically using IBM SPSS version 20 (USA). All statistical tests were conducted at $p < 0.05$. First, data were checked for normal distribution using Shapiro-Wilk test) and homogeneity of variance using Levene's tests. None of the datasets exhibited normal distribution, even after log transformation and were therefore analysed using non-parametric tests. The Kruskal-Wallis test was used to evaluate differences in the medians of SOC and GHGs under the varying wetland plant dominance and cropland Spearman's correlation was used to assess the relationship between GHGs and soil physical and chemical properties as well the relationship between SOC and soil physical and chemical properties in areas under different wetland plant dominance and cropland.

CHAPTER FOUR

RESULTS

4.1 Physical and chemical properties of soils in natural wetland sites and those transformed into sugarcane cultivation areas

The median values for the water level, water temperature, soil temperature, bulk density, TN, TP, C: N, NH₄-N and NO₃-N at the wetland areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, *Cyperus papyrus* and areas converted to sugarcane are presented in Table 4.1 below. Water level ranged from 0.00-50.00 cm in *Typha latifolia*, 0.00-30.00 cm in *Cyperus alopecuroides*, 15.50-100.00 cm in *Cyperus papyrus* and 1.00-30.00 cm in areas converted to sugarcane. A significant variation in the water level was observed among the sites (Kruskal-Wallis: H (3) = 44.34, adj. p = 0.001 (Table 4.1). Significantly different water levels were observed between areas dominated by *Cyperus alopecuroides* and *Cyperus papyrus* (Adj. p = 0.001), between sugarcane and *Cyperus papyrus* (Adj. p = 0.001) and between *Typha latifolia* and *Cyperus papyrus* (Adj. p = 0.009). Significantly higher water level was observed in areas dominated by *Cyperus papyrus* (81.00 cm) than in other sites. There was no significant difference in water level between areas dominated by *Typha latifolia* (8.00cm) and sugarcane (5.00 cm), and between *Cyperus alopecuroides* (0.00 cm) and sugarcane (adj. p > 0.05).

Water temperature within *Typha latifolia* ranged between 21.00-27.00 °C, 22.00-29.00 °C in *Cyperus alopecuroides*, 23.00-26.00 °C in *Cyperus papyrus* areas and from 22.00 to 27.00 °C in areas converted to sugarcane. However, there was no significant differences among the sites ((Kruskal-Wallis: H (3) = 4.52, adj. p = 0.210) (Table 4.1).

Soil temperature within *Typha latifolia* ranged from 20.00-24.00 °C, 19.00-24.00 °C in areas dominated by *Cyperus alopecuroides*, 22.00-25.00 °C in *Cyperus papyrus* and 21.00-25.00 °C in areas converted to sugarcane. Soil temperature varied significantly in areas of the wetland dominated by *Typha latifolia*, *Cyperus alopecuroides*, *Cyperus papyrus* and areas converted to sugarcane (Kruskal-Wallis: H (3) = 18.59, adj. p <.001) (Table 4.1). Significant differences in soil temperature were observed between *Cyperus alopecuroides* and sugarcane (Adj. p. = 0.003) and between *Cyperus alopecuroides* and *Cyperus papyrus* (Adj. p. = 0.001). The soil temperature was significantly lower in areas dominated by *Cyperus alopecuroides* (22.00) °C than in areas dominated by *Cyperus papyrus* (24.00) °C and sugarcane (24.00) °C (adj. p < 0.05). *Typha latifolia* dominated areas had intermediate soil temperature (23.00) °C. However, there was no significant difference in soil temperature between *Typha latifolia* and *Cyperus alopecuroides* and among *Typha latifolia*, *Cyperus papyrus*, and sugarcane dominated areas (adj. p > 0.05).

Soil moisture was between 26.81-242.17% within *Typha latifolia* dominated areas, 38.28-75.74% for *Cyperus alopecuroides* areas, 30.10-203.72% in *Cyperus papyrus* dominated areas and 40.66-119.13% in areas converted to sugarcane. The highest soil moisture was observed in areas dominated by *Cyperus papyrus* (62.28%). However, the difference was not significant among the sites (Kruskal-Wallis: H (3) = 5.16, adj. p= 0.16) (Table 4.1).

Bulk density (BD) ranged from 0.79-1.24 gm⁻³ in *Typha latifolia* dominated areas, 0.57-0.80 gm⁻³ in *Cyperus papyrus* dominated areas, 0.44-0.96 gm⁻³ in *Cyperus papyrus* areas and 0.65-0.84 gm⁻³ in areas converted to sugarcane. Bulk density varied significantly among the sites (Kruskal-Wallis: H (3) = 22.01, adj. p = 0.001 (Table 4.1). Significantly higher bulk density was recorded in areas dominated by *Typha latifolia* (0.99 gm⁻³) than in areas dominated by *Cyperus papyrus* (0.82 gm⁻³), *Cyperus alopecuroides* (0.78 gm⁻³) and sugarcane (0.73gm⁻³). There was no significant difference in bulk density among areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides* and sugarcane (adj. p > 0.05). (Kruskal-Wallis: H (3) = 5.16, adj. p= 0.16) (Table 4.1).

TN ranged from 0.03-0.22% within *Typha latifolia* dominated areas, 0.06-0.22% in *Cyperus alopecuroides*, 0.65-0.84% in *Cyperus papyrus* and 0.06-0.13 in areas converted to sugarcane. TN varied significantly among the sites (Kruskal-Wallis: H (3) = 11.69, adj. p = 0.009 (Table 4.1). The only significant difference for TN was observed between *Cyperus papyrus* and sugarcane (Adj. p = 0.006). Significantly higher TN was recorded in areas dominated by *Cyperus papyrus* (0.12%) than in areas dominated by sugarcane (0.08 %). There was no significant difference in TN among *Typha latifolia*, *Cyperus alopecuroides* and sugarcane dominated areas (adj. p > 0.05). TP ranged from 0.09-0.23% within *Typha latifolia* dominated areas, 0.09-0.27% in *Cyperus alopecuroides* dominated areas, 0.10-0.22% in areas dominated by *Cyperus papyrus* and 0.09-0.27% in sugarcane dominated areas.

Higher TP was observed in areas dominated by *Cyperus papyrus*, though the difference was not significant (Kruskal-Wallis: H (3) = 5.22, adj. p = 0.157) (Table 4.1). Carbon to nitrogen ratio (C: N) ranged from 23.65-111.83 within *Typha latifolia* dominated areas, 17.05-39.55 in *Cyperus alopecuroides* areas, 20.48-38.66 in *Cyperus papyrus* dominated areas and 14.58-30.74 in areas converted to sugarcane.

C: N varied significantly in areas of the wetland dominated by *Typha latifolia*, *Cyperus alopecuroides*, *Cyperus papyrus* and in areas converted to sugarcane (Kruskal-Wallis: H (3) = 25.97, adj. p = 0.001 (Table 4.1). Areas converted to sugarcane had significantly lower C: N value (23.09) compared to all the other areas. There was no significant difference in C: N values

among areas dominated by *Typha latifolia* (38.62), *Cyperus papyrus* (31.50) and *Cyperus alopecuroides* (31.90) (adj. $p > 0.05$).

NH₄-N and NO₃-N concentrations in the soil did not vary significantly among the sites ($p > 0.05$). NH₄-N ranged from 0.04-0.52 mg/kg in areas dominated by *Typha latifolia*, 0.04-0.68 mg/kg in *Cyperus alopecuroides* dominated areas, 0.04-0.55 mg/kg in *Cyperus papyrus* dominated areas and 0.04-0.17 mg/kg in areas converted to sugarcane. NO₃-N ranged from 0.20-3.18 mg/kg within *Typha latifolia*, 0.39-7.43 mg/kg in *Cyperus alopecuroides*, 0.67-6.24 mg/kg in *Cyperus papyrus* areas and 0.40-2.32 mg/kg in areas converted to sugarcane.

Table 4.1: Physical and chemical parameters in wetland areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, *Cyperus papyrus* and those converted to sugarcane (Median, n =36). Different letters denote significant difference, while similar letters indicate no significant difference. S.M= soil moisture, B.D = bulk density, C: N = carbon nitrogen ratio

Physico-chemical parameters	Sampling sites			
	<i>T. latifolia</i>	<i>C. alopecuroides</i>	<i>C. papyrus</i>	Sugarcane
Water level (cm)	8 ^b	0 ^a	81 ^c	5 ^{ab}
Water temp. (°C)	24	24	24	24
Soil temp. (°C)	23 ^{ab}	22 ^a	24 ^b	24 ^b
S.M (%)	58.36	57.35	62.28	50.95
B.D (gm ⁻³)	0.99 ^b	0.78 ^a	0.82 ^a	0.73 ^a
TN (%)	0.08 ^a	0.08 ^a	0.12 ^b	0.08 ^a
TP (%)	0.12	0.16	0.19	0.13
C: N	38.62 ^b	31.9 ^b	31.5 ^b	23.09 ^a
NH ₄ -N (mg/kg)	0.08	0.07	0.07	0.06
NO ₃ -N (mg/kg)	0.91	1.06	0.96	0.94

4.2 Variation in SOC in the wetland areas dominated by *Typha latifolia*, *Cyperus papyrus*, *Cyperus alopecuroides* and areas converted to sugarcane

The results on proportion of organic carbon in soil, SOC content, density, and storage potential in *Typha latifolia*, *Cyperus papyrus*, *Cyperus alopecuroides* and areas converted to sugarcane are presented in Figures 4.1-4.4. The proportion of organic carbon in soil ranged from 1.99 to 5.73% in *Typha latifolia* dominated soils, 1.36–6.42% in *Cyperus papyrus*, 2.29–7.55% in areas with *Cyperus alopecuroides* and 1.30–2.89% in areas converted to sugarcane. The proportion of organic carbon varied significantly among the sites (Kruskal-Wallis: H (3)

= 26.35, $p = 0.001$) (Figure 4.1). The highest median proportion of organic carbon in soil was observed in soils under *Typha latifolia* (3.50%), followed by *Cyperus papyrus* (3.09 %), *Cyperus alopecuroides* (2.98 %) and sugarcane (1.73%) (Figure 4.1). The proportion of organic carbon in soil in areas of the wetland converted to sugarcane was significantly lower than areas dominated by *Cyperus alopecuroides* (Adj. $p = 0.002$), *Typha latifolia* (Adj. $p = 0.000$), and *Cyperus papyrus* (Adj. $p = 0.000$). However, there were no significant differences in the proportion of organic carbon in the soils among the areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, and *Cyperus papyrus* (adj. $p > 0.05$).

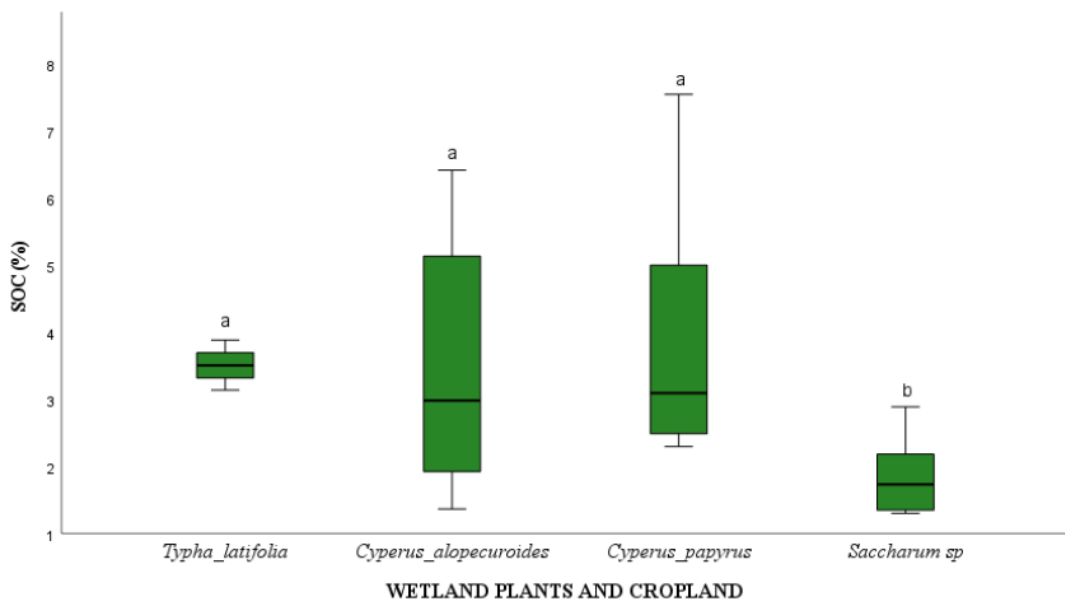


Figure 4.1: SOC proportions across areas with the different wetland plants and cropland

The median SOC content in *Typha latifolia* ranged between 19.89–57.27 g m^{-2} , *Cyperus papyrus* had 22.94–75.48 g m^{-2} , *Cyperus alopecuroides* soils ranges were between 13.64–64.17 g m^{-2} and sugarcane areas had between 12.98–28.88 g m^{-2} . The SOC content varied significantly in areas of the wetland dominated by *Typha latifolia*, *Cyperus alopecuroides*, *Cyperus papyrus* and areas converted to sugarcane (Kruskal-Wallis: $H(3) = 26.35$, $p = 0.001$) (Figure 4.2). *Typha latifolia* dominated soils had the highest median SOC content (35.05 g m^{-2}), followed by *Cyperus papyrus* (30.91 g m^{-2}), *Cyperus alopecuroides* (29.81 g m^{-2}) and lowest in sugarcane areas (17.3 g m^{-2}) (Figure 4.2). SOC content of sugarcane dominated soils was significantly lower than that of *Cyperus alopecuroides* (Adj. $p = 0.002$), *Typha latifolia* (Adj. $p = 0.000$) and *Cyperus papyrus* (Adj. $p = 0.000$). There were no significant differences in SOC content among the areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, and *Cyperus papyrus* (adj. $p > 0.05$).

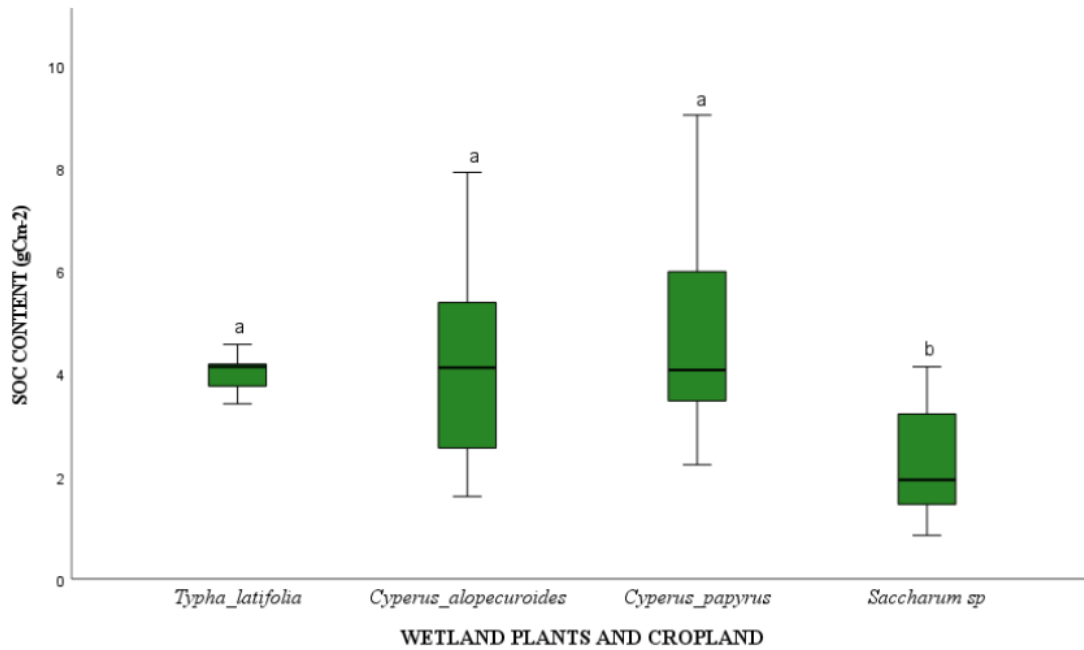


Figure 4.2: SOC content across areas with the different wetland plants and cropland

The results of SOC density within the *Typha latifolia* dominated areas ranged between 2.17–6.77 g kg⁻¹, 1.61–7.93 g kg⁻¹ in soils under *Cyperus alopecuroides*, 2.23–9.04 g kg⁻¹ for *Cyperus papyrus* soils and 0.85–4.14 g kg⁻¹ in sugarcane dominated areas. The SOC density varied significantly among the sites (Kruskal-Wallis: H (3) = 19.64, p = 0.001) (Figure 4.3). SOC density was highest in soils under *Typha latifolia* (4.14g C kg⁻¹), followed by *Cyperus alopecuroides* (4.12 g kg⁻¹), *Cyperus papyrus* (4.07 kg⁻¹), and lowest in sugarcane (1.93 g kg⁻¹) (Figure 4.3). Significantly lower SOC density was observed in areas under sugarcane than in areas dominated by *Typha latifolia* (Adj.p = 0.007), *Cyperus alopecuroides* (Adj.p = 0.008) and *Cyperus papyrus* (Adj.p = 0.000). However, there were no significant differences in SOC density among the areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, and *Cyperus papyrus* (adj. p > 0.05).

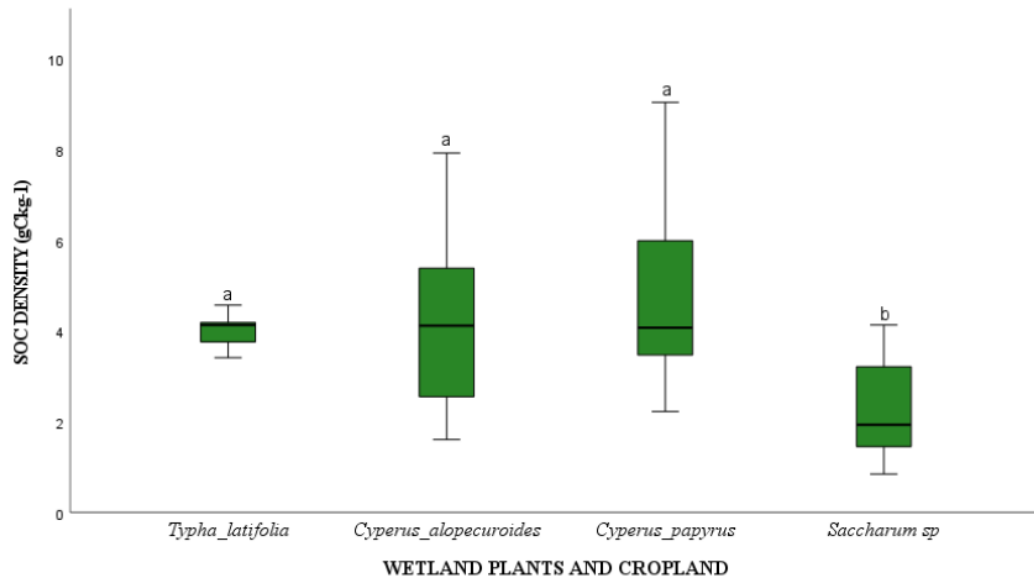


Figure 4.3: SOC density across areas with the different wetland plants and cropland

The SOC storage potential of *Typha latifolia* dominated soils ranged between 2,164.87–6,772.92 t km⁻², 1,605.67–7,925.17 t km⁻² in *Cyperus alopecuroides* soils, *Cyperus papyrus* had between 2,224.80–9,042.69 t km⁻² while areas converted to sugarcane had 847.42–4,135.07 t km⁻². The SOC storage potential varied significantly among the sites (Kruskal-Wallis: H (3) = 19.64, p = 0.001) (Figure 4.4). The median SOC storage potential was highest in soils under *Typha latifolia* (4,135.57 t km⁻²), followed by *Cyperus alopecuroides* (4,117.26 t C km⁻²), *Cyperus papyrus* (4,071.48 t km⁻²), and sugarcane (1,931.64 t km⁻²) had the lowest (Figure 4.4). SOC storage potential in areas under sugarcane was significantly lower than *Typha latifolia* (Adj. p = 0.007), *Cyperus alopecuroides* (Adj. p = 0.008) and *Cyperus papyrus* (Adj. p = 0.000). There was, however, no significant differences in SOC storage potential among the areas dominated by *Typha latifolia*, *Cyperus alopecuroides*, and *Cyperus papyrus* (adj. p > 0.05).

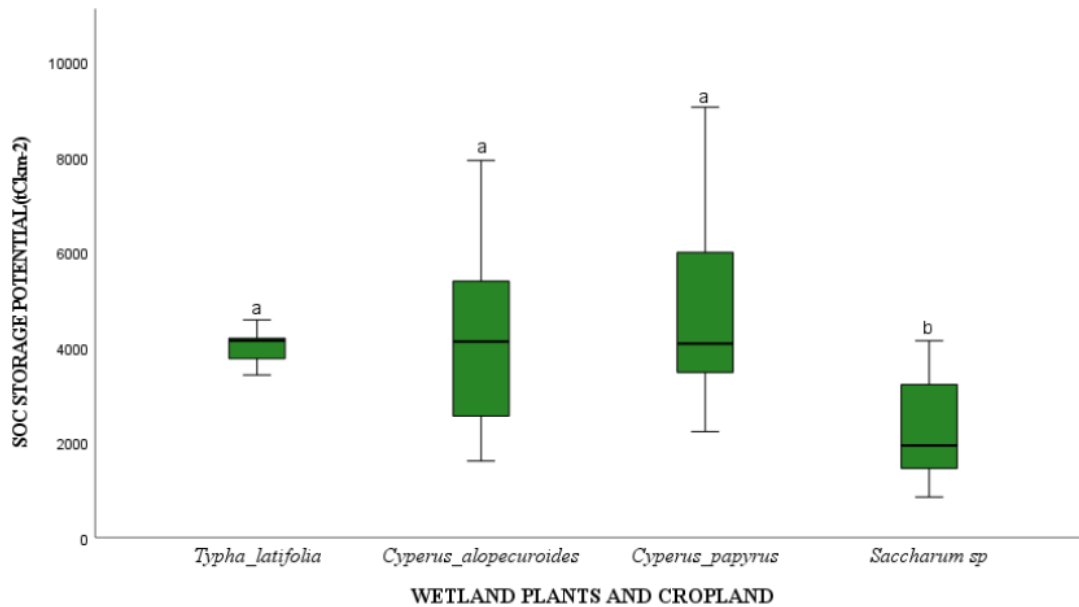


Figure 4.4: SOC storage potential across areas with the different wetland plants and cropland

4.3 Variations in GHG fluxes in wetland areas dominated by *Typha latifolia*, *Cyperus papyrus*, and *Cyperus alopecuroides* and areas converted to sugarcane

Results on GHGs recorded in areas dominated by *Typha latifolia*, *Cyperus papyrus*, *Cyperus alopecuroides* and areas converted to sugarcane are presented in Figures 4.5-4.7. Only CH₄ showed significant differences among the sites. CH₄ emissions ranged between -34.65-40.84 mg m⁻² h⁻¹ in *Typha latifolia* dominated soils, 0.08-95.99 mg m⁻² h⁻¹ in *Cyperus alopecuroides*, 0.093-187.05 mg m⁻² h⁻¹ in *Cyperus papyrus* and 0.31-23.96 mg C m⁻² h⁻¹ within sugarcane areas. The highest median CH₄ emissions were observed in *Cyperus alopecuroides* (27.19 mg m⁻² h⁻¹), followed by *Typha latifolia* (1.28 mg m⁻² h⁻¹), and sugarcane (0.31 mg C m⁻² h⁻¹). The lowest median CH₄ emissions were recorded in *Cyperus papyrus* (0.09 mg m⁻² h⁻¹) (Figure 4.5). CH₄ emissions varied significantly in areas of the wetland dominated by *Cyperus alopecuroides*, *Cyperus papyrus* and areas converted to sugarcane (Kruskal-Wallis: H (3) = 12.24, p = 0.007 (Figure 4.5). Significant differences were noted between *Cyperus alopecuroides* and *Cyperus papyrus* (Adj.p = 0.016) and between *Cyperus alopecuroides* and sugarcane (p = 0.029). CH₄ emissions were significantly higher in *Cyperus alopecuroides* areas compared to *Cyperus papyrus* and areas converted to sugarcane. CH₄ emissions in *Typha latifolia* (1.28 mg m⁻² h⁻¹) were not significantly different from those in other sites (adj. p > 0.05).

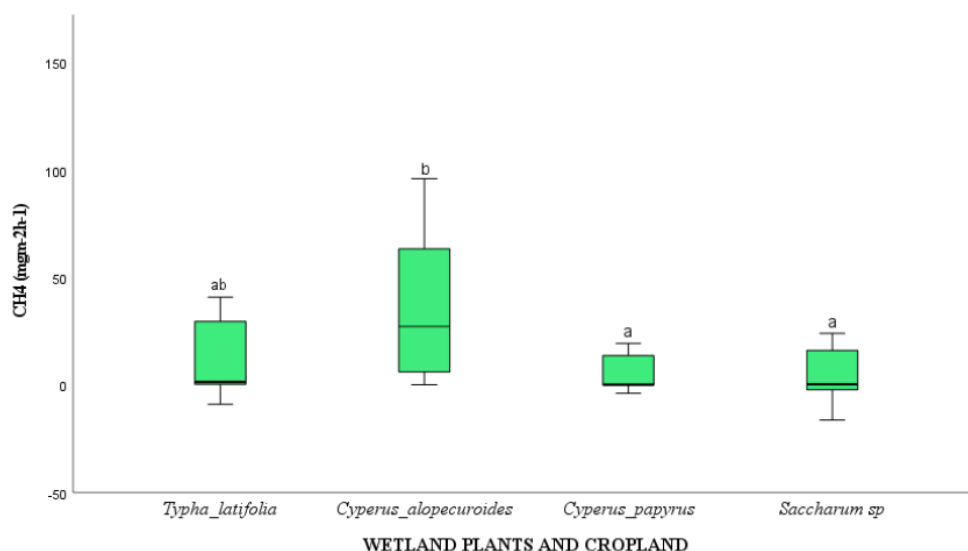


Figure 4.5: Variation in CH₄ emissions in wetland areas dominated by different plants and areas converted to sugarcane

CO₂ emissions in *Typha latifolia* were between 0.04-215.37 mg m⁻² h⁻¹, 7.50-216.17 mg m⁻² h⁻¹) in *Cyperus alopecuroides*, 14.12-154.37 mg m⁻² h⁻¹ in *Cyperus papyrus* soils and 7.37-81.82 mg m⁻² h⁻¹ from areas converted to sugarcane. Even though *Cyperus alopecuroides* emitted the highest CO₂ (67.63 mg m⁻² h⁻¹) and sugarcane the lowest (43.21, mg m⁻² h⁻¹) with *Typha latifolia* (54.22 mg m⁻² h⁻¹) and *Cyperus papyrus* (59.83 mg m⁻² h⁻¹) having intermediate values, CO₂ emissions showed no significant differences among the sites (Kruskal-Wallis: H (3) = 3.44, p = 0.33) (Figure 4.6).

Like CO₂, N₂O emissions did not vary significantly among the sites (Kruskal-Wallis: H (3) = 5.88, p = 0.118) (Figure 4.7). The N₂O emissions ranged between -4.70-11.76 μg m⁻² h⁻¹ in *Typha latifolia* area, -4.09-6.13 μg m⁻² h⁻¹ in *Cyperus alopecuroides*, -3.35-15.20 μg m⁻² h⁻¹ in *Cyperus papyrus* and -0.82-12.15 μg m⁻² h⁻¹ in areas converted to sugarcane. The highest median N₂O emissions were observed in *Cyperus papyrus* (3.94 μg m⁻² h⁻¹), followed by *Typha latifolia* (0.23 μg m⁻² h⁻¹), and sugarcane (0.24 μg m⁻² h⁻¹). N₂O uptake was observed in *Cyperus alopecuroides* (-0.60 μg m⁻² h⁻¹) dominated areas. Also, the negative range values observed in all the sites indicate N₂O uptake.

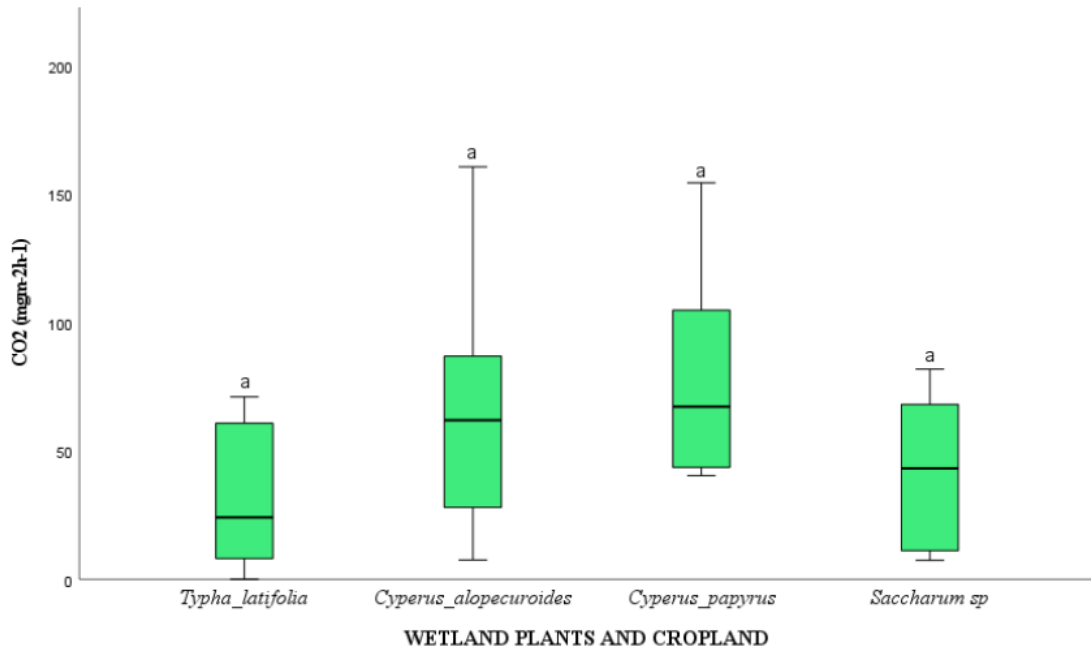


Figure 4.6: Variation in CO₂ emissions in wetland areas dominated by different plants and areas converted to sugarcane

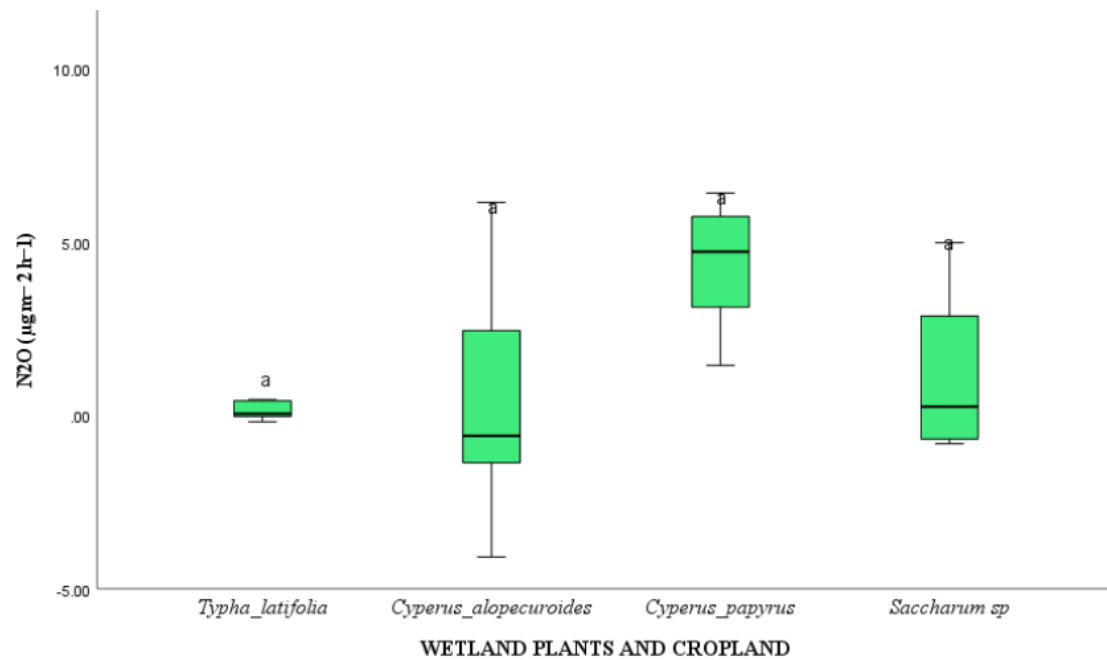


Figure 4.7: Variation in N₂O emissions in wetland areas dominated by different plants and areas converted to sugarcane

4.4 Relationships between SOC and soil physio-chemical characteristics and between GHGs (CO₂, CH₄, N₂O) and soil physico-chemical characteristics

4.4.1 Relationship between SOC and physico-chemical characteristics

Only TN had a significant correlation with SOC in all the three wetland plants dominated areas and, in the cropland (sugarcane) (Table 4.2). In *Typha latifolia* dominated areas, SOC was significantly positively correlated with TN ($r = 0.810$, $p < 0.001$) and significantly negatively with C: N ratio ($r = 0.580$, $p < 0.001$) (Table 4.2). In *Cyperus alopecuroides* dominated areas, SOC was significantly positively correlated with TN ($r = 0.850$, $p < 0.001$) and TP ($r = 0.683$, $p < 0.001$). In *Cyperus papyrus* dominated areas, SOC was significantly positively correlated with water temperature ($r = 0.783$, $p < 0.001$), soil temperature ($r = 0.488$, $p = 0.004$), water level ($r = 0.416$, $p = 0.002$), C: N ($r = 0.683$, $p < 0.001$) and TN ($r = 0.550$, $p = 0.02$) (Table 4.2). In sugarcane area, SOC was significantly positively correlated with soil moisture ($r = 0.560$, $p = 0.016$) and TN ($r = 0.633$, $p = 0.005$).

Table 4.2: Correlational relationship between SOC and physico- chemical parameters. S.M= soil moisture, B.D = bulk density, C: N = carbon nitrogen ratio

Parameter	<i>T. latifolia</i>		<i>C. alopecuroides</i>		<i>C. papyrus</i>		Sugarcane	
	Corr. Coeff.	p-value	Corr. Coeff.	p-value	Corr. Coeff.	p-value	Corr. Coeff.	p-value
Water temp	-0.162	0.548	0.247	0.324	0.738**	<0.001	-0.052	0.837
Soil temp	0.061	0.821	-0.298	0.229	0.488*	0.04	-0.06	0.812
Water level	-0.179	0.507	-0.4	0.1	0.292	0.24	0.051	0.841
S.M	0.467	0.068	0.182	0.469	0.029	0.909	0.560*	0.016
C: N	-0.580**	<.001	-0.264	0.054	0.422**	0.001	-0.105	0.448
B. D	0.19	0.48	0.133	0.598	0.683**	0.002	0.383	0.116
NH ₄ -N	0.379	0.148	-0.182	0.469	0.091	0.719	0.398	0.102
NO ₃ -N	0.302	0.256	-0.464	0.052	0.182	0.469	0.17	0.5
TN	0.810**	<0.001	0.850**	<0.001	0.550*	0.018	0.633**	0.005
TP	-0.143	0.598	0.683**	0.002	-0.167	0.509	0.418	0.084

* Correlation is significant at the 0.05 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

4.4.2 Relationship between soil GHGs and physico-chemical characteristics

In *Cyperus papyrus* and sugarcane sites, the two GHGs, CO₂ and CH₄ had a positive significant correlation with each other ($r = 0.762$, $p = 0.028$ and $r = 0.685$, $p = 0.029$ respectively) (Table 4.3). In *Typha latifolia* dominated areas, CH₄ showed a significant positive

-0.659, $p = 0.008$). No significant correlations with CH₄ were observed at areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides* and sugarcane.

CO₂ showed a significant positive correlation with NO₃-N ($r = 0.555$, $p = 0.049$) in *Typha latifolia* dominated areas. Within *Cyperus alopecuroides* dominated areas, CO₂ showed significant positive correlation with four out of the parameters tested i.e. water temperature ($r = 0.586$, $p = 0.013$), soil temperature ($r = 0.483$, $p = 0.05$), water level ($r = 0.595$, $p = 0.012$), and TP ($r = 0.658$, $p = 0.004$) (Table 4.3).

N₂O showed significant positive correlation with water temperature ($r = 0.759$, $p = 0.004$) and TP ($r = 0.799$, $p = 0.002$) in *Typha latifolia* dominated areas. No significant correlation was recorded in *Cyperus alopecuroides* dominated areas. Also, N₂O showed a significant negative correlation with soil temperature within *Cyperus papyrus* dominated areas ($r = -0.882$, $p = 0.004$) and for sugarcane dominated areas, only water level ($r = -0.809$, $p = 0.008$) showed a significant negative correlation with N₂O (Table 4.3).

Table 4.3: Correlational relationship between the GHGs and physico-chemical parameters. S.M= soil moisture, B.D = bulk density, C: N = carbon nitrogen ratio

Site	Parameter		CO ₂	CH ₄	N ₂ O	Water temp	Soil temp	Wate level	S. M	B.D	NH ₄ -N	NO ₃ -N	TN	TP	C: N
<i>T. latifolia</i>	CO ₂	Corr. Coeff.	1.000	0.441	-0.283	-0.445	0.120	-0.142	0.313	-0.227	-0.071	0.555*	0.478	-0.412	0.082
		p-value	.	0.152	0.460	0.128	0.696	0.644	0.297	0.456	0.817	0.049	0.099	0.162	0.810
	CH ₄	Corr. Coeff.	0.441	1.000	-0.345	-0.084	0.556*	0.117	0.357	-0.144	0.461	0.361	-0.045	-0.659**	0.500
		p-value	0.152	.	0.328	0.766	0.031	0.677	0.191	0.609	0.084	0.187	0.874	0.008	0.069
	N ₂ O	Corr. Coeff.	-0.283	-0.345	1.000	0.759**	-0.332	-0.496	0.175	-0.048	-0.441	-0.021	0.405	0.799**	-0.312
		p-value	0.460	0.328	.	0.004	0.292	0.101	0.587	0.881	0.152	0.948	0.192	0.002	0.380
<i>C. alopecuroides</i>	CO ₂	Corr. Coeff.	1.000	0.464	-0.700	0.586*	0.483*	0.595*	0.174	-0.087	0.400	0.056	-0.177	0.658**	0.209
		p-value	.	0.095	0.188	0.013	0.050	0.012	0.504	0.740	0.112	0.830	0.496	0.004	0.420
	CH ₄	Corr. Coeff.	0.464	1.000	-0.800	-0.077	0.280	-0.009	0.146	-0.059	-0.100	-0.196	0.110	0.347	0.142
		p-value	0.095	.	0.200	0.786	0.312	0.974	0.603	0.835	0.723	0.483	0.698	0.206	0.614
	N ₂ O	Corr. Coeff.	-0.700	-0.800	1.000	-0.353	-0.383	-0.270	0.143	-0.488	-0.657	0.143	0.290	0.058	-0.696
		p-value	0.188	0.200	.	0.492	0.454	0.604	0.787	0.326	0.156	0.787	0.577	0.913	0.125
<i>C. papyrus</i>	CO ₂	Corr. Coeff.	1.000	0.762*	-0.500	-0.259	-0.206	-0.163	-0.022	0.234	-0.473	-0.165	0.307	0.544	-0.506
		p-value	.	0.028	0.253	0.392	0.498	0.595	0.943	0.441	0.103	0.590	0.308	0.055	0.078
	CH ₄	Corr. Coeff.	0.762*	1.000	-0.400	0.520	0.459	0.124	-0.500	-0.101	-0.500	-0.282	0.274	0.164	0.210
		p-value	0.028	.	0.505	0.101	0.156	0.717	0.117	0.767	0.117	0.401	0.415	0.629	0.535
	N ₂ O	Corr. Coeff.	-0.500	-0.400	1.000	-0.124	-0.882**	-0.412	0.452	0.026	0.143	-0.143	-0.599	-0.192	-0.072
		p-value	0.253	0.505	.	0.770	0.004	0.310	0.260	0.952	0.736	0.736	0.117	0.649	0.866
<i>Sugarcane</i>	CO ₂	Corr. Coeff.	1.000	0.685*	-0.214	-0.410	-0.234	0.391	-0.133	-0.118	-0.098	0.287	0.165	-0.503	0.074
		p-value	.	0.029	0.610	0.186	0.463	0.208	0.681	0.714	0.762	0.366	0.607	0.096	0.819
	CH ₄	Corr. Coeff.	0.685*	1.000	-0.667	-0.302	0.360	-0.094	0.329	-0.372	-0.112	-0.105	0.092	-0.074	0.148
		p-value	0.029	.	0.071	0.340	0.250	0.772	0.297	0.233	0.729	0.746	0.777	0.818	0.646
	N ₂ O	Corr. Coeff.	-0.214	-0.667	1.000	-0.106	0.505	-0.809**	0.017	0.285	-0.350	0.100	-0.295	-0.051	0.127
		p-value	0.610	0.071	.	0.785	0.166	0.008	0.966	0.457	0.356	0.798	0.440	0.896	0.746

* Correlation is significant at the 0.05 level (2-tailed).

** Correlation is significant at the 0.01 level (2-tailed).

CHAPTER FIVE

DISCUSSIONS

5.1 Physico-chemical characteristics in the wetland areas and areas converted to sugarcane

Significantly higher water level was observed in areas dominated by *Cyperus papyrus* than in all the other sites. The presence of *Cyperus papyrus* in high-water level areas reflects its adaptability for deeper water conditions due to its unique characteristics, such as aerenchyma and rhizome systems, which allow it to thrive in waterlogged environments (Kansiime *et al.*, 2007; Rongoei & Outa, 2016). The *Cyperus papyrus* vegetation community has two growth forms which can grow either as emergent, where it is rooted in the sediment or floating and consisting of a mat above the water column particularly in areas with high water level (Kansiime *et al.*, 2007). This study area had both growth forms of *Cyperus papyrus*; however, the sampling points (Figure 3.1) were situated in the emergent form. Conversely, *Typha latifolia* and *Cyperus immensus* grow only as emergent, that is, they are rooted in the sediments, making them susceptible to being outcompeted by *Cyperus papyrus* in high-water level locations. The higher water level recorded in the *Cyperus papyrus* study area can also be attributed to the fact that River Yala had diverted back to its original channel, causing the site to be flooded. Kansiime *et al.* (2007) observed similar findings in Nakivubo wetland in Uganda, reporting significant higher water levels in *Cyperus papyrus* dominated wetlands as compared with other wetland types. Contrarily Were *et al.* (2021) in Naigombwa wetlands in Uganda reported nonsignificant water level differences between *Cyperus papyrus* and *Typha latifolia* dominated areas during the dry season and this was because apart from the two areas being hydrologically connected, the soils reached maximum saturation during the dry season, allowing water to flow between them and reducing differences in water levels.

Soil temperature was significantly lower in areas dominated by *Cyperus alopecuroides* (22.00) °C than in areas dominated by *Cyperus papyrus* (24.00) °C and sugarcane (24.00) °C. *Cyperus alopecuroides* have a denser or more extensive canopy compared to *Cyperus papyrus* and sugarcane, which could result in greater shading of the soil surface (Keddy, 2010). Increased shading reduces the amount of radiation from the sun reaching the soil, leading to lower temperature in the soil (Salmon *et al.*, 2008). Also, *Cyperus alopecuroides* have a higher leaf area index (LAI) compared to *Cyperus papyrus* and sugarcane (Asner *et al.*, 2010). A higher LAI increases the interception of sunlight by the plant canopy, lessening heat input into the soil.

Contrary to this study findings, Opio *et al.* (2016) reported higher soil temperature in areas dominated by *Cyperus alopecuroides* and *Cyperus papyrus*. This was attributed to reduced vegetation density in *Cyperus alopecuroides* areas, which allowed more direct sunlight to reach the soil surface. Also, Cerri (2021) examined soil temperature dynamics in sugarcane fields compared to natural wetlands in Brazil. His findings showed that soil temperatures in sugarcane fields were significantly higher than in wetlands dominated by *Cyperus alopecuroides*. This he attributed to reduced soil moisture and less shading in sugarcane fields, especially during the early stages of crop growth, concluding that agricultural practices, such as tillage and reduced soil cover during planting, exposed the soil to direct sunlight, leading to higher temperatures in sugarcane fields. Recently, Saunders *et al.* (2022) examined how the microclimatic effects of wetland vegetation on soil temperature in tropical wetlands dominated by *Cyperus alopecuroides* and *Cyperus papyrus*. In their findings, soil temperatures under *Cyperus alopecuroides* and *Cyperus papyrus* were found to have no significant differences. This was attributed to the dense canopy of both species.

Bulk density indicates soil compaction (Mouazen, 2003). It's a critical soil property influenced by both land use and vegetation cover whereby areas such as wetlands, forests or grasslands with dense root systems often have lower bulk density due to increased organic matter and porosity in the soil (Zhang *et al.*, 2023). Changes in bulk density are primarily driven by soil compaction from machinery and reduced organic inputs, which can negatively impact soil structure as well as plant growth (Wang *et al.*, 2021). In this study, areas dominated by *Typha latifolia* had significantly greater bulk density than areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides* and sugarcane. This may be due to *Typha latifolia* having a fibrous root system that can compact the soil more effectively than the root systems of *Cyperus papyrus*, *Cyperus alopecuroides* and sugarcane (Mitsch & Gosselink, 2000). This dense and shallow root mats of *Typha* can lead to higher soil bulk density by compressing soil particles and reducing pore spaces. Also, due to its lower lignin content and higher nutrient concentrations, which makes its biomass more readily decomposable by microbes, *Typha latifolia* tends to have faster decomposition rates (Mitsch & Gosselink, 2015). *Typha latifolia* dominated areas tend to have lower organic matter accumulation compared to areas dominated by *Cyperus papyrus* and *Cyperus alopecuroides* (Kadlec & Wallace, 2009; Tisdall, 2020). Organic matter in the soil leads to reduced bulk density by increasing porosity and decreasing particle packing. Therefore, this implies that if *Typha latifolia* contributes less organic matter to the soil, this could result in higher bulky density as per the findings of this study.

Significantly higher TN was recorded in areas dominated by *Cyperus papyrus* than in all the other sites. This may be due to *Cyperus papyrus* significant nutrient uptake and accumulation in its biomass. Papyrus usually has higher biomass compared to other macrophytes (Kansiime *et al.*, 2007). This allows it to store substantial amount of nitrogen which end up in the soil when papyrus die-off.

Soil TP content was observed not to be significantly different among the sites, though *Cyperus papyrus* and *Cyperus alopecuroides* had higher values. The natural wetland was characterised by dense stands of papyrus, with massive root volumes and this could be a reason for the higher content of TP observed. Vegetation and detritus accumulation are key among the factors that influence P accumulation (Reddy *et al.*, 2008).

The areas of the wetland that were being cultivated for sugarcane had significantly lower C: N value (23:1) compared to all the other areas. Two primary biological processes that are responsible for transformations of nitrogen in soil are assimilation and mineralization (Booth *et al.*, 2005). Assimilation of nitrogen by microorganisms and mineralization of organic N to ammonium. Higher NO₃-N and NH₄-N could be associated to mineralization, which tends to occur in more aerated conditions (Booth *et al.*, 2005). The lower C:N in areas under sugarcane cultivation could be explained by high mineralization in the converted area of the wetland. Microorganisms use carbon as an energy source and nitrogen for growth (Zhang *et al.*, 2017). C: N less than 30 stimulates/promotes net mineralization of organic nitrogen resulting in release of inorganic nitrogen, ammonium which is oxidised to nitrate through nitrification process. Under anaerobic conditions or in anaerobic microsites nitrate is reduced to N₂O. C: N greater than 30 favours net immobilization of inorganic nitrogen because of assimilation. A higher C:N ratio in *Typha latifolia* and other wetland plants dominated sites suggests slower organic matter decomposition, which reduces nitrogen mineralization and associated greenhouse gas emissions.

5.2 Soil organic carbon in the natural wetland areas dominated by different wetland plants and area converted to sugarcane

Wetlands typically have high organic matter in their soil due to the deposit of dead plant materials in anaerobic conditions, which slows down the rate of decomposition (Brown *et al.*, 2017). Generally, since plant biomass (above and below) is usually the main source of soil organic matter, highly productive plant species or communities typically lead to greater SOC accumulation because they provide more organic inputs (Wang *et al.*, 2017). According to Were *et al.* (2021), the average SOC content and densities in areas of a wetland dominated by

Cyperus papyrus are higher than those areas dominated by *Typha latifolia* and *Phragmites mauritianus*. This is due to differences in their primary productivities, with *Cyperus papyrus* having the highest production. However, the average SOC density of the three communities is comparable. However, in the present study, the proportions of organic carbon to dry soil, density, content and storage potential, were not statistically different among the wetland areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides* and *Typha latifolia*. This is because, all these wetland plant species were within the same study area with uniform environmental conditions (e.g., waterlogging, soil type, and nutrient availability), which factors could have overridden species-specific differences in SOC dynamics. In addition, these plants have relatively similar; productivity, root biomass, and litter quality (Lal, 2004). thus, their organic matter decomposes at similar rates, the accumulation and storage of SOC are as well also comparable

The proportion of organic carbon density and content in soil as well as storage potential, observed in this study were significantly lower in areas of the wetland converted to sugarcane than in areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides*, and *Typha latifolia*. This indicates that, regardless of vegetation community, drainage and conversion of wetlands, supplies oxygen into the soil allowing its SOM to decompose rapidly, hence little or no accumulation of carbon in the soil (Saunders *et al.*, 2016). Sugarcane cultivation often involves the removal of crop residues and frequent soil disturbance, resulting in reduction in the amount of organic carbon in the soil (Galdos *et al.*, 2009; Lal, 2004; Mitsch & Gosselink, 2007; Reddy *et al.*, 2008). Sugarcane cultivation involves frequent soil disturbance through ploughing, tilling, and harvesting. These practices increase soil aeration, which speed up the decomposition of organic matter, reducing SOC levels (Robertson & Vitousek, 2009). The drainage and cultivation of wetlands also create aerobic conditions that accelerate decomposition of SOC, consequently reducing carbon stock in the soil (Mitsch *et al.*, 2013; Ni *et al.*, 2009; Zhang *et al.*, 2025). Studies that document similar results regarding the significantly lower proportion of SOC in soils of wetland areas converted to sugarcane cultivation do exist. Don and Freibauer (2011) while studying how changes land use impacts the amount of organic carbon stored in the soil, concluded that, the conversion of wetlands as well as forests to agricultural systems, including sugarcane cultivation, results in a substantial decline in soil organic carbon stocks. Like this study, they found out that SOC losses were particularly high in tropical regions where wetlands were drained and converted to croplands. Similar results were reported in Brazil, where Galdos and Cerri (2009) found comparable trends in soil carbon stocks under sugarcane expansion. The study attributed SOC losses to the

removal of crop residues, soil erosion, and the rapid decomposition of organic matter under aerobic conditions in sugarcane fields. Similar findings were reported by Lal (2004) and Mitsch and Gosselink (2000). In contrast, while most studies support the observation that the wetland drainage and conversion to sugarcane cultivation leads to a significant reduction in SOC, there are some studies with contrary results. These studies suggest that under specific conditions or management practices, SOC levels in sugarcane-dominated areas may not be significantly lower or may even increase compared to natural wetland areas. For instance, Blair and Korschens (2006) reported that long-term sugarcane cultivation with proper residue management, for instance, leaving residues such as leaves on the field, can maintain or increase the amount of organic carbon in the soil. More studies like Six *et al.* (2002) and Graham and Meyer (2002), emphasized the significance of sustainable management practices in conservation of soil.

5.3 GHG fluxes in wetland areas dominated by different plants and areas converted to sugarcane

CH₄ emissions were significantly lower in areas converted to sugarcane and areas dominated by *Cyperus papyrus* than those dominated by *Cyperus alopecuroides*. The low CH₄ emissions in *Cyperus papyrus* wetlands can be due to the possibility of oxidation due to the high-water level in the area. High water levels create conditions where oxidation becomes the primary mechanism for reducing CH₄ emissions (Jiahui *et al.*, 2022). Also, sugarcane dominated soils, may experience more aerobic conditions due to better drainage or oxygen diffusion, which inhibits methanogenesis. In contrast, soils under *Cyperus alopecuroides* remain more waterlogged and anaerobic, creating favourable conditions for methanogenic activity and higher CH₄ emissions (Bridgham *et al.*, 2013; Wang *et al.*, 2020). In this study, both factors seem to have played a role in bringing about higher CH₄ in the *Cyperus alopecuroides* since both may lead to anaerobic conditions promoting methanogenesis and enhancing CH₄ emissions and at the same time decreases CO₂ emissions as it impedes aerobic organic matter decomposition (Brooker *et al.*, 2014; Gumaily *et al.*, 2016). According to Lunt *et al.* (2019), CH₄ emissions vary significantly depending on vegetation type and hydrological conditions and *Cyperus papyrus* dominated wetlands tend to emit less CH₄ which aligns with this study's findings. Also, consistent with the findings of this study, Hadi *et al.* (2005) found that CH₄ emissions were significantly higher in rice paddies compared to natural wetlands. The differences were attributed to changes in water management, and they concluded that land-use changes can significantly alter GHG emissions due to shifts in soil conditions, nutrient inputs,

and microbial processes. However, Were *et al.* (2021b) while studying CO₂ and CH₄ in Naigombwa wetland in Uganda, found out that except for Phragmites, there was no difference in either CO₂ or CH₄ emissions among the vegetation communities of the intact sections of the wetland. CH₄ uptake was observed in all sites, and this is because though wetlands are significant sources of CH₄ due to anaerobic decomposition of organic matter by methanogens, methane-oxidizing bacteria (methanotrophs) in the aerobic zones of wetlands, such as the soil-water interface or plant rhizosphere, can consume CH₄, reducing its emission (Reddy *et al.*, 2008).

Contrary to what was expected that CO₂ emissions would be higher in areas converted to sugarcane, variation in CO₂ emissions among the sites was not significant. The conversion of wetlands to agricultural fields usually disrupts the natural wetland carbon storage processes exposing organic-rich soils to oxygen which results to increased CO₂ emissions (Lal, 2004). In addition, SOC was significantly lower in the sugarcane area, also contributing to the low CO₂ emissions observed in sugarcane areas (Marín-Muñiz *et al.*, 2015). Similar findings were communicated by Marin-Muniz *et al.* (2015) in Mexican wetlands, where the variation in CO₂ fluxes among vegetation communities with similar amounts of organic carbon were insignificant. Additionally, Ondiek *et al.* (2021), considering only *Cyperus papyrus* dominated areas and areas converted to cocoyam in Anyiko wetland in Kenya, reported insignificant variation in CO₂ emissions between the sites. On the contrary, Were *et al.* (2020b), working in a tropical wetland in Uganda found that CO₂ emissions from areas of the wetland converted to rice paddy were substantially lower than from areas of the wetland that were still intact and this was attributed to differences in water levels during the dry season.

N₂O fluxes did not show significant differences among the sites. This may be due to different forms of nitrogen (NO₃-N and NH₄-N) which are some of the most important factors (substrates for nitrification and denitrification processes) controlling N₂O production did not vary significantly as well. Additionally, SOC content is a key factor in determining N₂O emissions as it provides electrons for heterotrophic denitrification (Marín-Muñiz *et al.*, 2015). In this study, SOC was significantly low in sugarcane areas compared to the other sites. Similar findings have been communicated in many studies. For instance, Baskerville *et al.* (2021) in their study along the riparian zones of the Canadian Washington Creek, reported non-significant differences of N₂O emissions. Also Were *et al.*, (2021b) found that average N₂O emissions from Naigombwa wetland in Uganda did not differ significantly among areas dominated *Cyperus papyrus*, *Typha latifolia* and *Phragmites mauritianus*, leading them to conclude that vegetation community type had no effect on N₂O emissions from natural

freshwater tropical wetlands under continuous flooding. In addition, insignificant difference in N₂O emission from swamp and marsh wetlands with varying vegetation community types in was reported by Marín-Muñiz *et al.* (2015) in Mexico. However, several researchers have come up with contrary results. For instance, Wang *et al.* (2008) noted significantly higher N₂O emissions from areas of the wetland dominated by *Zizania latifolia* compared to those dominated by *Typha latifolia* and *Phragmites australis*. Also, Liu *et al.*, (2014) found that the N₂O emissions from two coastal wetland zones, dominated by *Spartina spp.* and the other dominated by *Phragmites spp.* varied significantly, whereby, the one dominated with *Phragmites spp.* showed elevated N₂O flux. Liu *et al.*, (2014) attributed their observation to variations in biomass of the plant species, which influence the amount of organic matter and nitrogen enrichment in the soil. Meanwhile, Wang *et al.* (2008) attributed it to the variations in the root structure of the plant species, where the root structure of *Zizania latifolia* favoured growth of bacteria capable of oxidizing NH₄-N.

The nitrous oxide uptake was observed in *Cyperus alopecuroides* dominated areas as well as other sites as observed in range values in areas dominated by *Typha latifolia* (-4.70-11.76 $\mu\text{g m}^{-2} \text{h}^{-1}$), *Cyperus papyrus* (-3.35-15.20 $\mu\text{g m}^{-2} \text{h}^{-1}$) and sugarcane (-0.82-12.15 $\mu\text{g m}^{-2} \text{h}^{-1}$). The uptake can be attributed to specific soil conditions that favour the reduction of N₂O to nitrogen gas (N₂) during the denitrification process. *Cyperus alopecuroides* and other wetland vegetation are often found in waterlogged or poorly drained soils, which create anaerobic conditions. Anaerobic environments are essential for denitrification, a microbial process that reduces nitrate (NO₃) and nitrite (NO₂) to N₂O and ultimately to N₂. In such conditions, microbes capable of reducing N₂O to N₂ become active, leading to N₂O uptake (Butterbach-Bahl, *et al.*, 2013).

N₂O emissions are anticipated to rise in the wetland areas transformed into agriculture. The transformation of a wetland into cropland enhances the mineralisation of soil organic nitrogen. Furthermore, farming lowers the water table, enhancing oxygen exposure; hence, it is anticipated that N₂O emissions would be elevated in the sugarcane plantation relative to the natural wetland (Liengaard *et al.*, 2013; Ondiek *et al.*, 2021). According to studies by Owino *et al.* (2020), converting areas of a wetland dominated by *Cyperus papyrus* to paddy rice fields that are fertilized substantially increased N₂O emissions (4.37+3.18 $\mu\text{g m}^{-2} \text{h}^{-1}$) compared to paddy rice fields that are not fertilized (-3.59+2.56 $\mu\text{g m}^{-2} \text{h}^{-1}$). Zhang *et al.* (2017) also documented a positive link between fertiliser application and N₂O emissions from rice paddies, regardless of the rice development stage. In the current study, the farmer did not report any use of fertiliser in his sugarcane farm which partially explains why the N₂O were low. According

to Were *et al.* (2023), converting natural tropical freshwater marsh into unfertilized rice paddies have no impact on N₂O emissions particularly in cases where the amount of SOC and nitrogen are not substantially different between the sites.

5.4 Influence of soil physico-chemical characteristics on SOC and GHGs

5.4.1 Influence of soil physico-chemical characteristics on organic carbon

The only physico-chemical parameters that had a significant correlation with SOC were TN, TP, C: N, water temperature and soil temperature. TN had a significant positive correlation with SOC in all the three wetland plants dominated areas and cropland. This can be explained by the fact that both TN and SOC are derived from organic matter, such as plant residues, animal waste, and microorganisms and as organic matter decomposes, nitrogen and carbon are released thus the correlation. Microorganisms that fix nitrogen often rely on organic carbon as an energy source, linking TN and SOC. Denitrifying microorganisms consume organic carbon and reduce nitrogen oxides, further connecting TN and SOC (Bernal & Mitsch, 2008). Soils with high organic matter content tend to have higher cation exchange capacity, which can influence TN and SOC dynamics. The correlation between SOC, water and soil temperature observed in *Cyperus papyrus* dominated areas can be explained in term of higher soil temperatures enhancing microbial activity and organic matter decomposition, which increases SOC mineralization rates as well as stimulating plant productivity and organic matter inputs (Kayranli *et al.*, 2010).

There was a statistically significant correlation between C: N and SOC, however, this was only noted with the *Cyperus papyrus* dominated areas. Similar findings have been documented by Wang *et al.* (2018) where they refer to an earlier idea proposed by Christensen *et al.* (1999) the influence of C: N on SOC may be negligible in wetland soils with high amount of organic carbon.

5.4.2 Influence of soil physico-chemical characteristics on GHGs

Soil temperature, water temperature, water level, NH₄-N and TP are the only parameters that exhibited significant correlation with GHGs. In sugarcane and *Cyperus papyrus* study sites, CO₂ and CH₄ exhibited a significant positive correlation with each other. This is because CO₂ and CH₄ fluxes are interconnected through the carbon cycling processes where the net ecosystem exchange of CO₂ serves as a proxy for overall carbon dynamics, influencing CH₄ production (Reddy *et al.*, 2010). In this study it implies that some of the CH₄ produced in the anaerobic zones of the soil in both sites through methanogenesis were oxidized to CO₂ in the oxygenated zones through methanotrophy, leading to the positive relationship observed.

CH₄ exhibited significant negative correlations with soil temperature and TP in *Typha latifolia* dominated areas, these findings suggest that higher soil temperatures and phosphorus levels may suppress CH₄ emissions potentially due to shifts in microbial community composition or reduced methanogenesis under such conditions (van Lent *et al.*, 2020). CO₂ showed significant positive correlations with water temperature, soil temperature, water level and TP in *Cyperus alopecuroides* dominated areas. These correlations suggest that CO₂ emissions are greatly influenced by temperature, availability of water as well as nutrient levels, which are critical factors for microbial respiration and organic matter decomposition (Bond-Lamberty & Bond-Lamberty., 2009). Also, CO₂ had a significant positive correlation with NO₃-N in *Typha latifolia* dominated areas. This is because as the plant grows, it also takes up nutrients like NO₃-N from the soil or water. The positive correlation suggests that as CO₂ levels increase, the demand for NO₃-N might also rise, leading to higher uptake and possibly higher concentrations in certain areas due to plant activity or associated microbial processes (Reddy *et al.*, 2008).

N₂O exhibited significant positive correlations with water temperature and TP in *Typha latifolia* dominated areas. These results suggest that warmer water temperatures and higher phosphorus availability may enhance N₂O emissions, possibly by stimulating microbial activity or nutrient cycling. N₂O had a significant negative correlation with soil temperature and water level in areas dominated by *Cyperus papyrus* and sugarcane. These findings indicate that higher soil temperatures and water levels may suppress N₂O emissions in these ecosystems, potentially due to reduced nitrification or denitrification processes under such conditions.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

- i. The conversion of Yala wetland areas dominated by *Cyperus papyrus* (3.09%), *Cyperus alopecuroides* (2.98 %) and *Typha latifolia* (3.50%) to sugarcane cultivation (1.73%) significantly reduces SOC storage.
- ii. The conversion of Yala wetland areas dominated by *Cyperus papyrus*, *Cyperus alopecuroides* and *Typha latifolia* to sugarcane cultivation areas did not significantly impact GHG emissions expect for CH₄ in *Cyperus alopecuroides*
- iii. SOC storage in all the sites had significant correlations with different physico-chemical parameters, namely TN, TP, soil temperature, water temperature, C: N, and soil moisture. However, TN content in the soil positively influenced SOC storage in all the sites. GHGs as well had significant correlations with the different physico- chemical parameters. For instance, TP ($r = -0.659$, $p = 0.008$), soil temperature ($r = 0.483$, $p = 0.05$), water level ($r = 0.595$, $p = 0.012$).

6.2 Recommendations

- i. Reduced conversion of wetland areas to farmland and where farming is taking place, adoption of sustainable agricultural management practices like no-till or minimal tillage and addition of organic amendments (e.g., compost or manure) to increase SOC levels over time.
- ii. Regulation of activities within the wetland through wetland management committees as well as zoning to show which activity should be done in which section and in which season to reduce on CH₄ emissions associated with these activities.
- iii. Expand the study to include seasonal variations and long-term flux measurements for further understanding on how soil physico-chemical characteristics influence SOC storage and GHG emissions in Yala wetland and areas converted to cropland.

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APPENDICES

Appendix A: Photos showing soil sampling and GHGs nutrient analysis

i) Soil sampling, ii) GC Machine iii) spectrophotometer nutrient analysis, iii) GC Machine in operation



i



ii



iii



iv

Appendix B: Ethical Clearance

EGERTON

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**EGERTON UNIVERSITY INSTITUTIONAL SCIENTIFIC AND ETHICS REVIEW
COMMITTEE**

EU/RE/DIR/009

Approval No. EUISERC/APP/464/2025

8th July 2025

Mukiibi Jackson,
P.O.Box 536-20115,
Egerton- Njoro, Kenya
Telephone +251918636545 /+254716753980
Email: mukiibjack1991@yahoo.com

Dear Jackson,

**RE: EFFECT OF CONVERSION OF WETLAND DOMINATED BY DIFFERENT PLANT
COMMUNITIES TO CROPLAND ON SOIL ORGANIC CARBON AND GREENHOUSE
GAS EMISSIONS IN YALA WETLAND, KENYA**

This is to inform you that the *Egerton University Institutional Scientific and Ethics Review Committee* has reviewed and approved your above research proposal. Your application approval number is *EUISERC/APP/464/2025*. The approval period is *8th July 2025 – 9th July 2026*

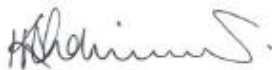
This approval is subject to compliance with the following requirements;

- i. Only approved documents including (informed consents, study instruments, MTA) will be used.
- ii. All changes including (amendments, deviations, and violations) are submitted for review and approval by *Egerton University Institutional Scientific and Ethics Review Committee*.
- iii. Death and life-threatening problems and serious adverse events or unexpected adverse events whether related or unrelated to the study must be reported to *Egerton University Institutional Scientific and Ethics Review Committee* within 72 hours of notification
- iv. Any changes, anticipated or otherwise that may increase the risks or affect safety or welfare of study participants and others or affect the integrity of the research must be reported to *Egerton University Institutional Scientific and Ethics Review Committee* within 72 hours.

- v. Clearance for Material Transfer of biological specimens must be obtained from relevant institutions.
- vi. Submission of a request for renewal of approval at least 60 days prior to the expiry of the approval period. Attach a comprehensive progress report to support the renewal.
- vii. Submission of an executive summary report within 90 days upon completion of the study to *Egerton University Institutional Scientific and Ethics Review Committee*.

Prior to commencing your study, you will be expected to obtain a research license from National Commission for Science, Technology and Innovation (NACOSTI) <https://oris.nacosti.go.ke> and also obtain other clearances needed.






Yours sincerely,



Prof. Kennedy N. Ondimu PhD
CHAIRMAN, EUISERC
KO/BK/



Appendix C: Research permit

 <p>REPUBLIC OF KENYA</p>	 <p>NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY & INNOVATION</p>
<p>Ref No: 799314</p>	<p>Date of Issue: 15/July/2025</p>
<p>RESEARCH LICENSE</p>	
	
<p>This is to Certify that Mr.. Jackson Mukibi of Egerton University, has been licensed to conduct research as per the provision of the Science, Technology and Innovation Act, 2013 (Rev.2014) in Siaya on the topic: EFFECT OF CONVERSION OF WETLAND DOMINATED BY DIFFERENT PLANT COMMUNITIES TO CROPLAND ON SOIL ORGANIC CARBON AND GREENHOUSE GAS EMISSIONS IN YALA WETLAND, KENYA for the period ending ; 15/July/2026.</p>	
<p>License No: NACOSTI/P/25/4176689</p>	
<p>Applicant Identification Number 799314</p>	
<p>Ag. Director General NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY & INNOVATION</p>	
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