UNIVERSITY OF CAPE COAST

ENHANCING SOIL HEALTH, SEQUESTERING CARBON AND REDUCING GREENHOUSE GAS EMISSIONS IN TROPICAL SOILS USING EMPTY OIL PALM FRUIT BUNCH BIOCHAR AND COMPOST

DORCAS BLANKSON

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BY

DORCAS BLANKSON

Thesis submitted to the Department of Soil Science, School of Agriculture, College of Agriculture and Natural Sciences, University of Cape Coast, in partial fulfilment of the requirements for the award of Doctor of Philosophy degree in Soil Science

MARCH, 2025

DECLARATION

Candidate's Declaration

I hereby declare that this thesis is the result of my original research and that no part of it has been presented for another degree at this University or elsewhere.

Candidate's Signature…………………………….. Date 26/03/2025

Name: DORCAS BLANKSON

Supervisors' Declaration

We hereby declare that the preparation and presentation of the thesis were supervised following the guidelines on supervision of theses laid down by the University of Cape Coast.

Principal Supervisor's Signature…………….Date 26/03/2025

Name: PROF. KWAME AGYEI FRIMPONG

Co-Supervisor's Signature…………………………….. Date 26/03/2025

Name: DR. KOFI ATIAH

Co- Supervisor's Signature…………………………….. Date……………………

Name: DR. EMMANUEL ARTHUR

ABSTRACT

The efficacy of the easily accessible empty oil palm fruit bunch (EFB) biochar and compost in improving soil quality, carbon sequestration, crop production and mitigating greenhouse gas emissions, following their application to soil is yet to be adequately investigated in Ghana. On this basis, the study was conducted to analyse the effect of pyrolysis and composting of EFB on the chemical properties and content of potential toxicant element of the derived biochars and composts, the EFB biochar loading capacity and potential phytotoxicity on four different tropical soils through incubation studies and pot experiments, and the effect of different application rates of sole EFB biochar, sole compost and biochar-compost combinations on soil properties, okra yield and nutrient use efficiency, carbon storage and greenhouse gas emssions. In the field experiments, there was a one-time application of EFB biochar (10 and 20 t ha^{-1}) [B10, B20], compost (20 t ha^{-1}) [CP20], biochar-compost combinations [B10CP20, B20CP20], an unamended control [B0], and mineral fertiliser (100 kg N ha^{-1}, 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}) [NPK] across two cropping cycles in a randomised complete block design with four replications. Notably, pyrolysis increased nutrient content in EFB biochar but also elevated polycyclic aromatic hydrocarbon (PAH) levels, surpassing safety thresholds. However, EFB composts showed low C/N ratios, high nutrient content, and negligible PAH and heavy metal levels. Furthermore, based on integrating soil chemical properties, phytotoxicity and nutrient uptake parameters,(amelioration score), the biochar application rate of 2.0% emerged as the ideal rate for Acrisol, Red Ferralsol and Vertisol soil types, whilst 1% was best suited for Brown Ferralsol. These rates enhanced the soils' chemical properties and increased nutrient uptake without exhibiting any inhibitory effect on maize germination. In the field studies, EFB amendments boosted okra pod yield by up to 283% over the unamended control. CP20 and B20CP20 increased yields by 58% and 100%, respectively, compared to mineral fertiliser in the first cycle. However, only B20CP20 maintained higher yields in the second cycle. All EFB treatments increased the soil's pH by 0.6 to 1.6 points, enhanced phosphorus uptake and recovery efficiency, while B20CP20 notably increased soil cation exchange capacity and available phosphorus and zinc content, relative to the unamended control. CP20 also increased arbuscular mycorrhizal fungi and gram-positive bacterial biomass than NPK and B20. Additionally, soil carbon management index was increased by the B10CP20 treatment, whiles B20 increased total organic carbon content and stock. Also, the combined treatments improved gas diffusion and convection by 32–89% and 28–30%, respectively, indicating enhanced soil pore organization. Despite the improved soil properties, the single compost and combined applications of biochar and compost led to significantly lower emissions of nitrous oxide (N_2O) (~74% decrease) and carbon dioxide (CO_2) (~50% decrease) compared to the unamended treatment. However, methane (CH_4) emissions were notably higher (87–685%) for all biochar and compost treatments compared to the unamended treatment. The increased soil pH, AMF biomass, air permeability and volumetric water content emerged as the major regulators of the decreased emissions of N_2O and CO_2 in the amended soils. In conclusion, the study demonstrated the potential of EFB biochar and compost in improving soil health, enhancing crop yields, and mitigating GHG emissions in tropical soils, offering promising solutions for sustainable agriculture in tropical regions. However, further research is needed to address the increase in methane emissions and to assess the long-term impacts on crop yields and soil health.

KEYWORDS

Arbuscular mycorrhizal fungi

Carbon management index

Gas flow by convection and diffusion

Greenhouse gas emissions

Nitrogen and phosphorus use efficiency

Okra yield

Soil fertility

Soil pore characteristics

Soil microbial biomass

Soil water retention

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DEDICATION

To Mr Albert Bruce Acquah, and my late father, Mr Joseph W. K. Blankson

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LIST OF ABBREVIATIONS AND ACRONYMS

Al	Aluminium
ANOVA	Analysis of Variance
ARE	Apparent recovery efficiency
C	Carbon
C/N	Carbon nitrogen ratio
Ca	Calcium
CBD	Convention on Biological Diversity
Cd	Cadmium
CEC	Cation exchange capacity
CH_4	Methane
Cl	Chlorine
CMI	Carbon management index
CO_2	Carbon dioxide
C_{org}	Organic carbon
Cr	Chromium
DAI	Days after incubation
d_B	Effective pore diameter
DL	Detection limit
dm	Dry matter
Dp/Do	Relative gas diffusivity
EBC	European Biochar Certificate
EC	Electrical conductivity
ECD	Electron capture detector
EDTA	Disodium ethylene-diamine-tetra acetic acid

EFB	Empty fruit bunch
ε	Total porosity
ε_a	Air-filled porosity
FAO	Food and Agriculture Organization
FC	Soil water content at field capacity
Fe	Iron
GC	Gas chromatograph
GDP	Gross Dometic Product
GHG	Greenhouse gas
GI	Germination index
H	Hydrogen
H_2O	Water
H_2S	Hydrogen sulphide
H_2SO_4	Sulphuric acid
hPa	Matric potential
IBI	International Biochar Initiative
IPCC	Intergovernmental Panel on Climate Change
K	Potassium
k_a	Air permeability
$KMnO_4$	Potassium permanganate
LI	Lability index
N	Nitrogen
n	Sample size
N_2	Dinitrogen (nitrogen gas)
N_2O	Nitrous oxide

Na	Sodium
NaOH	Sodium hydroxide
n_B	Number of air-filled pores in a soil cross-section
NH_3	Ammonia
NH_4^+	Ammonium
NLFA	Neutral lipid fatty acid
NO_2^-	Nitrite
NO_3^-	Nitrate
NUE	Nitrogen Use Efficiency
OH	Hydroxide
P	Phosphorus
P_2O_5	Phosphorus pentoxide
PAHs	Polycyclic aromatic hydrocarbons
PAW	Plant available water
Pb	Lead
PCA	Principal component analysis
PLFA	Phospholipid fatty acid
PO	Pore organisation
POXC	Permanganate oxidizable carbon
PWP	Permanent wilting point
RCBD	Randomized complete block design
RRE	Relative root elongation
S	Sulphur
SEM	Standard error means
SG	Relative seed germination

Si	Silicon
SOC	Soil organic carbon
SOM	Soil organic matter
SSA	Specific surface area
TN	Total nitrogen
TOC	Total organic carbon
UNCCD	United Nations Convention to Combat Desertification
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
VOC	Volatile organic carbon
WCFA	Whole cell fatty acid
WFPS	Water filled pore space
WHC	Water holding capacity
Zn	Zinc
θ	Volumetric soil water content
ρb	Bulk density
τ	Soil pore tortuosity

CHAPTER ONE

INTRODUCTION

1.1 Background to the Study

The importance of soil health cannot be overstated, as soil performs crucial ecosystem functions such as nutrient cycling, sustainable productivity, maintenance of biodiversity, and regulation of water dynamics, climate moderation, and buffering environmental changes (Nunes *et al*., 2021). According to Guo, (2021), soil's production function is indicated by high levels of crop yields and incomes, while its climate-regulative function is indicated by high levels of carbon stocks and slow rates of greenhouse gas emissions. These functions are achieved through various mechanisms and processes (Laishram *et al*., 2012). The capability of the soil to provide these ecosystem services is determined by its efficiency in performing intrinsic physical, chemical, and biological processes under specific geographic and climatic conditions, which is termed "soil health" (Guo, 2021). Furthermore, soil management significantly affects how well these functions are performed (Wienhold *et al*., 2006), and any land use or management practice that negatively alters soil properties subsequently impacts the soil's health (Guo, 2021).

Soil degradation poses a great challenge that threatens global food security (Zingore *et al*., 2015) particularly in sub-Saharan Africa where about 65% of the land area is classified as degraded (Vlek *et al*., 2008). Soil degradation leads to the disruption in the provision of essential ecosystem services, occurring due to the loss of intrinsic physical, chemical, and/or biological soil qualities, either through natural or anthropogenic processes

(Guo, 2021). Nunes *et al*. (2020) indicated that in sub-Saharan Africa, degradation of agricultural soils is often evidenced by organic matter decline, accelerated erosion, compaction, nutrient depletion, acidification, salinization, contamination, and loss of biodiversity. Additionally, earlier research by Zingore *et al*. (2015) showed that the region is characterised by some of the oldest, fragile, and inherently infertile soils with low nutrient contents and soil organic matter. The climate of Africa, South of the Sahel is characterised by high temperatures and high rainfall, further accelerates the degradation of these fragile soils (Adekiya *et al*., 2023). Consequently, severe soil degradation resulting from a combination of chemical, physical, and biological constraints is prevalent in this region. Under such conditions, effective, efficient, and multi-purpose management practices are essential to restore degraded agricultural soils to a "healthy" status capable of supporting crop production and delivering other essential ecosystem services, as suggested by earlier researchers such as Kibblewhite *et al*. (2008) and Zingore *et al*. (2015).

1.2 Statement of Problem

It is imperative to acknowledge the significance of scientific advancements and practical applications in the management of soil degradation (Guo *et al*., 2021). Research efforts have rightfully targeted the restoration of degraded tropical soils through various methods such as organic amendments, agroforestry, conservation agriculture, liming, reforestation, and crop rotation (Adekiya *et al*., 2023). These strategies are founded on the principles of minimizing soil disturbance, maximizing soil coverage, increasing crop diversity, and applying organic amendments as enunciated by Kibblewhite *et al*. (2008). However, it has been recognized by Zingore *et al*.

(2015) that addressing the issue of degraded soils in sub-Saharan Africa presents a unique challenge, particularly for resource-poor farmers with limited land for rehabilitation. Consequently, the use of organic amendments has been proposed as a feasible and sustainable approach to mitigate this problem. Despite this, accessing organic biomass or feedstock remains a challenge for small-holder resource-poor farmers (Adekiya *et al*., 2023).

Nevertheless, the waste generated by the oil palm industries in certain countries can serve as a potential organic resource to improve the degraded soils. According to Ghazali *et al*. (2018), the empty fruit bunch (EFB), which is the residue that remains after oil palm (*Elaeis guineensis*) fruits are removed from the fresh bunches, constitutes 25% of the oil palm biomass. Adu *et al*. (2022) reported that about 390 tonnes of EFB are generated daily from oil palm plantations in Ghana. This amounts to the annual production of approximately 397,000 tonnes of EFB by the oil palm industries, offering a significant opportunity for its utilization as organic amendments such as biochar, compost, or mulch to rehabilitate the soils of resource-poor farmers (Adu *et al*., 2022; Schuchard *et al*., 2008; Teh, 2016).

Composting can effectively reduce the volume of EFB, concentrate nutrients, and lower the C/N ratio, thereby enhancing the rate of decomposition and nutrient release (Kuo *et al*., 2004). Additionally, biochar, derived from the pyrolysis of biomass, offers valuable benefits such as carbon sequestration, soil improvement, resource use efficiency, and greenhouse gas mitigation (International Biochar Initiative, IBI, 2015). However, there is limited information available on the impact of EFB pyrolysis with different reactors and composting at different mixing ratios on the quality of EFB biochar or composts. Understanding how the transformation processes affect the levels of contaminants is crucial. Additionally, the potential of EFB biochar to enhance soil chemical properties or resolve infertility, phosphorus absorption, and acidity issues in different tropical soils has not been extensively studied to determine the maximum quantity suitable and safe for crop production. Furthermore, the potential benefits of EFB biochar, compost, or their co-applications on soil quality, crop yields, carbon storage, and greenhouse gas mitigation have not been extensively researched in Ghana.

Consequently, the current research aims to address these knowledge gaps by examining the effect of locally designed reactors and commercially established reactor produced biochars and mixing ratios of feedstocks on the chemical properties and potential toxic elements of EFB biochar and composts. Additionally, the potential phytotoxicity and short-term chemical response of different tropical soils to EFB biochar amendments were investigated. Furthermore, the research assessed the changes in soil chemical properties, carbon rehabilitation, crop yields, and nutrient use efficiency across two cropping cycles in response to the application of EFB amendments. Finally, the impact of single and co-applications of EFB biochar and compost on soil physical and microbial properties and greenhouse gas emissions were studied.

1.3 Research Objectives and Hypotheses

1.3.1 General objective

To examine the effect of empty oil palm fruit bunch (EFB) applied as biochar, compost and their combinations on soil quality, crop yield, carbon storage, and greenhouse gas emission.

1.3.2 Specific objectives

Specifically, the work sought to:

1. Examine the effect of pyrolysis and composting of EFB on the chemical composition and potential toxicant elements (PTE) of the resultant products.
2. Evaluate immediate chemical response and potential phytotoxicity of tropical soils to EFB biochar application.
3. Determine the extent to which changes in soil chemical and microbial properties affected by EFB amendments drive okra yield and nitrogen and phosphorus use efficiency.
4. Determine how carbon storage and management index regulate soil physical fertility after amending with EFB biochar and compost.
5. Examine the drivers of greenhouse gas emissions from EFB-amended soils.

1.3.3 Hypotheses

1. Pyrolysis and composting conditions of EFB significantly influence the chemical composition and concentrations of potential toxic elements in the derived biochar and compost.
2. The immediate chemical response and phytotoxicity of EFB biochar application differ across tropical soil types.
3. EFB biochar and compost enhance soil chemical and microbial properties which will drive okra yield and nitrogen and phosphorus use efficiency.
4. EFB biochar and compost application increase soil carbon storage and management index, which in turn improve soil physical fertility.

5. Soil amendment with EFB biochar and compost modifies soil physical and biochemical properties, leading to measurable changes in greenhouse gas emissions.

1.4 Justification of the Study

In tropical regions like Ghana, soils are often old, inherently infertile, and characterized by low soil organic matter and nutrient contents (Zingore *et al*., 2015). Acrisols and Ferralsols, which are dominant soil types in Ghana, are highly weathered and exhibit low native fertility, high acidity, and aluminium toxicity (Owusu *et al*., 2024). The decline in soil fertility is further exacerbated by nutrient mining, inappropriate land use management, and limited use of external inputs (Bello *et al*., 2021). To address the issue of soil infertility, a multipurpose management approach that enhances soil organic matter, optimizes pH, improves water and nutrient retention, and promotes biodiversity is necessary (Krull *et al*., 2004).

Oil palm is the second most important tree crop in Ghana, after cocoa, with an average fresh fruit bunch yield of 5.4 t ha^{-1}. About 350,000 ha of land are under oil palm cultivation in Ghana (Nature Economy and People Connected, 2017). The palm oil industry generates substantial waste from the mills. The waste, particularly empty fruit bunches (EFB), poses challenges for disposal and utilization due to its size and potential for pest outbreaks (Hambali *et al*., 2010). However, EFB contains essential nutrients and could be used as organic fertiliser to address soil infertility challenges in Ghana (Cheah *et al*., 2023; Rosli *et al*., 2023).

Previous studies have suggested that EFB could serve as a cost-effective and readily available organic fertiliser for restoring soil degradation

(Adu *et al*., 2022; Lim *et al*., 2015). Composting EFB can reduce its bulkiness, concentrate nutrients, and lower the carbon-to-nitrogen (C/N) ratio, thereby increasing the rate of decomposition and nutrient release in the field (Kuo *et al*., 2004). Additionally, biochar obtained from the thermochemical conversion of biomass can be used to improve soil quality, resource use efficiency, and mitigation of environmental pollution (International Biochar Initiative, IBI, 2015).

There is dearth of information regarding the impact of pyrolysis and co-composting methods on the properties of biochar and compost produced from EFB in Ghana. Understanding how the transformation processes affect the levels of contaminants is crucial. Again, the use of empty fruit bunch (EFB) biochar for small-scale crop production in Ghana is relatively new. Consequently, to optimise the use of EFB biochar in crop production, a better knowledge of the dynamics of its mineralisation and the release of mineral nutrients and toxic compounds on crop growth and yield depending on the soil type and the rate of application are critical. However, the sole use of either biochar or compost for improving soil fertility in tropical soils is not without constraints. Therefore, earlier researchers including Liu *et al*. (2012) and Abudjabha *et al*. (2016) have suggested their combined use, but there is dearth information regarding the optimum application rate and the duration of their effectiveness on soil carbon storage and crop yield. Furthermore, there is a lack of scientific evidence on the role of EFB biochar, compost and their combined applications in mitigating greenhouse gas emissions in tropical agro-ecosystems and its interaction with compost co-application. Consequently, further studies on the production and application of EFB biochar and compost and their impact on soil quality, crop yield, and carbon storage and management index and greenhouse gas mitigation are warranted.

1.5 Significance of the Study

Ghanaian agriculture is mainly composed of smallholder farmers who are vulnerable to crop failure caused by droughts, floods, higher incidences of pests and diseases and degraded farmlands (Ministry of Foreign Affairs of the Netherlands. 2018). In 2020, 6.8 million Ghanaians were estimated to be living on degraded agricultural land, which is projected to increase by 26% in a decade (Bakker *et al.,* 2021). Again, climate change has further exacerbated the on-going rapid land degradation through soil erosion, droughts and changes in biodiversity. Meanwhile, land degradation is also a contributor to greenhouse gas (GHG) emissions and therefore, a call for combatting land degradation can also contribute to climate adaptation and mitigation (Global Mechanism of the UNCCD, 2018). It is critical that sustainable practices are employed, focusing on efficient use of nutrients, protection of soil resources and minimizing impacts on climate change.

The research aims at the utilization of empty oil palm fruit bunch which otherwise poses a challenge for disposal by the oil palm industry to restore soil degradation in small-holder resource-poor farmer fields. Composting and pyrolysis of EFB are hardly practised in Ghana, thus scanty information exists on pyrolysis of EFB using two of the dominant reactors (continuous rotary reactor and locally designed kiln) in Ghana and composting methods on the elemental compositions, nutrient contents and potential toxicant (polycyclic aromatic hydrocarbons [PAHs] and heavy metals) elements in the resultant biochar or compost products. Therefore, the study will provide insights on the possibility of the introduction of contaminants such as heavy metals and PAHs into the soil, environment and the natural food system from the use of EFB biochar or compost.

Again, the use of EFB biochar for small-scale crop production is quite new in Ghana. Moreover, the potential of EFB biochar to improve soil chemical properties of different tropical soils is not widely researched to know its suitability for crop production in these soils. The findings will provide insights on the dynamics of EFB biochar mineralisation and the release of mineral nutrients and toxic compounds on seed germination, growth and yield of crops depending on the soil type and the rate of application. This knowledge is vital for scaling up empty fruit bunch biochar application as a soil conditioner in tropical soils and will provide insights for future research and management strategies aimed at utilizing oil palm empty fruit bunches to restore soil quality, rehabilitate organic carbon, and mitigate greenhouse gas emissions in tropical soils.

The study will also create awareness among the various EFB biochar and compost stakeholders such as farmers, oil palm plantations owners, agricultural extension officers, research scientists and agro-input dealers and build their capacities in EFB biochar and compost production and application. Additionally, the findings from this study will also serve as a stepping stone on which future research on thermo-conversion technologies for EFB biochar production will be built. Finally, this research will shed light on the factors driving increased crop yields, nutrient use efficiency, and greenhouse emissions from tropical soils amended with EFB biochar and compost.

1.6 Organization of Thesis

The thesis consists of five chapters. Chapter one provides a general introduction, including the study's background, problem statement, justification, objectives, hypotheses, and study significance. The second chapter offers a summary of pertinent literature supporting the research topic. Chapter three details the materials and methods of the entire research, and the chapter four presents and discusses the findings based of the specific objectives of the study. The first objective presents and discusses the effect of pyrolysis and composting on EFB quality and the second objective presents the results on the immediate response of different tropical soils to EFB biochar application. Objectives three through five present the results of the effects of single and co-application of EFB biochar and compost on soil chemical and microbial properties, okra yield and nutrient use efficiency, soil carbon rehabilitation and physical fertility, and soil gas transport, pore characteristics, and greenhouse gas emissions. Finally, the fifth chapter presents the summary, conclusions and recommendations from the study.

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

The potential of empty fruit bunch (EFB) biochar and compost in Ghana's smallholder agricultural systems has not been fully exploited, even though the disposal of empty fruit bunch waste poses a challenge to the oil palm industries in the country (Teh, 2016). Knowledge about the production, properties and utilization of EFB biochar and compost and their influence on crop production, carbon sequestration, greenhouse gas emissions mitigation and overall health of different tropical soils is crucial for up scaling their adoption in Ghana.

This chapter provides a comprehensive assessment of the current state of knowledge on tropical soil health, soil degradation and existing intervention strategies for restoration. With emphasis on the use of organic amendments to restore tropical soil health, this review focused on biochar and compost productions and the effect of their production conditions on the properties of the biochar and the compost, and how their application influences soil physical, chemical and biological properties as well as crop yield and nutrient use efficiency, carbon sequestration and greenhouse gas emissions mitigation, under tropical conditions. Also, existing literature on EFB as an available feedstock for biochar and compost production and their application on soil properties and crop yields were reviewed. Finally, key findings from the review have been interpreted to identify knowledge and research gaps in the use of EFB biochar and compost in crop production systems in Ghana.

2.2 What Constitutes a Healthy Soil

Soil health is a broad term encompassing the physical, chemical, and biological characteristics that influence soil function. A soil is considered healthy when it delivers ecosystem services at a level equal to or greater than those of undisturbed reference soils of the same type and region (Guo, 2021). Some researchers define soil health as the sustained ability of soil to function as a living system within ecosystem and land-use constraints. This includes supporting biological productivity, preserving air and water quality, and contributing to the well-being of plants, animals, and humans (Laishram *et al.*, 2012). Again, Andrews *et al*. (2004) refer to the ability of soil to function and provide ecosystem services based on its inherent characteristics (e.g. texture, mineralogy) and environmental conditions as soil health. Idowu *et al*. (2016) related the capacity of the soil to function and support important ecosystem services without having negative interaction with the environment as soil health. van Bruggen and Semenov (2000) also viewed soil health as a dimension of ecosystem health which is expressed based on the resistance and resilience of soil in response to various stresses and disturbances. Based on the above definitions, it can be synthesised that soil health is climate, geography and management bound, function-oriented under limitless time which is dependent on the intrinsic properties of the soil.

Soil performs multiple functions which can be categorized into five main ecological functions: (i) production function, (ii) biotic environmental function/living space function, (iii) climate-regulative function/storage function, (iv) hydrologic function and (v) waste and pollution control function Laishram *et al*. (2012). While soil performs numerous functions, five are recognized as fundamental to the Earth's system and essential for the sustainable development of human societies (Kopittke *et al*., 2024). Laishram *et al*. (2012) viewed the soil's production function as its capacity to supply water, nutrients and oxygen and to reduce crop yield losses due to pests and diseases. Soil as a biotic living space is exhibited by providing habitat for about 25% of global biodiversity (Bach & Wall, 2017), with more than 40% of living organisms in the terrestrial environment directly associated with soil, making soil the most biologically diverse habitat on Earth (Kopittke *et al*., 2024). Moreover, Laishram *et al*. (2012) related the climate-regulative function of the soil to high levels of carbon stocks and slow rates of greenhouse gas emissions. Again, the soil is the largest store of water in the terrestrial ecosystem, storing up to a maximum of 121,800 km^3 of water, averaging 17,000 km^3 at any given time showcasing water cycling as a central function of the soil (Webb *et al*., 1993). In the agricultural sphere, the concept of soil health is essentially an elaboration of the concept of soil productivity/fertility to deal with the multiple and complex problems faced worldwide. Guo (2021) stated that agricultural soils classified as healthy typically exhibit above-average crop yields, an adequate nutrient supply, balanced organic matter levels, proper tilth and drainage, a predominance of beneficial organisms over harmful ones, strong resistance to erosion and degradation, and an absence of contaminants. All in all the overall assessment about whether a soil is good or bad depends on the objective of the assessment and the net outcome of different soil processes and functions under any given conditions.

The soil functions can be weighted according to the relative importance of each function in fulfilling the management goals (Masto *et al*., 2007). Regulation of each function is determined by a large number of soil attributes (Laishram *et al*., 2012). Soil health is influenced by intricate biophysical and biochemical interactions that vary across both time and space, leading to the creation of a suitable environment that provides multiple ecosystem services (Norris *et al.*, 2020). Soil health will be dependent on soil properties and their relationship with specific soil functions, including complexity associated with the interactive effects of climate change (Tiwari, 2017). The interactions among inherent and dynamic soil biological, physical, and chemical properties and processes are complex and must be quantified when assessing management effects on soil health (Nunes *et al.*, 2021). The significance of a soil property in assessing soil health is typically determined by how specific soil functions respond to measurable changes in that property (Karlen *et al.*, 2019). Consequently, when the inherent physical, chemical, or biological attributes of soil are diminished or lost due to natural processes or human activities, leading to a decline or loss of key ecosystem functions, soil health deteriorates (Nunes *et al.*, 2020)

2.3 Soil Degradation in Sub-Saharan Africa

Globally, approximately 33% of soils are considered moderately to highly degraded, while 52% of agricultural land faces moderate to severe degradation (Zingore *et al.*, 2015). Sub-Saharan Africa contains some of the oldest and most inherently infertile soils, with many areas characterised by low nutrient content and soil organic matter due to intense weathering and excessive leaching (Zingore *et al*., 2015; Owusu *et al*., 2024). Again, about

65% of the land area in sub-Saharan Africa is classified as degraded (Vlek *et al*., 2008). Moreover, 5.4 million of agricultural lands in Ghana were classified as degraded as of 2010 (United Nations Convention to Combat Desertification [UNCCD], 2018). The degradation of agricultural soils is often indicated by a decline in organic matter, increased erosion, soil compaction, salinization, contamination, and reduced biodiversity (Nunes *et al*., 2020). However, in sub- Saharan Africa, soil acidity and aluminium toxicity, organic carbon depletion, nutrient depletion, soil erosion, and shallow soils are the major constraints in agricultural soils (Zingore *et al*., 2015). For instance, Buri *et al.* (2005) reported that most soils in Ghana have acidity problems and Acrisols and Ferralsols which are a few of the dominant soils in Ghana are characterized by low native fertility, aluminium toxicity, strong phosphorus sorption and acidity problems (Owusu *et al*., 2024).

Soil degradation may result from either natural, climatic, anthropogenic or the interaction of these factors. For instance, Guo (2021) indicated that Land use, disturbances, and management practices can modify soil properties, ultimately affecting soil health. In sub-Saharan Africa, soil degradation is primarily driven by water and wind erosion, as well as the deterioration of its physical, chemical, and biological attributes (Muchena *et al*., 2005). Hartemink and Barrow (2023) identified high rainfall, geology, clay mineralogy, soil texture and buffering capacity as the major factors that influence the development of soil acidity under tropical climates. The application of fertilisers to soil is now very essential for providing food for half of the world's population (Erisman *et al*., 2008), However, their over-application is causing acidification of soil, eutrophication of water bodies and the release of greenhouse gases (Snyder *et al*., 2009). In Ghana, important drivers of land degradation consist of land use conversions, industrial mining, overharvesting, trafficking of wild animal and plant species, illegal logging, urbanization, infrastructure development, invasive alien species and wildfires Bakkar *et al*. (2021). Rising temperatures and shifting rainfall patterns due to climate change can speed up the decomposition of organic matter, resulting in the loss of organic carbon in tropical soils (Zhao *et al*., 2021). Bakkar *et al*. (2021) iterate that climate change exacerbate the on-going rapid land degradation and desertification through soil erosion, droughts, changes in biodiversity and an increase in pests and diseases. Intensive cropping and organic matter mismanagement of agricultural soil also leads to soil degradation (Adekiya *et al*., 2023).

Soil health degradation is a global issue that poses a serious threat to food security (Guo, 2021). The intensive human exploitation of soil, along with the resulting decline in its multifunctionality, is significantly impacting planetary health (Kopittke *et al.,* 2024). In sub-Saharan Africa, soil degradation is a key factor contributing to low crop yields and widespread malnutrition, leading to an annual 3% decline in agricultural GDP (Sanchez, 2002). This has severe consequences for the majority of the population, who rely on agriculture for both food and income (FAO, 2010). For instance, food availability in Ghana is hampered by crop failure and reduced livestock productivity due to droughts, floods, higher incidences of pests and diseases and degraded farmlands (Bakkar *et al*., 2021). Also, yields of cereal crops in sub-Saharan Africa have stagnated at less than 1.5 t ha^{-1}, although the yield potential of most crop varieties exceeds 5 t ha^{-1} (FAO, 2010). Bakkar *et al*.

(2021) indicated that soil degradation has significant impacts on rural livelihoods in Ghana because it causes a reduction in food availability, soil fertility, carbon sequestration capacity, wood production, groundwater recharge and so on. Soil degradation not only reduces the capacity to produce food but is also expected to contribute to rising social and political instability (Kopittke *et al.*, 2024). For instance, as soil quality declines, competition for land intensifies, leading to reduced productivity and heightened conflicts (Bora *et al.*, 2010). Additionally, the deterioration of agricultural and forestry lands has been identified as a major source of greenhouse gas emissions in Ghana, accounting for 71% of the country's total emissions in 2012 (Global Mechanism of the UNCCD, 2018). Given the essential role of soil in sustaining both human and planetary health, addressing soil degradation is crucial to maintaining its multifunctionality for future generations. This underscores the need for adopting climate-friendly and cost-effective mitigation strategies.

2.4 Intervention Strategies for Restoring Tropical Soil Degradation

To restore and improve the health of degraded agricultural soils, effective soil management practices are essential. Degradation often results from improper land use, intensive crop cultivation, overgrazing, and excessive fertilization (Guo, 2021). The United States Department of Agriculture’s Natural Resources Conservation Service (USDA-NRCS, 2019) outlines five key principles for maintaining and enhancing soil health:

1. Soil cover: Keeping soil protected as much as possible using living plants, crop residue, compost or synthetic covers.

2. Reduce disturbance: Limiting mechanical, chemical and biological disruptions to preserve soil structure and function.
3. Crop diversity: Rotating and intercropping different plant species to manage pests and diseases while supporting soil microbial activity.
4. Enhanced soil biodiversity: Maintaining continuous live roots in the soil to promote a healthy soil ecosystem.
5. Livestock integration: Incorporating grazing animals for cover crop utilization, crop residue management and weed control.

The effectiveness of these soil health principles depends on several factors, such as soil characteristics, climate conditions, land management strategies, and plant-soil interactions (Adekiya *et al*., 2023). Research based on these five principles has explored multiple strategies for restoring degraded soils, including organic amendments, agroforestry, conservation agriculture, reforestation, cover cropping, and crop rotation, among other methods.

Conservation tillage is any planting systems in which at least 30% of the soil surface is covered by crop residues after planting to reduce erosion by water (Guo, 2021). Page *et al*. (2020) indicated that it is a sustainable agricultural practice that minimizes the disturbance of soil by reducing the intensity and the frequency of tillage operations. Strip-till, ridge-till, stubble mulch till, reduced till, and no-till are the different types of conservation tillage practices (Kumur *et al*., 2019). Tropical soils generally have lower organic carbon levels than those in temperate regions due to high temperatures, intense rainfall, and the rapid decomposition of organic matter (Adekiya *et al*., 2023). In such environments, tillage intensity plays a crucial role in soil health. Compared to conventional tillage, conservation tillage minimizes mechanical soil disturbance, helps maintain or increase organic matter, enhances soil structure, and reduces both erosion and greenhouse gas emissions (Fangueiro *et al.,* 2017). Additionally, conservation tillage lowers operational costs, decreases subsurface soil compaction, and improves both water infiltration and retention (Goldan *et al.,* 2023).

For instance, research by Pieper *et al*. (2015) found that vegetable plots under strip tillage exhibited greater soil aggregate stability, higher active organic carbon levels, increased nitrogen mineralization potential, and enhanced microbial activity. Similarly, Adekiya *et al.* (2019) observed that conservation tillage led to higher organic carbon content in tropical soils in Nigeria compared to conventional methods.

Despite strong evidence supporting the benefits of conservation tillage, challenges remain. Its implementation often requires higher investments in weed and pest control, carries an increased risk of nutrient runoff due to surface fertiliser application, and can limit root growth, particularly in heavy tropical soils, making adoption difficult in certain soil types and environmental conditions (Troeh *et al.*, 2004).

Furthermore, cover crops, crop rotation, and residue retention are widely recognized as sustainable agricultural practices that have been shown to have significant positive impacts on soil organic carbon. A key advantage of crop rotation is its ability to reduce pests and diseases in agricultural soils. Since different crops attract distinct pests and soil microorganisms, alternating crop species helps disrupt disease cycles and effectively manage soil-borne pests and pathogens (Berendsen *et al.*, 2012; USDA-NRCS, 2009).

Additionally, crop species vary in root structure, root exudates, nutrient demands, and their role in nutrient cycling. For example, research by Xiao *et al*. (2022) found that crop rotation enhanced soil physical properties, including soil structure, aggregation, and water infiltration. These improvements facilitated organic matter incorporation and protection, ultimately increasing soil organic carbon (SOC) levels. However, a drawback of crop rotation is the potential for new weed species to become problematic, as observed with Conyza species in rice-soybean rotations (Scherner *et al*., 2018). Again, economic factors can also limit rotation options, as certain crops may have advantages in specific regions due to environmental adaptation, input requirements, and market influences (Reeves, 1994). Additionally, adopting crop rotations may lead to reduced productivity for certain crops, such as soybean in rice-soybean rotations as observed by Scherner *et al*. (2018). Despite these drawbacks, crop rotation remains a valuable practice for improving sustainability and long-term profitability in agriculture, offering benefits such as reduced fertiliser and herbicide use, improved soil structure, and increased yield stability (Selim, 2019).

Cover cropping can also help to control soil erosion, maintain soil microbial pollution and diversity, and enhance soil health (Vukicevich *et al*., 2016). However, in regions with high rainfall, cover crops may decompose rapidly and their impact on SOC accumulation and soil health may be limited, as iterated by Rosolem, Li and Garcia (2016). Meanwhile difficulties in killing cover crops, potential pathogen hosting, and lack of immediate benefits can limit the inclusion of cover crops in a farming system (Sharma *et al*., 2018). There can also be a problem of increased direct costs and potentially reduced income if the cover crops interfere with other profitable crops (Snapp *et al*., 2005).

Intercropping, a traditional farming method, involves cultivating two or more crops simultaneously within the same field (Yang *et al*., 2020). This approach is particularly beneficial for improving soil fertility and overall soil health, especially in tropical regions. Yang *et al*. (2020) highlighted that intercropping enhances organic matter input by providing a continuous and diverse supply of crop residues and root exudates. Additionally, it promotes microbial diversity and activity, which are essential for organic matter decomposition and the formation of stable soil organic carbon pools. Furthermore, intercropping enhances nutrient cycling and improves nutrient use efficiency, contributing to better soil health (Kebede, 2021). For instance, Kebede (2021) found that integrating leguminous and non-leguminous crops in an intercropping system facilitates atmospheric nitrogen fixation. This process occurs through a symbiotic relationship between nitrogen-fixing bacteria and the root nodules of leguminous plants, enriching soil nitrogen levels.

The deliberate integration of trees with crops and/or livestock has gained recognition as a sustainable land-use practice that can provide multiple benefits, including improving soil health and increasing organic carbon content in tropical soils (Adekiya *et al*., 2023). Arshad *et al*. (2024) pointed the key mechanisms through which agroforestry systems improve the health of tropical soils. They indicated that increased biomass production and litter fall promote the formation of stable soil organic matter through the incorporation of recalcitrant carbon compounds into the soil matrix. Agroforestry also enhances biodiversity, improves soil health, and contributes to climate change mitigation through carbon sequestration (Salve *et al*., 2022). Moreover, agroforestry systems provide multiple socio-economic returns, including food, fuel, and additional income (Singh, 2021). Meanwhile, the implementation of agroforestry systems may face obstacles like land scarcity, lack of improved seeds/seedlings, and farmers' conservative attitudes (Orji *et al*., 2022). Additionally, there can be competition for resources between trees and crops (Arshad *et al*., 2024).

The challenge of restoring degraded soils in sub-Saharan Africa lies in the fact that it primarily impacts resource-poor farmers who have limited land available for rehabilitation. Many soil restoration techniques require land to be set aside or invested in practices that do not immediately contribute to food production, making adoption difficult for those who rely on every available portion of land for their livelihoods (Adekiya *et al*., 2023). In such areas, organic amendments promise to be more feasible intervention strategy to restore degraded soils and sustain crop production. Various organic materials, including manure, compost, crop residues, and biochar, can enhance soil physical, chemical and biological properties (Das *et al*., 2021). These amendments increase soil organic matter content and improve soil structure, water retention capacity, and nutrient availability (Matisic *et al*., 2024). They also promote beneficial microbial activity and carbon sequestration (Singh *et al*., 2024). However, the effectiveness of organic amendments varies depending on soil type, application rates, and cropping systems, necessitating tailored approaches (Matisic *et al*., 2024). While easily decomposable amendments may have immediate but transient effects, more stable organic materials can provide longer-lasting benefits (Das *et al*., 2021).

Organic amendments offer significant benefits for tropical soils in smallholder agriculture by increasing organic carbon content, pH, and nutrient availability, particularly in acidic soils (Doan *et al*., 2021). These amendments, including compost, biochar, and manure, have been shown to reduce phosphorus fixation by aluminium and iron, thereby enhancing phosphorus availability and uptake by crops (Rastogi *et al*., 2023). The effects of organic amendments are often site-specific, depending on local soil conditions and amendment types (Doan *et al*., 2021). However, Doan *et al*. (2021) suggested that while they generally improve soil chemical properties, though their impact on physical and biological parameters may be limited in the short term. Organic amendments also contribute to sustainable agriculture by promoting soil health, crop yield, and environmental conservation (Rastogi *et al*., 2023).

Long-term studies demonstrated sustained benefits of organic amendments on soil physical properties over time (Kaje *et al*., 2023). Application of amendments such as manures and compost was reported to increase porosity, water retention capacity, hydraulic conductivity, and structural stability while decreasing bulk density compared to unamended or conventionally managed soils in different soil types and climatic conditions (Benouadah *et al*., 2020; Çerçioğlu, 2017). For example, The Kaje *et al*. (2023) reported that the combination of different organic inputs, such as farmyard manure with green manures or compost with cattle slurry, resulted in the most significant improvements in soil physical properties that contributed to increased crop yields in some cases (Çerçioğlu, 2017).

Moreover, organic amendments also improve soil health by boosting microbial biomass, diversity, and enzymatic activity, potentially leading to disease suppression (Bonilla *et al*., 2012). Organic amendments contribute to climate change mitigation by enhancing soil carbon sequestration, helping to store carbon in the soil and reduce atmospheric greenhouse gas levels (Aytenew & Bore, 2020). While organic amendments offer numerous benefits, they may also introduce pollutants such as heavy metals, pathogens, and micro plastics, posing risks to environmental and human health (Urra *et al*., 2019). Other challenges such as quality variation and scalability need to be addressed for wider adoption in smallholder tropical agriculture (Rastogi *et al*., 2023). To maximize benefits and minimize risks, it is crucial to develop efficient strategies for their application in agriculture (Urra *et al*., 2019).

Lastly the requirement of a large number of organic resources for their production can pose a challenge for its use by small holder farmers, especially in Ghana. Sub-Saharan Africa has a diverse range of biomass resources, including agricultural crop residues, by-products, forestry residues, wood waste, the organic fraction of municipal solid waste (MSW), industrial wastewater, and animal manure (Duku *et al.,* 2011). However, reliable data on how these resources are allocated and utilized remain scarce. In reality, not all agricultural residues can be collected or repurposed for soil amendments due to technical limitations, their role in maintaining ecosystem functions, and competing uses (Duku *et al*., 2011). The suitability of particular biomass as a potential feedstock for organic amendment production depends upon various characteristics such as nutritional composition, moisture content, elemental composition, bulkiness, and ease of processing.

2.5 Compost as an Amendment for Restoring Degraded Soils

Compost is a carbon-rich, stabilized, and sanitized material formed through the aerobic biodegradation of organic matter by microorganisms (Insam & de Bertoldi, 2007). This process results in a humus-like, stable substance that is free of pathogens and weed seeds, making it suitable for use as an organic fertiliser or soil amendment (Fischer & Glaser, 2012). Kuo *et al*. (2004) emphasize that stabilizing organic waste before land application is crucial for reducing odour; ensuring nutrients are readily available for plants, and preventing phytotoxic effects on plant growth. Recycling waste through composting is an environmentally favourable substitute which can be used as a source of plant nutrients (Almendro-Candel *et al*., 2019).

Composting enhances nutrient concentration and reduces the carbon-to-nitrogen (C/N) ratio, which accelerates biomass decomposition and promotes faster nutrient release in the soil (Kuo *et al*., 2004). Composting is known to increase the ash, carbonate, and nitrogen and reduce the total carbon and C/N ratios of feedstocks (Teh, 2016). Earlier composting studies by Eghbal *et al*. (1997) show that during composting, the loss of carbon ranged from 46 to 62% and that of nitrogen is between 19 and 42% depending on type of composting system and composting conditions. Also, Ermadani and Arsyad (2013) found a higher content of nitrogen, phosphorus, potassium, calcium and magnesium in effluent compost compared to the fresh effluent. Tibu *et al*. (2019) noted that the nutrient concentration in the final compost results from the shrinking organic matter content during composting.

Composting progresses through three primary temperature-based stages; (1) mesophilic, (2) thermophilic, and (3) cooling and maturation phase

(Supriatna *et al.*, 2022). The duration of each stage is influenced by factors such as the initial composition of the material, moisture levels, aeration, and the diversity and abundance of microbial communities (Neklyudov *et al.*, 2006).

During the mesophilic stage, temperature and moisture levels increase as psychrophilic and mesophilic microorganisms become active, accelerating the breakdown of organic compounds. This phase sees temperatures exceeding 45°C, reaching a peak as microbial activity intensifies (Ayed *et al*., 2007). The thermophilic stage follows, with temperature rising between 55°C and 65°C, marking a critical sanitation phase where microbial activity is at its highest. Temperature increases in both the mesophilic and thermophilic stages occur when readily degradable carbon sources such as sugars, carbohydrates, hemicellulose, and cellulose are abundant, and the compost pile is well-aerated and insulated. This promotes vigorous microbial activity and rapid organic carbon decomposition.

The thermophilic phase is particularly essential for pathogen elimination and the destruction of fly larvae and weed seeds, provided high temperatures are maintained for at least three days (Kuo *et al*., 2004). The final phase, namely, ripening, is characterised by decrease in temperature up to 30°C and this is caused by the depletion of organic compounds in the compost, thereby making the C/N ratio potentially stable (Supriatna *et al*., 2022). The last stage of composting is curing which is a slow mesophilic process and may take several weeks to months depending the feedstock, method of composting, and the extent of the degradation of degradable organic C during the thermophilic decomposition phase. The curing phase is essential, allowing for the further breakdown of organic acids, humification, and the conversion of ammonium nitrogen into nitrate nitrogen through nitrification. Temperature regulation in the composting pile can be managed using various strategies, including adjusting the heap's size and shape, turning and watering the pile, or utilizing ventilation techniques (de Bertoldi *et al.*, 1982).

Various composting methods exist, but selecting the most suitable approach depends on multiple factors, including location, scale of operation, potential odour issues, available capital, feedstock characteristics, environmental regulations, and labour availability (Supriatna *et al.*, 2022). Additionally, Kuo *et al.* (2004) highlight that the choice and design of composting techniques varying in shape, size, complexity, and bulking agents are influenced by concerns related to odour control, phytotoxicity, nutrient availability, weed seed suppression, time and space constraints, cost, metal content in waste, and the potential impact on surface water quality. There are several criteria for categorising composting methods and based on the mode of aeration, it can either be passive or forced aeration, it can be open, covered, contained, or in-vessel based on the level of containment. The composting method can be termed as a static or turned pile depending on the degree of agitation and based on material movement the method can be batch or continuous composting (Kuo *et al.*, 2004; Supriatna *et al.*, 2022).

Several common and straightforward composting methods, such as passively aerated static piles and turned windrows, require little site modification, engineering, or financial investment. Another widely used and versatile approach is aerated static pile composting. In contrast, some composting facilities have adopted advanced technologies and automated systems to accelerate the composting process, optimize space usage, provide protection from weather conditions, and minimize odours (Bidlingmaier *et al*., 2000; Supriatna *et al*., 2022).

The quality of the final compost is influenced by factors such as feedstock selection, bulking agents, composting method, initial carbon-to-nitrogen (C/N) ratio, and the overall composting process. A wide range of organic waste materials can serve as feedstock, including sewage sludge, animal manure, yard waste, crop residues, municipal solid waste, fish scraps, food waste, and by-products from food processing. These materials vary significantly in their C/N ratios. According to USDA (2000), an optimal initial C/N ratio for composting is approximately 30:1. As a result, co-composting can be a viable option when materials are readily available, accessible, and cost-effective. High C/N ratio materials, such as crop residues, sawdust, wood chips, industrial solid waste, and municipal solid waste, can be effectively combined with nutrient-rich sources like animal manure or biosolids to balance the mixture for efficient composting (Liao *et al*., 1994).

According to Lim *et al*. (2015), the nutrient quality of compost is also affected by the mixing ratios of the initial feedstocks. Lim *et al*. (2015) reported that EFB compost with one-part EFB and three-part cow dung yielded higher total calcium, phosphorus, and magnesium levels compared to composts with less cow dung. Kamolmanit and Reungsang (2006) also observed higher percentages of total K and P in compost piles with more swine manure. Adding chicken manure in composting produces better compost with a higher N content and also resulted in degradation process faster than compost without chicken manure (Ermadani & Arsyad, 2013).

Compost can sometimes contain contaminants that negatively impact soil organisms when applied to the soil. The presence of carcinogenic substances poses potential health risks (European Biochar Certification [EBC], 2022). According to the International Biochar Initiative (IBI) (2015), toxicants in compost fall into two main categories: those originating from the feedstocks, such as heavy metals and polychlorinated biphenyls, and those formed during the biodegradation process, including polycyclic aromatic hydrocarbons and dioxins/furans. Additionally, compost may introduce phytotoxic substances like heavy metals, phenolic compounds, ethylene, ammonia, excessive salts, and organic acids, all of which can inhibit seed germination and hinder plant growth (Selim *et al*., 2012).

Heavy metal is any metal with a high density that is toxic at low concentrations, as it has the potential to accumulate in plants, animals, and water over time. According to Kuo *et al*. (2004), metal concentrations in finished composts tend to be higher than in the initial composting mix due to mass reduction from C loss as CO_2 or degradation of organic matter. Furthermore, the concentration of metals in the finished compost may vary from the feedstock depending on the types of bulking agent used. Although Kuo *et al*. (2004) reported a rapid decline of water-soluble lead and zinc during composting, Tibu *et al*. (2019) reported that the nickel and arsenic content of compost piles decreased, while the content of chromium, zinc, cadmium, and lead increased throughout the composting of various municipal wastes at different bulking ratios.

Again, according to Ermadani and Arsyad (2013), the usage of alkaline materials for composting is beneficial in reducing the soluble and exchangeable fractions of heavy metals through the formation of insoluble carbonates, adsorption of metals into particles of alkaline material or the formation of organo-metals with organic matter in soluble fractions during composting process so that the bioavailability of heavy metals is decreased.

Volatile organic compounds (VOCs), such as aliphatic and aromatic hydrocarbons, chlorinated compounds, and organic acids including formic, acetic, propionic, and butyric acids can develop during compost degradation when aeration is insufficient (Ramos *et al.,* 2002). Additionally, sulphur-containing compounds like hydrogen sulphide (H_2S), carbonyl sulphide (COS), and various methylated sulphur compounds [(CH_3) S, $(CH_3)_2S_3$, and $(CH_3)_2S_3$] may also form under these conditions. Compost made from straw or wood residues tends to have higher concentrations of low molecular weight aliphatic acids, which are by-products of amino acid degradation, particularly from methionine, cysteine, histidine, tyrosine, and phenylalanine (Baziramakenga & Simard, 1998).

2.6 Compost Application on Tropical Soil Health

2.6.1 Soil physical properties

Extensive research has demonstrated the various benefits of applying compost to soil. Compost amendments enhance soil physical properties by reducing bulk density, increasing water retention, and improving infiltration rates (Petersen *et al*., 1999). Compost application lowers soil density due to the admixture of low-density organic matter (OM) into the mineral soil fraction and this positive effect is typically associated with an increase in porosity because of the interactions between organic and inorganic fractions (Amlinger *et al*., 2007). However, Reynolds *et al*. (2015) observed a negligible effect of low rate (< 75 t ha^{-1}) of food waste compost on soil bulk density after a single application.

The addition of organic matter from compost is widely recognized for its role in reducing bulk density and enhancing water-holding capacity. Additionally, it significantly improves soil aggregate stability. Soil structural stability refers to the soil's ability to resist changes in pore and particle arrangement when subjected to stresses such as cultivation, compaction from trampling, and irrigation (Krull *et al*., 2004). Compost application to soil also increases aggregate stability of especially clayey and sandy soils through the supply of a well humified and as well as fresh, low-molecular organic matter that are responsible for micro-aggregates macro aggregates formation respectively (Fischer & Glaser, 2012). Soil macro-aggregates, which are primarily stabilized by fungal hyphae, fine roots, root hairs, and microorganisms, benefit from the carbon inputs provided by compost application (Amlinger *et al*., 2007).

According to Six *et al*. (2004), organic inputs stimulate microbial activity, leading to the production of by-products that act as binding agents between soil particles, promoting aggregate formation. Additionally, more resistant organic compounds introduced through compost contribute to the stabilization of smaller aggregates (Marschner & Flessa, 2006). Compost application also enhances soil structure by improving aggregate formation and pore properties, increasing the soil’s active surface area. This is because, the higher the specific surface area, the more intensive interactions can occur between soil fauna, microorganisms and root hairs under optimum conditions (e.g. sufficient humidity) (Amlinger *et al*., 2007). Liu *et al*. (2021) found that applying compost at a rate of 44.0 tonnes per acre every two years over an eight-year period led to a slight but notable increase in the large macro aggregate fraction and overall aggregate stability. Additionally, compost-derived organic matter enhances soil water conductivity (Carter *et al.,* 2004) by serving as a food source for soil organisms, which play a role in creating macro-pores and improving soil structure.

An essential indicator of soil physical fertility is its ability to store and supply both water and air for plant growth. Specifically, the balance between plant-available water and air-filled porosity at field capacity is commonly used to evaluate soil fertility (Peverill *et al.,* 1999). The movement, retention, and distribution of water in the soil are largely determined by pore volume and pore size distribution. Only pores with diameters smaller than 50 µm can retain water against gravity due to capillary forces, while those below 0.2 µm hold water so tightly that it becomes unavailable to plants (Peverill *et al.,* 1999).

Soil water-holding capacity is primarily influenced by the quantity and size distribution of pores, as well as the specific surface area of the soil (Haynes & Naidu, 1998). An increase in soil organic carbon (SOC) enhances aggregation and lowers bulk density, thereby increasing total pore space (Khaleel *et al*., 1981). Numerous studies have shown that organic amendments significantly improve soil water content (Carter *et al*., 2004). This improvement is linked to the formation of secondary pore structures, often created by plant roots and soil organisms. Additionally, organic matter can absorb water at a rate of three to twenty times its own weight. Research by Hudson (1994) found that raising total organic carbon (TOC) content from

0.5% to 3% led to a twofold increase in available water content. Similarly, Şeker and Manirakiza (2020) observed higher available water content in sandy clay loam soil treated with compost.

Soil aeration, greenhouse gas emissions, and the movement of volatile compounds are influenced by air permeability and relative gas diffusivity, which quantify the soil's ability to conduct air (Arthur *et al*., 2013). The air content in soil is determined by the difference between total pore volume and the portion occupied by water (Amlinger *et al.,* 2007). Research indicates that air permeability and exchange within soils are largely controlled by macropores.

Arthur *et al*. (2012) further noted that soil gas diffusivity, or relative gas diffusivity, is influenced by air-filled porosity, bulk density, and the tortuosity and connectivity of pore spaces. Soils with high clay content or compaction tend to have fewer macropores, which can restrict oxygen availability. In such cases, the application of organic matter through compost plays a crucial role in improving soil structure, enhancing stability, and promoting the formation of secondary macropores, particularly through root activity and soil organisms.

2.6.2 Soil chemical properties

Compost application to soil has proven to increase the cation exchange capacity (CEC) of soil. CEC is the measure of the total capacity of a soil to hold exchangeable cations and indicates the negative charge present per unit mass of soil (Peverill *et al*., 1999). A high cation exchange capacity (CEC) is considered beneficial, as it enhances a soil's ability to retain essential nutrient cations, making it a key indicator of soil fertility (Krull et al., 2004). Research by Kögel-Knabner *et al*. (1996) and Ouedraogo *et al*. (2001) found that compost application improves soil CEC by introducing stabilized organic matter (OM) rich in functional groups. Amlinger *et al*. (2007) further noted that soil organic matter (SOM) contributes between 20% and 70% of the CEC in many soils.

Krull *et al*. (2004) emphasized that in highly weathered tropical soils, such as those dominated by kaolinite; most of the CEC is linked to SOM. Therefore, maintaining high SOM levels through organic amendments is particularly crucial for sustaining soil fertility in tropical and sandy soils. Additionally, studies by Agegnehu *et al*. (2014) confirmed that compost application leads to an increase in CEC. Similarly, Ermadani and Arsyad (2013) reported that applying effluent compost to Ultisol soils resulted in higher CEC, with further increases observed as compost input levels rose.

Moreover, compost amendment is also known to optimize soil pH and improve the buffering capacity of a soil. Soil pH measures the acidity or alkalinity of soil and is expressed as the negative logarithm of hydrogen ion activity in a soil suspension. It plays a crucial role in crop cultivation, as most plants and soil organisms thrive within specific pH ranges, favouring either slightly acidic or alkaline conditions. Additionally, soil pH influences nutrient availability, directly impacting plant health and overall soil vitality (Fischer & Glaser, 2012). Adugna (2016) iterated that the liming effect of compost application is due to its richness in alkaline cations such as Ca, Mg and K which were liberated from OM due to mineralization.

Consequently, regularly applied compost material on the maintenance or enhancement of soil pH was reported by Kögel-Knabner *et al*. (1996) and Ouedraogo *et al*. (2001). Mohammad *et al*. (2004) observed a decline in soil pH following compost application. In contrast, Kluge (2006) found that even moderate compost additions led to a notable pH increase, with values rising from 6.4 to 6.8 when 10 Mg d.m. compost per hectare was applied. Similarly, Ermadani and Arsyad (2013) reported an increase in the pH of Ultisol soils after the application of effluent compost.

Soil buffering plays a crucial role in maintaining soil health by preventing drastic changes in pH and determining the amount of amendments required to adjust soil acidity or alkalinity (Krull *et al*., 2004). The presence of various functional groups in soil organic matter (SOM) from compost, including carboxylic, phenolic, acidic alcoholic, amine, and amide groups, helps regulate pH levels across a broad range of soil conditions. Cayley *et al*. (2002) documented a good correlation between buffering capacity and organic matter content, despite acidifying factors. Increased microbial activity and functionality from municipal solid waste compost can also stabilize potentially harmful components in sub-acidic polluted soils (Garau *et al*., 2019).

Compost is popularly termed as a multi nutrient fertiliser because it contains significant amounts of essential plant nutrients including N, P, K, Ca, Mg and S as well as a variety of trace elements (Adugna, 2016; Agegnehu *et al*., 2014). Alvarez *et al*. (1988) found that the OM content of compost is associated and available Ca, K, Mg, Na, P and exchangeable K and can be an alternative to chemical fertiliser to restore the soil nutrient balance. Soheil *et al*. (2012) reported that application of municipal waste compost increased the amount of available N, P and K and micronutrient concentrations in a pot experiment and observed that soil N, P and K content increased with increasing application rate. An insignificant effect of palm oil effluent compost on soil N was reported by Ermadani and Arsyad (2013). However, in that same experiment, the soil P, K, Ca and Mg were increased by the compost application. Again, Singer *et al*. (2004) observed that compost application increased soil P content to 198 % and soil K to 55 % compared to the unamended maize, soybean and wheat cropped field. Another major benefit of compost fertilization is attributed to its lasting effect on soil fertility due to a slow and gradual release of plant nutrients (Smith & Collins, 2007). There is also a much better protection of soil nutrient from leaching in compost fertilization compared to mineral fertilisers (Fischer & Glaser, 2012).

The total nutrient content in compost is not immediately available to plants, as nutrients exist in various binding forms within the organic matrix, leading to partial immobilization, particularly of nitrogen (Becker *et al*., 1995). This variability makes it challenging to accurately predict the fertilization effect and estimate soil nutrient balance when compost serves as the primary nutrient source (Becker *et al*., 1995). For example, Becker *et al*. (1995) estimated that only 10–20% of the total nitrogen in compost is mineralized within the first year of application. However, unlike nitrogen, phosphorus shows a fertilization effect of approximately 50%, potassium nearly 100%, and magnesium also demonstrates significant availability, as reported by Bidlingmaier *et al*. (2000).

2.6.3 Soil biological properties

Soil hosts a wide range of organisms, from large, visible species to microscopic life forms, each contributing to different soil functions (Adugna, 2016). Various fractions of organic matter (OM) influence microbial activity levels. While easily degradable organic compounds temporarily boost microbial activity and biomass, a sustained increase in microbial biomass requires the continuous input of stable OM, which is particularly enhanced through the application of mature compost (Fischer & Glaser, 2012).

One of the key benefits of compost amendment is its role in stimulating soil biological activity by providing a food source degradable carbon compounds for heterotrophic soil organisms (Blume, 1989). Additionally, compost improves soil biology by optimizing habitat conditions, such as water and air balance, increasing surface area, adjusting pH, and creating refuge areas for microorganisms. Compost application also introduces beneficial compost biota into the soil, acting as an inoculant (Amlinger *et al*., 2007). For instance, Brown and Cotton (2011) reported that microbial activity significantly increased in compost-amended soils compared to control soils, with compost-treated soils exhibiting 2.23 times greater microbial activity than untreated soils. Also, Paul (2003) reported that long-term manure compost application increased soil biological properties compared to mineral fertilisers. Lastly, Ding *et al*. (2021) observed that compost made from maize straw and sewage sludge could raise the soil pH that increased beneficial microorganisms and decreased the number of harmful microorganisms.

Arbuscular mycorrhizae fungi (AMF) are potential indicators of soil fertility due to their obligate symbiotic nature and their susceptibility to perturbation (Bending *et al*., 2004). Limited information and inconsistent results are presently available on the effect of complex heterogeneous organic matter, such as compost, on the AMF-plant system. For instance, it is not yet clear whether compost addition results in an increase, decrease, or invariability of mycorrhizal colonization of roots (Cavagnaro, 2014). Moreover, replacement of microbial C source from recalcitrant to labile soil organic matter was found to produce significant shifts in the composition of soil microbial communities, such as AMF, saprophytic fungi, and Gram+ and Gram− bacteria (Bardgett *et al*., 2007; Biasi *et al*., 2005).

A modification in the composition of phospholipid fatty acids (PLFA) with amendment to soil with manure was related to an increased availability of easily degradable dissolved organic C that directly stimulated growth of soil bacterial communities (Lazcano *et al*., 2013). Also, Cozzolino *et al.* (2016) indicated that the total amount of PLFA was significantly enhanced by all compost treatments. However, soil amendment with compost at varying degree of maturation reduced AMF spores and propagules and the quantity of C16:1ω5 NLFA, compared to the control (Cozzolino *et al.,* 2016). In fact, it was found that the decrease of AMF biomass and activity was concomitant to the reduction of plant growth and nutrient uptake in compost-treated soils.

2.6.4 Crop yield and nutrient use efficiency

Soil amendment with compost has proven to stabilize and increase crop yields and quality due to its multiple positive effects on the physical, chemical and biological soil properties (Adugna, 2016). Compost application affect plant productivity and nutrient use efficiency by intensifying essential interactions between root hairs, soil fauna and microorganisms due to an enhancement of specific surface area (Amlinger *et al*., 2007). According to Ndubuisi-Nnaji (2011), the enhancement of plant growth and yield as a result of compost application could be attributed to the content of plant hormone in compost that functions as stimulants. Aluminium toxicity in highly weathered tropical soil can be ameliorated by compost application through the formation of Al-organic compound with organic matter from compost and this can boost growth and yields of crops that are grown in these soils (Winarso *et al*., 2009). Research has consistently highlighted the economic benefits of incorporating compost into organic farm management (Forster *et al*., 2013; Adamtey *et al*., 2016). For instance, Bevacqua and Mellano (1993) reported that applying a total of 37 or 74 Mg ha^{-1} of biosolids compost over two years significantly increased the yields of turf grass, onion, and lettuce on sandy calcareous soil compared to untreated soil. Additionally, moderate applications of sugar beet vinasse compost to calcareous loamy sand soil improved plant nutrition and yield without posing significant risks of salinization or sodification in coarse-textured soils. Again, Zhang *et al.* (2016) demonstrated that after three years of compost treatment, maize grain yield increased.

Also, Hemmat *et al.* (2010) reported that municipal solid waste compost with sewage sludge compost acted as a source of nutrients with beneficial residual effect on nutrient contents of soil and increased crop yields. Further, a field trial by Courtney and Mullen (2008) demonstrated that soil amendments separately with a spent mushroom substrate and forced aeration compost applied at 25, 50, and 100 t ha^{-1}generated barley grain yields correlated with the amendment rate and remarkably higher than the control and comparable to inorganic NPK fertilization treatments.

However, in situations where compost replaces chemical fertilisers, a number of trials have revealed that yield decrease in the first years. This is attributed to the fact that the nutrients in applied compost are mainly not directly plant-available but will be released during the subsequent years (Baiano & Morra, 2017). On the other hand, Fischer and Glaser (2012) explained that compost use can also improve the quality of agricultural products in terms of dry matter yield, micro and macro nutrient contents of agricultural products. Fricke *et al*. (1990) observed a significant increase in the dry matter content of beetroot and a reduction in nitrate levels following compost application compared to plots treated with mineral fertilisers.

2.6.5 Soil carbon sequestration

The level of soil organic carbon (SOC) in a given soil is governed by the balance between organic carbon inputs such as vegetation, roots, and amendments and outputs, primarily CO_2 released through microbial decomposition. However, factors including soil type, climate, land management practices, mineral composition, topography, and soil biota, along with their interactions, influence the total SOC storage capacity of the soil (Krull *et al*., 2004). Land-use changes, soil degradation, and the impacts of climate change further complicate efforts to restore soil organic carbon (SOC) in tropical regions (La Scala Júnior *et al.,* 2012). However, compost application can be an effective strategy for replenishing organic carbon levels in these soils. However, the type, quantity, the frequency of application and the humification degree of compost coupled with soil properties such as soil type and clay content are the main factors that affect soil organic carbon enrichment from compost application (Fischer & Glaser, 2012).

Amlinger *et al.* (2007) suggested that maintaining adequate soil organic matter (SOM) in agricultural soils requires compost application at a rate of 7–10 Mg (dry matter) per hectare. However, for a long-term increase in SOM levels, more than 10 Mg per hectare would be necessary. A field study by Courtney and Mullen (2008) found that applying spent mushroom substrate and forced aeration compost at rates of 25, 50, and 100 t ha^{-1}led to increased soil organic carbon (OC) levels compared to untreated soil. Additionally, Prommer *et al*. (2020) highlighted that compost application enhances nutrient availability, promoting plant growth and biomass production, which subsequently increases organic matter input through plant residues, contributing to soil OC accumulation. Similarly, Bouajila and Sanaa (2011) reported that applying manure and household waste compost at 120 t ha^{-1} significantly boosted soil organic carbon, with increases of 1.74% and 1.09%, respectively, compared to control soils. Furthermore, in a study conducted over two growing seasons on the Tropical Island of Guam, Mohammad *et al*. (2004) observed that composted organic waste applications led to substantial improvements in soil organic matter content.

Krull *et al.* (2004) stated that for soil organic carbon (SOC) levels to increase effectively, the rate of carbon input must surpass the losses caused by decomposition and leaching. In tropical conditions, high radiation loads and high soil temperatures eventually lead to the total disappearance of the organic matter content from compost application, leaving only the mineral nutrients behind (Fischer & Glaser, 2012). As a result of this degradation kinetics, annual applications may be required in tropical soils to achieve optimum benefits of the compost on carbon sequestration and overall soil health (Kuo *et* *al*., 2004). This implies additional labour and increased costs for smallholder resource-poor farmers. Another major disadvantage associated with compost use is the introduction of heavy metals into the soil system (Kuo *et al*., 2004). Bünemann *et al.* (2018) noted that improper compost management, including excessive application rates or unbalanced nutrient composition, can result in nutrient losses and environmental pollution, potentially negating the benefits of organic carbon restoration. To ensure the sustainable use of compost and manure for enhancing organic carbon in tropical soils, it is crucial to adopt proper management strategies. This includes optimizing application rates based on soil nutrient requirements and maintaining a balanced nutrient supply (Adekiya *et al.,* 2023).

2.6.6 Greenhouse gas emissions

Greenhouse gases absorb and emit radiation within the thermal infrared spectrum (Pachauri & Reisinger, 2007). These gases include water vapour, ozone, carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), as well as hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulphur hexafluoride (SF_6) (Pachauri & Reisinger, 2007). Joseph *et al*. (2019) reported that CO_2, CH_4, and N_2O are the primary contributors to global climate change, accounting for approximately 60%, 20%, and 10% of the total impact among the six major greenhouse gases identified in the Climate Change Control Inventory. Soil is a significant source of greenhouse gas emissions, influenced by factors such as land use, agricultural practices, and natural terrestrial processes (Blagodatsky & Smith, 2012). Basheer *et al*. (2024) enumerated land use/land cover change, nutrient availability, humidity and temperature, the soil potential of hydrogen (pH), fertilisers, and organic amendments as the critical drivers of GHG emissions from agricultural soils.

Soil carbon dioxide emissions primarily originate from native soil organic matter (SOM), the decomposition of added carbon sources such as dead plant material, the breakdown of root exudates and decaying roots, as well as direct respiration from plant roots (Luo, Hui & Zhang, 2006). According to Fu *et al.,* (2023), key factors influencing CO_2 emissions from soil include temperature, moisture levels, and substrate availability. One approach to enhancing soil organic carbon is the application of organic amendments like compost and biochar (Martínez-Blanco *et al*., 2013). Compost contributes carbon to the soil, stimulating microbial activity, which in turn promotes microbial growth and encourages aggregate formation. This process aids in carbon stabilization and sequestration, either as microbial biomass or within soil aggregates. Carbon dioxide is emitted as a result of microbial decomposition of soil organic matter, root respiration, rhizomicrobial respiration and the priming effect, and from human activity including land use changes (Kuzyakov, 2006).

Therefore, while the carbon input from compost application shows positive influence on SOC content in agricultural soils, it also acts as a food source for soil microorganisms which leads to increase in microbial activity (Gaiotti *et al*., 2017). An increase in soil microbial activity such as respiration is also measured as carbon dioxide emissions and is also a typical response to increased SOM/SOC from compost addition (Rubio *et al*., 2013). For instance, Calleja-Cervantes *et al*. (2015b) found that the application of compost caused increased daily CO_2 fluxes, compared to the synthetic fertiliser treatment, but the cumulative CO_2 emissions following compost application were not significantly higher than the control or synthetic fertiliser treatments. Xu and Chang (2022) reported that a significant portion of the carbon from compost is released as CO_2 or CH_4 after being incorporated into the soil. Similarly, Gao *et al.* (2022) found that cumulative CO_2 emissions were higher from compost application compared to biochar over an entire growing season. Additionally, Verhoeven *et al.* (2017) noted that the presence of labile organic carbon in fresh compost can temporarily increase microbial activity, leading to a subsequent rise in CO_2 emissions. Fu *et al*. (2023) explained that the increase in CO_2 emissions following compost application is due to the improved conditions for microbial growth, which enhance microbial populations and nutrient availability (Zavalloni *et al*., 2011).

Soil CO_2 emissions from compost application vary based on factors such as compost type, maturity level, application rate, and frequency (Ray *et al*., 2020; Takakai *et al*., 2020). Ray *et al*. (2020) found that cumulative CO_2 emissions were higher in soils treated with chicken compost compared to those amended with dairy manure, regardless of the application rate. Additionally, Hoang and Maeda (2018) observed that commercial compost made from peat, which has a higher NH_4^+-N content, produced lower CO_2 emissions than chicken compost after being applied to the soil. Chau *et al*. (2023) also observed that CO_2 emissions from soil amended with mature kitchen compost were lower than those treated with immature ones because of less readily decomposable organic C in compost. As it stands little or no research exists on the potential emission of carbon dioxide from tropical soils amended with EFB compost, warranting research in this area.

Methane is produced when organic matter is decomposed under anaerobic soil conditions, in the absence of electron acceptors other than CO_2 (NO_3^-, Mn_4^+, Fe^{3+}, SO_4^{2-}) (Mosquera *et al*., 2007). According to Conrad (2007), CH_4 production takes place in all anoxic environments where organic carbon is microbially degraded and it is also influenced by soil bulk density (Basheer *et al*., 2024). Normally, when oxygen supply is adequate, most of the C in the organic matter converts to CO_2. However, in the absence of oxygen, decomposition is incomplete and C is released as CH_4 instead (Basheer *et al*., 2024). In the year 2000, agriculture contributed approximately 47% of total human-induced methane (CH_4) emissions, with rice paddies accounting for 30% of this share (Basheer *et al*., 2024). Kammann *et al*. (2017) identified two key biological processes that regulate the net exchange of methane between soils or ecosystems and the atmosphere. These processes include methane production by strictly anaerobic methanogenic Archaea (Methanogens) and methane consumption by methanotrophic bacteria (Methanotrophs).

Most research report increased CH_4 release after compost application to the soil. Takakai *et al*. (2020) found that methane (CH_4) emissions were higher in soils treated with livestock compost compared to those amended with rice straw compost. Gao *et al*. (2022) reported marginally higher than the unamended control and biochar, but insignificant effect of manure compost on cumulative methane emission under winter wheat. Meanwhile, Chau *et al*. (2023) reported a significant increase in CH_4 emissions from soil treated with 1% kitchen compost. Fu *et al*. (2023) also found increased CH_4 emissions from soil amended with organic materials, including compost. The increase in CH_4 emissions following compost application, compared to the control, may be due to the supply of electron donors from the compost, which supports microbial activity and enhances methane production. Additionally, the fermentation of organic amendments can contribute to higher CH_4 emissions and lower soil redox potential under anaerobic conditions (Conrad, 2007). This necessitates caution when compost is used for reversing soil degradation.

Nitrous oxide (N_2O) is a potent greenhouse gas with a global warming potential nearly 300 times greater than carbon dioxide (CO_2) (Wong, 2021). In agricultural soils, nitrification and denitrification are widely recognized as the primary processes responsible for N_2O production (Mosquera *et al*., 2007). Nitrification refers to the biological oxidation of ammonium (NH_4^+) into nitrite (NO_2 O) and nitrate (NO_3 O) under aerobic conditions, generating N_2O as a by-product (Mosquera *et al*., 2007). However, under anaerobic conditions, biological denitrification occurs where some bacteria utilize NO_3^- to release N_2 (Abao, 2000). A less common mechanism for nitrogen transformation in soils is assimilatory nitrate reduction, where microbes absorb nitrate (NO_3^-), reduce it to ammonium (NH_4^+), and incorporate it into their biomass. However, this process rarely occurs in agricultural soils due to its inhibition by the typically low concentrations of NH_4^+ (Dalal *et al*., 2003). Dalal *et al*. (2003) and Verhoeven *et al*. (2017) indicated that nitrous oxide emission from agricultural soils is closely related with soil aeration, moisture, presence of an active carbon and nitrogen sources. Therefore, it is plausible to infer that compost addition to the soil will produce significant impacts on N_2O emission. The mineralization of compost releases inorganic nitrogen, providing substrates for nitrogen transformations, which can lead to the production of nitrous oxide (N_2O) as a by-product.

However, its contribution to N_2O emissions through nitrification and denitrification is influenced by fluctuations in soil oxygen levels and moisture content (Congreves *et al*., 2018). A meta-analysis by Fu *et al*. (2021) found that organic amendments, including compost, increased N_2O emissions by 14% compared to unamended soils. This increase was attributed to higher nitrogen availability from the amendments and the stimulation of microbial activity, which likely enhanced nitrification and denitrification rates (Zhou *et al*., 2017). However, in a study conducted in a cornfield and walnut orchard, Pujol-Pereira *et al*. (2016) reported that compost application had no significant impact on N_2O emissions compared to non-composted treatments. Additionally, Peng *et al*. (2011) and Verhoeven *et al*., (2017) suggested that compost initially stimulates microbial activity due to its labile organic carbon content, resulting in a temporary increase in CO_2 and N_2O emissions. It is important to consider that the effects of organic amendments on N_2O flux can vary throughout the growing season or experimental period, highlighting the need for comprehensive measurements over time. For instance, Bass *et al*. (2016) realised that the initial rates of N_2O emissions were higher in amended plots than the control, but the pattern reversed after the first 4–6 week period. After the initial period, N_2O emissions from the amended plots were generally lower than synthetic fertiliser treated plot.

Similarly, Agegnehu (2017) noted in a field trial that in the first two months of amendments application, N_2O emission were generally higher in the compost, biochar and their combined application plus fertiliser treatments compared to sole synthetic fertiliser treatment, however, as the season progressed a reverse trend was observed. All in all, while compost offer benefits in terms of soil health and fertility, it is crucial to carefully manage their application to mitigate the potential drawbacks associated with GHG emissions.

2.7 Biochar and Pyrolysis Conditions on Biochar Properties

Biochar is a carbon-rich material produced through the pyrolysis of biomass under high temperatures and low oxygen conditions, primarily for biofuel production (Laird *et al*., 2009). While similar to charcoal, biochar is specifically intended for soil applications to provide environmental benefits (Lehmann & Joseph, 2009). The pyrolysis process involves breaking down organic matter at temperatures between 350°C and 1000°C in a low-oxygen environment (EBC, 2022).

Fischer and Glaser (2012) described biochar formation as a process of water elimination followed by increased aromatic condensation, characterized by decreasing atomic O/C and H/C ratios along the combustion spectrum. Biochar exhibits significant variation in its chemical and physical properties due to differences in pyrolysis conditions and feedstock composition (Downie *et al*., 2009; Schimmelpfennig & Glaser, 2012). As a result, biochar is not strictly defined by specific material thresholds but is instead considered part of a broader class of compounds along the combustion continuum (Verheijen *et al*., 2010). In previous versions of the European Biochar Certification, a limit value of 50% organic carbon content was applied to biochar and thus all pyrolysis products below this limit were considered as pyrogenic carbonaceous materials (EBC, 2022). Biochar derived from crop residues, such as straw and grain husks, has been found to be highly suitable for various agricultural and industrial uses.

However, its carbon content often falls below 50%, leading to a reassessment of the previous 50% threshold (EBC, 2022). Currently, there are more than 100 distinct biochar samples, varying in feedstock and production methods. Based on these variations, suggested elemental ratio thresholds for biochar include an oxygen-to-carbon (O/C) ratio of less than 0.4 and a hydrogen-to-carbon (H/C) ratio below 0.6 (Schimmelpfennig & Glaser, 2012).

Wiedner and Glaser (2013) identified three key characteristics of biochar that have significant ecological implications. The first is its poly-condensed aromatic structure, which is highly resistant to microbial decomposition, making biochar an effective tool for long-term carbon sequestration (Kuzyakov *et al*., 2009). The second feature is the presence of functional groups along the edges of biochar's polyaromatic backbone, formed through partial oxidation (Glaser, 2001). These functional groups contribute to soil quality by creating exchange sites for cationic soil nutrients (Glaser & Birk, 2012).

The third is its porous structure, which enhances water retention and increases surface area, improving the physical sorption of dissolved organic molecules and providing habitat for soil microorganisms. Additionally, biochar's high alkalinity, stable carbon content, and large surface area make it widely useful for enhancing soil fertility, sequestering carbon, promoting crop growth, and remediating contaminated soils (Lehmann & Joseph, 2015).. Biochar is very stable in soil compared to other organic amendment additions, making its application to soils a suitable approach for the build-up of SOM and thus, for C sequestration (Fischer & Glaser, 2012). However, Jones *et al.,* (2011) and Farrell *et al*. (2013) agued out that biochar is not completely biologically inert as it is considered to be because it contains a proportion of relatively small labile aliphatic C structures (Cheng *et al*., 2006; Liang *et al*., 2008).

Furthermore, the recent popularity of biochar lies in the fact that it can be produced from different biomass ranging from agricultural crop residues, forestry residues, wood waste, organic portion of municipal solid waste (MSW) to animal manures (Duku *et al*., 2011; Yaashikaa *et al.*, 2020). Its application has shown efficiency in different areas such as water treatment; thus, many studies have been dedicated to research biochar application for the removal of contaminants from aqueous solutions (Tan *et al*., 2015) and the treatment of polluted soils with heavy metals. Other application of biochar includes the reduction of greenhouse gas emissions and increased soil fertility in an environment friendly manner (Lee *et al*., 2019). Nevertheless, the suitability of biochar for a given application is dependent on pyrolysis conditions, together with feedstock characteristics that largely control the physical and chemical properties (e.g. composition, particle and pore size distribution) of the resulting biochar (Verheijen *et al*., 2010).

The characteristics and composition of biochar are shaped by various pyrolysis factors, including temperature, heating rate, retention time, kiln design, and feedstock type (Ippolito *et al*., 2020). The interaction of these elements creates a complex dynamic that influences biochar's ability to interact with soil, microorganisms, plants, water, and air (Domingues *et al*., 2017). Biochar is not a pure carbon, but rather, it consists of carbon (C), hydrogen (H), oxygen (O), nitrogen (N), sulphur (S), volatile matter, and ash content that contains plant macronutrients such as phosphorous, and potassium

(Zhang *et al*., 2015). Almutairi *et al*. (2023) and Domingues *et al*. (2017) reported that higher pyrolysis temperatures transform carbon into more stable, aromatic compounds while promoting the volatilization of oxygen, hydrogen, sulphur, and nitrogen. This process increases the ash content, liming potential, pH, and the concentration of oxides and carbonates, particularly in biochar derived from high-ash feedstocks.

Pyrolysis normally reduces hydrogen (H) and oxygen (O) content of biochar compared to the original feedstock. This reduction is due to processes such as dehydration, volatilisation of compounds (e.g., CO, CO_2, H_2O, and hydrocarbons), reduction of hydroxyl (− OH) functional groups, and condensation during pyrolysis as reported by Jindo *et al*. (2014), Souza *et al*. (2021), and Wijitkosum and Sriburi (2023), who observed decreases in H and O content in biochar produced from various biomass sources. Studies by Guizani *et al*. (2019) indicated that the aromatisation process begins at approximately 350°C and continues to increase. However, the temperature at which maximum aromatisation is achieved depends on the feedstock type, whether woody or non-woody (Tomczyk, Sokołowska, & Bogota, 2020).

During pyrolysis, a significant amount of nitrogen in feedstocks is lost through volatilization, leading to biochar with low nitrogen content, primarily existing in heterocyclic forms at higher temperatures (Gao *et al*., 2022). Liu *et al*. (2018) found that as pyrolysis temperature increases, labile nitrogen compounds such as protein-N, free amino acid-N, and alkaloid-N present in rice straw are converted into inorganic nitrogen forms (NH_4^+-N, NO_2^--N, and NO_3^--N) and more stable organic nitrogen compounds, including nitrile, pyridine, amino acids, and pyrrole groups, within the biochar. The reduced nitrogen availability in high-temperature biochars is attributed to the stability of the remaining nitrogen, which is predominantly found in pyridine-N or pyrrole-N forms within the charred matrix (Liu *et al*., 2017).

In addition to elemental composition, molar ratios such as H/C_{org}, O/C_{org}, (O + N)/C, and C/N are essential for characterizing biochar and distinguishing it from other carbonization products. The atomic ratios of O/C_{org}, H/Corg, and (O + N)/C are used to assess aromaticity, polarity, and stability (Schimmelpfennig & Glaser, 2012), while the C/N ratio helps predict biochar's potential for reducing greenhouse gas emissions (IBI, 2012). Additionally, the C/N ratio serves as a key indicator for determining whether biochar-amended soils will promote nitrogen mineralization or immobilization, particularly in the short term (Paiva *et al*., 2024). Schimmelpfennig and Glaser (2012) identified the molar H/C_{org} ratio as a measure of carbonization degree and biochar stability, making it a crucial factor in assessing its long-term carbon sequestration potential. A value above 0.7 suggests either non-pyrolytic char or inefficiencies in the pyrolysis process (EBC, 2022). Similarly, the O/C_{org} ratio is a distinguishing factor for biochar, with values below 0.4 considered optimal. However, this ratio can be influenced by post-pyrolysis treatments or co-pyrolysis with oxidative or catalytic additives (EBC, 2022; Schimmelpfennig & Glaser, 2012).

Moreover, the O/C and (O + N)/C ratios indicate the degree of polarity in biochar, which affects the presence of polar functional groups and determines its hydrophobic properties (Elmquist *et al*., 2006; Wijitkosum & Sriburi, 2023). Zhao *et al*. (2019) found that H/C, O/C, and (O+N)/C atomic ratios of various biochars decreased with increasing pyrolysis temperature, further supporting the influence of temperature on biochar properties. Again, Zhao *et al*. (2019) found that H/C, O/C and (O+N)/C atomic ratios of rice straw biochar, bamboo biochar, and cow manure biochar, charred at varying ranged temperatures decreased with increasing pyrolysis temperature from 300–700 °C. Souza *et al*. (2021) observed that H/C and O/C molar ratios of sewage sludge biochar reduced with increasing pyrolysis temperature.

The nutrient composition of biochar varies depending on the feedstock used and can make up as much as one-third of its total weight (EBC, 2022). For biochar intended for agricultural or livestock applications, nutrient content disclosure is a legal requirement (EBC, 2022). The pyrolysis temperature plays a crucial role in feedstock transformation, affecting carbonization levels, nutrient volatilization, and the presence of organic functional groups in the final biochar product (Domingues *et al*., 2017).

During biomass thermal decomposition, elements such as potassium (K), chlorine (Cl), and nitrogen (N) volatilize at relatively low temperatures, whereas calcium (Ca), magnesium (Mg), phosphorus (P), and sulphur (S) require significantly higher temperatures due to their greater stability (Amonette & Joseph, 2009). Knicker (2007) reported that potassium and phosphorus begin to volatilize at temperatures above 760°C, while magnesium and calcium are lost only when temperatures exceed 1107°C and 1240°C, respectively. Thus, nutrients are more likely to be retained in biochar produced at peak temperatures of around 500°C. Again, Naeem *et al*. (2014) observed that pyrolysis of wheat and rice straw at 300–500°C resulted in increased concentrations of all nutrients, except nitrogen, with the highest nutrient levels found in biochar produced at 500°C. Chan and Xu (2009), along with Naeem *et al*. (2014), attributed the increase in nutrient content at higher temperatures to the loss of volatile compounds (C, H, and O) and relatively minor losses of alkali nutrients in the gaseous phase.

The interaction between pyrolysis temperature and feedstock composition plays a crucial role in determining biochar properties, influencing its liming potential, potassium (K) and phosphorus (P) availability for crops, nitrogen (N) content, cation exchange capacity (CEC), and its role in regulating carbon (C) and nitrogen (N) dynamics in soil (Ippolito *et al*., 2020; Wang *et al*., 2012). Paiva et al. (2024) noted that the carbonization process induces alkalization within the biochar matrix, leading to significant pH differences compared to the original feedstock, depending on the pyrolysis temperature. Their study found that biochars produced at 300°C and 750°C exhibited pH increases of more than five units relative to their respective feedstocks. This rise in pH at higher pyrolysis temperatures is associated with elevated ash content and the accumulation of basic cations.

For high-ash feedstocks, increasing pyrolysis temperature results in the conversion of carbon into more stable, aromatic structures while promoting the volatilization of oxygen (O), hydrogen (H), sulphur (S), and nitrogen (N). This process enhances ash content, liming capacity, pH, and the concentration of oxides and carbonates in certain biochars (Almutairi *et al*., 2023; Domingues *et al*., 2017). In general, biochar pH tends to rise with higher temperatures and ash content, particularly in nutrient-rich feedstocks, though this may also lead to reduced nutrient retention, especially for volatile organic compounds (Ippolito *et al*., 2020). Conversely, some high-ash, lower-temperature biochars contain more polar organic functional groups, preserve non-volatilized nutrients in bioavailable forms, exhibit higher CEC, and retain greater levels of bioactive water-soluble carbon (Ippolito *et al*., 2020).

Nevertheless, various potentially toxic organic compounds, including polycyclic aromatic hydrocarbons (PAHs), heavy metals, polychlorinated dibenzo-p-dioxins and furans (PCDD/F) are formed during biomass pyrolysis (Buss *et al*., 2022). The Maximum Allowed Thresholds (MAT) indicates toxicant levels above which the material would not be considered acceptable. In order to meet the requirements of these IBI Biochar Standards, reported toxicant levels must be below the MAT that has been established in the area of jurisdiction where biochar is produced and/or intended for use (IBI, 2012). The maximum allowable concentrations for polychlorinated biphenyls (PCB) and polychlorinated dibenzo-p-dioxins and dibenzofurans (PCDD/F) are set at 0.2 mg kg^{-1} dry matter (DM) and 20 mg kg^{-1}, respectively, in accordance with soil protection regulations in Germany and Switzerland (EBC, 2022). These pollutant contents depend mainly on the chlorine content of the pyrolyzed biomass and that biomasses with low chlorine contents will likely have very low contents of these organic pollutants from the resulting biochar (EBC, 2022).

Heavy metals are a natural part of all ecosystems, and in biochar production, their concentration typically increases due to the pyrolysis process. Since more than 50% of the original biomass weight is lost during pyrolysis primarily through the release of carbon, hydrogen, and oxygen the heavy metals present in the feedstock become more concentrated in the resulting biochar (EBC, 2022). Wang *et al*. (2017) indicated that the concentrations of heavy metals in chicken manure and water-washed swine manure biochar produced at temperatures ranging from 200–800°C were all higher than those in their original feedstocks. Furthermore, Lu *et al*. (2015) found that the heavy metal contents in sewage sludge biochar were greater than in sewage sludge and that the heavy metal contents increased with the pyrolysis temperature.

Polycyclic aromatic hydrocarbons (PAHs) are organic compounds made up solely of carbon and hydrogen, consisting of at least two fused aromatic rings (Buss *et al*., 2022). These pollutants are widespread in the environment and are primarily produced through the incomplete combustion of organic materials (Creaser *et al.*, 2007). Many PAHs are known for their toxic, mutagenic, and carcinogenic properties. Due to their high lipid solubility, they are easily absorbed through the gastrointestinal tract in mammals. While PAHs can cause short-term harm to both human and plant health, their long-term carcinogenic, mutagenic, and teratogenic effects are of greatest concern (Buss *et al*., 2022). PAHs are generally characterized by a high melting and boiling points (therefore they are solid), low vapour pressure, and very low aqueous solubility and their solubility in water tend to decrease with increasing molecular weight which rather tend to increase resistance to oxidation (Creaser *et al*., 2007).

Among the many polycyclic aromatic hydrocarbons (PAHs), the United States Environmental Protection Agency (USEPA) has designated 16 as "priority pollutants," with eight classified as potentially carcinogenic by the European Food Safety Authority (EFSA). The 16 priority PAHs identified by the USEPA include naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, pyrene, fluoranthene, benzo(a)anthracene, chrysene, benzo (b) fluoranthene, benzo (k) fluoranthene, benzo (a) pyrene (B[a]P), indeno (1,2,3-cd) pyrene, dibenz (a,h) anthracene, and benzo (ghi) perylene (IBI, 2012). These individual PAH differ in the type and degree of toxicity which depends on the molecular structure, the concentration, the bioavailability, the exposure route, and the temporal course of the exposure (Hilber *et al*., 2017).

Earlier studies have reported that residence time of hot volatile compounds, feedstock type and particle size, pyrolysis equipment design, and catalyst are the major factors that affect PAHs contents in biochars (Li *et al*., 2023). Again, it has been shown that the presence of PAHs in biochar originate from the formation of biochar skeleton structure and the adhering of tar to the pores and surface of the biochar but little is known about effect of pyrolysis temperature on PAHs content of biochar (Lehmann *et al*., 2015). Freddo *et al*. (2012) found that the sum of 16 EPA-PAH levels in the biochar samples obtained from redwood, bamboo, maize and rice at 300–600 ºC decreased as the temperature increased. Hale *et al*. (2012) also reported a similar PAH trend when they analysed 50 biochar samples obtained in a temperature range of between 250–900 ºC. Greco *et al*. (2021) attributed aromatisation, cyclisation, dehydrogenation and dealkylation as the main reactions involved in PAH formation at relatively low pyrolysis temperatures (below 500 ◦C). On the other hand, Weidemann *et al*. (2018) found that wheat straw-derived biochars produced at 700 °C contained higher concentrations of PAHs than those produced at 550 °C. The increase in concentration of PAHs is because they become more concentrated as other components of raw material release the solid matrix and the amount of solid matrix decreases. Research has shown that slow pyrolysis with a longer residence time tends to produce lower PAH concentrations compared to fast pyrolysis with a shorter residence time (Wang *et al*., 2017). Additionally, the PAH content in biochar decreases as the carrier gas flow rate increases (Buss, 2022).

Other works demonstrated that the most important factor yielding high contents of toxic PAHs in biochar was the pyrolysis unit design. During biochar production, PAHs are typically released with pyrolysis gases and are eliminated when these gases are combusted to generate thermal and electrical energy. However, depending on process conditions, some of the released PAHs may be reabsorbed by the biochar during production. Additionally, if biochar cools in the presence of PAH-rich pyrolysis gases, substantial amounts of PAHs can condense onto its surface and within its porous structure (Buss, 2022). To prevent this, it is recommended that biochar and pyrolysis gases be separated at temperatures high enough to prevent PAH condensation and absorption onto the biochar. Some works have proposed controlled vapour quenching as feasible means to support the avoidance of PAH accumulation in biochar (EBC, 2022).

Apart from kiln design, post-pyrolysis PAH contamination can also increase the PAHs content of biochar. Buss *et al.* (2022) noted that the pyrolysis process can effectively separate PAHs from solid biochar, with most PAHs remaining in the pyrolysis liquids and gases (Dai *et al*., 2014; Fagernäs *et al*., 2012). However, contamination can occur during the post-pyrolysis stage when pyrolysis vapours condense in cooler areas outside the furnace. In these conditions, PAHs do not undergo further conversion but instead physically adsorb onto the biochar surface, leading to contamination (Buss *et al.*, 2022).

Biochar intended for soil amendment must fulfil certain property- and composition-related requirements, to prevent harm to the ecosystem (EBC, 2020). Therefore, several guidelines with suggested threshold values for contaminants (including PAHs, PCDDs, or PCDFs) in biochar have been established, and the importance of contaminant analysis has been emphasized (IBI, 2015; EBC, 2020). In order to attain biochar with low concentration of contaminants the selection of suitable feedstock and the controlling of pyrolysis conditions in the unit as well as the post pyrolysis condition are necessitated. Therefore, knowledge of the variation in different process variables associated with different pyrolysis units is essential for producing high quality biochar suitable for a given application.

2.8 Biochar on Tropical Soil Health

2.8.1 Soil physical properties

Biochar's unique physical properties, including high surface area, porosity, and low density, influence soil structure, creating favourable conditions for root development, nutrient absorption, and plant growth (Fu *et al.*, 2019; Oladele, 2019; Diatta *et al.*, 2020). Its application affects various soil physical characteristics, such as water infiltration, moisture retention, aggregate stability, aeration, porosity, bulk density, and soil compaction (Wang *et al.*, 2020).

Research by Blanco-Canqui (2017) found that biochar enhances soil porosity by reducing compaction and bulk density while promoting soil aggregation. Similarly, Zhang *et al*. (2017) reported that biochar application increased capillary and total soil porosity by 23% and 24%, respectively, compared to untreated soil. Additionally, Adekiya *et al*. (2020) observed a 65% improvement in soil porosity with biochar application relative to the control. Studies showed that the margin of increase of soil porosity differed even when the same rate of biochar was applied and this is likely a consequence of differences in soil type and soil textural classes in the different studies (Alghamdi, 2018).

Blanco-Canqui (2017) identified two primary mechanisms through which biochar application reduces soil bulk density. First, soil typically has a higher bulk density (~1.25 g cm^{-3}) compared to biochar (< 0.6 g cm^{-3}), meaning that incorporating biochar lowers soil bulk density through a dilution effect. Second, over time, biochar interacts with soil particles, enhancing aggregation and porosity, which further contributes to bulk density reduction. Experimental evidence supports these findings. Liu *et al*. (2015) reported a decrease in soil bulk density from 1.66 to 1.53 g cm^{-3} following biochar application. In a three-year field study, Sushant *et al*. (2020) observed reductions in bulk density of the 0–7.5 cm soil layer by 4.5% and 6.0% at application rates of 0.23 kg m^{-3} and 0.45 kg m^{-3}, respectively. Similarly, Zhang *et al*. (2019) found that applying biochar over two years resulted in a 9.4% (± 2.2%) decrease in soil bulk density. Laboratory studies also support these observations. Adekiya *et al*. (2020) reported that applying 30 Mg ha^{-1} of biochar reduced soil bulk density by up to 75% compared to untreated soil.

Soil texture plays a crucial role in determining the extent of bulk density reduction following biochar application. Alghamdi (2018) found that biochar had a greater impact on reducing bulk density in coarse-textured soils than in fine-textured soils. Liu *et al.* (2017) observed the most significant bulk density reduction (31%) in sandy soils, followed by 14.2% in coarse-textured soils and 9.2% in fine-textured soils. These findings align with Blanco-Canqui (2017), who reported similar reductions of 14.2% in coarse soils and 9.2% in fine-textured soils after biochar application. Therefore, the extent of bulk density reduction in biochar-amended soils is influenced by soil-specific properties.

Soil aeration plays a vital role in crop growth, as a soil's ability to facilitate air movement is essential for plant productivity. The movement of air within soil is influenced by physical properties such as the balance between water-filled and air-filled pore spaces, as well as pore size and connectivity Obour *et al*. (2019). Unlike air permeability (k_a), the critical threshold for relative gas diffusivity in supporting plant growth varies, with estimates ranging between 0.005 and 0.02 (Sun *et al*., 2013). Biochar, due to its porous structure, enhances soil pore space, improving aeration. Obour *et al*. (2019) found that after three years of applying rice straw biochar at a rate of 30 t ha^{-1}, air permeability (k_a) increased at a matric potential of−300 hPa. Sun *et al*. (2013) reported that the application of 20 Mg ha^{-1} birch wood biochar increased air-filled porosity by 28 to 34%, 53 to 161% for relative gas diffusivity and 69 to 223% for air permeability. However, Garg *et al*. (2019) reasoned that additional water that occupies the intra-pores of biochar can cause reduction in air conductive channels for gas migration, causing the decrease in air flow by convection in biochar amended soils, especially with higher biochar content. Again, after seven months application of 20 t ha^{-1}

birch wood biochar, decrease in the air permeability of a sandy loam was reported by Kumari *et al.* (2014) and they attributed it to localized high water contents in biochar-rich soil.

Soil water retention indicates the maximum quantity of water that a soil can retain or hold. Several field studies have demonstrated that application of biochar increased soil water retention by positively affecting the soil porosity and other structural and textural properties of soil (Chan *et al*., 2008). Asai *et al*. (2009) proposed that incorporating biochar into soil alters the pore system due to its highly porous structure and diverse binding sites, which enhance water retention by adsorbing additional water molecules. Other researchers have attributed the increased water-holding capacity in biochar-amended soils to an expansion in specific surface area (Esmaeelnejad *et al*., 2016). However, the impact of biochar on soil moisture retention varies depending on soil texture.

Tryon (1948) observed that biochar application improved water availability in sandy soils, had no noticeable effect on loamy soils, and reduced moisture retention in clay-rich soils. Similarly, a field experiment conducted by Asai *et al*. (2009) found that adding biochar to upland rice paddies enhanced both soil water permeability and holding capacity, ultimately increasing water availability for plants. Chan *et al*. (2008) reported that soil field capacity improved with higher biochar application rates, but significant increases were only observed at 50 t ha^{-1} and 100 t ha^{-1} treatments.

However, biochar's effect on available water content (AWC) is not always beneficial. Jeffery *et al*. (2011) noted that its influence can be variable.

Mukherjee *et al.* (2014) found that applying powdered biochar to fine-textured soils actually reduced AWC. Since biochar tends to improve moisture retention primarily in coarse-textured soils (Sohi *et al*., 2010), selecting the right biochar-soil combination is crucial. Additionally, not all retained water may be accessible to plants, as water held in extremely small pores is unavailable for uptake (Sohi *et al*., 2010).

Other physical properties that are influenced by biochar amendments also include aggregate stability, specific surface area, tensile strength, soil consistency limits, compressibility, and penetration resistance. Biochar reduced the tensile strength and cracks of a surface soil (Mandal *et al*., 2020). Biochar application has been shown to reduce soil shrinkage by enhancing water retention, thereby improving soil structure (Fu *et al*., 2019). Research has also identified hydrophilic functional groups on the surface and within the pores of biochar, which exhibit a strong affinity for water. This property allows biochar to increase soil water retention more effectively in sandy soils compared to loamy or clay soils (Mandal *et al*., 2020).

Additionally, biochar has a positive influence on soil surface area, although the extent of this effect depends on the type of biochar used (Tomczyk *et al*., 2020). For instance, Tomczyk *et al*. (2020) reported that soil amended with 10% biochar had a surface area three times larger than untreated soil. In field studies on coarse-textured soils, Obia *et al*. (2016) found that biochar improved aggregate stability by 7–9% and 17–20% per percent biochar applied after two growing seasons. Overall, regardless of soil type, experimental conditions, biochar characteristics, or application rates, biochar consistently demonstrates beneficial effects on soil physical properties.

2.8.2 Soil chemical properties

Biochar addition to soil is reported to enhance soil chemical properties, such as soil pH, electrical conductivity (EC) cation exchange capacity (CEC), soil total carbon (TC), and C/N ratio, ensuring nutrient retention and soil fertility (Sun *et al*., 2022) Some studies have reported varying effects of biochar on soil chemical properties, with results differing due to factors such as climate variations, soil characteristics, cropping systems, and differences in biochar composition (Abujabhah *et al*., 2016).

Soil pH is a key factor influencing soil fertility and crop growth (Šimek & Cooper, 2002). Biochar application can affect soil pH in both the short and long term, depending on its source of alkalinity, which may originate from organic functional groups, soluble organic compounds, carbonates, or other inorganic alkalis (Cheah *et al*., 2014). Bolan *et al*. (2022) attributed the liming effect of biochar to the dissolution of carbonates, oxides, and hydroxides within its ash fraction. Additionally, a meta-analysis by Sun *et al*. (2022) found that variations in soil pH after biochar application were influenced primarily by initial soil pH (31.6%), biochar application rate (27.0%), and biochar pH (12.1%). However, Van Zwieten *et al*. (2010) noted that biochar has a more pronounced effect on acidic soils than on alkaline soils.

The application of biochar has been shown to enhance soil cation exchange capacity (CEC) (Cheng *et al*., 2006). This improvement is attributed to the presence of carboxylic and phenolic functional groups in biochar, which create electrostatic attraction for positively charged ions on its surface, thereby increasing the availability of cationic nutrients in the soil (Ding *et al*., 2016;

Tian *et al*., 2018). On the other hand, the effect of biochar is influenced by the initial CEC value of soil, the interaction between clay content and type, soil organic matter (SOM) and biochar amendment (Gul *et al*., 2015). Generally, the effect biochar on CEC in soils is more pronounced in a low fertility soils (Agegnehu *et al*., 2016). Zhang *et al*. (2017) observed a 21% increase in soil cation exchange capacity (CEC) following biochar application compared to untreated soil. Similarly, El-Naggar *et al*. (2018) found that applying biochar derived from rice straw, silvergrass residues, and umbrella tree to sandy soils resulted in CEC increases of 906%, 180%, and 130%, respectively, relative to the control. Additionally, Adekiya *et al*. (2020) reported an improvement in soil CEC after applying biochar at a rate of 30 Mg ha^{-1}. Domingues *et al*. (2020) reported that an increase in soil CEC relies on biochar CEC and rate rather than alterations in soil pH.

However, not all biochar-soil combinations result an increase in CEC because no or minimal changes in CEC have also been observed after certain biochar additions to soils (Nguyen *et al*., 2010). For instance, Eucalyptus sawdust and sugarcane bargass biochars did not increase the soil CEC and in the other hand, a statistically significant reduction in CEC of soil was observed for some of the soil-biochar combinations (Domingues *et al*., 2020). The reduction in the CEC of biochars prepared at high pyrolysis temperatures were enunciated as the reasons for the insignificant effects of such biochars on soil CEC.

Soil nitrogen mineralization is strongly impacted by biochar amendments due to its interaction with soil carbon, pH, moisture, biotic and abiotic adsorption and the activity of microbial communities in soil (Clough *et* *al*., 2013). Mukherjee, Lal and Zimmerman (2014) observed increased nitrogen mineralization rates after the short-term application of biochar, which they attributed to the biochar's higher hydrogen-to-carbon (H/C) ratio, representing less recalcitrant biochar and thus it is more likely to be decomposed and release N trapped in the biochar into the mineral nitrogen pool. Lone *et al*. (2015) found that biochar reduces inorganic nitrogen leaching by enhancing nutrient retention through cation and anion exchange reactions, as well as by immobilizing inorganic nitrogen due to its labile carbon content. Similarly, Abujabhah *et al*. (2016) suggested that biochar promotes organic carbon decomposition, which in turn lowers nitrogen availability in red loam soil. Nguyen *et al*. (2017) reported that plant-derived biochars led to reductions in soil ammonium and nitrate levels. Overall, biochar application can have varying effects on soil inorganic nitrogen availability, resulting in positive, negative, or neutral outcomes depending on specific conditions (Gao & DeLuca, 2016; Nguyen *et al*., 2017).

In agricultural systems, biochar application has been shown to improve phosphorus availability in the soil, making it more accessible for plant uptake (Gao & DeLuca, 2018). Tan *et al*. (2017) explained that phosphorus in biochar can undergo oxidation by aluminium and iron in the soil, increasing its availability. However, in some cases, biochar's alkalinity and the presence of exchangeable calcium (Ca), magnesium (Mg), and iron (Fe) on its surface can facilitate phosphorus precipitation or sorption, thereby reducing its availability in the soil (Tian *et al*., 2018; Xu *et al*., 2014). Lone et al. (2015) reported that incorporating maize residue biochar into calcareous soil at rates of 1–2%

(w/w) resulted in an increase of total nitrogen by up to 41% and available phosphorus by as much as 165% compared to untreated soil.

Generally, biochars are known to contain small quantities of N, P, K, Ca, Mg, S, and other nutrients required for plant growth (El-Naggar *et al*., 2019; Liu *et al*., 2018). For example, the application of maize residue biochar in calcareous soil at 1–2% (w/w) increased K up to 160%, and manganese (Mn), Fe, Zn, and Cu up to 21, 17, 42, and 10% compared to the control, respectively (Lone *et al*., 2015). Adekiya *et al*. (2020) found that biochar application increased the concentrations of nitrogen (N), phosphorus (P), potassium (K), sulphur (S), calcium (Ca), and magnesium (Mg) in soil compared to untreated soil.

However, Yao *et al*. (2017) observed that while total soil nitrogen increased, total phosphorus decreased following the application of maize stalk biochar at rates between 50 and 200 Mg ha^{-1}. Similarly, Mclennon *et al*. (2020) reported that biochar, whether applied alone or in combination with nitrogen fertilisation, did not improve nutrient concentration or nutrient removal, including N and P. On the other hand, biochar was found to enhance Ca and Mg availability, contributing to increased crop yields (Hossain *et al*., 2020). These varying results indicate that biochar's effects on soil chemical properties are influenced by factors such as biochar type, soil characteristics, environmental conditions, and the duration of the experiment.

2.8.3 Soil biological properties

Soil amendment with biochar can increase microbial biomass, stimulate soil microbial activity and change microbial community in soil

(Lehmann *et al*., 2011). Several mechanisms by which biochar influence soil biological properties have been outlined by earlier research. According to Lehmann *et al*. (2011), biochar affect soil microbial community structure due to their high sorption capacity, their effect on soil pH as well as modification of microbial environment. Again, seven major mechanisms underlying biochar's effects on microbes and their related soil functions and processes have been enunciated: (1) Biochar pore structures and surfaces provides shelter for soil microbes (Quilliam *et al*., 2013). (2) Nutrients and ions adsorbed on biochar particles supplies nutrients to soil microbes (Joseph *et al*., 2013). (3) Triggers potential toxicity with environmentally persistent free radicals. (4) Modifies microbial habitats by improving soil aeration water content, pH etc. (Quilliam *et al*., 2013). (5) Induces changes in enzyme activities that affect soil elemental cycles related to microbes (Stefaniuk & Oleszczuk, 2016). (6) Interrupts intra and interspecific communication between microbial cells via a combination of sorption and the hydrolysis of signalling molecules (Gao *et al*., 2016) and (7) Reduces the bioavailability and toxicity soil contaminants to microbes through sorption and degradation (Stefaniuk & Oleszczuk, 2016).

However, it should be noted that the underlined mechanisms are specific to the type and functions of the microbe in the soil, biochar property, soil type, plant species, and the environmental conditions. Soil fungi are heterotrophic eukaryotic organisms with diverse morphology, cytology, and phylogeny, and their populations are significantly influenced by biochar application (Wiedner & Glaser, 2013). Research suggests that biochar can serve as a habitat for mycorrhizal fungi (Xin *et al.*, 2022). Nishio (1996) proposed that biochar application enhances native arbuscular mycorrhizal fungi (AMF) in soil, promoting plant growth by modifying soil physical and chemical properties, altering plant-AMF signalling, and protecting AMF hyphae from fungal grazers (Lehmann *et al*., 2011).

The beneficial effects of biochar on AMF may also result from its ability to influence soil microbial communities in ways that support or hinder their growth, modify plant-mycorrhizal fungi interactions, or detoxify allelochemicals, thereby affecting root colonization by AMF. Additionally, biochar may provide a protective environment for AMF hyphae, shielding them from grazing by other soil organisms (Warnock, 2009). Warnock (2009) reported a negative impact of biochar on arbuscular mycorrhizal (AM) fungi. The application of three different biochar types lodgepole pine, peanut shell, and mango wood led to a significant decline in AM fungi abundance. With increasing biochar application rates from 0 to 116.1 tonnes biochar-C ha^{-1}, hyphal density decreased from 16.7 to 4.50 m hyphae per cm^3 in lodgepole pine biochar, 2.12 to 1.33 m hyphae per cm^3 in peanut shell biochar, and 19.2 to 4.45 m hyphae per cm^3 in mango wood biochar.

In contrast, Matsubara *et al*. (2002), DeLuca *et al*. (2006), Gundale and DeLuca (2006) found that biochar amendments altered soil nutrient availability, potentially improving host plant performance and increasing AMF colonization of plant roots (Ishii & Kadoya, 1994). Beyond AMF, Matsubara *et al*. (2002), observed that two saprophytic white-rot fungi, Pleurotus pulmonarius and Trametes versicolor, colonized the surface and, to a lesser extent, the interior of biochar, exhibiting different growth patterns. The researchers suggested that the physical structure of biochar and the presence of nutrients such as phosphorus (P), potassium (K), and calcium (Ca) on its surface likely facilitated fungal colonization.

The impact of biochar on soil bacterial diversity and community composition is influenced by factors such as biochar type, soil characteristics, and agricultural practices, including crop selection and planting duration (Abujabhah *et al*., 2016). Chen et al. (2015) found that applying biochar at rates of 20 and 40 t ha^{-1} increased the relative abundance of Betaproteobacteria and Deltaproteobacteria in soil with a pH of 4.89 and an organic carbon content of 17.7 g kg^{-1}. However, in soil with a pH of 5.99 and organic carbon content of 20.1 g kg^{-1}, Betaproteobacteria abundance declined while Chloroflexi increased. No significant microbial changes were observed in soils with a pH of 6.21 and organic carbon levels of 18.8 g kg^{-1}. Additionally, Azeem *et al*. (2020) reported that biochar application led to an increase in bacterial abundance (16S rRNA), as well as Gram-positive and Gram-negative bacteria, in soils cultivated with mash beans but not in those planted with wheat. This finding underscores the role of crop species in shaping microbial populations in biochar-amended soils.

2.8.4 Crop yield and nutrient use efficiency

The effects of biochar application on crop yields vary widely, with outcomes ranging from positive to negative or negligible (Chan *et al*., 2007). These variations are likely influenced by differences in biochar types, soil properties, climatic conditions, and management practices (Schulz *et al*., 2013). In cases where positive effects were observed, maize yield increased by 98–150%, with water use efficiency improving by 91–139% following the application of manure-derived biochar (Uzoma *et al*., 2011). Similarly, wheat grain yield rose by 18% with the use of oil mallee biochar (Solaiman *et al*., 2010). Additionally, Zhang *et al*. (2017) reported that rice yields increased by 8.8–14% after biochar application. Likewise, the addition of wood-derived biochar enhanced wheat yield by up to 30% and maintained productivity for two consecutive seasons, even without further biochar application in the second year (Vaccari *et al*., 2011).

Also, earlier works by (Awodun *et al*., 2007) showed that EFB biochar application increased crop growth and yield and also enhanced phosphorus uptake by increasing pH of acidic soils. Abdulrazzaq *et al*. (2015) further reported that the applications of 15 and 30 t ha^{-1} EFB biochar increased the shoot dry weight of sweet corn by about three and six times, respectively, for both biochar rates. A meta-analysis of Jeffery *et al*. (2011) comprising of 177 individual studies revealed an average increase of crop productivity of around 10% as an effect of biochar amendments. Anwar *et al*. (2021) found that the application of biochar produced a higher number of okra fruit $plant^{-1}$ and yield compared to unamended control. Biochar application has been shown to enhance wheat yield and fertiliser use efficiency, particularly in dry land farming systems in Australia (Blackwell *et al*., 2010). Improved fertiliser efficiency has also been observed in biochar-amended soils with low fertility (Arif *et al*., 2017).

Kalu *et al*. (2022) found that applying spruce biochar increased plant nitrogen uptake, nitrogen use efficiency, crop biomass, and grain yield over two years compared to fertilized control plots. Similarly, Haider *et al.* (2020) reported that spruce biochar helped retain nitrate nitrogen (NO_3^--N), making it more available for plant uptake. In a pot experiment, Agegnehu *et al.* (2015)

observed that maize grown in soil treated with acacia biochar and synthetic fertiliser had higher nitrogen (N), phosphorus (P), and potassium (K) uptake than plants in soils receiving only fertiliser or left unamended. Numerous studies have demonstrated that improvements in plant growth and yield following biochar application are largely due to enhanced nutrient availability (Agegnehu *et al*., 2016a; Gaskin *et al.,* 2010). This is sometimes through direct nutrient additions via ash when biochar is applied to soils (Fischer & Glaser, 2012).

Another agronomic benefit of biochar is its ability to increase yield beyond fertiliser effect. Brassard *et al*. (2019) indicated that biochar often has high carbon content, pH, high porosity and surface area and a high potential to increase nutrient availability beyond a fertiliser effect. For example, Steiner *et al*. (2008) reported that biochar application to soil significantly increased plant growth and nutrition and improved the efficiency of N fertilisers. Again, effects of biochar on wheat and soybean were not significant in the absence of fertiliser, (Van Zwieten *et al*., 2010). Co-application of biochar and fertiliser increased N uptake in wheat grown in a Ferrosol, compared to the control and this was attributable to improved fertiliser use efficiency (Agegnehu *et al*., 2017). Yamato *et al*., (2006) recorded a 200% maize yield increase in soil amended with biochar prepared from the bark of Acacia mangium and synthetic fertiliser. They attributed the yield to increase in N and P availability, mycorrhizal fungi colonization and reduction of exchangeable Al^{3+}.

Despite the potential benefits of biochar, yield reductions have also been reported in some studies. Asai (2009) observed a decline in grain yields ranging from 10% to 23.3% following biochar application. Similarly, a field experiment by Bass *et al*. (2016) showed a significant decrease in banana bunch weight, dropping from 33 kg in the fertiliser-treated control to 27 kg in plots receiving both biochar and fertiliser. Additionally, the same study found that the proportion of unusable fruit increased in biochar-treated plots, and the number of deformed bananas per bunch was 50% higher compared to the fertiliser-only control. In another location, the combined application of biochar and fertiliser had no significant impact on papaya yield (Bass *et al*., 2016).

One possible explanation for the limited effectiveness of biochar is that pure biochar does not directly supply nutrients to the soil. Instead, its high carbon-to-nitrogen (C/N) ratio can lead to nitrogen immobilization, limiting nutrient availability for plants (Lehmann & Joseph, 2009). Hui (2021) also suggested that the presence of toxic or harmful substances in certain biochar types could negatively affect the soil environment, reducing nutrient uptake and inhibiting plant growth.

However, variations in crop yields and nutrient use efficiencies across different studies may be explained by differences in soil types, which might vary in terms of texture, bulk density, water retention capacity, nutrient content, pH, organic carbon content, and effective cation exchange capacity. For example, Omara *et al*. (2020) found that yield response to biochar and nitrogen application was more pronounced in sandy, low-yielding environments. A meta-analysis by Adu *et al*. (2022) found that the effect of empty fruit bunch (EFB) biochar on plant growth and yield varied depending on soil type. The study showed that crop growth and yield improved in EFB-amended soils compared to unamended soils, with the greatest increase observed in sand (0.566), followed by sandy loam (0.411), clay loam (0.384), loamy sand (0.207), sandy clay loam (0.118), and a slight decline in clay soils (−0.065). These findings suggest that plants grown in coarser-textured soils responded more positively to EFB application than those cultivated in finer-textured soils.

Finally, apart from differences in biochar and soil types, experimental duration also accounts for margin of effect of biochar application on crop yield. Major *et al*. (2010) reported that maize grain yield showed no significant increase in the first year after applying 20 t ha^{-1} of biochar. However, in the subsequent three years, yields improved by 28%, 30%, and 140% compared to the control. These findings suggest that biochar has long-term benefits for both soil fertility and crop productivity. Also, Frimpong *et al*. (2021) reported an increased okra N and P concentration from residual sole corncob biochar, compared to sole compost and unfertilized treatments in an Acrisol. Despite the extensive research that has gone into plant growth and yield increases due to soil biochar application, the studies are more skewed to cereals, grains, and legumes with few touching on vegetables warranting research concerns into those plant species.

2.8.5 Soil carbon sequestration

Soil organic matter serves as a vital reservoir of plant nutrients and plays a crucial role in maintaining and enhancing various soil properties (Krull *et al*., 2004). Tropical soils, which are often low in organic carbon and deficient in nutrients, can significantly benefit from biochar application (Adekiya *et al*., 2023). Biochar's stability in the environment is attributed to its condensed aromatic structures, making it a viable option for long-term carbon sequestration. The effectiveness of biochar in storing carbon has been demonstrated by the terra preta soils, which have retained high levels of organic matter for over 2,000 years (Glaser, 2007). This makes biochar particularly beneficial in tropical soils, as it can effectively sequester organic carbon, reducing its decomposition and limiting atmospheric losses (Adekiya *et al*., 2023). However, the extent to which biochar contributes to carbon sequestration depends on its stability in soil and its influence on the decomposition of native soil organic carbon, known as the priming effect (Wang *et al*., 2017).

Previous studies have shown that the mineralization of native soil organic carbon (SOC), which regulates carbon exchange between soil and the atmosphere, undergoes inevitable changes following biochar application (Yang *et al*., 2022). However the turnover of native soil organic carbon induced by biochar addition is affected by the extents to which biochar will alter physicochemical properties, microbial abundance and activities (Yang *et al*., 2022). Contrasting interactions of biochar with native SOC have been reported; these contrasting effects stem possibly from the differences in biochar characteristics, soil types, prevailing climatic conditions and production systems. Chen *et al*. (2021a) iterated that under specific conditions, the priming effect is more often driven by a combination of mechanisms between properties of biochar and soil (e.g., the composition and structure of organic carbon, pH and microbes).

Wang *et al*. (2017) reported a positive priming effect, where biochar application increased the mineralization rate of native soil organic carbon (SOC). A short-term incubation study by Maestrini *et al*. (2015) found that labile carbon fractions in biochar stimulated microbial activity, leading to enhanced SOC decomposition. However, Yu *et al*. (2020) observed a negative priming effect, while Nguyen *et al*. (2014) found no significant impact. Whitman *et al*. (2014) explained that biochar's labile carbon fraction initially causes a short-term positive priming effect. Over time, however, a negative priming effect dominates as microbes shift from decomposing native SOC to utilizing fresh organic inputs or alternative carbon sources. Negative priming effects associated with biochar are attributed to several mechanisms, including CO_2 adsorption onto biochar surfaces, reduced activity of carbon-mineralizing enzymes, increased microbial carbon use efficiency, and nitrogen immobilization (Lehmann *et al*., 2011; Agyarko-Mintah *et al*., 2017).

Meanwhile, previous works showed that the responses of soil properties including SOC to biochar addition would change with time (Yang *et al*., 2022). For example, in a field experiment by Lu *et al*. (2022), biochar produced at 300 °C was reported to induce a positive-negative-positive priming effect on the native SOC turnover during the entire 182 days, whereas biochar produced at 500 °C had a negative-positive-negative priming effect. Consequently, for accurate assessment of the effects of biochar on native SOC mineralization, long term investigations of the priming effect induced by both fresh and aged biochar under variable climatic conditions, soil types, crop specie and production systems are urgently warranted.

The addition of biochar to agricultural soils is regarded as an effective strategy for carbon sequestration due to its high carbon content, chemical stability, and resistance to decomposition (Kavitha *et al*., 2018). Zhao *et al*. (2019) reported a notable increase in soil organic carbon and humic fractions after incorporating biochar pyrolyzed at 300 and 500°C into Earth-cumuli-

Orthic Anthrosol. Similarly, Rahman *et al.* (2022) found that biochar produced from sugarcane bagasse increased soil organic carbon content from 42% to 58%.

Yang *et al*. (2018) reported that biochar enhanced carbon sequestration rates three years after its application to Hapli-Udic Cambisol. In a 12-year long-term field study, Xu *et al*. (2022) found that soil total carbon increased with biochar application, with the effect varying by soil depth. The researchers explained that after biochar is added to the soil, only a small portion is broken down by microorganisms, while the majority remains, directly contributing to increased total soil carbon. Additionally, Farooqi *et al*. (2024) linked the rise in soil organic carbon storage due to biochar application to the formation of organic soil colloids, which promoted organo-mineral complexes and aggregate formation, helping to protect soil carbon.

Soil labile organic carbon, a key component of total soil organic carbon (SOC), is highly responsive to soil management practices and serves as an important indicator of soil quality and productivity (Morrissey *et al*., 2014). Amoakwah *et al*. (2021) found that applying corn cob biochar significantly increased labile carbon content in soils, with the most pronounced effect observed at a biochar application rate of 30 t ha^{-1}. Similarly, Zhang *et al*. (2017) reported a significant rise in the labile carbon pool after incorporating straw biochar, pyrolyzed at 350–550°C, into silty clay loam at a rate of 16 t ha^{-1}. The carbon management index (CMI) is used to assess changes in soil carbon dynamics relative to a stable reference soil and it is recognised as an indicator of soil carbon restoration. The CMI accounts for both labile and total organic carbon in its calculation (Blair *et al*., 1995). Amoakwah *et al.* (2021)

observed an increase in CMI in Haplic Acrisol following the application of 30 t ha^{-1} of corn cob biochar. Likewise, Qiu *et al*. (2023) reported that applying high amounts of peanut shell biochar improved the CMI of sandy loam over time. Overall, findings from the reviewed studies suggest that biochar plays a significant role in enhancing stable SOC, making it a valuable tool for mitigating climate change in tropical soils.

2.8.6 Greenhouse gas emissions

Currently, literature suggests that biochar may play a role in reducing of greenhouse gas (GHG) emissions; biochar can affect GHG emissions directly following its application to soils, and indirectly by adding carbonized instead of non-carbonised residue or manures which usually have higher emissions following application (Kammann *et al*., 2017). Extensive studies have been conducted to establish the role of biochar in regulating soil carbon dioxide emissions, yet the results are diverse and quite complicated (Wang *et al*., 2023). Research on the impact of biochar amendment on soil CO_2 emissions has yielded mixed results, with studies reporting increases (Johnson *et al*., 2017), decreases (Gascó *et al*., 2016), or no significant effect (Zhou *et al*., 2017). He *et al*. (2016) and Song *et al*. (2016) observed significant increases in CO_2 emissions by 19% and 22%, respectively, while Liu *et al*. (2016) found no measurable impact. These variations are attributed to factors such as study design, duration, soil texture, disturbance intensity, soil pH, biochar characteristics, and application rate (Bruun *et al*., 2014). In the long term, biochar may influence soil CO_2 emissions by modifying microbial communities through changes in habitat structure and by chemically or physically protecting native soil organic matter (Song *et al*., 2016).

The impact of biochar on soil CO_2 emissions can be attributed to several processes. These include microbial utilization of biochar, its decomposition, and the release of biochar-derived carbon, such as carbonates (Bruuns *et al*., 2008). Additionally, interactions between biochar and native soil organic matter may stimulate the mineralization of existing soil carbon, known as the priming effect (Zimmerman *et al*., 2011). The direct introduction of labile carbon from biochar can also contribute to increased CO_2 emissions (Zhang *et al*., 2020).

Changes in soil carbon and nitrogen availability following biochar application are another factor influencing greenhouse gas emissions. Li *et al*. (2013) suggested that reduced CO_2 emissions may be due to biochar adsorbing dissolved organic carbon, thereby limiting its availability as a microbial substrate. However, other studies have shown that biochar can increase labile carbon content and enhance the availability of mineral nitrogen, leading to greater CO_2 emissions (Troy *et al*., 2013). Soil pH is a key chemical parameter regulating greenhouse gas emissions after biochar application. The alkalinity of biochar raises soil pH, which can lead to the sequestration of CO_2 in soil solution as carbonates, thereby reducing emissions (Li *et al*., 2013). Maucieri *et al*. (2017) also confirmed a negative correlation between soil pH and CO_2 emissions.

Furthermore, biochar influences microbial activity by altering the composition of soil microbial communities. It has been found to increase the ratio of Gram-positive to Gram-negative bacteria, which may enhance CO_2 fluxes through organic carbon decomposition Khan *et al*. (2022). Lower nitrogen availability has also been linked to reduced microbial activity, subsequently decreasing CO_2 emissions (Maucieri *et al.*, 2017). Additionally, Khan *et al*. (2022) identified a strong relationship between soil CO_2 emissions and the activity of β-glucosidase and dehydrogenase enzymes, further emphasizing the role of microbial processes in biochar-amended soils.

Biochar influences soil greenhouse gas (GHG) emissions through its impact on soil physical properties, including improved aeration, increased water-holding capacity, and enhanced adsorption potential. Troy *et al*. (2013) observed that biochar application led to higher CO_2 emissions due to improved soil aeration, which accelerated the mineralization of soil organic matter. Additionally, the high porosity and large surface area of biochar contribute to the direct adsorption of GHGs (Li *et al*., 2013). Khan *et al*. (2022) also reported that biochar physically adsorbs CO_2, thereby reducing its release into the atmosphere.

The net exchange of methane (CH_4) between soil and the atmosphere is governed by two key biological processes: methane production by anaerobic methanogenic archaea (methanogens) and methane consumption by methanotrophic bacteria (methanotrophs) (Fu *et al*., 2023). Methanogens obtain energy from hydrogen (H_2) and carbon dioxide (CO_2) or from acetate, formate, methanol, and other methylated compounds (Brasseur & Chatfield, 1991). In general, CH_4 production by soil microbes occurs under anaerobic conditions (Yang *et al*., 2022). Increased soil aeration and oxygen availability can suppress CH_4 emissions by limiting anaerobic conditions (Li *et al*., 2013).

Methanotrophic bacteria, particularly α- and γ-proteobacteria, are responsible for CH_4 oxidation, using methane as their sole carbon source while requiring oxygen for activity (Conrad, 2007). A meta-analysis by Fu *et al.* (2023) found that biochar application reduced CH_4 emissions, attributing this effect to an increased abundance of methanotrophic bacteria (pmoA) and a decline in methanogenic archaea (mcrA). This response is linked to the sensitivity of methanogenic microorganisms to elevated soil pH (Muñoz *et al.*, 2019). Additionally, biochar enhances soil aeration, thereby improving redox potential and increasing CH_4 oxidation rates (Chen *et al.*, 2018).

Earlier studies show that responses of CH_4 emissions to biochar depend on the soil type, biochar properties, and the agricultural practices including fertilization and water management. For example, Rondon *et al.* (2006) found suppressions of CH_4 emissions from soils treated with biochar in a grass stand and soybean grown on a tropical soil. Qin *et al.* (2016) also reported that biochar amendment significantly decreased CH_4. The reductions in CH_4 emission is primarily ascribed to an increase in soil pH, enhanced adsorption by soil particles, and the stimulated biodiversity and abundance of methanotrophic (Feng *et al.*, 2012).

Some studies have reported either increased CH_4 emissions or no significant impact following biochar application. Jia *et al.* (2012) found no effect on CH_4 emissions in a vegetable production system. However, Zhang *et al.* (2012) reported that total soil CH_4 emissions increased by 34% with nitrogen fertilization and by 41% without it. Lehmann *et al.* (2011) attributed the rise in CH_4 emissions to the presence of labile organic substrates in biochar, which created favourable conditions for microbial activity. Additionally, biochar can influence greenhouse gas emissions by altering soil microbial communities and enzymatic activity (Maucieri *et al.*, 2017).

Changes in soil pH also play a role in CH_4 emissions. Li *et al*. (2013) suggested that increased soil alkalinity due to biochar amendment could enhance CH_4 production.

In addition to changes in soil properties, Wang *et al*. (2012) observed increased CH_4 emissions in soils where rice and wheat were cultivated. The study suggested that biochar-enhanced rice growth may have contributed to the higher CH_4 emissions. However, the specific relationship between plant biomass and greenhouse gas emissions remains unclear and requires further investigation. On the contrary, Schimmelpfennig *et al*. (2014) reported that Miscanthus straw biochar reduced CH_4 emissions from the soil with Lolium perenne. Moreover, biochar from bamboo leaves and corn residue could also inhibit CH_4 productions and emissions in the soil with the growth of Chinese chestnut and corn plants (Yang *et al*., 2020). This is attributed to the biochar-induced soil aeration and promotion of CH_4 oxidation. These literature findings suggest that crop species also contribute to the emission of methane in biochar amended soils.

Biochar application significantly influences soil N_2O emissions, though the extent of its impact varies across studies (Case, 2014). Some research has found no significant difference in N_2O emissions between biochar-treated and control soils while others have reported increased emissions following biochar application (Troy *et al*., 2013). Field trials frequently show no statistically significant differences in N_2O emissions between biochar-amended and untreated soils (Jones *et al*., 2012). One possible explanation for these inconsistent findings is that factors such as biochar application rate, uneven particle distribution, and variability in soil and plant conditions contribute to high fluctuations in N_2O fluxes, reducing the likelihood of detecting a clear effect (Kammann *et al*., 2017).

Case (2014) enunciated five mechanisms by which biochar application affect N_2O emission: Increased water holding capacity and decreased bulk density of the soil; increasing soil aeration (Karhu *et al*., 2011) can reduce the activity of denitrifying micro-organisms. Secondly, immobilisation of soil inorganic N via adsorption to biochar surface or increased microbial immobilisation (Spokas *et al*., 2012) will lead to reduction of N substrate for nitrifying and denitrifying enzymes and eventually reduce enzymatic activity. Thirdly, increased soil pH (Van Zwieten *et al*., 2010) thereby decreasing N_2O: N_2 emission ratio produced during denitrification. Last but not least, increased labile C added to the soil (Bruun *et al*., 2011) that also reduces N_2O: N_2 product ratio of denitrification. Lastly, substances that may inhibit microbial activity, are emitted by the biochar, such as ethylene, α-pinene, PAHs, VOCs (Clough *et al*., 2010) inhibiting the activity of soil nitrifying/denitrifying organisms.

Obia *et al*. (2015) investigated NO, N_2O, and N_2 emissions under strictly anaerobic conditions using high-resolution gas kinetics and found that emissions increased with cacao shell biochar but not with rice husk biochar in acidic soil. Additionally, Zhang *et al*. (2020) suggested that a reduction in total denitrification following biochar application could be attributed to improved soil aeration. Biochar has also been shown to lower the N_2O/N_2 ratio due to its alkalizing effect on soil (Obia *et al*., 2015). Furthermore, biochar can influence nitrification rates by adsorbing phenolic compounds, which may inhibit or reduce nitrification (Kammann, 2017). In another study, Wells and

Baggs (2014) found that biochar primarily affected ammonia oxidation rather than N_2O reduction or production by denitrifiers, leading to a 27% increase in N_2O emissions. Additionally, Rohe *et al*. (2014) suggested that biochar plays a significant role in N_2O production by influencing fungal co-denitrification or fungal denitrification processes.

The long-term persistence of N_2O emission reductions following biochar application remains an area of on-going research. It is still unclear whether aged biochar-rich soils, which undergo changes in their physical and chemical properties, will have a greater or lesser potential for mitigating N_2O emissions compared to soils without biochar (Kammann *et al*., 2017). Some studies have reported sustained reductions in N_2O emissions with biochar over multiple seasons. Hagemann *et al*. (2016) found that biochar continued to significantly suppress N_2O emissions in the third season compared to a control site without biochar. Similarly, Keith *et al*. (2016) observed that fresh biochar suppressed N_2O emissions in laboratory conditions and maintained this reduction in a sunflower cropping system for three years after field application.

Fidel *et al*. (2019) suggested that the long-term suppression of N_2O emissions in biochar-amended soils is linked to changes in microbial communities due to shifts in physical habitat, the protection of organic carbon and nitrogen through physical and chemical interactions with biochar, and modifications in micro-scale soil redox conditions influenced by the electrochemical properties of biochar. However, longer-term field studies, such as that conducted by Spokas (2012), indicated that soil N_2O emissions were not consistently reduced with biochar application up to three years post-application. Overall, the ability of biochar to mitigate N_2O emissions depends on multiple factors, including biochar characteristics, soil type, environmental conditions, and study duration. These factors must be carefully considered when developing and implementing biochar-based strategies for N_2O mitigation.

2.9 Co-application of biochar and compost as promising alternative for restoring tropical soil health

In many tropical areas with good annual rainfall, soils are mostly acidic mainly because of intensive leaching, continuous cropping and use of fertilisers like ammonium sulphate (Ssali *et al*., 1986). In addition, nitrogen and phosphorus are generally the most limiting nutrients in these soils. In many cases where the supply of nitrogen is adequate, phosphorus is found to be the most limiting nutrient due to intense weathering that has decreased soils total phosphorus coupled with phosphorus sorption and thus render it less available for plant uptake (Owusu *et al*., 2024). Moreover, Owusu *et al*. (2024) reported that the magnitude of phosphorus fixation is generally related to the soil pH, the iron and aluminium oxides in the soil, and exchangeable aluminium which is typical of highly acid and weathered clayey soils. Again, many soils of the tropics have pH-dependent charges that preferentially hold monovalent cations; polyvalent cations like calcium and magnesium are easily lost through leaching in areas of high rainfall (Ssali *et al*., 1986). According to Nweze *et al*. (2020), the general nutrient poverty of tropical soils is attributable to the more intensive weathering due to high temperature and moisture together with extensive agricultural practices, imbalanced nutrient management, overgrazing, burning of crop residues, and other land use changes. Such a condition demands a feasible multi-purpose intervention strategy, though studies have investigated several approaches to restore SOC in tropical soils, including the use of organic amendments, agroforestry, conservation agriculture, and reforestation, crop rotation among others (Adekiya *et al*., 2023). However, the success of these approaches depends on various factors, including soil characteristics, climate conditions, land management practices, and their interactions (Adekiya *et al*., 2023).

The application of manure or compost is a traditional and common practice that has been in existence for quite a while for improving soil fertility and yields in low-input farming systems. However, in tropical conditions with high radiation loads and soil temperatures, the benefits of compost to the soil tend to dissipate within one to two years of application (Fischer & Glaser, 2012). This is so because composts are rapidly mineralized after soil application within months or a few years (Schulz *et al*., 2013) and the nutrients released may be subjected to leaching or gaseous losses. Also for soil carbon sequestration, because of the readily degradable and high labile carbon it contains, compost effects diminishes due to rapid decomposition by micro-organisms (Figueiredo *et al*., 2002). This necessitates annual applications in tropical soils to achieve and maintain these benefits (Dick *et al*., 1993) implying additional labour and increased costs for smallholder resource-poor farmers. These limitations associated with the exclusive use of compost need to be mitigated by combining it with a more stable carbon source.

In recent times, biochar has been identified as an alternative/ complimentary amendment to compost in agricultural soils (Wang *et al*., 2023). Interest in biochar has rapidly increased over the last decade due to the potential benefits it offers to carbon sequestration (Lehmann & Joseph, 2009) and improvement in soil properties from physical, chemical and biological aspects (Spokas *et al*., 2012). Incorporation of biochar into soil increases soil water retention, soil aeration, nutrient retention and availability, pH and soil carbon content, and hence may improve soil fertility and productivity and reduce greenhouse emissions (Biederman & Harpole, 2013; Cayuela *et al*., 2015). Despite the potential benefits of biochar, its sole application to soil for enhancing fertility and crop yield has limitations, particularly with plant-based biochar, which may be deficient in nitrogen and phosphorus and it is also resistant to degradation (Adekiya *et al*., 2020; Phares *et al*., 2020). Combined biochar and compost application can compensate for the shortcomings of each other such that their interactive effect is likely to improve soil quality and crop yield.

Liu *et al*. (2012) demonstrated that applying biochar and compost together in the field provided synergistic benefits, enhancing soil organic matter (SOM) content, nutrient availability, and water-holding capacity. Similarly, Abudjabha *et al*. (2016) reported an increase in soil microbial abundance due to improved macropore formation and bioturbation following the combined application of biochar and compost. Duarte *et al*. (2022) found that co-applying sewage sludge compost with beech wood biochar significantly improved the water-holding capacity of sandy soil. Likewise, Frimpong *et al*. (2021) observed that amending Acrisol soil with a combination of corn cob biochar and maize straw compost over three cropping cycles led to improvements in soil pH, cation exchange capacity (CEC), total organic carbon (TOC), nitrogen (N), and potassium (K). However, Iacomino *et* *al*. (2022) reported that a single application of olive mill waste compost (20 t ha^{-1}) and beech wood biochar (30 t ha^{-1}) did not produce significant changes in soil pH within the first year of application.

For crop yield and nutrient uptake, Iacomino *et al*. (2022) reported that a mixture of compost from olive mill wastes and orchard pruning residues and biochar from beech wood proved to be the best treatment for the growth of Fennel and Rape with an increase of 39% and 100%, respectively. Kheir *et al*. (2023), observed a yield increase of 13.6% of faba bean seed after combined corn waste compost and biochar application, compared to their single applications. Frimpong *et al*. (2021) also reported an increased okra N and P concentration from biochar plus maize straw compost treatments, compared to sole compost and unfertilized control in a field experiment. Agegnehu et al. (2015) observed increased soil water and nutrient retention as well as water and nutrients uptake by the plants from the combined application of compost and biochar, though they found little or no synergistic effect. Kuzyakov *et al*. (2009) related the result to increased stabilization of the compost and better surface oxidation of the biochar to enhance the capacity of biochar to chemisorb nutrients and minerals gradual release and leaching losses.

Though studies on greenhouse gas emissions, especially in tropical systems are still rare, some of the few studies on GHG emissions were reported by Gao *et al*. (2023) who showed that biochar co-compost significantly decreased CO_2 emissions but had an insignificant impact on N_2O and CH_4 emissions over one growing season of winter wheat. Additionally, a meta-analysis by Fu *et al*. (2023) found that the co-application of biochar and organic amendments increased soil greenhouse gas emissions, with an antagonistic effect on CO_2 but an additive effect on N_2O and CH_4 emissions. Bass *et al*. (2016) also reported that soil CO_2 emission was increased with combined compost biochar treatments, but had no effect on N_2O emissions. Since soils are varied so do climatic conditions, management practices, plant species and organic amendments, therefore it very difficult to generalise conclusion from findings. Combined biochar and compost application still seem to be a promising alternative for improvement of tropical soil quality, increased agronomic output, carbon sequestration and greenhouse emission mitigation, although more research into the optimal application rate and sustenance duration on crop yield, carbon sequestration and greenhouse mitigation in tropical soils is warranted.

2.10 Availability of Empty Oil Palm Fruit Bunch as a Source of Feedstock for Biochar and Compost Production

Although all biomass containing carbon can be used as organic amendments such as compost and biochar, it is suggested that feedstock must be financially and environmentally viable (Hadi & Norazlina, 2021). In Ghana, there is an abundance of waste materials from palm oil mills that can be converted into biochar or compost such as empty fruit bunch (EFB). Oil palm (*Elaeis guineensis* Jacq.) is largely produced in Asia, Africa, and Latin America, but its origin is traced to the tropical rain forest and equatorial region of Africa (Awere *et al*., 2022). Palm oil is the most widely consumed vegetable oil globally. In 2012 and 2013, it accounted for 39% of total vegetable oil consumption. With global demand for vegetable oil projected to rise to between 201 and 340 million tonnes by 2050, the significance of palm oil in meeting this demand is expected to grow (Corley, 2009). In Ghana, oil palm is the second most important economic crop after cocoa (Angelucci, 2013; Ofosu-Budu & Sarpong, 2013). The total area currently under oil palm in Ghana is estimated at 330,000 ha and this is about 5% of the land area suitable and available to grow oil palm (Rhebergen, 2019). About 40,000 ha (~12%) of this is managed in smallholder and out-grower schemes, about 140,000 ha (~42%) is cultivated by independent medium- and small-scale farmers, and the remaining 150,000 ha (~45%) is found in wild oil palm groves (Ofosu-Budu & Sarpong, 2013). The palm oil milling industry is a major source of income and employment to many women in the rural areas of the forest agro-ecological zone which had employed over two million people as at 2015. As at 2021, the country's crude palm oil production had increased to 375,000 tonnes, doubling the production in a decade (Awere *et al*., 2022).

The palm oil industry produces large quantities of waste and residues, which can be classified into two main categories: those generated in plantation fields and those resulting from the milling process (Teh, 2016). Plantation-based biomass includes oil palm fronds (OPF) and trunks (OPT), which are produced through routine pruning of mature trees and the replanting of older trees (Teh, 2016). Meanwhile, the milling process generates approximately 0.21 tonne of empty fruit bunches (EFB), 0.15 tonne of mesocarp fiber, 0.6 tonnes of kernel shells, 0.2 cubic meters of palm oil mill effluent (POME), and between 0.6 and 1.2 cubic meters of wastewater per tonne of fresh fruit bunches processed (Hambali *et al*., 2010).

Empty fruit bunches (EFB) constitute the waste generated during the sterilisation and stripping of the fresh fruit bunches during the milling process, accounting for larger quantities of waste as compared to POME, fibre, and shells (Cheah *et al*., 2023). Yearly, about 99 million metric tonnes of EFB are produced globally (Geng, 2013). Averagely, approximately 397,000 tonnes of EFB is produced annually in Ghana (Adu *et al*., 2022). Oil palm plantations in Ghana produce approximately 390 tonnes of EFB daily (Adu *et al*., 2022). The high moisture content of EFBs prevents their immediate use as solid fuel. Some small-scale mills also dry and use EFBs and palm press fibres as solid fuel for boiling the palm fruits or incinerate it to dispose of them (Osei-Amponsah *et al*., 2012). So far, the small-scale mills in Ghana use the EFB as a source of fuel. The large-scale mills in Ghana use EFB as soil mulching material (Osei-Amponsah *et al*., 2012). A notable disadvantage of EFB has always been its large physical size, making it costly to store and to transport back to the plantation fields. EFB is also more difficult to apply in the fields (Teh, 2016). Heaps of EFB in the plantations pose a danger of pest outbreak (Schuchard *et al*., 2008). EFB can be utilized as biomass source for the production of biochar or compost by small-holder rural farmers to improve their production (Adu *et al*., 2022).

The chemical composition of the EFB fibre consists mainly of lignin, cellulose, and hemicellulose. It contains approximately 64.17%, moisture 48.6% C, 0.87% N, 0.05% P, 1.3% K, 0.2% Mg, 1.2% Ca and 29.45% lignin on dry weight basis (Teh, 2016). Adu *et al*. (2022) and Lim *et al*., (2015) inferred that EFB may be a cheap and easily accessible organic fertiliser for restoring soil degradation, especially for small-holder farmers because it can improve soil organic carbon and other soils' physical and chemical properties. Using EFB as fertiliser can be highly advantageous because they contain many essential nutrients plants need for growth. To avoid abundant of EFB, the best way to dispose of EFB is to compost or pyrolysed EFB and this will help to reduce the bulkiness, and concentrate nutrients before soil application (Supriatna *et al*., 2022).

2.11 Composting of Empty Oil Palm Fruit Bunch and Effect on Soil Health

The use of raw empty fruit bunch as fertiliser comes with notable disadvantages of high cost of transportation, storage and field application because of the large physical size, bulkiness and high moisture content (Teh, 2016). Composting empty fruit bunches (EFB) can reduce their volume by up to 85% from the initial amount, making field application more efficient and cost-effective (Salètes *et al*., 2004). Composting EFB also concentrate the nutrients and lowers the C/N ratio that increases the rate of biomass decomposition and nutrients release in the field (Redshaw, 2003). Again, Compost production from EFB can reduce its bulkiness and volume and also concentrate mineralized nutrients and eliminate odour (Kuo *et al*., 2004; Schuchard *et al*., 2008). Composting empty fruit bunches (EFB) is considered a sustainable practice, as it helps protect the environment while also providing economic benefits by reducing the reliance on chemical fertilisers that contribute to land degradation (Saputra et al., 2022). To address the high initial carbon-to-nitrogen (C/N) ratio of EFB, which can hinder the composting process, co-composting with palm oil mill liquid waste, decanter cake, and animal manure such as chicken, cow, and goat dung is recommended (Siddiquee *et al*., 2017; Yahya *et al*., 2010). For instance, composting EFB with palm oil mill effluent (POME) at a typical 1:3 ratio (EFB:POME) by weight has been carried out in open windrows measuring 3 m in width and 2 m in height, with regular turning to improve aeration (Redshaw, 2003). The duration required for complete composting varies between 10 and 22 weeks, depending on specific composting conditions. These include achieving an initial C/N ratio between 20 and 40, maintaining moisture content of 45–65%, process temperatures between 43–65°C, an oxygen level of 5%, and ensuring a particle size of less than 50 mm (Lord *et al*., 2002).

Empty fruit bunch compost has high potential to be used as an organic fertiliser because it can improve soil organic matter content, restore the structure and soil aeration, and provide sufficient nutrients for growth (Vakili *et al*., 2015). For example, Tohiruddin and Foster (2013) composted EFB with POME (1:3 ratio by weight), after soil application, they observed that 10 to 20 t ha^{-1} yr^{-1} for three years increased oil palm yields by 16 to 21%, increased soil K and Mg by 133 to 150%, and increased leaf N, P, and Mg levels by 2 to 9%. Also, a meta-analysis by Adu *et al*. (2022) showed that EFB applied as compost increased growth and yields of crops by 30.9% compared with crops grown on the unamended soils. Caliman *et al*. (2001) found that applying empty fruit bunch (EFB) compost to soil led to an increase in soil pH within 60 days. Krishnan *et al*. (2017) reported that co-composting EFB with palm oil mill effluent or other additives could reduce greenhouse gas (GHG) emissions by up to 76%. This reduction is attributed to the decreased methane emissions from uncontrolled EFB dumping and the replacement of synthetic fertilisers with organic soil amendments. Zaharah and Lim (2000) suggested that since EFB is composed of more than 50% carbon, its use as compost or biochar could contribute to GHG mitigation through soil carbon sequestration.

Utilizing the benefits of EFB compost in small-scale agricultural systems in Ghana could enhance soil fertility and support sustainable crop production.

2.12 Pyrolysis of Empty Oil Palm Fruit Bunch and Effect on Tropical Soil Health

Empty oil palm fruit bunch can also be converted into biochar to be used as a soil conditioner in agricultural soils due to its high carbon content, pH, stability, porosity, and surface area (Teh, 2016). EFB biochar is produced by combusting under temperatures above 300 °C in the absence or under low levels of oxygen. The material being pyrolysed undergoes chemical transformation into refractory molecular structures, making the end product, the biochar, extremely inert (Teh, 2016). Biochar production from EFB can be chemically described by water elimination followed by increasing aromatic condensation associated with decreasing atomic ratios of O/C and H/C along the combustion continuum (Fischer & Glaser, 2012). Schimmelpfennig and Glaser (2012) suggested that for pyrolysed EFB to be considered as biochar, it has elemental ratio thresholds of $O/C < 0.4$ and $H/C < 0.6$. Therefore, EFB biochar can be an option for long-term C sequestration and maintaining soil fertility under tropical agriculture. However, differences in the pyrolysis condition being it temperature, residence time, heating rate can affect properties of biochar. Claoston *et al*. (2014) investigated the pyrolysis of EFB at three different temperatures 350°C, 500°C, and 650°C and found that higher temperatures resulted in lower biochar yields, reduced cation exchange capacity (CEC), and decreased carbon (C) and nitrogen (N) concentrations in the biochar. Several studies have emphasized that pyrolysis temperature plays a crucial role in determining the physicochemical properties of EFB-derived biochar (Adu *et al*., 2022). The production of EFB biochar has the potential to be carbon-negative, making it a sustainable option. Additionally, EFB biochar serves as an effective soil amendment, particularly in addressing soil toxicity and acidity issues (Teh, 2016).

Several studies in Asia have shown that biochar application ameliorates low soil nutrient status, and cation exchange capacity while increasing pH, which often leads to higher nutrient holding capacity and plant nutrient use efficiency due to its liming effects in acidic soils (Adekiya *et al*., 2020). For example, studies by Rabileh *et al*. (2015) and Rosenani *et al*. (2015) showed that EFB biochar is an effective soil amendment, particularly for soils with toxicity or low pH problems, which is often a key problem for highly weathered tropical soils. EFB biochar can improve crop growth and yield by enhancing soil physical and chemical properties, suppressing weeds and controlling erosion (Chiew & Rahman, 2002). Also, earlier works by (Awodun *et al*., 2007) showed that EFB biochar application increased crop growth and yield and also enhanced phosphorus uptake by increasing pH of acidic soils. Abdulrazzaq *et al*. (2015) reported that applying empty fruit bunch (EFB) biochar at rates of 15 and 30 t ha^{-1} increased the shoot dry weight of sweet corn by approximately three and six times, respectively. The improvements in soil pH and reductions in aluminum (Al) and iron (Fe) content were identified as key factors contributing to enhanced sweet corn growth in biochar-amended soil.

In a pot experiment on acid sulfate soil, Rosenani *et al*. (2015) found that adding EFB biochar at 10, 20, and 40 t ha^{-1} increased wet rice yield by 141–472% while raising soil pH from 3.5 (control) to 6.0 at the highest biochar application rate. The study concluded that EFB biochar effectively mitigated Al^{3+} toxicity by lowering its concentration in floodwater. Similarly, Rabileh *et al*. (2015) observed that applying up to 20 t ha^{-1} of EFB biochar increased soil pH, reduced Al^{3+} toxicity, and improved maize growth parameters, including plant height, total root length, and total dry weight, in Ultisol soil under glasshouse conditions. A meta-analysis by Adu *et al*. (2022) further confirmed that EFB biochar application led to an approximately 78.4% increase in crop growth and yield compared to unamended soils. As a known fact, there is accelerated decline in organic matter, extent of degradation of tropical soils, the climate under which they are found, and the adverse impact of climate change, converting EFB into biochar is one promising option to sequester soil carbon and mitigate the effect of climate change on tropical soil health, but its benefits yet to be exploited in Ghanaian soils and smallholder farming settings.

2.13 Research Gaps in the Use of Empty Oil Palm Fruit Bunch as Biochar or Compost in Ghana's Crop Production System

Most soils in the tropics are very old and are inherently infertile with low soil organic matter and nutrient contents (Zingore *et al*., 2015). Another constraint to crop productivity in these soils is soil acidification and aluminium toxicity (Zingore *et al*., 2015). For example, Acrisols and Ferralsols which are few of the dominant soils in Ghana are highly weathered soils characterized by low native fertility, aluminium toxicity, strong phosphorus sorption and acidity problems (Owusu *et al*., 2024). The decline in soil fertility is further exacerbated by nutrient mining, inappropriate land use management, and use of conflicting resources and limited use of external inputs in both organic and inorganic forms (Bello *et al*., 2021). In such soils, adoption of multipurpose management option that increases soil organic matter, optimizes pH and at the same time improves water, nutrients retention and biodiversity is needed to resolve this imminent problem of soil chemical infertility (Krull *et al*., 2004). The application of soil amendments has been proposed as a soil management intervention to rejuvenate degraded soils to ensure agricultural productivity in these soils (Amoakwah *et al*., 2021). However, the sole use of manures or composts has limitation due to their accelerated decomposition that stabilizes only a small fraction of organic carbon overtime when applied to the soil (Agegnehu *et al*., 2017). Glaser *et al*. (2001) revealed that biochar application has a potential to reverse degradation of these soils by increasing organic carbon, levels of plant available nutrients and correcting of acidity, but the requirement of large number of organic resources for the production of biochar can pose a challenge for its use especially, in Ghana.

Regardless of the numerous benefits EFB tends to offer due to its availability, easy accessibility, cheapness and its convertibility to various forms, the efficacy of EFB as compost and biochar on soil quality, carbon sequestration, crop productivity and greenhouse gas emissions mitigation has not been fully exploited in Ghana. Biochar and composts intended for soil application must fulfil certain property and composition-related requirements, to prevent harm to the ecosystem (EBC, 2022). Therefore, several guidelines with suggested threshold values for contaminants (including PAHs, heavy metals, PCDDs, or PCDFs) in biochar have been established, which is similar to composts and fertilisers and the importance of contaminant analysis has been emphasised (IBI, 2015; EBC, 2022). Limited information exists regarding the impact of different conversion methods and conditions on the properties of biochar and compost produced from EFB in Ghana. Understanding how the conversion process affects the reduction of contaminants is crucial.

Also the use of empty fruit bunch (EFB) biochar and compost for small-scale crop production in Ghana is relatively new. Consequently, to optimize the use of EFB biochar in crop production, a better knowledge of the dynamics of its mineralisation and the release of mineral nutrients and toxic compounds on crop growth and yield depending on soil and environmental conditions and the rate of application are critical. This knowledge is vital for scaling up EFB biochar application as a soil conditioner. Again, to date, little exploration has been done single or combined EFB biochar and compost application on soil quality, agronomic output, carbon sequestration and greenhouse emission despite the potential effect it can offer to Ghana's agricultural systems.

2.14 Conclusion

The review revealed several promising results from certain studies regarding biochar and compost's effect on crops and soil. However, there are risks and a high degree of uncertainty surrounding biochar's effects on crops and soil under practical farming conditions. Also, biochar or compost properties depend on the feedstock and the pyrolysis or composting conditions, resulting in a huge diversity of biochar and compost types and their influence after soil application. Moreover, the quality of these amendments needs to be ensured because they could contain heavy metals and polycyclic aromatic hydrocarbons that can be introduced to the soil environment when applied to the soil.

Furthermore, the review showed that little exploration has been done on the co-application of biochar and compost on crop production and soil health, despite the majority of the studies giving positive and promising feedback. Meanwhile, intensive research will be required in the production and use of EFB biochar and compost as soil amendments to sustain tropical smallholder agricultural production, ensuring their quality, guiding their application rates and evaluating their impact on crop yields, resource use efficiency, carbon sequestration and greenhouse gas mitigation.

CHAPTER THREE

MATERIALS AND METHODS

In all, the research was in three main phases: EFB product development phase; product testing phase; and field application phase, as illustrated in Figure 1. Therefore, the materials and methods employed and the objectives set were specific to the different phases. However, some protocols such as preliminary soil sampling and analysis are common to the various phases of the project.

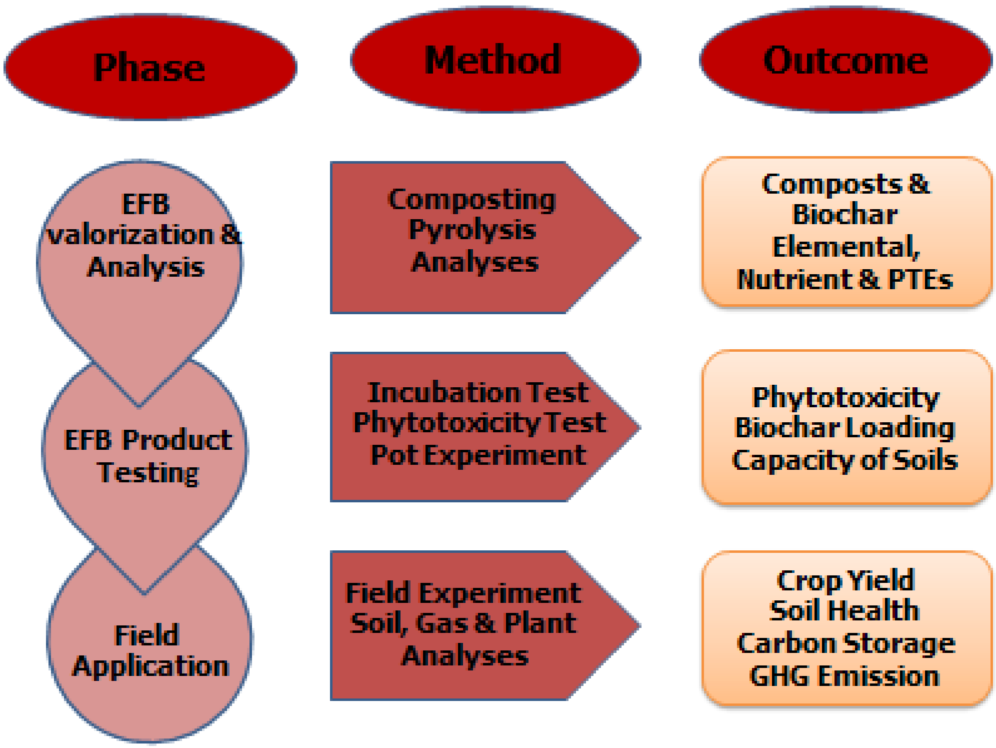

Figure 1: The empty oil palm fruit bunch project framework.

3.1 Study Area

Compost production, laboratory incubation studies, pot and field experiments were conducted at the A. G. Carson Technology Village (5° 7′ 48.9″ N, 01° 17′ 21.27 W) within the School of Agriculture, University of

Cape Coast, Ghana, from September 2020 to May 2023. The area was characterized by a bimodal rainfall pattern, comprising two rainy seasons and a dry season, with an average annual rainfall of 1400 mm. The mean monthly temperature ranged from 24^0 to 28^0, with a maximum temperature of 31^0. The vegetation is made up of grass, shrubs and few trees. The soil in the study area was classified as Haplic Acrisol (IUSS Working Group WRB, 2015) with a sandy clay loam texture, comprising 58% sand, 17% silt, and 25% clay. Key soil characteristics that were determined included a pH of 5.5, organic matter content of 1.7%, effective cation exchange capacity of 6.87 cmol $_{(+)}$ kg^{-1}, and base saturation of 31% (Blankson *et al.,* 2025). The soil's total nitrogen content was 0.62 g kg^{-1} and available phosphorus was 4.15 mg kg^{-1}. Exchangeable potassium, calcium, and magnesium were 0.36, 1.20, and 0.46 cmol $_{(+)}$ kg^{-1}, respectively (Blankson *et al.,* 2025).

3.2 Phase 1: Biochar and Compost Production and their Characterization

3.2.1 Empty oil palm fruit bunch biochar production

The empty fruit bunch that was used to produce the biochar was obtained from Juaben Oil Mills Limited (6°48'55.4"N, 1°24'49.8"W) in the Ashanti Region of Ghana. Two types of biochar were produced from EFB biomass using two dominant technologies in Ghana; continuous rotary pyrolysis reactor (with temperature range between 400–550^0 and residence time of 30 minutes) and a locally manufactured kiln (static chamber) (with temperature range between 300–400^0 and residence time of 8–10 hours). Biochar 1 (B1) was produced by cutting EFB biomass into smaller sizes (15–30 cm) and sun-dried to 40–45% moisture content before pyrolysis. The sun-dried bunches were pyrolyzed using the continuous rotary pyrolysis reactor. The EFB biochar yield ranged between 27% and 30%.

Biochar 2 (B2) was produced using the locally manufactured kiln, after sun-drying the whole bunches to 40–45% moisture content. The biochar yield from the locally manufactured kiln ranged from 20 to 25%. After carbonization, the biochar was left in the reactors or kiln for cooling. Samples from each reactor were collected, grounded, sieved through 2 mm mesh size and stored for laboratory analyses.

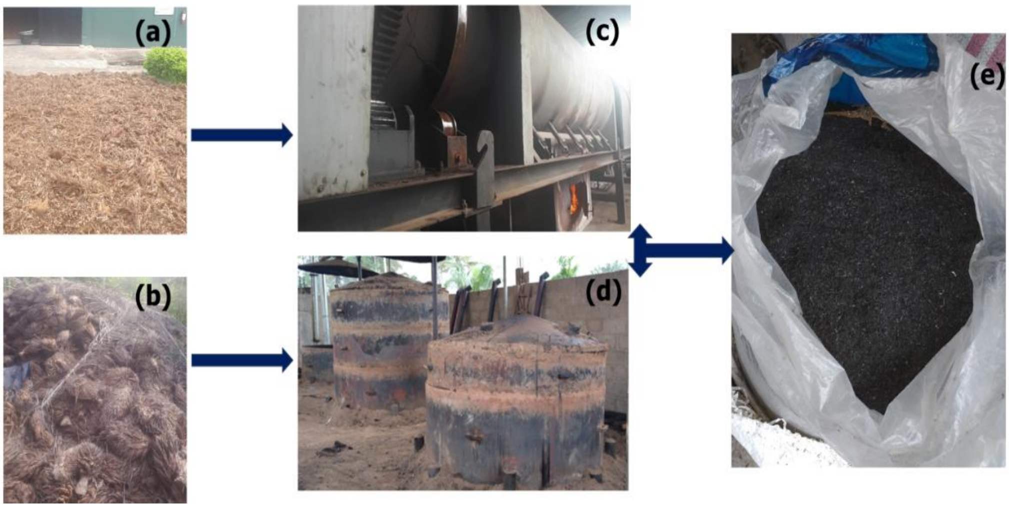

Figure 2: Illustration of empty oil palm fruit bunch biochar production process; (a) sun-dried shredded EFB; (b) Air-dried EFB bunches; (c) continuous rotary reactor; (d) locally manufactured kiln; (e) EFB biochar.

3.2.2 Empty oil palm fruit bunch compost production

Three types of composts were prepared by combining EFB and poultry manure at three different mixing ratios (Table 1). The poultry manure was obtained from the School of Agriculture Research Farm, University of Cape Coast, Cape Coast, Ghana. A chainsaw machine was used to cut the fresh EFB that were tightly packed in a wooden box with spaces through which the bar of the chainsaw could pass (Figure 3). The fruit bunches were cut into sizes of 2–

10 cm. Samples of the feedstocks were analyzed for total carbon, total nitrogen and moisture content before the calculation of the bulking ratios (Table 1).

Table 1: Description of mixing ratios of the different composts

Treatment	EFB: PM mixing ratios (w/w)	C/N ratio of starting pile
C1	1:1.5	30:1
C2	2:1	35:1
C3	3:1	40:1

EFB, empty fruit bunch; PM, poultry manure; C/N ratio, carbon-nitrogen ratio

Figure 3: Illustration of empty oil palm fruit bunch co-compost production process.

The compost was prepared from a combination of shredded EFB biomass and poultry manure using the pit method. Poultry manure was chosen as an additive in the composting empty fruit bunches to readjust the C/N ratio of the initial feedstock. The EFB biomass had moisture, carbon and nitrogen contents of 20.5%, 44.0% and 0.9% respectively, while the poultry manure had moisture, carbon and nitrogen contents of 20.0%, 35.8% and 1.6% respectively. Three bulking C/N ratios of 30:1, 35:1 and 40:1 subsequently referred to as C1, C2 and C3 respectively, were used. The bulking C/N ratios were calculated based on the moisture, carbon and nitrogen contents of feedstocks using *a compost carbon nitrogen calculator* developed by Cornell University (Cornell Waste Management Institute, 1996). The weight of the starting pile ranged from 680–800 kg on a fresh weight basis, and the mixing ratios of EFB and poultry manure were 1:1.5, 2:1 and 3:1, respectively, for C1, C2 and C3.

Long narrow EFB piles of about 1.0 m high and 1.5 m wide with a length of 3.0 m were made in a shallow pit of about 50 cm depth. The pile was mixed to reflect the mixing ratios. Piles were done in duplicates and monitored throughout the composting period. The compost piles were covered with black polythene sheets and piles were turned every three days in the first two weeks and subsequently done at bi-weekly intervals. The moisture content of the piles was maintained at 50– 65% water holding capacity (WHC) by the application of 100–180 L of water to each pile at each turn (Figure 3). The temperature and moisture of the piles were monitored at every turn from three spots at the surface and about 50 cm deep into the piles (Figure 4).

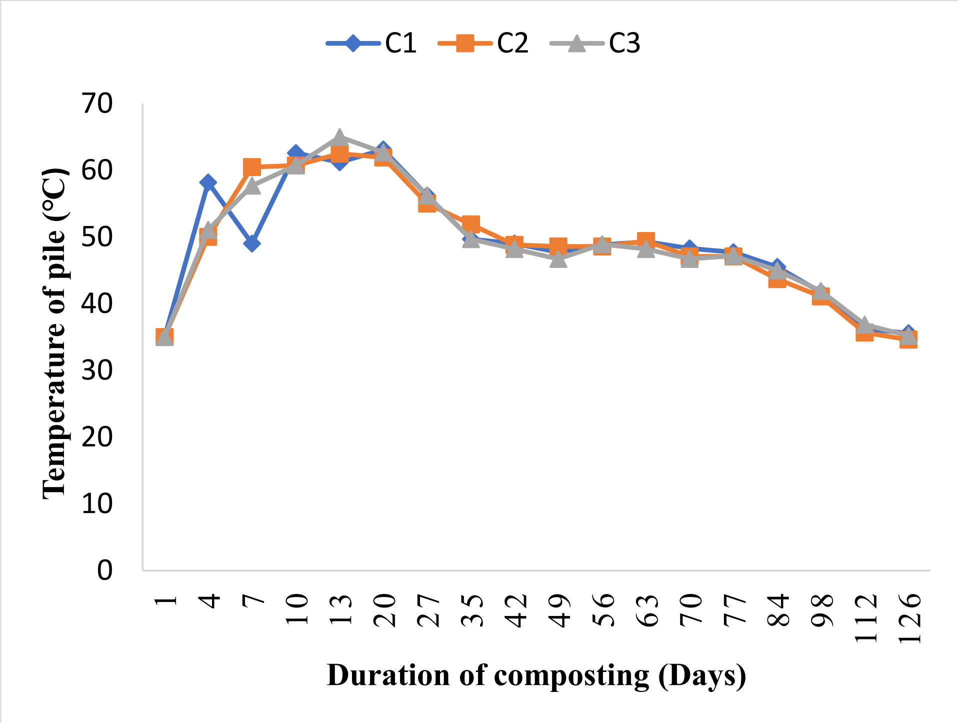

Figure 4: Mean temperature trend of compost piles during the composting period.

The compost maturity was assessed by physical observation using colour and the disappearance of odour (Khan *et al*., 2009) and self-heating test was done by measuring the temperature of the pile to determine the peak temperature. The compost was considered matured and cured when the peak temperature was between 30–40^{0} on the 18th week of composting. The compost was cured for 5 weeks till a constant temperature of $\leq$ 30^{0} was attained before harvesting. Samples of biochar and compost were collected and stored as shown in Figures 3 for analyses.

3.2.3 Empty oil palm fruit bunch biomass, biochar and compost analyses

3.2.3.1 Proximate and elemental analyses

Samples of biochar, oven-dried (65^{0}) EFB biomass and composts were analysed at Eurofins Umwelt Ost GmbH laboratory, Bobritzsch-Hilbersdorf, Germany. The ash content, organic carbon (C_{org}), total inorganic carbon (TIC), carbonate-CO_2 carbon (C), hydrogen (H), oxygen (O), nitrogen (N) and total sulphur (S) contents were analysed using standard methodology. Briefly, ash content was determined by a gravimetric method, oxygen (O) content by pyrolysis and infrared detection, carbonate-CO_2 through acid titration, and carbon (C) and nitrogen (N) by combustion analysis. Total sulphur (S) was analysed through high-temperature oxidation. Hydrogen (H) was analysed by an elemental analyser (Vario ELIII, Elementar, Germany). The elemental ratios of the EFB biomass, biochar and composts were calculated from their various elemental compositions. The carbon storage class, known as stock BC+100 (sBC+100), was obtained by multiplying the BC+100 value by the C_{org} contained in the amendments.

3.2.3.2 Nutrient content analyses

The EFB biochar samples analysis was performed according to guidelines for the sustainable production of biochar - EBC, Version 10.1E - of 10/01/2022. The biochar and compost samples were analysed in duplicate for nutrient compositions that included manganese, zinc and boron content after micro wave pressure digestion, while calcium, iron, potassium, phosphorus and sulphur content were determined in borate digestion of ash at 550°C followed by spectroscopic analysis.

3.2.3.3 Heavy metals and polycyclic aromatic hydrocarbon (PAHs) analyses

The heavy metals content of the EFB biomass, biochar and composts including arsenic, lead, cadmium, copper, nickel, mercury, chromium and silver were analyzed after microwave pressure digestion with an Atomic

Absorption Spectrophotometer. A total of 16 PAHs were considered under this study and they include; Naphthalene, Acenaphthylene, Acenaphthene, Fluorene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benz(a)anthracene, Chrysene, Benzo(b)fluoranthene, Benzo(k)fluoranthene, Benzo(a)pyrene, Indeno(1,2,3-cd) pyrene, Dibenz(a,h)anthracene and Benzo(g,h,i)perylene. The contents in the EFB biomass and composts were determined from the original substance. The PAHs contents of the EFB biochar were determined after toluene extraction using gas chromatography-mass spectrometry (GC-MS; Agilent 7890A/jms-Q1000GC MKII, USA).

3.2.4 Statistical analyses

Data normality was assessed using the Shapiro-Wilk test, and variance homogeneity was evaluated with the Brown-Forsythe test before analysis and all the data met these assumptions. One-way ANOVA was performed to test the effects of EFB pyrolysis and composting on the chemical and toxicant levels in the EFB feedstock. The Tukey posthoc test was used for multiple means comparison at a significance level of 0.05. Analysis of variance (ANOVA) was performed in GenStat 12th edition, version 12.10.3338.

3.3 Phase 2: Evaluation of Empty Oil Palm Fruit Bunch Biochar Loading Capacity of Different Tropical Soils using Incubation, Germination and Pot Experiments

3.3.1 Soil sampling for incubation and pot experiments

Acrisols and Ferralsols are part of the dominant and most cultivated soils in the tropics. They are known to be chemically poor, of low natural fertility and have the tendency to fix phosphorus (Ghartey *et al.,* 2012). However, Vertisols are considered among the most fertile soils of the dry tropics, but sometimes they can be slightly acidic and contain high levels of exchangeable aluminium. These different tropical soils vary in their deficiencies and needs, necessitating the establishment of the application rates of EFB biochar to address their specific issues. Topsoil (0–20 cm) from four contrasting soil types (Figure 5), which were broadly characterised as Acrisol, Brown Ferralsol, Red Ferralsol and Vertisol (IUSS Working Group WRB 2015), were collected from Cape Coast (5º 7′ 48.9′ N 1º 17′ 21.278' W) in the Central, Anyinase (5°58′60′ N 1 º 4′ 0′W) in the Western, Offinso (6°56′ N 1°40′W) in the Ashanti, and Kpong (6°8.112' N 00°4.578' E) in the Greater Accra regions of Ghana, respectively for this experiment. The soil samples were air-dried and sieved through a 4 mm mesh for the incubation experiment. The soil samples used for physicochemical characterisation were sieved through a 2 mm sieve, packaged and stored at room temperature for analysis.

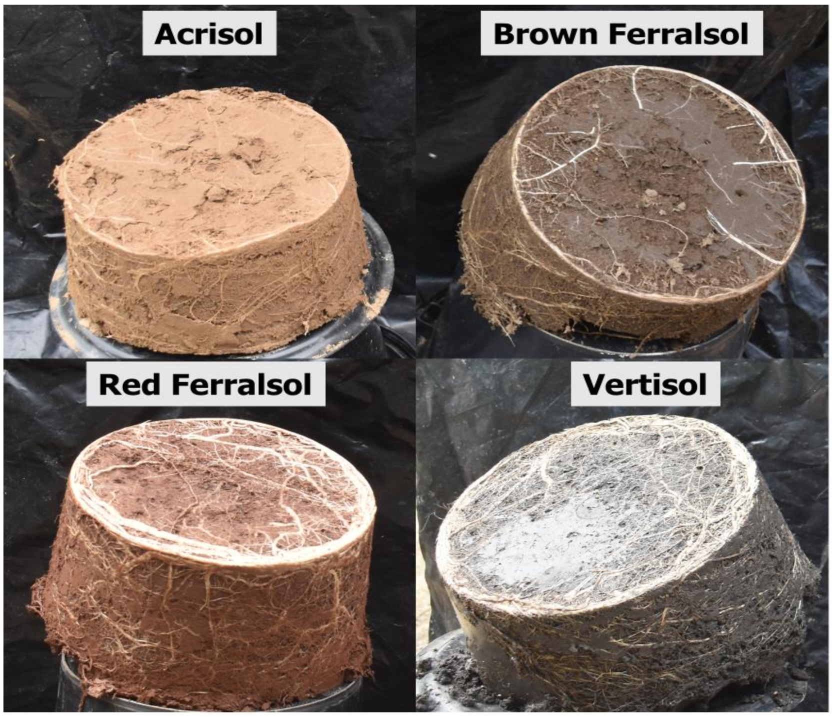

Figure 5: The four different tropical soil types that were used for the incubation, phytotoxicity and pot experiments.

3.3.2 Pre-treatment soil analyses

Soils collected from the different locations across the country were characterized for texture, pH, organic matter content, total nitrogen, available phosphorus, exchangeable bases (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) and exchange acidity (H^{+} and Al^{3+}).

3.3.2.1 Soil pH measurement

Soil pH was measured in a 1:2.5 soil-water ratio using a glass electrode pH meter following the method of Rowell (1994). Approximately 10 g of air-dried soil was weighed in triplicate into a plastic bottle with a screw cap; A 25 mL of distilled water was added and shaken for 15 minutes on a mechanical shaker. After calibrating the pH meter with buffers of pH 4.0 and 7.0, the pH was measured. The average pH for each sample was read and recorded.

3.3.2.2 Soil particle size distribution analysis

The particle size distribution of the soil was determined using the pipette sampling technique described by Rowell (1994). The organic matter content of soil was destroyed with H_2O_2 under gentle heating, stirring and boiling. This was followed by dispersion, shaking (overnight) dilution of the peroxide treated soil. The sampling of silt was done by drawing 20 mL of suspension with a special pipette, after thorough mixing with a plunger and subsequent settling time of 32 seconds was allowed. After 8 hours of standing, the clay was sampled at a depth of 10 cm into labelled, weighed beakers and was subsequently dried at 105˚C. After drying, the beakers were cooled in a desiccator and reweighed.

Figure 6: Soil particle size determination using the pipette sampling technique.

These gave the mass of silt and clay, plus a small residue of the dispersing agent. After another 8 hrs, sand fraction was sampled by pouring away most of the supernatant liquid and quantitatively transferring the sediment into a beaker. Stirring, settling and decanting were done repeatedly until the supernatant was clear. The sand was also oven-dried, cooled and reweighed to calculate the mass of the oven-dried soil. The textural class of the soil was determined using the textural triangle, after calculating the percentage of each particle size in the samples (Equations 1–3).

$$\text{Sand (\%)} = \frac{MS \times 100}{MdS} \quad [1]$$

$$\text{Silt (\%)} = \frac{TSi \times 100}{MdS} \quad [2]$$

$$\text{Clay (\%)} = 100 - (TSi + TS) \quad [3]$$

Where:

MS = mass of sand; MdS = mass of oven-dried soil; TSi = total silt in soil sample; TS = total sand in soil

3.3.2.3 Total organic carbon determination

The organic carbon content of the soil was determined using the Walkley-Black method (Walkley & Black, 1934). A 0.5 g of soil sample was weighed in duplicates and transferred into a 500 ml Erlenmeyer flask; a blank was also included, and the weights were recorded, 10 mL of 0.167 M potassium dichromate ($K_2Cr_2O_7$) was added and gently swirled. Twenty millimetres of concentrated H_2SO_4 was then added and the flask was allowed to stand for 30 minute before diluting to 200 mL with distilled. Ten millilitres and 0.2 g of H_3PO_4 and NaF, respectively, were added before the addition of a diphenylamine indicator. The excess Cr_2O_7 was back titrated with 0.5 M ferrous solution to a green end point and TOC was calculated (Equation 4).

Total organic carbon (TOC) (%) = $= \frac{(B-S) \times M\ F3+}{wt \times \left(\frac{100}{77}\right)} \times 100$ [4]

B = Blank titre value (mL); S = sample titre value 0.300 mL = 12/4000 = milli-equivalent weight of carbon; Wt = weight of soil (g); M = morality; 100/77 = the factor converting the carbon oxidized to total carbon; 100 = the factor to change from decimal to percentage.

Organic matter (%) $= \frac{\text{TOC} \times 100}{58}$ [5]

Where: TOC = Total organic carbon

3.3.2.4 Total nitrogen determination

Soil total nitrogen was determined by the modified Micro-Kjeldahl method as described by Rowell (1994). A 0.5 g of soil was digested with 0.2 g catalyst and 30 mL of concentrated H_2SO_4 at 380 ˚C, for two hours on a block digester; two blanks were included. On completion of digestion, the content was diluted with 100 mL distilled water. After flushing, a steam distillation apparatus, a 100 mL conical flask containing 5 mL of boric acid indicator was placed under the condenser of the apparatus, 20 mL aliquot of the sample digest was transferred to the reaction chamber, and 10 mL of alkali mixture was added to collect 50 mL of the distillate. The distillate was then titrated against 1/140 M HCl from a green to a wine-red end point and total nitrogen was calculated using Equation 6.

$$\text{Total nitrogen (\%)} = \frac{(S-B)\times V}{100 \times v \times wt} \qquad [6]$$

Where:

S = Sample titre value (mL); B = Blank titre value (mL); wt = sample weight (g); V = solution volume (mL); v = volume of aliquot (mL)

3.3.2.5 Available phosphorus determination

Soil available phosphorus was determined using Bray No. 1 method after soil extraction with $NH4^{+}$ and 0.5 M HCl. Two millilitres aliquot of the extract was mixed with a reagent (ammonium molybdate, potassium antimony tartarate, 2.5 M H_2SO_4 dissolved in ascorbic acid) and diluted with a distilled water. The colour was allowed to develop for 15 minutes to read the absorbance on a spectrophotometer at 882 nm. The concentrations of the soil samples were extrapolated using Equation 7.

$$\text{Available phosphorus } (\mu g\ g^{-1}) = C \times df \qquad [7]$$

Where:

C = concentration obtained from the graph; df = dilution factor

3.3.2.6 Exchangeable cations analyses

Exchangeable basic cations, Ca^{2+}, Mg^{2+}, K^{+}, Na^{+} were extracted with 1 M NH4OAc (FAO, 2008) and aliquots of the extract were used for the determination of these cations. Exchangeable potassium and sodium were determined by flame analysis using a flame photometer. Exchangeable Ca^{2+} and Mg^{2+} were determined by titration with ethylene diaminetetra-acetic acid (EDTA), using solochrome black indicator and EBT indicator, respectively.

The acidic cations Al^{3+} and H^{+} were extracted with KCl solution and titrated to pink endpoint with NaOH, using phenolphthalein as an indicator. The effective cation exchange capacity (ECEC) of the soil was calculated by summing the basic cations (Equation 8–13).

$$K^{+}\ (\text{cmol}_{(+)}\ \text{kg}^{-1}) = \frac{(C \times 10)}{(wt \times 39.1)} \qquad [8]$$

$$Na^{+}\ (\text{cmol}_{(+)}\ \text{kg}^{-1}) = \frac{(C \times 10)}{(wt \times 22.99)} \qquad [9]$$

$$Ca^{2+}\ (\text{cmol}_{(+)}\ \text{kg}^{-1}) = \frac{(4 \times T)}{wt} \qquad [10]$$

$$Mg^{2+}\ (\text{cmol}_{(+)}\ \text{kg}^{-1}) = \frac{(4 \times T)}{wt} \qquad [11]$$

$$H^{+} + Al^{3+}\ (\text{cmol}_{(+)}\ \text{kg}^{-1}) = \frac{(2 \times T)}{wt} \qquad [12]$$

$$\text{ECEC}\ (\text{cmol}\ (+)\ \text{kg}^{-1}) = Ca^{2+} + Mg^{2+} + K^{+}\ Na^{+} + Al^{3+} + H^{+} \qquad [13]$$

Where:

C = concentration of extract from standard curve; T = sample titre value (mL); wt = of soil (g)

3.3.3 Incubation experiment

An incubation experiment was performed to establish the maximum rate of EFB biochar that can be safely applied to the different soil types without compromising their functions. Six biochar application rates of 0, 0.25, 0.5, 1.0, 1.5 and 2.0% w/w, which translated to 0, 2.5, 5, 10, 15 and 20 g kg^{-1}, were premixed with 5 kg of soil and packed into 5 L perforated plastic pots to stimulate a bulk density of 1.5 g cm^{-3}. Distilled water was added, and soil moisture was maintained at 65% field capacity. Amended soils were incubated at room temperature of 28 ± 2 °C (Figure 7) for four weeks to allow the biochar to react with soil (Abdulrahman *et al*. 2016) and re-initiate microbial activity after storage.

Figure 7: Activities under the incubation experiment; (a) Filled pots (b) incubation of pots and (c, d) soil sampling for analyses during the incubation experiment.

Soil samples were taken after 7 and 28 days of incubation to measure ammonium (NH_4^+) and nitrate (NO_3^-)-nitrogen, available phosphorus (P), pH, electrical conductivity (EC) and total organic carbon (TOC) contents to establish the chemical properties affecting seed germination when the two-week soil-biochar equilibration time (delay to the sowing of seeds after application of organic amendment to the soil) (Oyeyiola et al. 2020) is not reached or exceeded.

3.3.4 Phytotoxicity test

The seed germination testing method outlined by Solaiman et al. (2012) with slight modifications was adopted in this study. Soil samples collected at 1 and 7 days after soil-biochar incubation was used for the incubation test to evaluate the generally accepted two-week equilibration period and determine the extent to which seed germination was impacted when this equilibration time was not reached. After 1 and 7 days of incubation, 15 g of each soil-biochar mixture was sampled from each treatment and placed in a petri dish lined with filter paper.

Figure 8: Activities under the germination tests; (a) plated seeds, (b) germinated seedlings and (c, d) measuring of radicle length during the germination experiment.

The soils without biochar were used as controls for the respective soil types. All treatments were replicated four times, making a total of 96 pots. Ten maize seeds were placed in each petri dish that contained biochar-amended soil, and using a pipette, 10 mL of water was added to each petri dish and then covered with fitting lids (Figure 8a). The petri dishes for germination were placed in a dark, cool and dry room and maintained at 28 ºC (± 2 ºC).

The number of seeds that germinated in each petri dish was recorded from day one to seven. A seed with a radicle length of at least 2 mm was considered germinated (Kebrom *et al*., 2019). After 7 days, germination was deemed to be complete and soil was washed off from the root and the root length of germinated seeds was measured using a ruler and was reported as a sum for each petri dish (cm/petri dish) (Figure 8). Relative seed germination (SG), relative root elongation (RRE) and germination index (GI) were calculated (Equations 14, 15, and 16).

$$\text{SG}\ (\%) = \frac{\text{Gb}}{\text{Gs}} \times 100 \qquad [14]$$

$$\text{RRE}\ (\%) = \frac{\text{Rb}}{\text{Rs}} \times 100 \qquad [15]$$

$$\text{GI}\ (\%) = \frac{\text{SG} \times \text{RRE}}{100} \qquad [16]$$

Where:

Gb = number of seeds germinated in soil-biochar mixture; Gs = number of seed germinated in soil only; Rb = mean root length in soil-biochar mixture; Rs = mean root length in soil only

3.3.5 Pot experiment

A pot experiment was conducted at the A. G. Carson Technology village (N 5º 7′ 48.9″, W 01º 17′ 21.2784″) of the School of Agriculture, University of Cape Coast, Ghana. After four weeks of soil incubation, pots with the different soil types were seeded with 3 maize seeds (Obatanpa variety). The pots were arranged in a Completely Randomized Block Design (RCBD) with four replications under a transparent polythene shed, exposed to natural conditions apart from rain, for 8 weeks (Figure 9).

Figure 9: Activities undertaken during the pot experiment; (a) Maize at various stages of growth and (b) fertiliser application during the pot experiment.

Following germination, seedlings were thinned to two plants per pot and soil moisture was maintained at 65% field capacity with top-up done by difference through weighing. Inorganic fertiliser was applied at the rate of 140 kg N ha^{-1}, 80 kg P_2O_5 ha^{-1} and 80 kg K_2O ha^{-1}. The fertiliser was applied in two splits, a basal application was done 10 days after germination, using N: P: K 23:10: 5, Triple super phosphate and Multi K (containing 13% N and 46% of K). The second split of fertiliser was applied four weeks after the basal application using urea.

3.3.6 Data collection

Maize plants were harvested at the tassel stage when they were eight weeks old (Figure 10). The maize shoots were cut at the soil surface using a knife and each pot was emptied into an aluminium pan so that the roots could be carefully removed and washed of soil. The fresh shoots and roots were weighed before being oven-drying at 65 °C till constant weight was attained. The dry shoot and root biomass yield of each plant (g plt^{-1}) were measured to estimate the dry biomass yield per plant.

Figure 10: Biomass data collection under pot experiment.

3.3.7 Laboratory analyses

Soils that were sampled after incubation were split into two sub-samples; first sub-sample was used for the determination of ammonium and nitrate in fresh soil. The fresh sample was extracted with 2M KCl and ammonium and nitrate- nitrogen contents were quantified with the indophenol blue method and trans-nitration of salicylic acid, respectively (Tzollas *et al*., 2010). The second sub-sample was air-dried for the measurements of pH, electrical conductivity, organic carbon, total nitrogen, and available phosphorus contents. The soil pH and electrical conductivity were determined at soil-to-water ratio of 1:2.5 using a combined pH and EC meter (Hanna

Instruments, Kungsbacka, Sweden), organic carbon by the Walkley-Black method, total N by the Micro-Kjeldahl method, available phosphorus by Bray No. 1 method, exchangeable calcium, magnesium, sodium and potassium, effective cation exchange by Silver-Thiourea extraction method and exchangeable acidity by barium chloride-Triethanoamine method (Black, 1965).

3.3.7.1 Soil ammonium and nitrate content determination

Soil ammonium-nitrogen content was quantified with the indophenol blue method. A 6 g of moist soil was weighed into a labelled extraction bottle and mixed with 50 mL of 2 M KCl. A blank containing no soil was also included. The content was shaken for an hour on a mechanical shaker and was filtered through a Wattman filter paper. The moisture content in the sample was determined by drying a similar sample in an oven at 105 °C to a constant weight. One millilitre of disodium ethylene-diamine-tetra acetic acid (EDTA) solution was added to an aliquot of a 2 M KCl extract of soil followed by the addition of phenol-nitroprusside and buffered hypochlorite reagents. The solution was allowed to stand for an hour for colour development at room temperature (23°C) before determining the absorbance on the spectrophotometer at 650 nm (Figure 11).

Figure 11: Ammonium and nitrate-nitrogen determination after soil-biochar incubation.

The nitrate-nitrogen content was determined by trans-nitration of salicylic acid. Briefly, 1.0 mL of the soil extract was mixed with 2 mL of salicylic acid and was mixed concentrated H_2SO_4. The reaction was allowed to take place for about 20 minutes. Thereafter, approximately 100 mL of 2 M HCl was added to raise the pH of the solution to above 12. The solution was cooled to room temperature before the absorbance was determined at 410 nm. Nitrate-nitrogen standards of 0 to 60 μg NO_3^--N were analysed with each set of samples. Excel software was used to plot a calibration curve, using the concentrations and absorbance of the standard solutions; from the curve, the concentrations of the soil samples were extrapolated and expressed on oven-dried soil basis.

3.3.7.2 Biomass nitrogen and phosphorus analyses

The total N in dry shoot and root was determined using the micro-Kjeldahl method. The oven-dried maize biomass samples harvested at tassel were ground to a very fine powder. Approximately 0.2 g of the ground sample was weighed into a 100 mL Kjeldahl flask, 4.5 mL of concentrated H_2SO_4

(digestion reagent) was added and the samples digested at 360 ˚C for two hours. Blank digestions (digestion of the digestion mixture without plant sample) were carried out in the same way. After the digestion, the digests were transferred quantitatively into 100 mL volumetric flasks and made up to volume.

The steam distillation apparatus was set up and steam was passed through it for 20 minutes. After flushing the apparatus, a 100 mL conical flask containing 5 mL of boric acid indicator was placed under the condenser of the apparatus. A 20 mL aliquot of the sample digest was pipetted and transferred to the reaction chamber through the trap funnel and 10 mL of alkali mixture was added, commencing the distillation to collect 50 mL of the distillate. The distillate was then titrated against 1 M HCl from green to wine red end point. The nitrogen content of the smaple was calculated using Equation 17.

$$N\ (\%) = \frac{(\mathrm{S-B}) \times \text{solution volume}}{100 \times \text{aliquot} \times \text{sample weight}} \qquad [17]$$

Where:

S = Sample titre value (mL); B = Blank titre value (mL)

Total phosphorus content of the shoot and root samples was measured using the ascorbic acid method after wet oxidation. One millilitre aliquot of the digested sample was pipetted into 25 mL volumetric flasks and 10 mL of distilled water was added. Next, 4 mL of reagent B (Reagent A containing a mixture of 12 g ammonium molybdate in 250 mL distilled water, 0.291 g of potassium antimony tartarate in 100 mL distilled water and I L of 2.5 M H_2SO_4 made up to 2 L with distilled water. Reagent B was prepared by dissolving 1.06 g of ascorbic acid in every 200 mL of reagent A) was added, and their volumes made up to 25 mL with distilled water and mixed thoroughly. Phosphorus standards were also prepared. The flasks and contents were allowed to stand for 15 minutes for colour development, after which the absorbance of the standards and samples were determined using a spectrophotometer, at a wavelength of 882 nm. A calibration curve was plotted using their concentrations and absorbance. The concentrations of the sample solutions were extrapolated from the standard curve and the concentration was calculated using Equation 18.

$$\mu g\ P\ g^{-1} = \frac{C \times solution\ volume}{sample\ weight} \qquad [18]$$

Where: C = concentration of the sample solution

The N and P uptakes were calculated using Equation (19) (Li et al. 2016).

Nutrient uptake (g plant^{-1}) = (Nutrient content (%) × biomass yield (g plant^{-1}))/100 [19]

3.3.8 Determination of optimal biochar rates for the different soils

To determine the optimal EFB biochar rate that conditions the soil for enhanced nutrient uptake without any harmful effect, scores ranging from 0 to 1 (at 0.2 intervals) were assigned to measured parameters based on their contribution to soil amelioration (Fouladidorhani *et al.*, 2024). The soil chemical properties (EC, pH, NH_4^+, NO_3^-, available P and TOC measured at 28 DAI), phytotoxicity and plant parameters (GI and RRE measured at 1 DA1, biomass yield, N and P uptake) were used. A high value was considered optimal for all the measured parameters and it was assumed that each parameter contributed equally to soil amelioration. Since the biochar rates were six, the range was partitioned into 6 intervals from 0 to 1, with increments of 0.2. For a particular soil type, scores were assigned to each biochar rate based on the parameter value; a score of 0 was assigned to the rate with the lowest parameter value and score of 1 was assigned to the rate that had the highest parameter value. Thereafter, the average score for each rate for each soil type was computed and the rate that had the highest average score was regarded as the best rate suited for that particular soil type.

3.3.9 Statistical analysis

Data collected in the incubation, phytotoxicity and pot experiments were analysed using GenStat 12th edition, version 12.10.3338. Normality and homogeneity of variance test were checked using the Shapiro Wilk test and Brown-Forsythe respectively and data were log-transformed when necessary. One-way analysis of variance was performed to test the effects of EFB biochar rates on germination indices, soil chemical properties, yield and N and P uptakes for each soil type separately. The Tukey post hoc test was used for multiple means comparison at a significance level of 0.05. Two sample t-tests were performed to compare the two incubation periods on the chemical properties and the germination index of maize in the different soil types. The relationship between biomass yield, nitrogen and phosphorus uptakes with soil chemical properties at planting (28 days after incubation) from pooled data from the four different soil types were established with Pearson correlation analysis.

3.4 Phase 3: Field Experiments

3.4.1 Pre-treatment soil sampling

Before setting up the field experimental, soil samples were collected at a depth of 0–20 cm in a Z pattern for the determination of soil physico-chemical properties. The samples were bulked and composite samples picked for laboratory analyses. The bulked soil samples were air-dried, sieved through a 2 mm mesh, and stored for the physico-chemical analyses. Also, intact soil cores were taken from the experimental field by taking eight core samples at a depth of 10 to 15 cm, using metal core samplers (6.1 cm diameter, 3.4 cm high and a volume of 100 cm^3).

3.4.2 Experimental design, amendments incorporation and sowing

A randomized complete block design of seven treatments (Table 2) and four replications, making a total of 28 plots was used in this study with local okra genotype as a test crop. The plot sizes were 9 m^2 with 2 m and 1 m spacing between blocks and plots respectively. Each plot was manually tilled and amendments incorporated into 20 cm depth using a hoe for all biochar and compost treatments with a hoe (Figure 12). The inorganic fertilisers were applied at later stages of the okra growth. Details of the different amendment combinations are indicated in Table 2.

The local okra (*Abelmoschus esculentus*) genotype known as *Baabo* was obtained from the Crop Science Department of University of Cape Coast, Ghana. Two weeks after the application of amendments, the okra was sown at spacing of 60 cm × 45 cm and to 3 cm depth. Two seeds were sown per hole, and thinning and refilling were done one week after germination.

Table 2: Description of experimental treatments

Treatment code	Application rate	Application (kg) per plot
B0	Unamended	0
NPK	Mineral Fertiliser: 100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1}and 60 kg K_2O ha^{-1}	NPK 23:10:5 = 0.196 TSP = 0.075, MOP = 0.074, Urea = 0.098
B10	Biochar 10 t ha^{-1}	9
B20	Biochar 20 t ha^{-1}	18
CP20	Compost 20 t ha^{-1}	18
B10CP20	Biochar 10 t ha^{-1} + Compost 20 t ha^{-1}	9 +18
B20CP20	Biochar 20 t ha^{-1} + Compost 20 t ha^{-1}	18 +18

TSP, Triple super phosphate; MOP, Muriate of potash.

Mineral fertiliser was applied to the NPK plots using NPK 23:10:5 (fortified with sulphur, magnesium and zinc at 3%, 2% and 0.3% respectively), Triple super phosphate, Muriate of potash and urea. The application was done in two splits; the first split was done at sowing by drilling in between rows of crops at the rate N, P and K 50:60:60 kg ha^{-1}. The second split application was done using urea at the 50% flowering stage.

Figure 12: Activities undertaken during the field experiments; (a) Field clearing, (b) amendment incorporation and (c) sowing.

The first okra cropping cycle lasted from June to October 2022 and the second crop cycle was from February to May 2023. The mean monthly rainfall and temperatures of the experimental period are presented in Figure 13. The amendments were applied only once in June 2022, thus, no new biochar or compost was added in the second cycle.

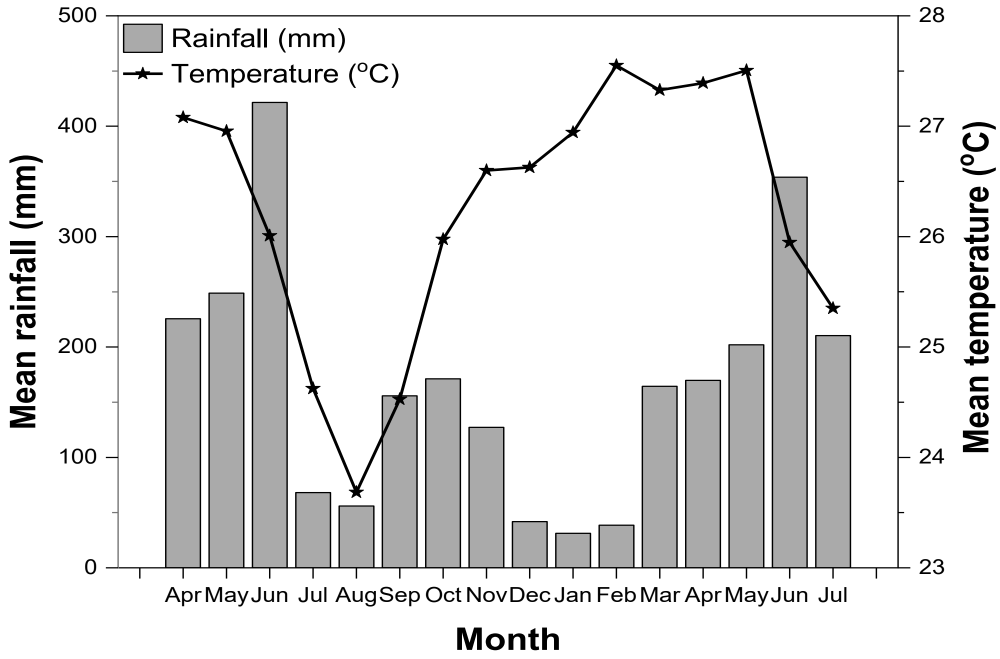

Figure 13: Mean monthly rainfall and temperatures during the experimental period (April 2022–July 2023).

3.4.3 Agronomic practices

Agronomic practices for okra including irrigation, weeding, and pest control were observed. Supplemental irrigation was done as and when needed (Figure 14). Within the crop cycle, weeding was done four times. The major pests of okra were controlled with K optimal (active ingredients: Lambda-Cyhalothrin 15 g L^{-1} + Acetamipride 25 g L^{-1}) and Lambda-Cyhalothrin at the recommended application rates.

Figure 14: Agronomic practices that were undertaken during the field experiment; (a) irrigation, (b) weeding and (c) pesticide application.

3.4.4 Growth and yield data collection

Growth data collection started three weeks after germination. In each plot, eight plants were randomly selected from the middle rows for the growth data. All growth measurements were done in 4 growth stages of the okra (seedling, vegetative, flowering and fruiting stage) (Figure 15). The height of each plant was measured from the soil surface to the tip of the main branch of the plant using a measuring ruler (Figure 16). The stem diameter of each plant was measured at 5 cm above the soil by using a digital vernier calliper.

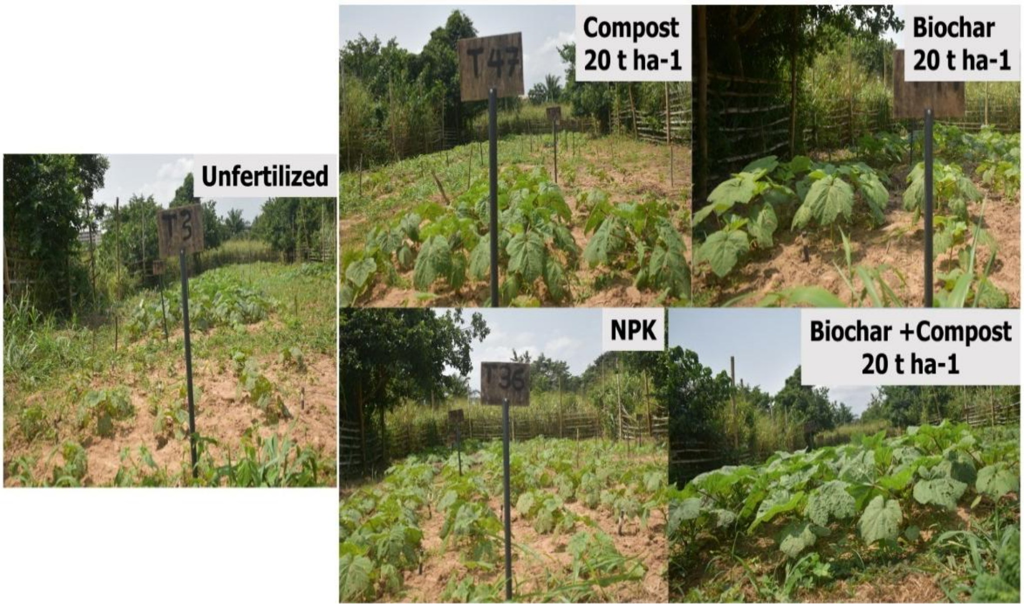

Figure 15: Growth pattern of okra plant under the different soil amendments.

At the 50% flowering stage (i.e. half of plants on each plot were flowering), aboveground biomass was harvested by randomly selecting and cutting four plants from each plot with a knife above the surface of the soil. The fresh biomass was chopped into pieces, weighed and oven-dried at 70 °C till constant weight. The dried shoot biomass from the four plants was weighed and the biomass yield per plot was calculated using the plant population from each plot. Representative biomass samples were ball-milled for nitrogen and phosphorus analyses.

Figure 16: Growth and yield data collection during field experiment.

At maturity, eight plants from the middle of each plot were selected and tagged for harvesting, which was carried out twice a week when pods were 2 to 4 inches long, typically 5 to 6 days after flowering. The number of pods from tagged plants was counted and weighed for the estimation of yield per plot. The number of pods from tagged plants was counted and weighed for the calculation of yield per plot at each time of harvest and cumulative yield at the end of the harvesting period was estimated (Figure 17).

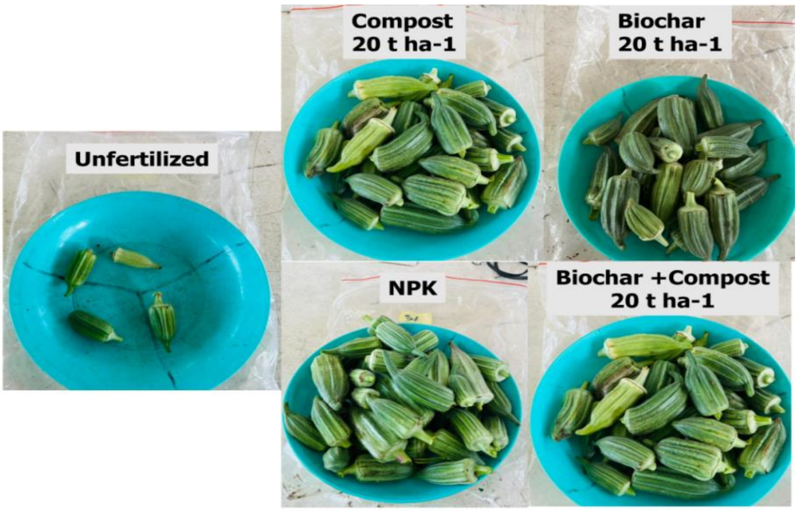

Figure 17: Okra pod yield from each treatment plot at a harvest.

3.4.5 Post field treatment soil sampling

3.4.5.1 Soil sampling for nematode extraction, signature fatty acid and chemical analyses

In each plot, 10 sub-samples of soil were randomly collected in the upper 15−20 cm by walking in a W pattern over the plot. These sub-samples were bulked together, for a composite sample per plot (Figure 18). Between each plot, the auger was carefully cleaned, and surface sterilized with ethanol. In the laboratory, 100 g of the fresh soil was weighed and used for the extraction of nematodes. A subsample of approximately 40−50 g soil was taken and stored in a freezer at 20^0 for analysis. Before the whole cell fatty acid analysis, the frozen soil was freeze-dried and ball-milled (Figure 18). Bulk soil samples from each plot were air-dried and ground to pass through a 2-mm sieve and stored in a Ziploc bag for chemical analyses (Figure 18).

Figure 18: Soil sampling, weighing, freeze-drying and milling for microbial analyses.

3.4.5.2 Soil sampling for carbon analyses

Calculation of carbon management (CMI) requires samples of the soil of interest and a sample collected from a reference site. The reference site represents an area where change in soil carbon dynamics is likely to be slow relative to the disturbed area (Sodhi *et al*., 2009). Bulk soil samples were collected from each experimental plot after the first and second okra crop cycles in October, 2022 and May, 2023 respectively, and from an adjoining area that has been relatively undisturbed since 2003 (20-year-old orchard plantation adjacent to the experimental field). In each plot and reference plot, five sub-samples of soil were randomly collected in the upper 15–20 cm by walking in a W pattern over the plot. These sub-samples were bulked together, for a composite sample per plot. The soil samples were air-dried and ground to pass through a 2-mm sieve and stored for analysis.

3.4.5.3 Soil sampling for physical properties and greenhouse gas analyses

Soil sampling for water retention, air permeability, gas diffusion, bulk density and greenhouse gas measurements was done at the end of the second okra crop cycle (i. e. of 12 months of amendment application) in May 2023. Intact soil cores were sampled from 3 out of 4 blocks for each of the seven treatments. In each plot, three core samples were taken at a depth of 10 to 15 cm using metal core samplers (6.1 cm diameter, 3.4 cm high and a value of 100 cm^3). In total, 63 intact soil cores were sampled and used for the subsequent measurements.

3.4.6 Shoot biomass analyses

The okra shoot biomass were analysed for total nitrogen and total phosphorus content. Total nitrogen content of the ground oven-dried okra shoot was measured with LECO CNS-1000 according to DIN EN 15407: 2011-05. The total phosphorus content of the plant material was measured by borate acid digestion of previously ashed biomass at 550°C according to DIN 51729-11: DIN EN ISO 11885 (E22):2009-09. The total nutrient uptake in the shoot dry matter was calculated by multiplying the dry matter yields (g $plot^{-1}$) by the shoot N and P content (g kg^{-1}). The nitrogen or phosphorus apparent recovery efficiency was calculated using Equation 20 adopted from Sarkar and Baishya (2017).

$$\text{ARE (\%)} = \frac{\text{Uf} - \text{Uc}}{Qf} \times 100 \qquad [20]$$

Where:

ARE = apparent nutrient recovery efficiency (%); Uf = nutrient uptake in shoot (above ground biomass) in the amended plot (g $plot^{-1}$); Uc = nutrient uptake in shoot (above ground biomass) in the unamended plot (g $plot^{-1}$); Qf = quantity of nutrient applied per plot (g $plot^{-1}$) (N and P in this case).

3.4.7 Soil nematode extraction and counting

Using the extraction tray method (Whitehead & Hemming, 1965), 100 g of fresh soil was weighed and placed in a 3 mm plastic sieve/mesh lined with two layers of tissue paper (Figure 19). The soil sample in the sieve was placed in a plastic bowl that contained 200 mL of distilled water. The setup was allowed to stand for three days to allow the nematodes to crawl out of the soil into the water. After three days, the sieve containing the soil was removed from the plastic bowl and the water was transferred into a plastic tube and was sent for counting. The nematode counting was done using a stereomicroscope by placing the extract in a demarcated counting dish.

Figure 19: Soil nematode extraction and counting.

3.4.8 Whole-cell fatty acid (WCFA) analysis

Whole-cell fatty acid analysis is a technique used to identify and characterise microorganisms based on the fatty acid composition of their cell membrane. One gram of soil from each freeze-dried and milled sample was saponified with a 2.0 mL NaOH-methanol mixture under heat to kill and lyse the cells. Heating was done by placing the sample tubes in a water bath at 100 (+/−2°C) for 30 minutes by vortexing in between heating (Figure 20). After heating, samples were cooled in a vessel with cold tap water. After cooling, the sample was methylated by adding 4.0 mL HCl-methanol, vortex for 10 minutes and was heated in an 80°C +/−1°C water bath for 10 minutes. The sample was allowed to cool to room temperature by placing the tubes in a vessel with cold water.

Subsequently, the sample was extracted with 2.5 mL of hexane/methyl tert-butyl ether and amending with 100 µl 19:0 (0.25 µg μL^{-1}) as internal standards. The tube was placed in a *blood turner* programmed at 50 rpm for 10 minutes and later centrifuged at 2500 rpm for 5 min. A clean Pasteur pipette was used to pipette the upper phase of the sample into a new labelled test tube. A 6 mL of NaOH solution was added to the sample to remove free fatty acids and remaining reagents from the organic extract. The tube was capped, placed on a blood turner for 5 minutes before centrifuging for five minutes at 2500 rpm in order to make the interface between the phases clearer when a separation is present (Figure 20). The organic upper phase of the sample was carefully removed with a Pasteur pipette and was transferred to a 4-mL GC vial. The sample was stored under frost. The sample was prepared for GC analysis by transferring 300 µL into a GC vial and evaporating under N_2 stream. The evaporated sample was further dissolved in 75 µL of hexane, vortexed, and transferred into an inset. The gas chromatography vial was filled with 200 µL of hexane before placing the inset into it to seal with a cap (Figure 20). An Agilent 6890 series gas chromatograph was used for the analysis.

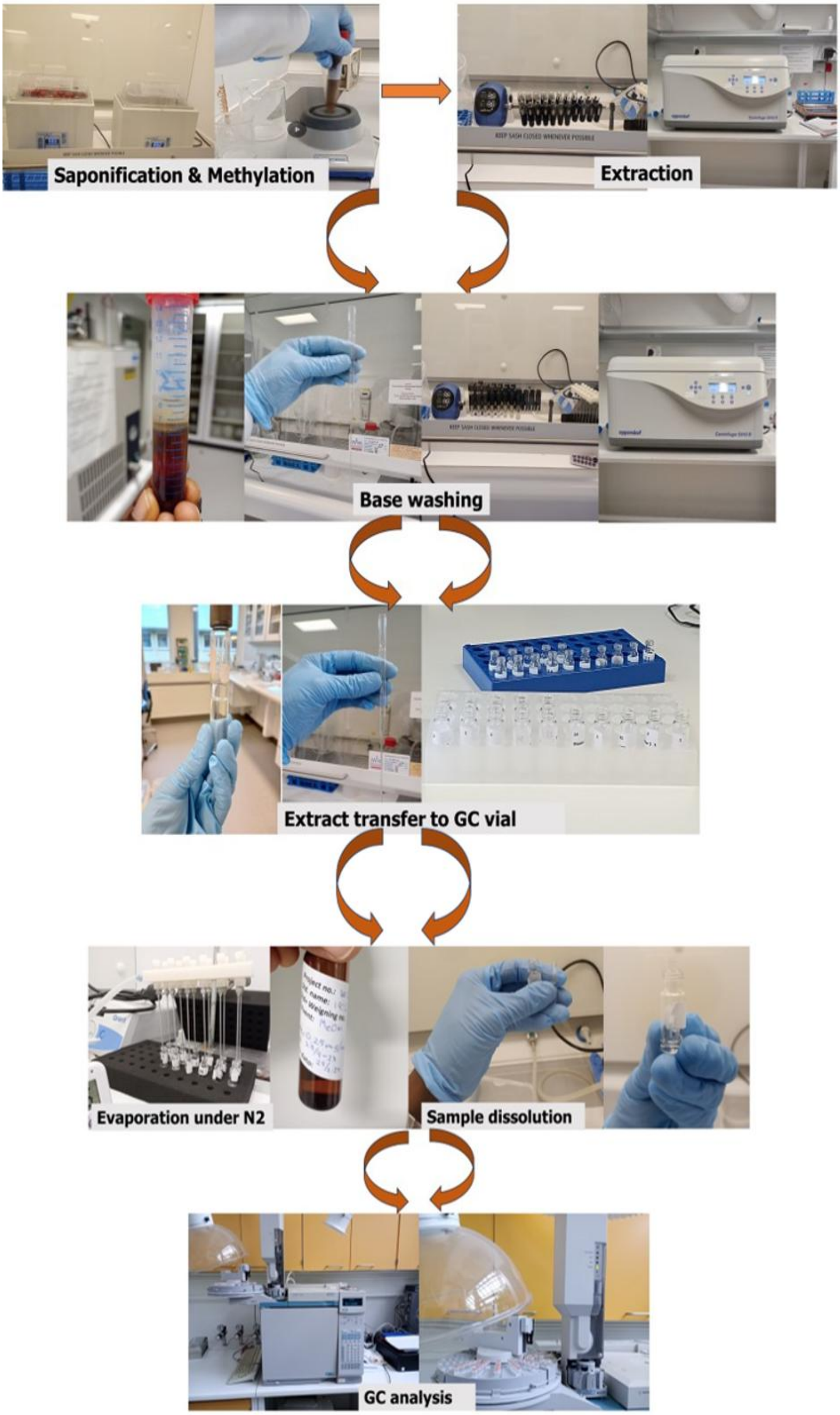

Figure 20: Illustration of Whole-Cell Fatty Acid (WCFA) analysis protocol.

Gas chromatography analysis and the identification of the fatty acids were performed according to the software library TSBA41. The peaks used as markers for gram-positive bacteria were iso 15:0, anteiso 15:0, iso 16:0, iso 17:0, anteiso 17:0 and iso 19:0 (White *et al*., 1996). The biomarker for arbuscular mycorrhizal fungi (AMF) was 16:1w5 (Olsson, 1995; Yang *et al*., 2020) and for saprophytic soil fungi, the marker was summed 18:1ѡ9c and 18:2ѡ6,9c (Kaiser *et al*., 2010).

3.4.9 Soil chemical analyses

Soil chemical analyses were conducted using standard protocols. Briefly, total organic carbon and nitrogen in the experimental soils were determined using a CHNS Elemental analyser as described by Sodhi, Beri and Benbi (2009). Soil pH was determined in a soil-to-water ratio of 1:2.5 using an electrode pH meter. Soil phosphorus content was determined using the Bray No. 1 method. The soil's exchangeable potassium was determined by flame photometry and magnesium with atomic absorption spectrophotometry at 285 nm after extraction of soil with ammonium acetate solution (Section 3.3.2).

The soil zinc and iron analyses were performed by extracting the soil samples in a DTPA, TEA and $CaCl_2$ solution buffered to 7.3 with HCl (Dissolution of 1.967 g DTPA, 14.919 g TEA and 1.470 g $CaCl_2$ in 900 mL deionized and the pH adjusted to 7.3 ± 0.2 HCl at 25 °C temperature) (FAO, 2022). Specifically, about 10.0 g of air-dried soil sample was weighed into 100 mL extraction bottle, 20 mL of DTPA extraction solution was added and shaken for exactly 2 hours at 25 °C. The extract was filtered using a Watmann filter paper. Standard solutions within optimum ranges were prepared for both zinc and iron their amount in the solution were quantified with atomic absorption spectrophotometry at 214 nm and 259 nm, respectively.

3.4.10 Soil cation exchange capacity (CEC) estimation

Soil cation exchange capacity (CEC, $cmol_{(+)}\ kg^{-1}$) was estimated from hygroscopic water content (wh, w/w%) at 40% relative humidity using the regression model (Equation 21) developed by Arthur *et al.* (2017) for kaolinite-rich soil samples. This method of CEC estimation is rapid for a large scale with no prior measurements of other soil properties. Again, this method is convenient because only a relative humidity (RH) meter and a large room with constant relative humidity are required. However, the type of clay in the soil should be known in order to select a model that is best suited for a particular soil. The hygroscopic water content of the soil sample was determined by air drying soil samples under room temperature for two weeks before oven drying at 105 °C for two days. The weight difference between oven dried soil weight and air dried soil weight expressed on oven-dried soil basis was used for the calculation.

$$CEC = 2.53 + 6 \times wh \qquad [21]$$

Where: wh = hygroscopic water content (% w/w) at 40% relative humidity

3.4.11 Total organic carbon and permanganate oxidizable carbon (POXC) determination

Total soil organic carbon (TOC) for the soil of the experimental sites and reference site soil was determined using a CHNS Elemental analyser, as described by Sodhi *et al.* (2009). The labile carbon fraction of concern is permanganate oxidizable carbon (POXC) because it represents the organic carbon pool that is more sensitive to the input of organic matter than the other labile carbon fractions (Bongiorno *et al*., 2019). The permanganate oxidizable carbon (POXC) was analysed following the procedure of Weil *et al*. (2003) with slight modification by Bongiorno *et al*. (2019).

About 2.50 g (± 0.05 g) of air-dried soil was weighed into a labelled 50 mL centrifuge tube and 18.0 mL of deionized water was added. Using a 1.0-10.0 mL pipette, 2.0 mL of 0.2 M $KMnO_4$ stock solution was added. The tube was tightly covered before arranging on a horizontal shaker and shaken at 180 oscillations per minute for 2 minutes. After 2 minutes, the sample was removed from the shaker and inverted vigorously to ensure that there was no soil clinging to the sides of the tube. The soil was allowed to settle for ten minutes. Four solution standards (0.005, 0.01, 0.015 and 0.02 M) were prepared from the $KMnO_4$ stock solution.

While the sample was settling, 49.5 mL of deionized water was added to a labelled dilution tube. Once the ten minutes settling period has passed, 0.5 mL of supernatant was quickly transferred from the reaction tube to the corresponding dilution tube containing 49.5 mL of water. The dilution tube was capped and was inverted to mix. The working standard solutions were also diluted following the same procedure. The standards and sample were quantified by dispensing 200 microliters of each standard and unknown sample into cuvettes and measuring the absorbance at 550 nm using spectrophotometer. Deionized water was used as a blank. The POXC was calculated according to an Equation 22 by Weil *et al*. (2003).

POXC (g kg^{-1}) = $[0.02\, mol\, L^{-1} - (a + b \times Abs)] \times (9\, g\, C\, mol^{-1}) \times (0.02\, L\, solution\, Wt^{-1})$[22]

Where: 0.02 mol L^{-1} = initial solution concentration; a = intercept of the standard curve; b = slope of the standard curve; Abs = absorbance of unknown; 9 = g of carbon oxidized by 1 mole of MnO4 changing from $Mn7^{+}$ - Mn^{4+}; 0.02 L = volume of stock solution reacted; Wt = weight of air-dried soil sample in kg

The carbon management index was calculated as per the procedure of Blair *et al*. (1995).

$$Carbon\ pool\ index, CPI = \frac{Sample\ total\ carbon\ mg\ kg}{Reference\ total\ carbon\ mg\ kg} \quad [23]$$

$$Lability\ of\ Carbon, LC = \frac{Carbon\ oxidized\ by\ KMnO4}{Carbon\ remaining\ unoxidized\ by\ KMnO4} \quad [24]$$

$$Lability\ index, LI = \frac{Lability\ of\ carbon\ in\ sample\ soil}{Lability\ of\ carbon\ in\ reference} \quad [25]$$

Carbon Management Index CMI = CPI × LI × 100 [26]

Total carbon stock was calculated by multiplying the concentrations of the various carbon pools by the respective bulk densities of each experimental plot and the sampling depth of the soil (15 cm).

Total carbon stock t ha^{-1}

$$= [\rho_b (\mathrm{g\ cm^{-3}}) \times \mathrm{soil\ depth\ (cm)} \times \mathrm{total\ carbon\ content\ (g\ kg^{-1})}] \times 100 \quad [27]$$

Where: ρ_b = the soil bulk density

3.4.12 Soil physical analyses

3.4.12.1 Determination of soil water retention at near saturation, at field capacity and dry limit

Measurement of water retention at near saturation was performed at constant temperature of 20 °C. The intact core samples were placed in a sand box and saturated with water from underneath prior to imposition of suction levels. Suctions of−10 and−100 hPa were applied after saturation to establish matric potentials. The soil water retention at field capacity (−300 hPa) was determined by first weighing and slowly saturating the samples from below in a sandbox.

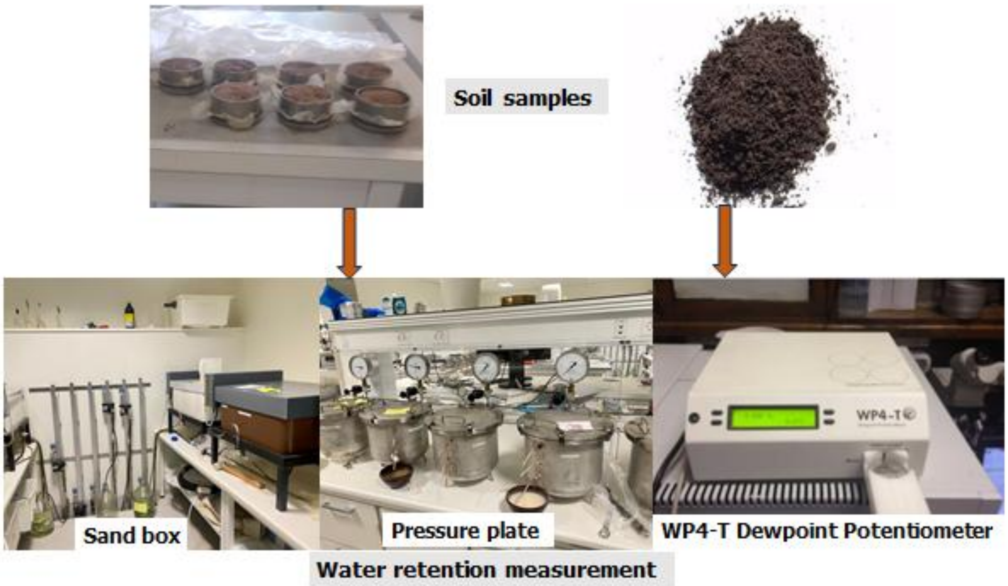

Figure 21: Experimental procedure used for determination of soil water retention for the wet end (sandbox and pressure plate) and permanent wilting point (Potentiometer).

After that, the samples were placed in a *Richard pressure plate* apparatus (Figure 21) and equilibrated for about three weeks at matric potential of−300 cm.

The soil water retention at matric potential of −15000 cm (wilting point) was determined on the air-dried and sieved < 2 mm samples with a temperature compensated WP4-T Dewpoint Potentiometer (METER Group Inc., Pullman, WA, USA), (2002). Air-dry subsamples from composite samples were oven-dried to determine the prevailing water content. Approximately 2 g of an air-dried subsample was placed in the instrument (Figure 21) and the matric potential and soil mass were simultaneously measured. The measurements were done on six replicates. After the measurement, soil samples were oven-dried at a temperature of 105 °C for 24 h to determine the gravimetric water content.

3.4.12.2 Soil air permeability measurement

After soil equilibration at −10, −100 and −300 hPa, soil air permeability was measured by the Forchheimer approach described by Schjønning and Koppelgaard (2017). Four corresponding values of pressure difference at values around 5, 2, 1 and 0.5 hPa, were applied across the soil sample placed in an air permeameter (Figure 22), and the resulting air flow was measured. For the purpose of quality control, standard tests with actual pressure difference at target air flow, was performed prior to soil samples measurement, in four series of steps. Darcy's law (Equation 28) was then used to calculate k_a in a steady state.

$$Q = KA\,(\Delta p/L) \qquad [28]$$

Where: Q = the volumetric flow rate; K = permeability of the medium; A = cross-sectional area; Δp = pressure drop, and L = given distance over which the pressure drop is computed.

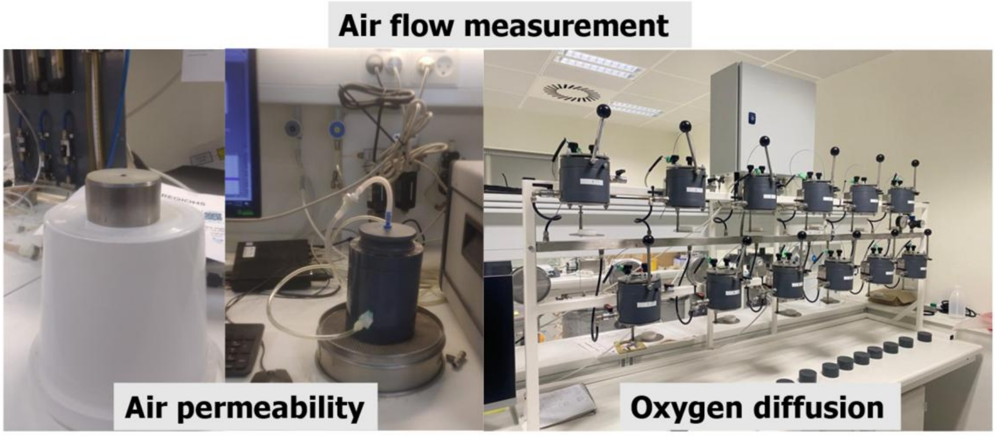

Figure 22: Measurement of soil air permeability and oxygen diffusivity.

3.4.12.3 Soil gas diffusion measurement

The oxygen diffusion in the soil was measured according to Taylor (1949) with an apparatus developed by Schjonning (1985). Briefly, the bottom of the core sample was fitted with a mesh cup and was placed in an air tight diffusion chamber (Figure 22). The chamber was then flushed with 100% N_2 gas to remove all oxygen. The top of the soil core was exposed to the atmosphere to allow atmospheric air to enter into the chamber through the soil sample. Further, the increase in oxygen concentration was measured by an electrode mounted on the chamber wall. The oxygen diffusion coefficient in soil (D_p) was calculated based on the method proposed by Rolston and Moldrup (2002). Relative gas diffusivity (D_p/D_o) was quantified as the ratio of D_p to the oxygen diffusion coefficient in free air ($D_o = 2.05\times10^{-5}$ m^2 s^{-1}).

3.4.12.4 Soil specific surface area estimation

For soil specific surface area determination, bulk soil samples from each plot were oven-dried for 48 hours to determine the hygroscopic water content after air drying at room temperature for a week. The specific surface area (SSA) of the soil was estimated from desorption of soil water content (%)

at a relative humidity level of 40% (RH) using a kaolinite-rich sample regression model (equation 29) developed by Yan *et al*. (2023).

$$SSA = 24.1 + (16.85 \times wh\ \%) \qquad [29]$$

Where: SSA= specific surface area in $m^2\ g^{-1}$, wh = the hygroscopic water content at a 40 (%) relative humidity

3.4.12.5 Modelling soil pore structure

The soil pore organization (PO), a parameter introduced by Groenevelt *et al*. (1984) to characterize pore size distribution and continuity, was calculated using the following equation:

$$PO = k_a/\varepsilon_a \qquad [30]$$

Additionally, soil pore tortuosity (τ), effective pore diameter (d_B), and the number of air-filled pores per cross-section area (n_B) were quantified using the tube model proposed by Ball (1988):

$$\tau = \left\{\frac{\varepsilon_a}{D_p/D_o}\right\}^{1/2} \qquad [31]$$

$$d_B = 2\left\{\frac{8k_a}{D_p/D_o}\right\}^{1/2} \qquad [32]$$

$$n_B = \frac{\sqrt{\varepsilon_a} \times (D_p/D_o)^{3/2}}{8\pi k_a} \qquad [33]$$

Where: k_a, = air permeability; ε_a, = air-filled porosity; D_p/D_o, = relative gas diffusivity

3.4.13 Greenhouse gas sampling and analysis

The potential greenhouse gas emissions were assessed using an intact soil core ($100\ cm^{-3}$) collected after two okra cropping cycles (12 months) following amendment application. The core samples were saturated in a sandbox and equilibrated at matric potentials of−10 and−100 (hPa). The mass difference between the two suctions was calculated. To optimize conditions for GHG emission, C and N nutrient sources were added to the soil samples, which were presumed to be low in carbon and nitrogen contents due to the time of sampling (12 months after amendment application). Specifically, a-D-glucose ($C_6H_{12}O_6$) at a rate of 160 kg ha^{-1} C and potassium nitrate (KNO_3) at 50 kg ha^{-1} N were applied as C and N sources respectively. Chloramphenicol was included in the solution to inhibit enzymatic activities that could potentially consume the added N source. The nutrient solution was prepared in a water volume corresponding to the mass difference between the two suctions, and slowly applied to the soil samples from the top using a pipette.

The soil cores were immediately transferred to 1 L glass jars with air-tight lids sealed with rubber gaskets and vacuum grease. Immediately after closure, gas samples were collected from the headspace at 0 (closure), 20, 40, and 60 minutes thereafter. A volume of 10 mL gas was extracted and transferred to 6 mL pre-evacuated exetainer vials. The gas samples were analysed using an Agilent 7890 gas chromatograph (GC), and emission fluxes (CO_2, N_2O and CH_4) were calculated using the HMR package in R (Flux estimation with static chamber data, R package version 0.31) (Pedersen, 2010). The fluxes of all the gases were set to zero if the difference between maximum and minimum concentrations measured during closure was smaller than the detection limit.

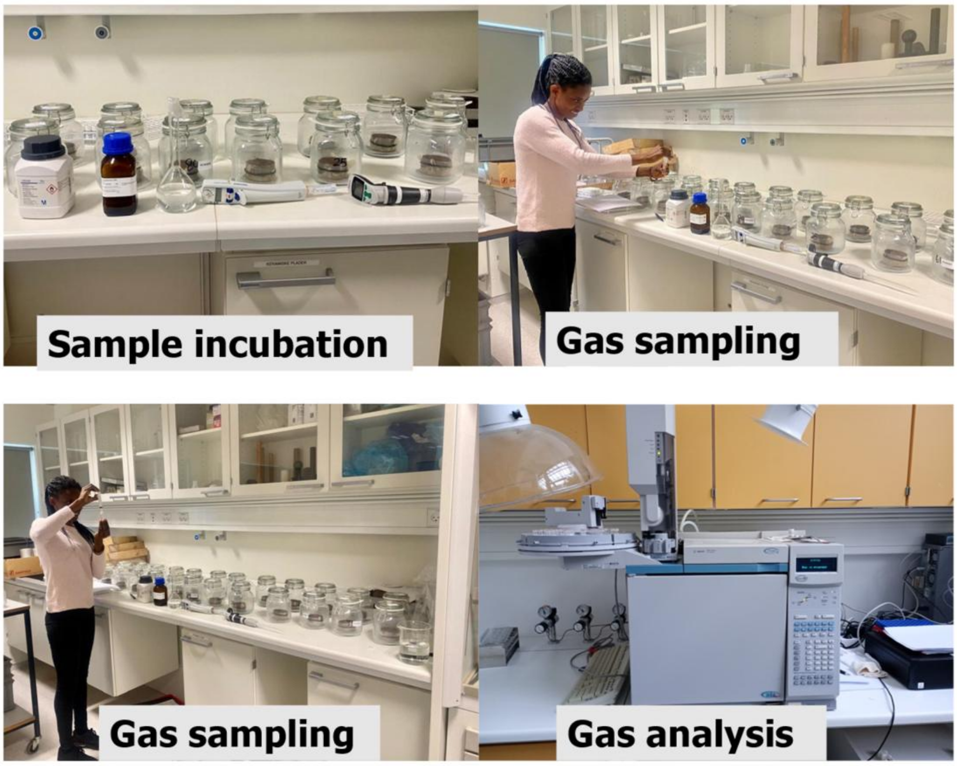

Figure 23: Incubation, sampling and analysis protocol for measuring greenhouse gases (N_2O, CO_2 and CH_4).

3.4.14 Bulk density and porosity determination

After water retention gas diffusion determination and GHG measurement, the soil cores were oven-dried at 105°C for 24 hours. The bulk density was calculated as the ratio of the oven-dried mass to the total volume of each soil core. The total porosity was estimated from the bulk density, assuming a particle density of 2.65 Mg m^{-3}. The volumetric water contents at−100,−300 and−15000 hPa were determined by multiplying the gravimetric water content by the corresponding bulk density of each sample. The air-filled porosity (ε_a) at−100,−300 and−15000 hPa was calculated as the difference between total porosity and volumetric water content at the same matric potential. The plant available water content (PAW cm^3 cm^{-3}) was calculated as the difference between the water content at−300 and−15000 hPa.

3.4.15 Statistical analyses

Data normality was assessed using the Shapiro-Wilk test, and variance homogeneity was evaluated with the Brown-Forsythe test before analysis. The soil air permeability and pore organization data were log-transformed to meet normality assumption before analyses. Subsequently, one-way ANOVA was performed to test treatment effects on soil chemical and microbial properties, gas transport, water retention, pore structures, bulk density, and potential greenhouse gas emission at−100 matric potential. Two-way ANOVA was also performed to establish the interactive effects of cropping cycles and EFB treatments on growth, yield and nutrient use efficiency parameters, POXC, TOC, CMI, LI, and carbon stock. The Tukey posthoc test was used for multiple means comparison at a significance level of 0.05.

Subsequently, the interrelationships between soil chemical and microbial properties with growth, yield and nutrient use efficiency parameters were determined using Pearson correlation analysis. Principal component analysis (PCA) was performed on a standardized correlation matrix with only soil chemical and microbial parameters that mostly correlated with okra yield and nutrient use efficiency in each crop cycle. Again, the relationships between soil physical, chemical, and microbial properties with greenhouse gas emission parameters were determined using Pearson correlation and Principal Component analyses (PCA). ANOVA was performed in GenStat 12th edition (VSN International, Hemel Hempstead, UK.), and correlation and principal component analysis were performed in OriginPro, (2024) (OriginLab Corporation, Northampton, MA, USA).

3.5 Chapter Summary

In order to test the five hypotheses outlined in the current study, the thesis was organised into three main phases. The first phase examined the elemental composition, molar ratios, nutrient content, heavy metal and polycyclic aromatic hydrocarbon (PAH) concentrations in EFB feedstock, two EFB biochar types; produced at $400–550^0$ for 30 minutes with a rotary reactor (B1); $300–400^0$ for 8 hours with locally designed kiln (B2) and three EFB compost produced from co-composting of EFB with poultry manure at mixing ratios of 1:1.5 (C1);1:2 (C2) and 1:3 (C3), respectively. All elemental, nutrient and toxicant analyses were conducted using standard methods approved by the International Biochar Initiative (IBI). The second phase of the research examined the effect of EFB biochar application loading capacity on soil chemical properties, maize phytotoxicity and nutrient uptake in four tropical soils (Acrisol, Brown and Red Ferralsols and Vertisol) in an incubation study, germination and pot experiments.

The third phase of the study was about application of EFB biochar and compost that examined the effect of EFB amendments on soil physical, chemical and microbial properties, okra yields and nutrient use efficiency, carbon storage, carbon management index and potential greenhouse emission after two okra cropping cycles. In a randomized complete block design, seven treatments: an unamended treatment [B0], sole EFB biochar applied at rates of 10 and 20 t ha^{-1} [B10, B20], sole EFB compost applied at 20 t ha^{-1} [CP20], combination of biochar and compost [B10CP20, B20CP20], and inorganic fertiliser [NPK], were evaluated in field experiments. Data on soil chemistry, soil nematodes, okra growth and yield were assessed for each crop cycle, and at the end of the second cycle, composite soil samples were taken for soil microbial biomass analysis, using the *Whole-Cell Fatty Acid* (WCFA) method.

The calculation of carbon management index (CMI) required samples of the soil of interest and a sample collected from a reference site. In our case, a 20-year-old orchard plantation adjacent to the experimental field was chosen. Since the continuity of carbon supply depends on both the total organic carbon pool size and the lability which is an estimate of turnover rate, both total organic carbon (TOC) and permanganate oxidizable carbon (POXC) were determined for the calculation for CMI for both crop cycles.

Bulk and intact 100 cm^3 core samples were extracted 12 months after application and physical properties (e.g., water retention, relative gas diffusivity, air permeability at matric potentials of−100 and−300 hPa) were measured. Subsequently, GHG emissions were measured on the soil cores at a soil water potential of−0 hPa after the addition of KNO_3 and glucose as nitrogen and carbon sources, respectively. Modelling for pore size distribution and architecture were done using standard models or equations.

Finally, all statistical test and analyses were performed using GenStat 12th edition (VSN International, Hemel Hempstead, UK.), and correlation and principal component analysis were performed in OriginPro, (2024) (OriginLab Corporation, Northamptonne, MA, USA).

CHAPTER FOUR

RESULTS AND DISCUSSION

The results and discussion section is presented according to the order by which the objectives or hypotheses appear. This chapter presents the results and discusses the effect of two different pyrolysis conditions and three EFB co-compost bulking ratios on the elemental, chemical and nutrients composition as well as the levels of PAH and heavy metals of the resultant EFB biochar and composts products. Also, the short-term chemical response and the potential phytotoxicity of different tropical soils to EFB biochar application are also discussed. Finally, this chapter presents the results and discussion on the effect of sole and combined application of EFB biochar and compost on soil physical, chemical and microbial properties, okra yield and nutrient use efficiency, carbon storage and greenhouse gas emission in a Haplic Acrisol.

4.1 Effect of Pyrolysis and Composting of Empty Oil Palm Fruit Bunch on Chemical Composition and Potential Toxicant Elements of Resultant Products

The differences in physicochemical characteristics of biochar stem from variations in the temperature, residence time and the design of the of pyrolysis reactor (Li *et al*., 2021) and for composts, the difference emanate from microbial degradation processes such as type of composting system, bulking C/N ratio and composting conditions (Eghball *et al*., 1997). This objective aims to provide insights into how these transformation methods of EFB into soil amendments could be optimised for informed decision on optimise processing and amendment choice for soil health and crop productivity. The first objective of the study examines the influence of pyrolytic reactors and feedstock mixing ratios on the chemical properties and levels of toxicants in EFB biochar and composts.

4.1.1 Differences in the elemental and chemical composition of the EFB products

The results on the elemental and chemical composition of EFB feedstock, biochar and co-composts are presented in Table 3. Pyrolysis of EFB using two different reactors and co-composting of EFB and poultry manure at different mixing ratios significantly ($P < 0.01$) affected the elemental composition of the biochar and composts. Pyrolysis and the composting conditions also significantly affected the organic carbon, carbonates, volatile matter and ash contents of the EFB feedstocks, biochar and compost types. Compared to the EFB feedstock, the contents of hydrogen (H) and oxygen (O) of the EFB biochar and compost types were significantly ($P < 0.001$) lower. The elemental carbon (C) of the EFB feedstock was significantly ($P < 0.001$) lower than the biochar 2 but was significantly ($P < 0.001$) higher than the compost types. However the trend of the elemental nitrogen (N) content of the feedstock, biochar and compost types was not consistent. Also, the elemental carbon content of the EFB biochar was significantly higher than the EFB composts.

Table 3: Elemental and chemical composition of the empty oil palm fruit bunch feedstock, biochars and co-composts

	Element (% d.m.)							Chemical properties (% d.m.)		
Amendments	**H**	**O**	**C**	**N**	**S**	**C_{org}**	**TIC**	**Carbonate**	**VM**	**Ash**
FS	5.7 ± 0.4a	37.4 ± 2.6a	44.1 ± 3b	0.9 ± 0.06c	0.10 ± 0.01b	44.1 ± 3b	--	--	42.9 ± 3.0a	6.7 ± 0.5c
BI	1.05 ± 0.07c	7.9 ± 0.4d	44.8 ± 2.3b	0.82 ± 0.45c	0.14 ± 0.01b	44.0 ± 2.3b	0.8 ± 0a	2.95 ± 0.15a	8.9 ± 0.5c	41.6 ± 2.1b
B2	2.45 ± 0.15b	13.8 ± 0.4cd	63.1 ± 3.9a	1.47 ± 0.09a	0.24 ± 0.02a	62.7 ± 3.9a	0.5 ± 0.05d	1.5 ± 0.0e	14.6 ± 1.0bc	16.4 ± 1.1c
C1	0.97 ± 0.04c	19.6 ± 0.9bc	10.5 ± 0.45c	1.10 ± 0.05bc	0.21 ± 0.01a	9.9 ± 0.45c	0.6 ± 0b	2.15 ± 0.05b	18.7 ± 0.8b	66.7 ± 2.9a
C2	1.54 ± 0.07bc	21.5 ± 0.9b	14.2 ± 0.6c	1.29 ± 0.06ab	0.24 ± 0.01a	13.7 ± 0.55c	0.6 ± 0b	2.15 ± 0.05b	22.1 ± 1.0b	59.8 ± 2.5a
C3	1.36 ± 0.5c	14.6 ± 0.5bcd	12.8 ± 0.7c	1.14 ± 0.05bc	0.20 ± 0.01a	12.4 ± 0.45c	0.4 ± 0e	1.55 ± 0.05d	15.6 ± 0.6bc	68 ± 2.5a
***P* value**	**< 0.001**	**< 0.001**	**< 0.001**	**0.0017**	**0.001**	**< 0.001**	**0.0026**	**0.001**	**< 0.001**	**< 0.001**

Each value is the mean (± standard error) of three replicates; Different letters in a column indicate significant differences (Tukey test at $P < 0.5$). FS, (EFB feedstock); B1, (biochar 1); B2, (biochar 2); C1, (30:1 compost); C2, (35: 1 compost); C3, (40:1 compost); H, hydrogen; O, oxygen; C, carbon; N, nitrogen; S, sulphur; Corg, organic carbon; VM, volatile matter; TIC, total inorganic carbon; d.m., dry matter.

Moreover, between the two EFB biochar types, H, C, N and S of B2 (locally manufactured kiln with temperatures that range between 300–400 °C and residence 8–10 hours) were 1.3, 0.4, 0.79 and 0.7 folds, respectively, significantly higher than the corresponding values in B1 (continuous rotary pyrolysis reactor with temperatures range between 400–550 °C and residence time of 30 minutes). However, the O content did not differ significantly ($P > 0.05$) between the two biochar types. Among the three EFB composts, C2 (C/N 35:1) had the highest elemental H, O, C, N and S contents but were not significantly ($P > 0.05$) different from C1 (C/N 30: 1) and C3 (C/N 40:1).

The organic carbon (C_{org}) content was highest in the B2, followed by the feedstock and BI and the composts type C1 recorded the least among the amendments. The TIC and carbonate content of EFB feedstock were not detectable. The highest total inorganic carbon (TIC) and carbonate contents were found in the BI and was significantly ($P < 0.01$) higher than the B2 and the compost types. Again, the EFB feedstock had the highest volatile matter (VM) but the lowest ash contents, compared to the EFB biochar and compost types.

The significant ($P < 0.001$) reduction in the elemental H and O content of EFB biochar types compared to the EFB feedstock is similar to the findings of Jindo *et al*. (2014), Souza *et al*. (2021) and Wijitkosum and Sriburi (2023), who observed decreases in H and O content in biochar produced from various biomass sources. This reduction was likely due to processes such as dehydration, volatilisation of compounds (e.g., CO, CO_2, H_2O, and hydrocarbons), reduction of hydroxyl (− OH) functional groups, and condensation during pyrolysis.

The biochar produced at 300–400°C for 8 hours (B2) exhibited significantly higher elemental H, C, N, C_{org}, and S content compared to biochar produced at 400-550°C for 30 minutes (B1). In this study, biochar produced at 400°C had the highest C content, but a decrease was observed when the pyrolysis temperature exceeded 500°C, which is consistent with the results of Wijitkosum and Sriburi (2023). Previous studies by Guizani *et al*. (2019) indicated that the aromatisation process begins at approximately 350°C and continues to increase. However, the temperature at which maximum aromatisation is achieved depends on the feedstock type, whether woody or non-woody (Tomczyk *et al*., 2020). Nitrogen loss mainly occurs through the volatilisation of different nitrogen groups such as NH_4-N or NO_3-N at low temperatures and pyridine at temperatures > 600 °C (Souza *et al*., 2021). The loss of S from the biochar is due to sulphur-containing volatile organic compounds such as carbonyl sulphide. The nitrogen content of biochar was also reported to increase with the increment of residence time at a lower pyrolysis temperature.

The decrease in the volatile matter but increase in ash, total inorganic carbon (TIC) and carbonate contents with increasing pyrolysis temperature as observed agreed with the findings of Keiluweit *et al*. (2010), Elaigwu *et al*. (2014) and Zama *et al*. (2017), who produced biochar from various feedstocks at varying pyrolysis temperatures. They related the disappearance of acidic functional groups (–COOH) and the appearance of basic functional groups with increasing temperature contributed to the higher TIC and carbonate content of the biochars.

The EFB composts also had relatively low H, O, and C compared to the EFB feedstock, indicating that volatilisation of organic compounds such as aliphatic, aromatic and sulphur-containing compounds (H_2S, COS, CH_3S, $(CH_3)_2S_2$), as well as NH_3, might have occurred during microbial degradation of the feedstocks. Similarly, Kuo *et al*. (2004) found a reduction of the carbon content of the composts compared to their feedstocks and attributed it to the release and incorporation of organic C as CO_2 into microbial cells during the composting process. The increased N and ash contents of composts compared to the EFB feedstock perhaps stemmed from the mineralization of organic matter and progressive concentration of inorganic constituents during microbial degradation, together with additional N from the poultry manure, as observed by de Bertoldi (1982). The study suggested that differences in pyrolysis techniques and composting conditions of feedstocks eventually affect the elemental volatile and ash composition of the final products.

4.1.2 Effect of pyrolysis and composting on the elemental ratios of the different EFB products

The effect of the pyrolysis techniques and co-composting of EFB on the elemental ratios are presented in Table 4. Pyrolysis and composting of EFB significantly ($P < 0.001$) affected the elemental ratios compared to the EFB feedstock. Specifically, the H/C ratio (aromaticity index) was significantly ($P < 0.001$) lower in the biochar and compost types, compared to the EFB feedstock. Also, pyrolysis decreased the H/C ratio by an average of 3.3 times compared to composting of EFB. Between the biochar types, B1 had a significantly lower H/C ratio compared to B2 and among the composts, C1 recorded the least H/C ratio of 1.11 which was significantly ($P < 0.001$) lower than C2 and C3.

Generally, the O/C ratio (hydrophobicity index) was significantly ($P <$ 0.001) increased by composting but decreased by the pyrolysis of EFB (Table 4). The B1 recorded the lowest O/C ratio of 0.013 which was significantly (P < 0.001) lower than B2 (0.16), the compost types and the feedstock. Among the compost types, the C1 recorded the highest O/C ratio, followed C2 and C3.

Moreover, the H/C_{org} ratio (carbon stability index) was highest in the EFB feedstock, followed by the composts and then the biochar types. The H/C_{org} ratio of the B1 was 0.28 which was significantly ($P < 0.001$) lower than the H/C_{org} ratio value of 0.47 obtained in the B2. Also, the H/C_{org} ratio varied significantly ($P < 0.001$) among the compost types, with the highest in the C2. The lowest (O+N)/C ratio (polarity index), was recorded in B1, which was significantly ($P < 0.001$) lower than B2. Composting of the EFB significantly ($P < 0.001$) increased (O+N)/C ratio of all the composts compared to the feedstock and biochar types.

Moreover, the C/N increased significantly ($P < 0.001$) from 49.0 ± 0.07 in the feedstock to 54.9 ± 0.23 in BI and declined to 42.9 ± 0.02 in B2. Composting on the other hand significantly decreased the C/N ratio of the EFB, where C1 had the lowest C/N ratio which was significantly ($P < 0.001$) lesser than C2 and C3 (Table 4). Meanwhile, EFB pyrolysis increased significantly ($p < 0.001$) the H/C_{org} molar ratio and C_{org} content as an index of stock sBC+100 or carbon storage value from 211 g kg^{-1} in the feedstock to 308 g kg^{-1} and 313 g kg^{-1} in B1 and B2 respectively.

Table 4: Elemental ratios of the empty oil palm fruit bunch feedstock, biochars and co-composts

EFB Amendments	H/C	O/C	H/C_{org}	(O+N)/C	C/N	sBC+100 (g kg^{-1})
EFB feedstock	1.56 ± 0.01a	0.64 ± 0.0d	1.55 ± 0.0a	0.87 ± 0.0d	49.0 ± 0.07b	211 ± 1.0
B1	0.28 ± 0.0e	0.13 ± 0.0f	0.28 ± 0.0e	0.19 ± 0.0f	54.9 ± 0.27a	308 ± 1.5
B2	0.46 ± 0.0d	0.16 ± 0.0e	0.47 ± 0.01d	0.24 ± 0.0e	42.9 ± 0.03c	313 ± 2.2
C1	1.11 ± 0.01c	1.41 ± 0.0a	1.18 ± 0.0c	1.98 ± 0.0a	9.5 ± 0.02e	--
C2	1.28 ± 0.0b	1.13 ± 0.0b	1.34 ± 0.06b	1.60 ± 0.0b	11.1 ± 0.01d	--
C3	1.26 ± 0.0b	0.86 ± 0.0c	1.31 ± 0.0b	1.23 ± 0.0c	11.3 ± 0.01d	--
p value	**< 0.001**	**< 0.001**	**< 0.001**	**< 0.001**	**< 0.001**	

Each value is the mean (± standard error) of three replicates; Different letters in a column indicate significant differences (Tukey test at $P < 0.5$). B1, (biochar 1); B2, (biochar 2); C1, (30:1 compost); C2, (35: 1 compost); C3, (40:1 compost); H/C, hydrogen-carbon ratio; O/C, oxygen-carbon ratio; H/C, hydrogen-carbon ratio; H/ C_{org} , hydrogen-organic carbon ratio; (O+N)/C, oxygen and nitrogen-carbon ratio;C/N carbon-nitrogen ratio.

In the current study, pyrolysis of EFB significantly ($P < 0.001$) decreased the elemental ratios of the EFB. The molar H/C and H/C_{org} ratios indicate the degree of carbonisation and biochar stability. A biochar is considered to be completely pyrolyzed if the H/C_{org} ratio is < 0.7 (Schimmelpfennig & Glaser, 2012). The low H/C of the EFB biochar types was due the increased aromatisation of carbon structures from hydrocarbons into aromatic rings during pyrolysis (Schmidt *et al*., 2021). Aromaticity serves as a predictor of the longevity of biochar, and earlier works by Crombie *et al*. (2015) and Wijitkosum and Sriburi (2023) have established that an H/C ratio lower than 0.7 indicated that the biochar possesses better fused aromatic ring structures, making it highly stable and resistant to oxidation. The comparison of two biochar types; B1 (H/C ratio of 0.28) and B2 (H/C ratio of 0.46), suggested that increased pyrolysis temperature led to an increased aromatisation process. In contrast, composting of EFB only decreased the H/C atomic ratios to a certain extent, failing to confer resistance to oxidation due to their atomic H/C ratios being greater than 0.7.

Furthermore, increasing pyrolysis temperature correspondingly led to lower O/C and (O + N)/C ratios of the EFB biochar types. Agreeably, Zhao *et al*. (2019) found that H/C, O/C and (O+N)/C atomic ratios of rice straw biochar, bamboo biochar, and cow manure biochar, charred at varying ranged temperatures decreased with increasing pyrolysis temperature from 300−700 °C. They attributed it to the decarboxylation, carbonisation, or the formation of solid carbon structures as temperature increases during pyrolysis. Additionally, based on the spectrum of the O/C molar ratio, biochar with O/C < 0.2 is considered highly stable with a half-life >1000 years and may be highly hydrophobic (Elmquist *et al*., 2006).

The C/N ratio is a valuable indicator for evaluating the capacity of biochar to mineralize or immobilize nitrogen (N) when applied to the soil (Paiva *et al*., 2024) and it was significantly increased by increasing pyrolysis temperature and lower resident time in the current study. Consistent with the current study, Wijitkosum (2022) noted that the C/N ratio was increased by increasing pyrolysis temperature. However, a contradictory result was obtained by Wijitkosum and Sriburi (2023) where the C/N ratio decreased with increasing temperature when disposable bamboo-chopsticks waste was charred. According to Paiva *et al*. (2024), the carbon-to-nitrogen (C/N) ratio of a biochar reflects the impact of both the pyrolysis conditions which included temperature, pressure and residence time on their carbon (C) and nitrogen (N) contents, highlighting the dissociation between losses of C and N compounds during pyrolysis.

C/N ratio is a key indication of N loss from a composting process. It was also one of the factors used to indicate compost maturation (Paul *et al*., 1989). The C/N ratios of the composts were significantly lower relative the EFB feedstock. Among the composts, the C/N ratio of the C1 was significantly lower than the C/N ratio of C2 and C3, indicating that the starting C/N of compost affects the C/N ratio of the final compost. According to Paul *et al.* (1989), a C/N ratio of 20 or less is accepted as mature compost and the all the C/N ratios of the EFB composts from current study were below 20. Contrary to the results from the current study, Paul *et al.* (1989) found no difference in the changes of C/N ratio in the compost pile with the same initial C/N ratio.

They indicated that the initial C/N ratio was the main factor affecting the time to reach maturity of the compost. Evaluation of C and N transformations from organic amendments is very essential for their better use as a source of N in crop production (Antil *et al*., 2011). Consistent with the results from a study by Kuo *et al*. (2004), who reported that composting decreases the C/N ratio of organic materials suggesting the potential use of compost as a source of nitrogen for crop production.

The carbon storage class is a measure of the estimated long-term (i.e., 100 year) soil carbon storage potential of a biochar. Biochars are classified by the quantity of organic carbon (C_{org}) in grams per kilogram estimated to remain in soil for at least 100 years (BC+100) (IBI, 2012). Notably, both B1 and B2 biochars fall within the proposed classification system identified by the IBI Stable Carbon Protocol as Class 2 (sBC+100 ranging between 300 and 400g kg^{-1}) (IBI, 2012), demonstrating their similar recognition despite differing Corg and H/Corg ratios. Similar to the current studies, biochar produced from corn stover, bull manure and willow at 350 and 400^0 fell into the IBI Stable Carbon Protocol Class 2 (Pereira *et al*., 2011; Enders *et al*., 2012). The study suggested that differences in the pyrolysis affect the aromaticity, polarity, hydrophobicity and carbon storage value of biochars from the same feedstocks but the temperature range at which these differences occur may vary for different feedstocks.

4.1.3 Nutrient content and pH of the EFB biochars and composts

The plant nutrient content and pH of EFB biochars produced using two different pyrolysis techniques and co-composting of EFB and poultry manure at different C/N ratios are presented in Table 5. The potassium and sulphur content of B2 were respectively, 102% and 96% higher than B1. Conversely, the sodium, boron, sulphur, calcium, iron, zinc and manganese contents of B1 were higher than the B2. Among the EFB compost, the nutrient contents of the C3 were generally lower than the C1 and C2.

Table 5: Plant nutrient and pH contents of the empty oil palm fruit bunch biochar and co-composts

Plant nutrients (g kg^{-1} d.m.)	**B1**	**B2**	**C1**	**C2**	**C3**
Phosphorus (P_2O_5)	6.6	4.3	13.69	9.62	5.1
Potassium (K_2O)	35.5	71.8	12.58	12.2	10
Magnesium (MgO)	15.2	11.2	5.18	4.81	2.96
Calcium (CaO)	24.9	10.4	35.9	21.85	21.19
Sodium (Na_2O)	1.5	0.3	2.22	1.85	1.48
Boron	0.041	0.028	0.025	0.027	0.026
Sulphur (SO_3)	2.7	5.3	4.07	2.96	2.22
Manganese	0.268	0.056	0.243	0.216	0.196
Iron (Fe_2O_3)	10.21	0.3	6.29	7.4	5.9
Zinc	0.170	0.085	0.153	0.129	0.102
Silicon (SiO_2)	137	17.5	253.87	270.1	290.8
pH ($CaCl_2$)	9.9	10.1	8.1	8.3	8.4

B1, (biochar 1); B2, (biochar 2); C1, (30:1 compost); C2, (35: 1 compost); C3, (40:1 compost).

The C1 was superior to the other composts in terms of phosphorus, calcium, sulphur and manganese content. Meanwhile, the trend of effect of pyrolysis and co-composting of EFB on nutrient content were not consistent, for instance, the magnesium and potassium content of the biochars were higher than the composts while the content of silicon was higher in the composts than the biochar (Table 5). The biochars and composts produced from the EFB were all alkaline in nature with pH ($CaCl_2$) ranging between 8.1 and 10.1 but the pH of the biochars was higher than the composts (Table 5).

Pyrolysis or composting alters the physical and chemical characteristics of an organic material, including its nutrient content, thereby influencing its effects on soil functions (Naeem *et al*., 2014). Among the macronutrients, higher pyrolysis temperatures decreased the K_2O and SO_3 and increased the P_2O_5, CaO, and MgO levels in the EFB biochars. The content of micronutrients in biochar also increased with higher pyrolysis temperatures (400–500 °C). Similarly, Hadi and Norazlina (2021) found that EFB pyrolysed at 350°C, 500 °C, and 750 °C had the highest levels of P, K, and Mg at 500 °C, with a marginal decline at 750 °C. Knicker (2007) explained that potassium and phosphorus vaporize at temperatures above 760 °C, while magnesium and calcium are lost only at temperatures exceeding 1107 °C and 1240 °C, respectively. Thus, nutrients are more likely to be retained in biochar produced at peak temperatures of around 500°C. Naeem *et al*. (2014) observed that pyrolysis of wheat and rice straw at 300–500°C resulted in increased concentrations of all nutrients, except nitrogen, with the highest nutrient levels found in biochar produced at 500°C. Chan and Xu (2009), along with Naeem *et al*. (2014), attributed the increase in nutrient content at higher temperatures to the loss of volatile compounds (C, H, and O) and relatively minor losses of alkali nutrients in the gaseous phase.

For the EFB composts, the sample with the highest poultry manure content and an initial C/N ratio of 30:1 had the highest nutrient content. Agreeably, Lim *et al*. (2015) reported that EFB compost with one-part EFB and three-part cow dung yielded higher total calcium, phosphorus, and magnesium levels compared to composts with less cow dung. Kamolmanit and Reungsang (2006) also observed higher percentages of total K and P in compost piles with more swine manure. Tibu *et al*. (2019) suggested that the nutrient concentration in the final compost results from the shrinking organic matter content during composting. This phenomenon may also explain the higher potassium and magnesium content in biochar compared to compost. Examining the nutrient contents of a feedstock after their conversion in to biochar or compost is very important, but a deeper understanding of their impact on nutrient release and availability after incorporation in the soil is also affected by other chemical and physical properties of both the amendment and soil.

4.1.4 Heavy metal concentration of the EFB feedstock, biochars and composts

The heavy metal contents of the EFB biochars and composts are presented in Table 6. Most of the heavy metals including in the EFB feedstock (FS) were below the detection limits (< 0.8 for arsenic, < 0.2 for lead and cadmium, < 0.05 for mercury, < 5 for silver and < 38 for zinc). Moreover, the chromium and nickel contents of the feedstock were higher than the biochar and composts. However, the copper content of the biochars and composts were on average 233% and 76% respectively, higher than the EFB feedstock.

Between the two biochar types, the heavy metals were higher in the B1 compared to the B2. For instance, the zinc content in the BI was 2.0 folds higher compared to that of B2. Composting on the other hand, increased the arsenic, chromium, lead and nickel contents compared to the biochar types. The copper and zinc contents of the compost were higher than the EFB feedstock. Generally, the heavy metals did not vary substantially among the three different composts (Table 6).

Table 6: Heavy metal contents of the empty oil palm fruit bunch feedstock, biochar and composts

Heavy metals (mg kg^{-1} d.m.)	**FS**	**B1**	**B2**	**C1**	**C2**	**C3**
Arsenic (As)	< 0.8	1.6	< 0.8	2.3	2.6	1.9
Lead (Pb)	< 0.2	5	< 2	11	11	10
Cadmium (Cd)	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2	< 0.2
Chromium (Cr)	31	9	< 1	22	26	24
Copper (Cu)	15	51	50	30	27	22
Nickel (Ni)	19	< 1	1	7	6	7
Mercury (Hg)	< 0.05	< 0.07	< 0.07	<0.05	<0.05	<0.05
Silver (Ag)	< 5	< 5	< 5	< 5	< 5	< 5
Zinc (Zn)	< 38	170	85	153	127	102

FS, (Feedstock); B1, (biochar 1); B2, (biochar 2); C1, (30:1 compost); C2, (35: 1 compost); C3, (40:1 compost).

Pyrolysis of EFB in the continuous rotary reactor at 400-550^0 increased the contents of arsenic, lead, copper, and zinc but decreased the chromium content in the biochar (B1). Similarly, Wang *et al*. (2017) also observed that the concentrations of heavy metals in chicken manure and water-washed swine manure biochar produced at temperatures ranging from 200–800^0 were all higher than those in their original feedstocks. EBC (2022), also noted that the weight of the original feedstock is reduced by over 50% during pyrolysis due to the loss of carbon, hydrogen, and oxygen, leaving behind the heavy metals, leading to increased concentration.

Moreover, the co-composting EFB amplified the contents of lead, copper, arsenic, manganese, and zinc compared to the original feedstock but the cadmium, silver, and mercury contents of the compost remained unchanged. According to Kuo *et al*. (2004), metal concentrations in finished composts tend to be higher than in the initial composting mix due to mass reduction from C loss as CO_2 or degradation of organic matter. Conversely, Kuo *et al*. (2004) reported a rapid decline of water-soluble lead and zinc during composting. Tibu *et al*. (2019) reported that nickel and arsenic content of compost piles decreased, while the content of Cr, Zn, Cd, and Pb increased throughout the composting of various municipal wastes at different bulking ratios. Moreover, the poultry manure addition to EFB at different ratios increased varying zinc and copper concentrations. This suggested that the concentration of metals in the finished compost may vary from the feedstock depending on the quantity of bulking agent used. Nonetheless, the concentrations of all the nine heavy metals in the EFB feedstock, biochars, and composts were below the threshold limit set by IBI (2015) and EBC (2022) for agro-organics, offsetting the danger of heavy metal contamination when they were applied to the soil.

4.1.5 Polycyclic aromatic hydrocarbon (PAH) content of the EFB feedstock, biochars and composts

The polycyclic aromatic hydrocarbon (PAH) contents in the EFB feedstock were all below detection limit apart from the content of Phenanthrene (Table 7). Also, the PAH contents in the three compost types C1, C2 and C3 were all below detection limit. Pyrolysis of EFB on the hand, generally increased the PAH contents with the exception of benzo(b)fluoranthene, indeno (1,2,3-cd) pyrene and dibenz(a,h)anthracene of the biochars (Table 7). The most dominant PAH was naphthalene with a value of 13 mg kg^{-1} d.m and 14 mg kg^{-1} d.m in B1 and B2, respectively. Between the two biochar types, acenaphthene and fluorene increased by five and four-folds, respectively in the B2 compared to the B1, but the acenaphthylene content of B1 was 3-folds higher than the B2.

Table 7: Polycyclic aromatic hydrocarbon (PAH) contents of the empty oil palm fruit bunch feedstock, biochar from two reactors and co-composts

PAHs (mg kg^{-1} d.m.)	FS	B1	B2	C1	C2	C3
Naphthalene	< 0.1	13	14	< 0.1	< 0.1	< 0.1
Acenaphthylene	< 0.1	0.6	0.2	< 0.1	< 0.1	< 0.1
Acenaphthene	< 0.1	0.1	0.5	< 0.1	< 0.1	< 0.1
Fluorene	< 0.1	0.3	1.2	< 0.1	< 0.1	< 0.1
Phenanthrene	0.2	2.7	3.2	< 0.1	< 0.1	< 0.1
Anthracene	< 0.1	0.6	0.7	< 0.1	< 0.1	< 0.1
Fluoranthene	< 0.1	0.9	1.1	< 0.1	< 0.1	< 0.1
Pyrene	< 0.1	0.9	1.0	< 0.1	< 0.1	< 0.1
Benz(a)anthracene	< 0.1	0.3	0.4	< 0.1	< 0.1	< 0.1
Chrysene	< 0.1	0.4	0.4	< 0.1	< 0.1	< 0.1
Benzo(b)fluoranthene	< 0.1	0.2	0.3	< 0.1	< 0.1	< 0.1
Benzo(k)fluoranthene	< 0.1	< 0.1	0.1	< 0.1	< 0.1	< 0.1
Benzo(a)pyrene	< 0.1	0.2	0.3	< 0.1	< 0.1	< 0.1
Indeno(1,2,3-cd)pyrene	< 0.1	< 0.1	0.1	< 0.1	< 0.1	< 0.1
Dibenz(a,h)anthracene	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Benzo(g,h,i)perylene	< 0.1	< 0.1	0.2	< 0.1	< 0.1	< 0.1
Total 8 EFSA EPA	-	1.2	1.8	nc	nc	nc
Total 16 EPA-PAH	0.2	20.3	23.7	nc	nc	nc

EPA, Environmental protection agency; PAHs, polycyclic aromatic hydrocarbons; EFSA, European Food Safety Association; nc, non-calculable; B1, (biochar 1); B2, (biochar 2); C1, (30:1 compost); C2, (35: 1 compost); C3, (40:1 compost).

Furthermore, the sum of 16 PAHs in the EFB feedstock was 0.2 mg kg^{-1} d.m. Those of the EFB composts were below detection limit. However, pyrolysis increased the sum of 16 PAHs concentration from 2 mg kg^{-1} d.m in the EFB feedstock to 20.3 and 23.7 mg kg^{-1} d.m. in the B2 and B1 respectively. Again, among the eight more complex (carcinogenic) PAHs, five out of eight were found in B1 while seven out of eight were detected in the B2, with sum of European Food Safety Authority's (EFSA) 8 PAHs of 1.2 and 1.8 mg kg^{-1} on dry weight basis, respectively.

In the current study, most of the PAHs in all three EFB co-composts were below the detection limit of < 0.1 mg kg^{-1} (d.m.), even though phenanthrene was present in the EFB feedstock. Similarly, Amir *et al*. (2005) reported a PAH decrease of about 75% when lagooning sewage sludge was composted. They attributed the reduction in PAHs to biodegradation due to the intense microbial activity during the thermophilic phase of composting.

However, both EFB biochars contained total PAH levels above the International Regulatory Maximum Allowed Thresholds (MATs) of 6 mg kg^{-1} (d.m), with the highest level recorded in B2 at 23.7 mg kg^{-1} (d.m). Previous studies have shown that, besides the type of feedstock, pyrolysis conditions such as temperature, residence time, and reactor design, play a significant role in the concentrations of PAHs in biochar (Bucheli *et al*., 2015; Wang *et al*., 2017). Feedstock was found to have negligible effects on the formation of PAHs, as the detection limit for 15 out of the 16 'priority pollutants' in the EFB feedstock was below 0.1 mg kg^{-1} (dry matter). Similarly, studies by Bucheli *et al*. (2015) and Buss *et al*. (2022) make it apparent that PAHs in biochar cannot be associated with feedstock types. Buss *et al*. (2022) further noted that a large variety of feedstocks, from plant residues, sewage sludge, and demolition wood yielded biochars with increased levels of PAHs under certain production conditions.

The differences in the levels of PAHs in the two EFB biochars (B1 and B2) may be due to the different production conditions. B1 was produced in a rotary reactor at 400–550°C with a residence time of 30 minutes, while B2 was pyrolysed in a locally designed kiln at 300–400°C with a residence time of 8–10 hours, with no separate chamber for the removal of syngas and tar compounds and oils. This corroborated with Freddo *et al*. (2012) who found that the sum of 16 EPA-PAH levels in the biochar samples obtained from redwood, bamboo, maize and rice at 300–600 ºC decreased as the temperature increased. Hale *et al*. (2012) also reported a similar PAH trend when they analysed 50 biochar samples obtained in a temperature range of between 250–900ºC. Greco *et al*. (2021) attributed aromatisation, cyclisation, dehydrogenation and dealkylation as the main reactions involved in PAH formation at relatively low pyrolysis temperatures (below 500 ◦C). However, Weidemann *et al*. (2018) found that wheat straw-derived biochars produced at 700°C contained higher concentrations of PAHs than those produced at 550°C. They explained that the concentrations increased with increasing temperature because they became more concentrated as other components of the raw material were released from the solid matrix and the amount of solid matrix decreased.

Buss *et al*. (2022), on the other hand observed that the condensation effect of pyrolysis vapours in cooler zones during pyrolysis operations was of great significance and it surpassed the effect of temperature, carrier gas flow, or feedstock on PAH concentrations in biochar. Therefore, it was likely that the B2 biochar, produced from the local kiln, was contaminated with pyrolysis vapours containing PAHs. Similarly, José *et al*. (2016) observed 10 times increase in soil PAH content in kiln wood biochar. The increase in the PAH content in B2 could also be attributed to the usage of traditional kilns in which syngas and tar oils are not removed. Similarly, work by Buss (2022) demonstrated that the most important factor yielding high contents of toxic PAHs in biochar was the pyrolysis unit design, suggesting the importance of considering the various production conditions and their potential impacts on the levels of PAHs in biochar.

4.1.6 Summary

The chemical, nutrient, and toxicant contents in soil amendments, particularly biochars and composts are influenced by pyrolysis or composting conditions, and these factors impact the quality of the amendment with environmental ramifications when the products with high PTEs are applied to the soil. In this study, the elemental composition, elemental ratios, nutrient content, heavy metal, and PAHs levels of Empty Fruit Bunch (EFB) feedstock, two EFB biochars produced from two different but locally available reactors, and three EFB composts produced from co-composting of EFB with poultry manure at different mixing ratios were assessed.

The study results revealed that pyrolysis of EFB at 300–400^0 for 8 hours (B2) resulted in significantly higher H, C, N, Corg, and S content compared to EFB pyrolysis at 400–550^0 for 30 minutes (BI). Pyrolysis under increasing temperature resulted in lower H/C, O/C, O/Corg atomic ratios of the biochar, indicating high aromatization, polarity, and hydrophobicity. Higher pyrolysis temperature decreased K_2O and SO_3 content but increased

P_2O_5, CaO, MgO, and micro nutrient contents in the biochar. However, analyses of toxicant levels revealed that pyrolysis of EFB at 400– 550^0 increased the contents of arsenic, lead, copper, and zinc. Additionally, both biochars had PAHs levels above allowable thresholds, with B2 having the highest levels. The high PAHs content in B2 was likely due to contamination during the cooling stage.

On the other hand, the EFB composts had relatively low H, O, C contents and a high N and ash content compared to the original EFB feedstock. The compost with the highest poultry manure content had the highest nutrient content and lowest C/N ratio, suggesting that the N and C/N ratio of the starting pile greatly affects the final C/N ratio of compost. In summary, EFB feedstock had high carbon content, low toxicant concentrations, and a considerable amount of essential macro and micro nutrients. However, the conversion pathway, whether pyrolysis or composting can alter its elemental composition, nutrient content, and toxicant content, impacting its use for soil health restoration.

4.2 Evaluation of Immediate Chemical Response and Potential Phytotoxicity of Different Tropical Soils to Empty Oil Palm Fruit Bunch Biochar Application

The impact of different types of biochar on soil properties and crop yields can vary depending on the soil type. Verheijen *et al*. (2009) introduced the concept of 'biochar loading capacity', which refers to the maximum amount of biochar that can be safely applied to a specific soil type without compromising its functions. Therefore, the second objective aimed to investigate the loading capacity of EFB biochar for different tropical soils. The study focused on the immediate chemical response, potential phytotoxicity and nutrient uptake at varying application rates. The goal was to establish the maximum rate best suited to each soil type. It was hypothesised that the immediate effect of EFB biochar on soil chemical properties differs across soil types, the potential phytotoxicity on maize germination varies with different application rates and soil types, and the co-application of EFB biochar and inorganic fertiliser affects nitrogen and phosphorus uptake differently across soil types. These hypotheses were tested in incubation, germination tests, and pot experiments using six different application rates (0, 0.25, 0.5, 1.0, 1.5, and 2.0% w/w) of EFB biochar on four different tropical soils, namely Acrisol, Brown Ferralsol, Red Ferralsol, and Vertisol.

4.2.1 Properties of EFB biochar and experimental soils

The empty fruit bunch biochar (B1) used in this study contained total carbon, ash, carbonate and inorganic carbon contents of 47%, 43.7%, 3.1% and 0.8%, respectively (Table 8). The biochar was very low in nitrogen (0.85%) and phosphorus (as P_2O_5 = 0.66%) but contained a high amount of potassium (3.5% as K_2O), calcium (2.49%) and magnesium (1.59%). It had low sulphur and trace elements except iron (1.02%). The biochar had a pH of 10.1 (water 1: 5 w/v) with an electrical conductivity (EC) of 3.8 dS cm^{-1}. The contents of all potentially toxic metals detected in the biochar were within the EBC threshold. The total concentration of 16 priority PAHs regulated by the United States Environmental Protection Agency (USEPA) in the biochar was 20.3 mg kg^{-1} and the total concentration of 8 priority PAHs set by the European Food Safety Authority was 1.2 mg kg^{-1}.

Table 8: Chemical properties, heavy metals and polycyclic aromatic hydrocarbon contents of the empty oil palm fruit bunch biochar used in the study

Property	**Value**
Ash (%)	43.7
Total carbon (%)	47.0
Total inorganic carbon (%)	0.8
Carbonate as CO_2 (%)	3.1
Nitrogen (%)	0.85
Phosphorus as P_2O_5 (%)	0.66
Potassium as K_2O (%)	3.54
Calcium as CaO (%)	2.49
Magnesium as MgO (%)	1.59
Sodium as Na_2O (%)	0.15
Sulphur SO_3 (%)	0.027
Iron (g kg^{-1})	10.2
Manganese (g kg^{-1})	0.27
Boron (g kg^{-1})	0.04
pH (water 1: 5 w/v)	10.1
Electrical conductivity (dS cm^{-1})	3.8
Cadmium (mg kg^{-1})	< 0.2
Zinc (mg kg^{-1})	170
Copper (mg kg^{-1})	50
Total 8 EFSA-EPA (mg kg^{-1})	1.2
Total 16 EPA-PAH (mg kg^{-1})	20.3

EFSA: European Food Safety Authority; EPA, Environmental Protection Agency; PAH, polycyclic aromatic hydrocarbons

The physical and chemical properties of the soils used for the experiments are presented in Table 9. The particle sizes differed considerably among the different soils with the clay content of the Vertisol (58%) being the highest while the Acrisol (25%) showed the lowest clay content. The pHs of soils used in the study ranged from 5.5 in the Acrisol to 6.8 in the Red Ferralsol. The TOC contents followed a decreasing order: Vertisol (1.69%) >

Brown Ferralsol (1.33%) > Acrisol (0.99%) > Red Ferralsol (0.77). Effective cation exchange capacity and contents of all the exchangeable cations apart from exchangeable potassium were highest in Vertisol and least in Brown Ferralsol.

Table 9: Characteristics of the different soil types before amendment application

	Soil type			
Property	**Acrisol**	**Brown Ferralsol**	**Red Ferralsol**	**Vertisol**
pH (water 1: 2.5 w/v)	5.5 ± 0.07c	5.8 ± 0.06c	6.8 ± 0.03a	6.1 ± 0.09b
Electrical conductivity (μs cm^{-1})	30.0 ± 0.0a	36.7 ± 3.2a	13.3 ± 2.9b	26.7 ± 2.8ab
Organic carbon (%)	0.99 ± 0.02c	1.13 ± 0.02b	0.77 ± 0.02d	1.69 ± 0.04a
Organic matter (%)	1.70 ± 0.04c	1.95 ± 0.04b	1.33 ± 0.4d	2.91 ± 0.06a
Total nitrogen (g kg^{-1})	0.62 ± 0.03b	0.21 ± 0.01d	0.42 ± 0.02c	1.01 ± 0.0a
Available phosphorus (mg kg^{-1})	4.15 ± 0.86c	2.85 ± 0.61c	23.3 ± 0.46a	8.02 ± 0.93b
Potassium (cmol $_{(+)}$ kg^{-1})	0.36 ± 0.01b	0.19 ± 0.01c	0.66 ± 0.02a	0.21 ± 0.01c
Calcium (cmol $_{(+)}$ kg^{-1})	1.20 ± 0.06c	1.08 ± 0.05c	8.01 ± 0.14b	14.2 ± 0.28a
Magnesium (cmol$_{(+)}$ kg^{-1})	0.46 ± 0.01c	0.41 ± 0.01c	2.98 ± 0.9b	5.67 ± 0.08a
Sodium (cmol $_{(+)}$ kg^{-1})	0.11 ± 0.01c	0.10 ± 0.01c	0.19 ± 0b	0.68 ± 0.02a
Sum of bases (cmol $_{(+)}$ kg^{-1})	2.12 ± 0.05c	1.77 ± 0.07c	11.9 ± 0.24b	20.8 ± 0.34a
Exchangeable acidity (cmol $_{(+)}$ kg^{-1})	0.31 ± 0.04a	0.26 ± 0.01a	0.0 ± 0.0b	0.20 ± 0.02a
ECEC (cmol $_{(+)}$ kg^{-1})	6.87 ± 0.79c	4.67 ± 0.27c	13.7 ± 0.35b	39.3 ± 2.34a
Base saturation (%)	31.3 ± 2.73c	38.0 ± 1.15c	87.0 ± 1.15a	53.3± 3.71b
Clay (%)	25 ± 0.6c	26 ± 0.3c	34 ± 0.9b	58 ± 2.3a
Silt (%)	17 ± 0.6b	7 ± 0.6c	29 ± 1.9a	7 ± 2.8c
Sand (%)	58 ± 0.0b	67 ± 0.3a	37 ± 3c	35 ± 1.4c
Textural class	Sandy clay loam	Sandy clay loam	Clay loam	Clay

Values are arithmetic mean ± standard error from three replicates. Different letters in a horizontal indicate significant differences among the different soils (Tukey test at $P < 0.05$), ECEC, effective cation exchange capacity.

Total nitrogen contents ranged from 0.21 g kg^{-1} in the Brown Ferralsol to 1.01 g kg^{-1} in the Vertisol. The Red Ferralsol had significantly ($P < 0.001$)

higher available phosphorus and potassium contents of 23.3 mg kg^{-1} and 0.66 $cmol_{(+)}$ kg^{-1} than the three other soils. The base saturation of 87% recorded in the Red Ferralsol was significantly ($P < 0.05$) higher than the 53.3%, 38.0%, and 31.3 % recorded in the Vertisol, Brown Ferralsol and Acrisol, respectively.

4.2.2 Immediate chemical response of the different soils to EFB biochar application across the incubation periods

The different tropical soils responded differently to EFB biochar application between the two incubation periods (Figure 24). The effects of biochar on the pH of the different soil types are presented in Figure 24a,b. Relative to the unamended soils, the application of EFB biochar significantly ($P < 0.001$) increased the pH of the Acrisol, Brown Ferralsol and Red Ferralsol at both incubation times except the Vertisol at 28 days after incubation (DAI). However, the margin of pH increase varied for the different soils, ranging from −0.1−0.2 points in the Vertisol to up to 2−0.7 points in the Red Ferralsol.

Also, the application of EFB biochar significantly ($P < 0.001$) increased the electrical conductivity (EC) of the Acrisol, Red and Brown Ferralsols and Vertisol, compared to their respective unamended soils, where the EC increased with increasing biochar application rates (Figure 24c, d). The highest biochar application rate, 2% w/w, increased the EC of the Acrisol, Brown and Red Ferralsols and Vertisol about 134%, 219%, 27% and 41% at 7 DAI and 126%, 182%, 56.6% and 70% at 28 DAI respectively higher than the unamended soils.

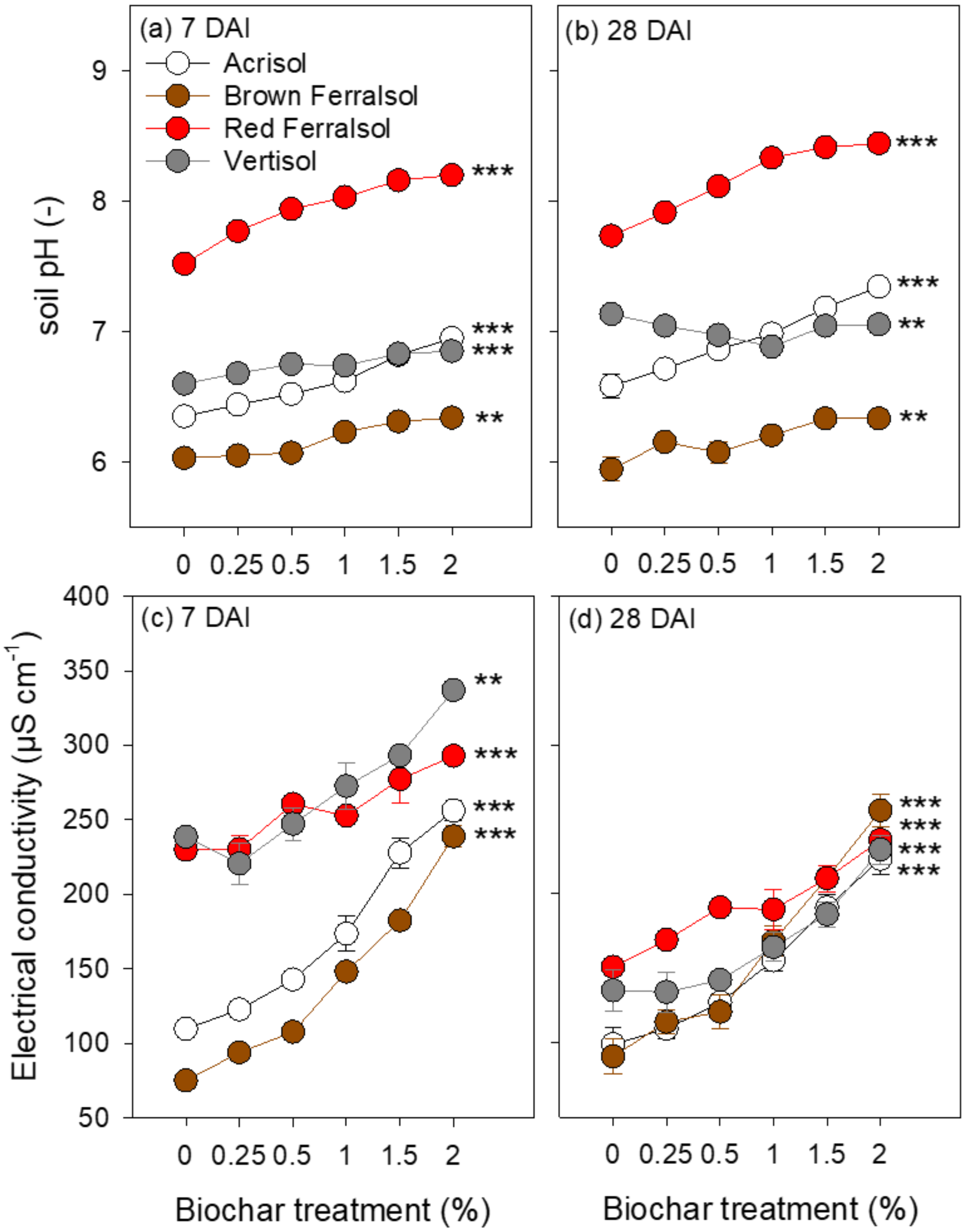

Figure 24: Effect of empty oil palm fruit bunch biochar applied at six rates (0 to 2% w/w) on (a, b) pH and (c, d) electrical conductivity of different soil types at 7 and 28 days after incubation (DAI). Error bars show the standard error of means from three replicates. Asterisks *, **, ***, denote significant difference among the biochar rates at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively, ns indicates no significant difference among the biochar rates.

The application of biochar significantly ($P < 0.05$) affected the available phosphorus contents in all the soil types (Figure 25a, b). At 7 DAI, percentage increases in available P content following biochar application ranged from 21–73.7%, 19–42% and 19–33% in the Acrisol, Brown Ferralsol and Vertisol, respectively, compared to their respective unamended soils.

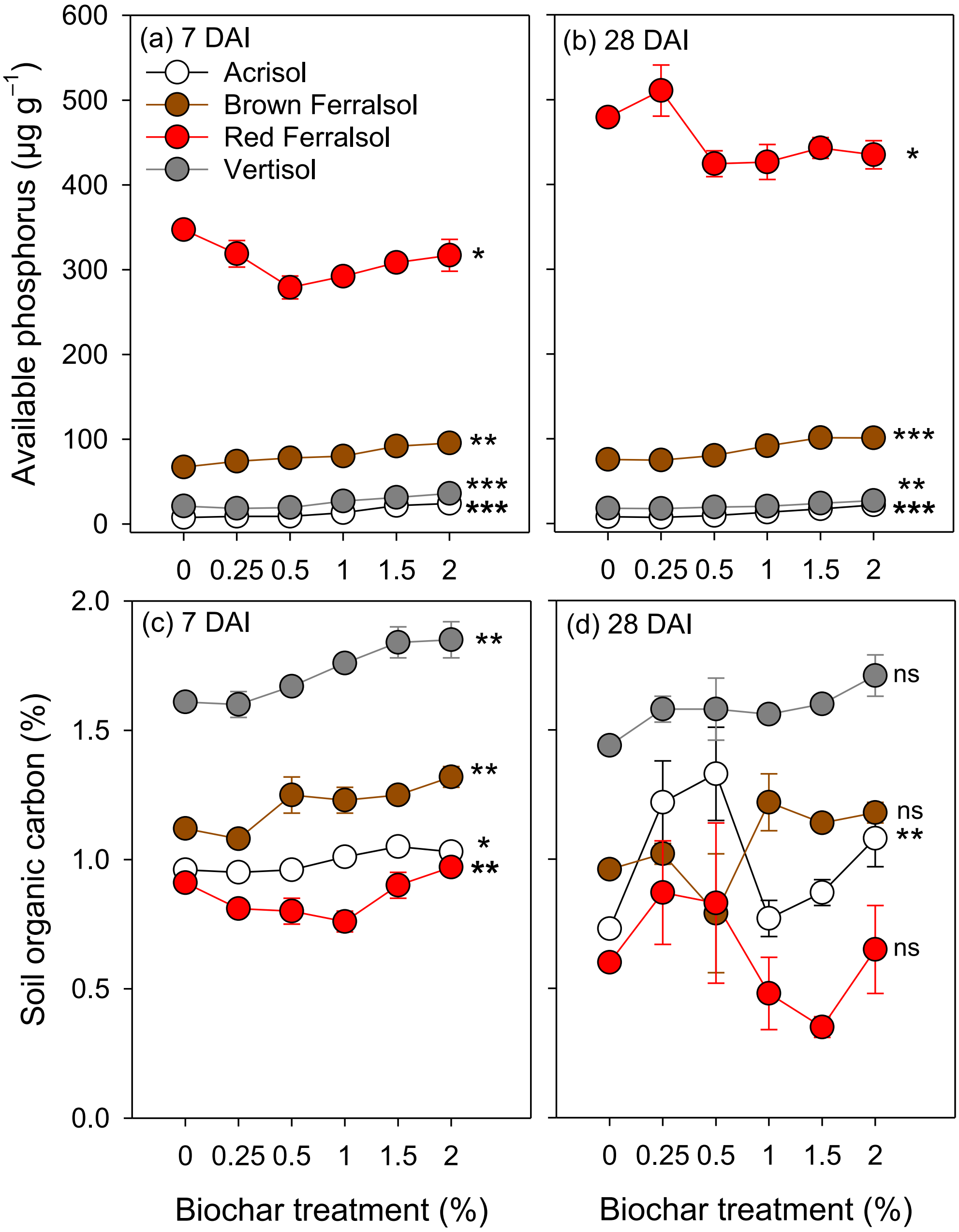

Figure 25: Effect of empty oil palm biochar applied at six rates (0 to 2% w/w) on (a, b) available phosphorus and (c, d) total organic carbon content of different soil types at 7 and 28 days after incubation (DAI). Error bars show the standard error of means from three replicates. Asterisks *, **, ***, denote significant difference among the biochar rates at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively, ns indicates no significant difference among the biochar rates.

Meanwhile, in the Acrisol and Brown Ferralsol, significant positive biochar effects were only observed after 0.5% w/w but after, 1% w/w in the Vertisol at 28 DAI. However, in the Red Ferralsol, biochar application beyond 0.25% decreased the available P content average by 13.8% and 9.8% at 7 and 28 DAI, respectively, compared to the unamended soil.

In the Acrisol, biochar application at the rate of 1.5% w/w, significantly ($P < 0.05$) increased total organic carbon (TOC) content compared to the unamended soil at 7 DAI, but at 28 DAI, a significant increase in soil TOC was observed at 0.5% w/w application rate (Figure 25c, d). Moreover, at 7 DAI, biochar application at 2% w/w significantly ($P < 0.01$) increased the TOC content by 18.5% in the Brown Ferralsol and 15% in the Vertisol compared to their respective unamended soils. Whereas in the Red Ferralsol, the biochar application rate at 0.5 and 1% w/w had significantly ($P < 0.05$) lower TOC content than the unamended soil at 7 DAI. However, no significant increase in soil TOC content was observed at 28 DAI in Ferralsols and Vertisol.

The effect of EFB biochar application on ammonium (NH_4^+) and nitrate (NO_3^-) contents of the different soils at the two incubation periods are presented in Figure 26. Ammonium contents of all the soil types were not significantly ($P > 0.05$) affected by biochar application at 7 DAI. Also, at 28 DAI (Figure 26a, b), the NH_4^+ content of the Acrisol, Brown Ferralsol and Vertisol were not significantly increased by biochar application and in the Red Ferralsol, the unamended soil had higher NH_4^+ content than all the biochar treatments. Moreover, NO_3^- content in the Acrisol and Brown Ferralsol were significantly ($P < 0.05$) decreased by biochar application at 7 DAI (Figure

26c). However, at 28 DAI biochar application did not significantly ($P > 0.05$ affect the NO_3^- content of these same soil types (Figure 26d).

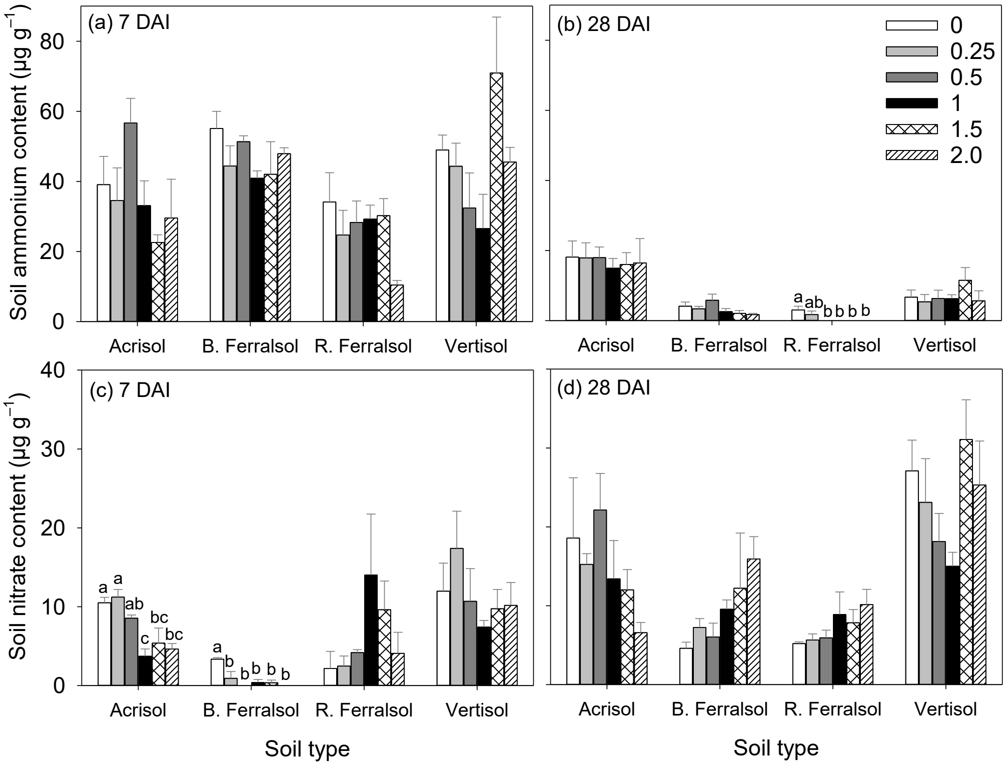

Figure 26: Effect of empty oil palm biochar application on (a, b) ammonium and (c, d) nitrate content of different soil types at 7 and 28 days after incubation (DAI). Error bars show the standard error of means from four replicates. Bars with different letters are significantly different (Tukey HSD test at $P < 0.05$); bars with no letters are statistically similar. B. Ferralsol (Brown Ferralsol); R. Ferralsol (Red Ferralsol).

The effect of incubation times on soil chemical properties of the different soil types is represented in Table 10. A two-sample t-test showed a significant increase in soil pH but a decline in EC and TOC content of all the soil types from 7 DAI to 28 DAI, except the pH and EC of the Brown Ferralsol and the TOC of the Acrisol. However, the trend of change of the available P was inconsistent for the different soil types.

Table 10: Effect of incubation times on soil chemical properties of the different soil types

Soil type	Property	Incubation time (days)	Means	Test statistic t	*P* value
Acrisol	pH	7	6.6 ± 0.1	−3.90	***
		28	6.9 ± 0.2		
	EC (µS cm^{-1})	7	172.0 ± 28.3	1.22	ns
		28	150.7 ± 24.9		
	Available P (µg g^{-1})	7	14.3 ± 3.5	0.73	ns
		28	13.0 ± 2.8		
	TOC (%)	7	0.99 ± 0.1	−10	ns
		28	1.0 ± 0.2		
	NH_4^+ (µg g^{-1})	7	35.9 ± 8.9	4.81	***
		28	16.9 ± 4.0		
	NO_3^-(µg g^{-1})	7	7.3 ± 1.8	−3.65	***
		28	14.7 ± 4.6		
Brown Ferralsol	pH	7	6.2 ± 0.1	0.09	ns
		28	6.2 ± 0.1		
	EC (µS cm^{-1})	7	140.8 ± 29.2	−0.96	ns
		28	160.0 ± 30.7		
	Available P (µg g^{-1})	7	81.08 ± 6.4	−1.77	ns
		28	87.45 ± 6.2		
	TOC (%)	7	1.2 ± 0.1	2.86	**
		28	1.05 ± 0.1		
	NH_4^+ (µg g^{-1})	7	47.0 ± 5.2	20.18	***
		28	3.29 ± 1.2		
	NO_3^-(µg g^{-1})	7	0.83 ± 0.7	−5.82	***
		28	9.3 ± 3.5		
Red Ferralsol	pH	7	7.9 ± 0.1	−2.53	*
		28	8.2 ± 0.2		
	EC (µS cm^{-1})	7	256.2 ± 0	6.67	***
		28	191.0 ± 0		
	Available P (µg g^{-1})	7	310 ± 15.7	−12.54	***
		28	453 ± 23.1		
	TOC (%)	7	0.85 ± 0.1	2.95	**
		28	0.63 ± 0.2		
	NH_4^+ (µg g^{-1})	7	26.2 ± 6.4	9.67	***
		28	0.83 ± 0.8		
	NO_3^-(µg g^{-1})	7	6.1 ± 4.0	−0.66	ns
		28	7.2 ± 1.7		
Vertisol	pH	7	6.8 ± 0.6	−5.17	***
		28	7.0 ± 0.3		
	EC (µS cm^{-1})	7	259.5 ± 18.4	7.3	***
		28	165.1 ± 19.5		
	Available P (µg g^{-1})	7	25.7 ± 3.9	2.52	*
		28	21.1 ± 2.4		
	TOC (%)	7	1.7 ± 0.07	3.58	***
		28	1.6 ± 0.07		
	NH_4^+ (µg g^{-1})	7	44. 8 ± 11.0	8.23	***
		28	7.0 ± 2.5		
	NO_3^-(µg g^{-1})	7	11.2 ± 3.5	−5.05	***
		28	23.3 ± 4.8		

Values are arithmetic mean ± standard error of three replicates. Asterisks *, **, ***, denote significant difference between the two incubation periods at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively, ns means no significant difference between the two incubation periods. EC, electrical conductivity; $NH4^+$, Ammonium; NO_3^-, nitrate, Available P, available phosphorus; TOC, total organic carbon.

Moreover, the NH_4^+content of all the soil types decreased from 7 DAI to 28 DAI and a reverse trend was observed for the NO_3^- in the different soil types, except that no significant change in the $NO_3^-$was observed between the two incubation periods in the Red Ferralsol.

In this study, the short-term impact of EFB biochar application on the chemical properties of Acrisol, Brown and Red Ferralsols, and Vertisol, exhibited varying physico-chemical characteristics. The findings revealed a rise in soil pH with increasing biochar application rates for Acrisol and Ferralsols (Figure 24a,b). Similar to the current studies, Rosenani *et al.* (2015) observed that the addition of EFB biochar at rates 10 t ha^{-1} and 20 t ha^{-1} increased soil pH from 3.5 (in control) to 6.0 (at maximum biochar rate) in a pot experiment with acid sulphate soil. Also, Rabileh *et al.* (2015) observed that applications of EFB biochar up to 20 t ha^{-1} increased soil pH and decreased Al^{3+} toxicity of Ultisol soil under glasshouse conditions. The increase in soil pH following biochar addition can be attributed to the alkaline nature and liming effect of biochar resulting from residual ash produced during carbonisation and partial calcination of feedstocks used for biochar production (Chia & Liu, 2016; Prapagdee & Tawinteung, 2017; Rees *et al.,* 2014; Uchimiya *et al.,* 2010). According to Bolan et al. (2022), the liming effect of biochar on soils is due to the dissolution of carbonates, oxides, and hydroxides in the biochar ash fraction.

Moreover, the biochar addition affected the pH of the four soil types differently. The pH response of the four soils to EFB biochar application was more pronounced in the Acrisol and Ferralsols and was less in Vertisol. These differences may be due to the high clay (58%), organic matter (2.91%), and effective cation exchange capacity (39 cmol $_{(+)}$ kg^{-1}) contents of the Vertisol, potentially increasing its buffering capacity compared to the other soil types. Agreeably, Cao *et al.* (2017) reported that rice hull biochar significantly increased the pH of Red soil but decreased the pH of sandy soil and Coastal Solonshack. Again, Zhu *et al.* (2015) observed after an incubation experiment that two different rice straw biochars improved the pH of red soil only, but a pH increase was not observed in chaotu soil, black soil, loess, and purple soil. Van Zwieten (2009) reasoned that the liming effect of biochar depends on soil acidity and Al toxicity, and thus, it is more obvious for strongly acidic soils such as Ferralsols and Oxisols compared to Vertisols.

Furthermore, an increase in the EC for all the four soil types after biochar application may be attributed to the high ash content (43.7%) of the EFB biochar used and the release of soluble bound cations such as Ca^{2+}, K^{+}, and Na^{+}, associated with salts, ashes, and weak acid functional groups on the biochar into the soil solution (Chan *et al.,* 2007; Ng *et al.,* 2022; Prapagdee & Tawinteung, 2017). Agreeably, Chintala *et al*. (2013) reported that applying corn stover and switchgrass biochar significantly increased the EC of acidic soil by 21% to 156%. They attributed the increase in EC of the acidic soil to the release of weakly bound nutrients (cations and anions) of biochar into the soil solution for plant uptake. The immediate elevation of soil pH and EC following biochar application suggested the potential for EFB biochar to address the challenge of inherent soil acidity, particularly in tropical soils. However, the loading capacities may vary among different soils due to differences in their buffering capacities.

The increased available P content in the Acrisol, Brown Ferralsol, and Vertisol after EFB biochar application may be attributed to enhanced chemical and biochemical processes such as mineralisation, solubilisation, and mobilisation of organic or previously fixed phosphorus in the acidic soils, following the increased pH from biochar application. In the current study, the additive effect from the phosphorus contained in the biochar can be negligible because the EFB biochar used in this study had low phosphorus content (0.66% as P_2O) to might have directly increased the available phosphorus content in the three soils. Contrary to the results from the current study, Prapagdee and Tawinteung (2017) observed no increase in available phosphorus content with increasing rates of cassava stem-derived biochar in a pot experiment. Moreover, the pH rise of the Red Ferralsol above 7 after biochar application might account for the lower available P content observed in this soil type, as the calcium and magnesium released from EFB biochar decomposition at pH levels exceeding 7 might have adsorbed the available P in the soil solution as suggested by Bornø *et al.* (2018). Again, Chintala *et al.* (2013) also attributed the reduction of soil-available P by biochar to the exchange between phosphorus anions in solution and the oxygenated positive functional groups on the surface of biochar. The findings from this study suggested that EFB biochar can increase the availability of fixed phosphorus in tropical acidic soils through soil pH optimisation. However, EFB biochar can potentially adsorb phosphorus in the soil after excessive release of calcium and magnesium from EFB biochar decomposition under basic pH conditions.

The increase in TOC content in Acrisol, Brown Ferralsol, and Vertisol at 7 DAI might be partly due to the high carbon content of EFB biochar and the initial exposure of fresh biochar to the soil, resulting in a high microbial response and turnover of biochar's labile carbon content as indicated by Wang *et al*. (2015). Agreeably, Jing *et al*. (2020) attributed the increase in the organic carbon content of soil after biochar application to the high organic carbon content of biochar. Moreover, the adsorption, distribution, dissolution, and mobility of organic matter and nutrients driven by soil particle sizes and differences in the initial organic carbon contents of the different soils could explain the varying increases in these soils. Gross *et al*. (2021) illustrated that soil with a high sand content can increase aeration and drainage, leading to the decomposition of organic matter, as seen in Acrisol and Brown Ferralsol. However, soils with high clay contents, like Red Ferralsol and Vertisol, can protect organic matter from decomposition through the formation of aggregates or complexes with it (Jing *et al*. 2020). For example, the Red Ferralsol, rich in iron and aluminium oxides, could form strong bonds with organic carbon. Despite this, there was a decline in TOC content, possibly due to increased soil pH after EFB biochar application. According to Reichenbach *et al*. (2021), this pH shift could weaken the bonds between sesquioxides and organic matter, exposing the soil's organic matter to microbial attack and intensifying its decomposition rate.

The application of EFB biochar did not significantly affect the NH_4^+ content of any of the soil types except the Red Ferralsol, where the NH_4^+ content was higher in the unamended soils than the biochar-amended soils. The result from the current study contradicts the findings of Taghizadeh-Toosi *et al*. (2012), who observed that biochar can adsorb NH_4^+-N at pyrolysis temperatures up to 500°C, facilitating subsequent release for plant uptake, thereby reducing nitrogen losses. Again, the unamended soils exhibited a significantly higher NO_3^- content than the biochar-amended Acrisol and Brown Ferralsol. Contrarily, Cao *et al*. (2017) reported that rice hull biochar addition resulted in higher NO_3^--N levels in the red and sandy soils, whereas NO_3^- -N levels fell in coastal solonchak. Again, Nessa (2021) found that nitrogen nitrification and mineralisation were lower, resulting in significantly lower soil NH_4^+-N and NO_3^--N in the biochar-amended soils compared to the control soils on day five following incubation. The underlying mechanisms of the reduced $NO_3^-$and NH_4^+ content of the biochar-amended soils can be partly attributed to the adsorption of NH_4^+-N onto biochar surfaces, including the presence of negatively charged functional groups on its surface leading to the adsorption of positively charged NH_4^+-N or the immobilisation of N rather than stimulating N mineralisation by biochar (Zheng *et al.*, 2013). This suggested that EFB biochar could reduce the solubility and availability of mineral nitrogen contents, especially in nitrogen-deficient soils.

Moreover, between the two incubation periods, the EC and content of NH_4^+ and TOC decreased while pH and NO_3^-increased from 7 DAI to 28 DAI in the different soils. The decline in TOC over time might be due to microbial preference for easily degradable carbon pools, depleting this fraction of carbon. Moreover, the trend of change of the NH_4^+ and the NO_3^- between the incubation periods could be attributed to the incorporation of depleted N into the NH_4^+-N pool, whereas NO_3^- enrichment emanating from N loss processes such as leaching, nitrification or denitrification, which is typical of nitrogen mineralisation processes that occur in nature (Wang *et al.,* 2020). Similarly, Nessa *et al.* (2021) reported that NH_4^+-N was lower than NO_3^--N on day 5 following the incubation.

4.2.3 Phytotoxicity of EFB biochar on maize germination

Effects of EFB biochar application on relative root elongation (RRE) and germination indices (GI) of maize in the different soils are presented in Figure 27. After a day of soil-biochar incubation, biochar application at 0.25% increased the RRE by 45% in the Acrisol (Figure 27a). However, it was the highest biochar rate (2% w/w) that significantly ($P < 0.05$) increased the RRE by 50% in the Vertisol relative to the unamended soil.

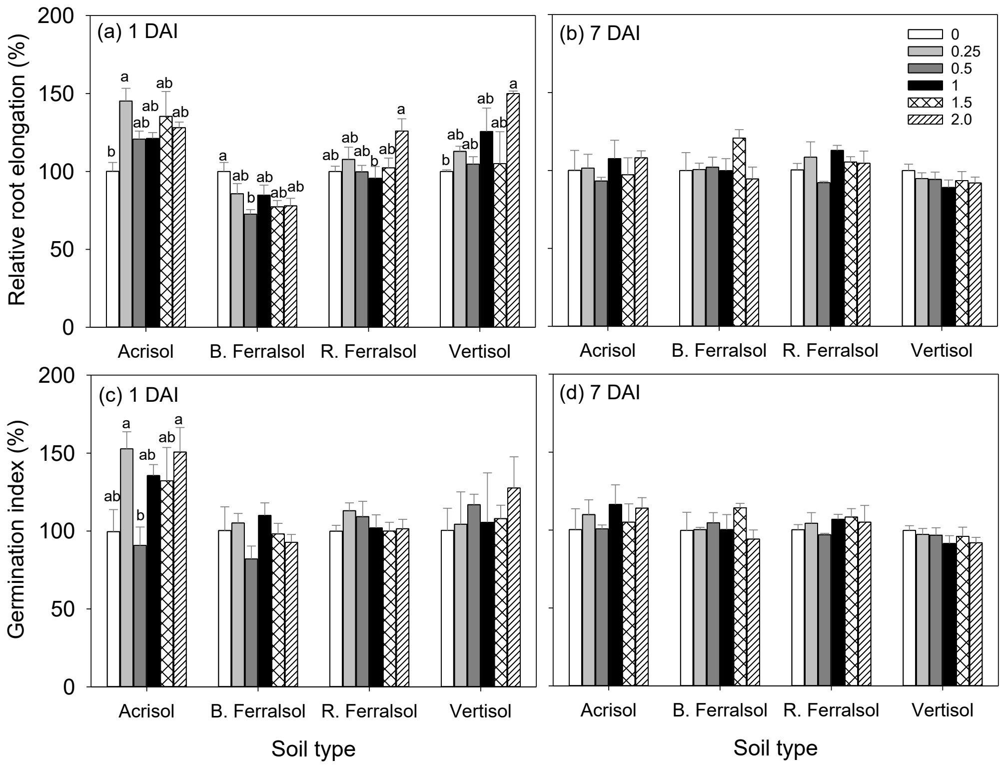

Figure 27: (a, b) Relative root elongation and (c, d) germination index of maize in various soil types after 1 and 7 days after incubation (DAI) with empty oil palm fruit bunch biochar at six application rates (0 to 2% w/w). Error bars show the standard error of means from four replicates. Bars with different letters are significantly different (Tukey HSD test at $P < 0.05$); bars with no letters are statistically similar. B. Ferralsol (Brown Ferralsol); R. Ferralsol (Red Ferralsol).

Moreover, in the Brown Ferralsol, the biochar application rate at 0.5% decreased RRE by 27% relative to the unamended soil. Meanwhile, the RRE was not significantly ($P > 0.05$) affected by EFB biochar application in all the soil types at 7 DAI (Figure 27b). The EFB biochar application did not significantly ($P > 0.05$) increase GI in the Acrisol after one day of incubation compared to the unamended control. However, the application rate of 0.5% significantly decreased the GI compared to the highest application rate of 2% (Figure 27c). In the rest of the soil types, EFB biochar application did not significantly affect the germination indices at both incubation periods and in all cases, the germination indices were greater than 80% (Figure 27d).

The mean RRE and GI of the Acrisol and Vertisol at 1 DAI were significantly ($P < 0.05$) higher than 7 DAI (Table 11).

Table 11: Effect of incubation times on relative root elongation and germination index of maize in the different soil types

Soil type	**Parameter (%)**	**Incubation time (days)**	**Means**	**Test statistic t**	***P* value**
Acrisol	RRE	1	125.1 ± 4.2	4.35	***
		7	101.3 ± 3.5		
	GI	1	126 ± 6.8	2.45	*
		7	107.8 ± 3.9		
Brown Ferralsol	RRE	1	83 ± 2.6	−4.8	***
		7	103 ± 3.2		
	GI	1	98.1 ± 3.7	−0.88	ns
		7	103.3 ± 2.9		
Red Ferralsol	RRE	1	111 ± 7.1	0.44	*
		7	103.7 ± 1.7		
	GI	1	104.3 ± 2.6	0.18	ns
		7	103.7 ± 2.3		
Vertisol	RRE	1	116.4 ± 5.2	0.73	***
		7	94 ± 1.8		
	GI	1	111 ± 7.1	2.1	*
		7	95.5 ± 1.7		

Values are arithmetic mean ± standard error of four replicates. Asterisks *, **, ***, denote significant difference between the two incubation times at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively, ns means no significant difference between the two incubation times. RRE, relative root elongation; GI, germination index.

Whereas in the Brown Ferralsol, the mean RRE at 1 DAI was 20% ($P < 0.001$) less than the RRE at 7 DAI, but in the Red Ferralsol, the mean RRE did not differ between the two incubation periods. The mean germination indices recorded at 1 DAI did not differ significantly ($P > 0.05$) from 7 DAI in the Red Ferralsol and Brown Ferralsols but significantly decreased in the Acrisol and Vertisol.

Pearson correlation performed with pooled data from the four different soils revealed that RRE positively correlated with soil EC (r = 0.3, $P < 0.01$) and NO_3^- content (r = 0.3, $P < 0.5$) (Figure 28). Meanwhile, GI did not correlate significantly ($P > 0.05$) with any of the soil chemical properties.

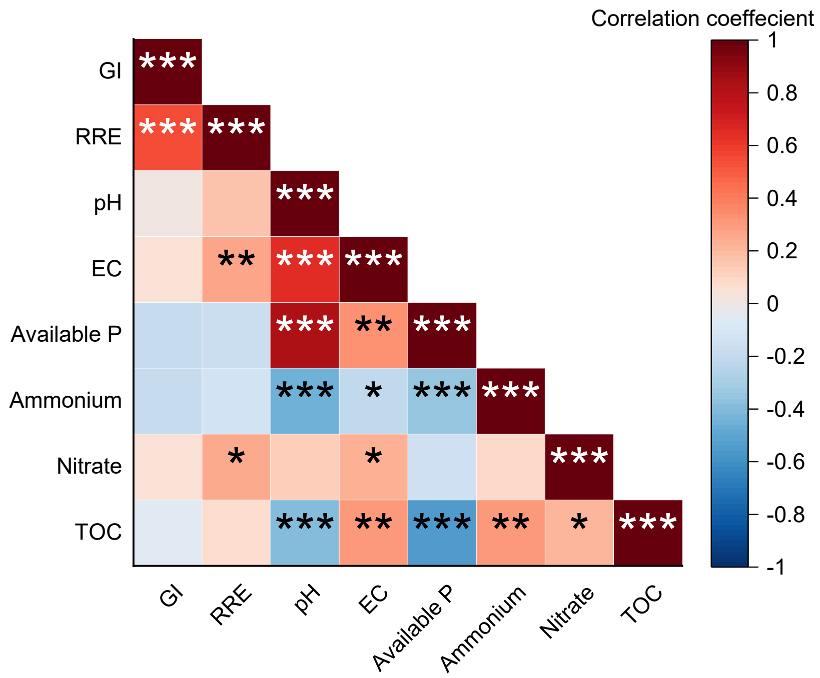

Figure 28: Correlation matrix of germination parameters with soil chemical properties after 7 days of incubation. Asterisks *, **, ***, denote significant correlation at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively. GI, germination index; RRE, relative root elongation; available P, available phosphorus; EC, electrical conductivity; TOC, total organic carbon.

Germination experiments were carried out to assess the potential phytotoxicity of EFB biochar after incubating with four different tropical soils for one and seven days. The germination index (GI), which combines relative germination and relative root elongation (RRE), was used to identify the phytotoxicity of organic amendments and their suitability for soil application (Barral & Paradelo, 2011). According to Barral and Paradelo (2011), a GI of less than 50% indicates high toxicity, a GI between 50% and 80% indicates moderate toxicity, a GI from 80% to 100% indicates the absence of toxicity, and a GI greater than 100% indicates that the amendment is a phytonutrient or phyto-stimulant.

In the current study, the application of EFB biochar to the Acrisol, Red Ferralsol, and Vertisol did not significantly decrease the RRE of maize compared to the unamended soils at both 1 and 7 DAI, and the highest rate of biochar application stimulated root growth in these soils. It is also plausible that the stimulatory effects observed in the different soils are due to the positive effect of biochar on soil NO_3^- and TOC contents and pH optimisation that stimulate germination and seedling growth or reactions that deactivate heavy metals and phytotoxic organic compounds, as suggested by Joseph *et al.* (2021) and Ni *et al.* (2020). Similarly, Li *et al.* (2015) observed that applying corn stover biochar at low/moderate dosages promoted seedling growth, which they attributed to the nutrients contained in the biochar. Luo *et al.* (2018) suggested that the inhibitory effects of biochar on plant germination and root growth can be caused by the presence of polycyclic aromatic hydrocarbons, heavy metals, organic carbon, pH, salinity, and volatile fatty acids that may be released during biochar decomposition.

Despite the total of 16 priority PAHs in the EFB biochar (20.3 mg kg^{-1}) being higher than the maximum permitted limit of 12 mg kg^{-1}, in no significant inhibitory effect on seed germination and root elongation was observed in three out of the four soil types in the current study. Graber *et al*. (2010) reasoned that the pollutants present in the biochar, such as heavy metals and PAHs, may have caused chemical facilitation (Hormesis effect) at low levels and stimulated plant growth. However, biochar application at 0.5% to the Brown Ferralsol had a moderate inhibitory effect (RRE > 50% < 80%) on root elongation at 1 DAI. This same soil type generally had the highest NH_4^+ content, least NO_3^- content and was slightly acidic (pH= 6.2) among the soil types after amending with EFB biochar, and this could account for the root inhibition observed in this soil type. The decline in the stimulatory effect of EFB biochar from 1 DAI to 7 DAI can be related to the decline in nutrient availability for root growth with time in the different soils. This was evident in the significant positive correlation between RRE and soil EC and NO_3^- content, showcasing increased nutrient availability as the major driver of the stimulatory effect observed in the biochar-amended soils. This study shows that EFB biochar has no toxic effect on maize germination but has a photo-stimulatory effect on seedling growth when applied to soils. However, the margin of effect differs for different tropical soils.

The application of EFB biochar to the different soils (Acrisol, Brown Ferralsol, Red Ferralsol, and Vertisol) did not significantly affect germination indices during the incubation periods (1 and 7), regardless of the application rates. The study found that the germination of maize seeds was less sensitive to biochar application in the different soils. This finding is consistent with previous studies by Bai *et al*. (2015), Di-Salvatore (2008) and Paradelo *et al*. (2008), who explained that the germination stage of plants is relatively insensitive to many contaminants such as heavy metals and PAHs because the embryo is isolated from the environment and many chemicals are not absorbed by the seed, which supplies the necessary nutrients to the embryo. Contrarily, Li *et al.* (2017) observed inhibition of germination rate and root growth at the highest sawdust biochar dosage of 80.0 g kg^{-1} compared to the control. The study suggests that seed germination will not be negatively impacted even if seeds are sown before the generally accepted two-week equilibration period for soil-biochar incubation.

4.2.4 Effect of co-application of EFB biochar and mineral fertiliser on maize biomass yield and nitrogen and phosphorus uptake

The effects of the co-application of different rates of EFB biochar and mineral fertiliser on maize biomass yield in the different soils are presented in Table 12. Maize biomass yields did not vary significantly ($P > 0.05$) with different EFB biochar rates in all the soil types. Though the highest biomass yield was recorded at 2.0 % biochar rate in all the soil types, the margin of differences (218.6%, 42.7%, 9.3% and 22.2% for the Acrisol, Brown Ferralsol, Red Ferralsol and Vertisol, respectively) varied considerably among the different soil types compared to their respective unamended controls.

Table 12: Effect of co-application of empty oil palm fruit bunch biochar and inorganic fertiliser on maize biomass yield (g plant^{-1}) in the different soil types

Biochar application rate (%)	Maize biomass yield at different soil types (g plant^{-1})			
	Acrisol	Brown Ferralsol	Red Ferralsol	Vertisol
0	9.7 ± 1.6a	20.6 ± 5.3	29.3 ± 2.7	38.1 ± 2.4
0.25	16.7 ± 6.1a	21.2 ± 5.3	32.1 ± 1.2	44.0 ± 4.5
0.5	28.6 ± 6.1a	23.7 ± 2.1	30.9 ± 3.8	38.1 ± 3.5
1.0	26.7 ± 4.8a	20.9 ± 1.6	26.4 ± 3.7	37.0 ± 3.1
1.5	27.3 ± 4.3a	23.8 ± 5.6	28.5 ± 3.2	40.0 ± 1.7
2.0	30.9 ± 8.2a	29.4 ± 2.1	32.3 ± 2.0	49.0 ± 3.1
P value	*	ns	ns	ns

Values are arithmetic mean ± standard error. Asterisks * denote significant differences among the biochar rates at $P < 0.05$ and ns means not significant among the biochar rates. Different letters indicate significant differences, and the same letters mean there are no differences (Tukey test at $P < 0.05$).

The co-application of EFB biochar and mineral fertiliser significantly ($P < 0.05$) increased nitrogen and phosphorus uptake in the Acrisol, and increasing biochar rates led to increasing maize nutrients uptake (Figure 29). Biochar application to Brown Ferralsol and Red Ferralsol did not significantly ($P > 0.05$) affect maize N and P uptake. Meanwhile, biochar application to Vertisol significantly affected maize phosphorus but not nitrogen uptake. The biochar application at 0.25% w/w to the Vertisol recorded the highest P uptake of 0.17 g plant^{-1}, and was only significantly ($P < 0.05$) higher than the application rates at 1% and 1.5% w/w. Again, the margin of differences in N and P uptake varied considerably among the soil types, with Acrisol being more responsive than the other soil types.

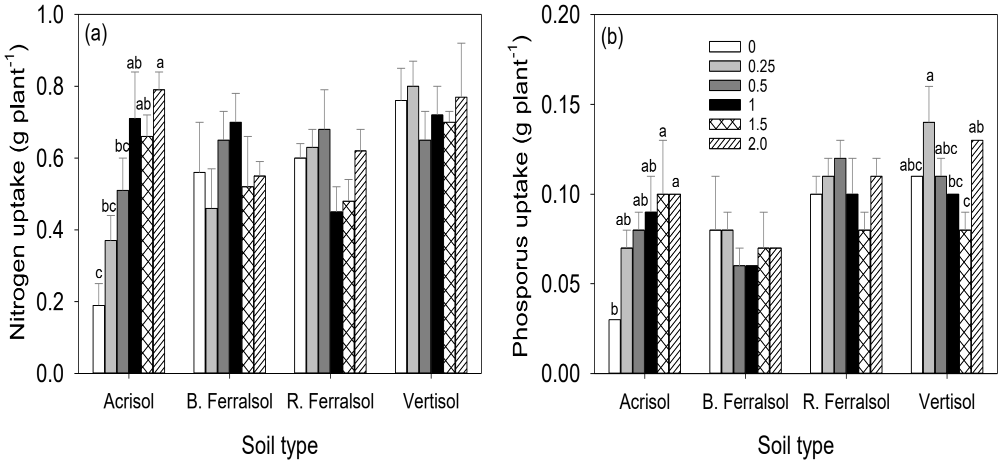

*Figure 29: (*a) Nitrogen and (b) phosphorus uptake of maize grown in different soil types amended with six rates of empty oil palm fruit bunch biochar and inorganic fertiliser. Error bars show the standard error of means from four replicates. Bars with different letters are significantly different (Tukey HSD test at $P < 0.05$); bars with no letters are statistically similar. B. Ferralsol (Brown Ferralsol); R. Ferralsol (Red Ferralsol).

Pearson correlation performed with pooled data from the four different soils also revealed that biomass yield had a significant positive correlation with soil pH (r = 0.3, $P < 0.05$), EC (r = 0.3, $P < 0.01$), NO_3^- (r = 0.5, $P < 0.05$) and TOC (r = 0.5, $P < 0.001$) contents (Figure 30). Nitrogen uptake positively correlated (r = 0.4, $P < 0.01$) with soil TOC content, while phosphorus uptake had a significant positive correlation with soil pH (r = 0.4, $P < 0.001$) and EC (r = 0.2, $P < 0.5$).

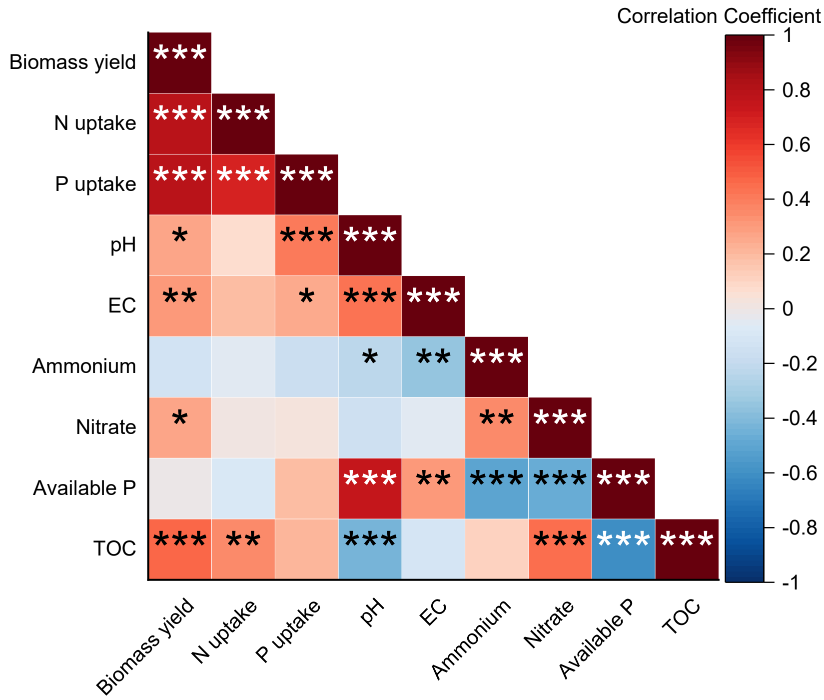

Figure 30: Correlation matrix of biomass yield, nutrient uptakes and soil chemical properties after 28 days of incubation. Asterisks *, **, ***, denote significant correlation at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively. N uptake, nitrogen uptake; P uptake, phosphorus uptake; EC, electrical conductivity; available P, available phosphorus; TOC, total organic carbon.

The observed non-significant effect of co-application of EFB biochar and fertiliser, irrespective of the soil type, on maize biomass yield in the is similar to the findings of Mavi *et al*. (2018), who reported that the addition of rice-residue biochar at different rates, with N fertiliser, did not significantly influence wheat biomass yield in two soils contrasting in texture. Also, Zhu *et al*. (2015) reported that out of the different soils, only the Phinthosols under rice straw biochar and NPK application showed a positive effect on maize biomass and that the biomass was even reduced significantly in the Gleyic Cambisols, Chernozems, Calcic Cambisols, and Regosols. Again, Syuhada *et al*. (2016) observed that adding EFB biochar to a sand Podzol soil did not increase the growth and yield of corn, despite the increase of CEC by biochar in soil. Contrarily, Abdulrazzaq *et al*. (2015) reported that the applications of

15 and 30 t ha^{-1} EFB biochar increased the shoot dry weight of sweet corn by about three and six times, respectively. Again, in a pot experiment with acid sulphate soil by Rosenani *et al.* (2015), the addition of EFB biochar at rates 10, 20, and 40 t ha^{-1} increased wet rice yield by 141 to 472%. Also, Rabileh *et al.* (2015) observed that applications of EFB biochar up to 20 t ha^{-1} increased plant growth parameters such as height, total root length, and total dry weight of maize grown on an Ultisol under glasshouse conditions. The plausible reasons for the non-significant effect of biochar on maize biomass may be related to slower biochar degradation and its interaction with soil, consequently delaying its beneficial effects on plant productivity (Fang *et al.*, 2015). Again, the negligible effect of EFB biochar on maize growth could be caused by limited N availability, as evidenced by the reduced NO_3^- content and N uptake observed in some of the soil types.

Though insignificant, the observed yield margin across the soil types may be attributed to variations in their inherent physical and chemical properties, including clay content, effective cation exchange capacity, and their interactions with the added biochar. Moreover, the highest biomass yields were observed at the maximum biochar application rate (2% w/w) across all soil types. This increase may result from improved nutrient solubility, availability, and uptake, likely driven by optimised soil pH and increased soil organic carbon following biochar application. A positive correlation was also found between dry biomass yield and soil chemical properties at planting. Agreeably, the results of a meta-analysis by Adu *et al*. (2022) showed that the effect of EFB on plant growth and yield was affected by the soil types, where crop growth and yield increased in EFB-amended soils over unamended soils in the order of sand (0.566), sandy loam (0.411), clay loam (0.384), loamy sand (0.207), sandy clay loam (0.118), and clay (−0.065) soils, suggesting that plants cultivated on coarser textured soils with low organic matter and low nutrients content, did relatively better than plants cultivated on EFB amended finer-textured soils.

In the current study, the Acrisol, characterised by lower pH, base saturation, and clay content, responded more positively to biochar application, reflecting increased N and P uptake than other soil types. Agreeably, Zhu *et al*. (2015) observed that the co-application of rice straw biochar and fertiliser significantly increased the nitrogen content and use efficiency of maize in an acidic and highly weathered red soil. However, the biochar did not increase the pH of three alkaline soils due to the buffer of soil pH, which was shown in the lack of effect on nutrient use efficiency and maize biomass in these soils.

Correlation analysis showed a significant positive relationship between nitrogen uptake and soil organic matter content, as well as between phosphorus uptake and soil pH (Figure 30). These findings align with previous research by Abdulrahman *et al*. (2016), Arif *et al*. (2017), and Uzoma *et al*. (2011) who identified increased soil organic carbon as a primary factor for improved crop yields. However, variations in N and P uptake across the different soils may be explained by differences in soil texture, pH, organic carbon, and effective cation exchange capacity (Table 10). Omara *et al*. (2020) also found that yield response to biochar and nitrogen application was more pronounced in sandy low-yielding environments. This study suggested that the co-application of EFB biochar and inorganic fertiliser enhances nutrient retention, solubility, and availability, particularly in Acrisol, leading to increased biomass yield and nutrient uptake. However, the responses of other soils varied, indicating that the agronomic effects of co-application of biochar and fertiliser are soil-specific.

4.2.5 Optimal rate of EFB biochar for the different tropical soil types

To determine the optimal rate of EFB biochar for tropical soil types, this study integrated soil chemical variables, phytotoxicity indices, plant biomass yield, and nutrient uptake into a comprehensive amelioration score (0–1) (Table 13). The results showed that the highest biochar application rate of 2.0 % was optimal for Acrisol, Red Ferralsol and Vertisol soils, with scores of 0.83, 74 and 0.84, respectively.

Table 13: Scoring for the empty oil palm fruit bunch biochar rate best suited for each of the different soil types

Biochar application rate (%)	**Amelioration score**			
	Acrisol	**Brown Ferralsol**	**Red Ferralsol**	**Vertisol**
0	0.13	0.50	0.32	0.36
0.25	0.52	0.51	0.73	0.54
0.5	0.42	0.31	0.62	0.41
1	0.51	**0.64**	0.38	0.37
1.5	0.59	0.51	0.36	0.51
2	**0.83**	0.57	**0.74**	**0.84**

However, the optimal EFB biochar rate for Brown Ferralsol was 1%. Generally, at these rates, soil chemical properties improved, and biomass yield and nutrient uptake increased without inhibiting maize germination or root elongation.

According to Verheijen *et al.* (2009), biochar can only be safely applied up to a certain limit, beyond which other soil functions may be compromised. Results showed that the highest application rate of 2.0% was optimal for Acrisol, Red Ferralsol and Vertisol soils, with scores of 0.83, 74 and 0.84, respectively. In the Red Ferralsol, application higher than 0.25 % raised the pH beyond the 7 (optimum range) and decreased TOC, available P, and NH_4^+ content, but biochar rate at 2% increased the NO_3^- content and enhanced root elongation after 1 DAI and maize biomass yield. In the Brown Ferralsol, rates above 1% improved soil chemical properties and nutrient uptake but reduced germination and root elongation. These findings support Gul and Whalen's (2016) assertion that soil properties such as clay type, carbon content, and CEC influence biochar's ameliorative effects. The study suggests that the optimal EFB biochar application rate varies across tropical soils, necessitating caution when using it as a soil conditioner.

4.2.6 Summary

Biochar is a potential multipurpose management option that can resolve imminent problems of acidity, phosphorus sorption, low organic carbon and infertility status of tropical soils. The second objective of the study investigated the loading capacity of empty oil palm fruit bunch biochar for different tropical soils, focusing on potential phytotoxicity, immediate chemical response, and nutrient uptake at varying application rates. The results showed that the application of empty fruit bunch biochar significantly increased pH across all soil types except the Vertisol. This means that Vertisol may require biochar application at a rate higher than 2% w/w due to its high buffering capacity.

Again, the biochar application increased the available phosphorus and organic carbon content of all soil types except the Red Ferralsol. In the Red Ferralsol, phosphorus retention was observed and this was attributed to phosphorus adsorption to calcium and magnesium ions when the pH of the soil was elevated beyond 7. Acrisol showed a significant increase in nutrient uptake from the co-application of empty fruit bunch biochar and mineral fertiliser, while the other soil types exhibited no effect. These different effects were driven by the extent to which the application of empty fruit bunch biochar improved the organic carbon content and the pH of these soils. Collectively, the highest application rate of 2.0% emerged as the ideal rate for Acrisol, Red Ferralsol and Vertisol soil types, while rate of 1% was best suited for the Brown Ferralsol. These rates enhanced the chemical properties and increased nutrient uptake without exhibiting any inhibitory effect on maize germination, giving a clue on the empty fruit bunch biochar loading capacity of these dominant and most cultivated soil types in the tropics. Long-term field studies are recommended for further validation.

4.3 Changes in Soil Chemical and Microbial Properties Affected by Empty Oil Palm Fruit Bunch Amendments Drive Okra Nutrient Use Efficiency and Yield

Biochar application offers the potential to increase soil fertility, sequester carbon, and enhance soil nutrient cycling while ameliorating nutrient leaching, bioavailability of contaminants, and reducing greenhouse gas emissions (Park *et al*., 2023). The third objective of the study examined the effects of one-time biochar and/or compost application on soil chemical and microbial properties, nutrient use efficiency, and the yields of okra in two cropping cycles. It was hypothesized that a one-time application of EFB biochar and compost significantly increased okra yield and nutrient use efficiency, across two cropping cycles, through significant improvement in the soil's chemical and microbial environment.

4.3.1 Response of okra growth and yield to soil amendments across the cropping cycles

Okra growth and yield components varied significantly ($P < 0.001$) between the first and second crop cycles (Table 14 and Figure 31). Plant height and stem girth were generally higher in the first cycle compared to the second cycle, except for the unamended plants (Table 14).

Table 14: Effect of treatments on okra plant height and stem girth at the flowering stage

Treatment	Plant height (cm)		Stem girth (mm)	
	First cycle	Second cycle	First cycle	Second cycle
B0	29.1 ± 2.3c	24.2 ± 1.7bc^{ns}	8.0 ± 0.7b	6.6 ± 1.5bc^{ns}
NPK	49.2 ± 2.39ab	16.6 ± 2.7c*	14.0 ± 1.6a	6.1 ± 1.2c*
B10	39.8 ± 3.8b	20.50 ± 2.2c*	12.8 ± 1.0ab	6.9 ± 0.5bc*
B20	42.8 ± 2.5ab	24.50 ± 3.2bc*	13.6 ± 1.2a	7.9 ± 0.6abc*
CP20	50.1 ± 1.4a	37.25 ± 0.6a*	16.4 ± 1.0a	10.9 ± 0.2a*
B10CP20	48.3 ± 0.9ab	32.43 ± 2.7ab*	16.0 ± 1.0a	10.1 ± 0.5ab*
B20CP20	47.8± 1.3ab	36.25 ± 0.6a*	15.8 ± 1.3a	11.13 ± 0.5a*
P value	< 0.001	< 0.001	< 0.001	< 0.001
Crop cycle × Amendments	< 0.001		< 0.001	

Each value is the mean (± standard error) from four replicates. For the second crop cycle, * indicate a significant difference from the first crop cycle, "ns" indicates no difference ($P > 0.05$). Different letters in a column indicate significant differences among treatments within the same crop cycle and same letters mean there are no differences ($P < 0.05$). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar); CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

In the first cycle, plant height and stem girth of the compost (CP20), biochar (B10 and B20), and combined biochar and compost applications (B10CP20 and B20CP20) were comparable to the chemical fertiliser (NPK) but were significantly ($P < 0.001$) higher than the unamended control (B0). However, in the second cycle, only the B20CP20 significantly ($P < 0.001$) increased okra growth compared to the B0 (Table 14).

The result showed that pod numbers per plot varied significantly ($P < 0.01$) among the treatments in both cycles (Figure 31a). Regardless of the treatment, pod numbers were generally higher in the first cycle than in the second cycle. In the first cycle, more significant ($P < 0.01$) pod numbers were harvested from B20, CP20, B10CP20 and B20CP20 than the B0. However, pod numbers of NPK treatment did not differ from the EFB amendments. In the second crop cycle, only the number of pods harvested from the CP20 was significantly ($P < 0.05$) higher than the B0.

Generally, pod yields of all the treatments significantly ($P < 0.01$) decreased on average in the second crop cycle by threefold (Figure 31b). Moreover, relative to the B0, okra pod yield was significantly ($P < 0.001$) increased by B20, CP20, B10CP20 and B20CP20 treatments in the first cycle crop cycle (Figure 31b). The increase in pod yield in the first cycle followed this order; B20 < B10CP20 < CP20 < B20CP20, which corresponds to 222%, 255%, 306% and 348%, respectively, higher than the B0. Also, the yield from CP20 and B20CP20 were 58% and 100%, respectively, higher than the NPK treatment. However, in the second cycle, only the pod yield from B20CP20 was significantly ($P < 0.05$) higher than that obtained in the B0.

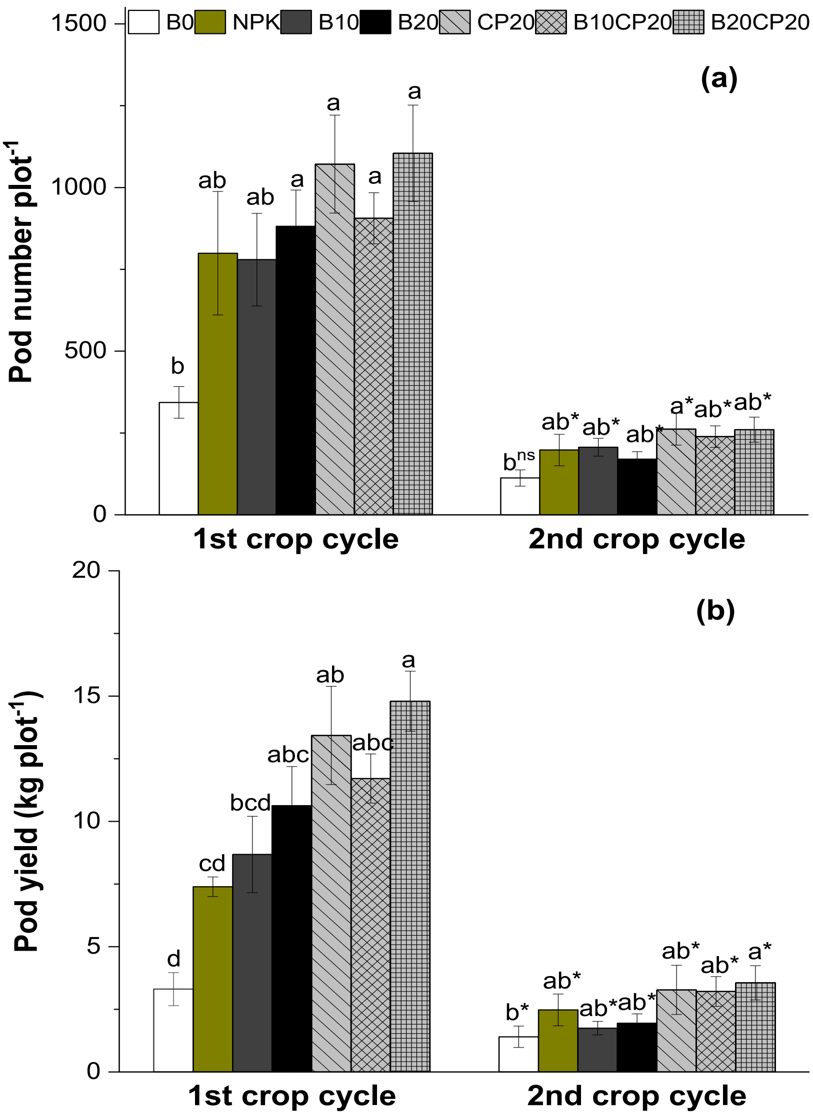

Figure 31: (a) Okra pod number and (b) pod yield in response to one-time application of amendments across two cropping cycles. For the second crop cycle, bars with * indicate a significant ($P < 0.05$) difference from the first crop cycle. Bars with "ns" indicate no difference ($P > 0.05$). Bars with different lowercase letters mean differences among amendments over the same crop cycle ($P < 0.05$). Error bars show the standard error of means from four replicates. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

The study assessed the effect of sole application of EFB biochar and compost applied singly and combined on okra growth, pod yield, N and P contents, uptakes and use efficiency across two cropping cycles. The result revealed that sole biochar, compost and co-application of biochar and compost increased okra pod yield compared to the unamended control in the first crop cycle. Similarly, Agegnehu *et al*. (2016a) reported that mean grain yield responses of barley to the single and combined compost and biochar application to a Nitisol were 30–78% higher compared to the unamended soil at two sites in central Ethiopian highlands. In contrast, Frimpong *et al*. (2021) found that sole corn cob biochar did not increase maize grain yield. However, They observed that sole NPK, sole compost, combined biochar and compost additions increased maize yields by 56.2%, 69.8% and 75.3%, respectively, relative to the unamended soil.

The positive effect of organic amendments on the growth and yield of crops could be attributed to improvement in percolation, reduced loss of nutrients, better nutrient absorption, buffering capacity and increased water storage capacity and soil water retention, which helped plants better overcome critical climate conditions, like droughts (Adugna, 2016), and supply of essential plant nutrients, including N, P, K, Ca, Mg and S, and a variety of trace elements (Agegnehu *et al*., 2014). In contrast to our findings, Syuhada *et al*. (2016) observed that adding oil palm empty fruit bunch biochar to a sand Podzol soil increased the CEC but did not increase the growth and yield of corn. They argued that EFB biochar addition did not significantly improve the growth and yield of corn since the biochar, which has highly stable carbon content, decomposed more slowly in the soil than the compost. In this study, the co-application of biochar and compost showed the highest yield, which indicates a positive synergistic effect between biochar and compost on okra growth and yield. The results agree with the findings by Iacomino et al. (2022), who observed that a mixture of olive mill wastes, orchard pruning compost, and beech wood biochar increased the growth of fennel and rapeseed by up to 100% in a field experiment.

According to Kuzyakov *et al*. (2009), the increased stabilisation of the compost by biochar promoted the chemisorption of nutrients for gradual release for plant uptake. In the current study, only the B20CP20 treatment produced a significantly higher pod yield than the unamended control in the second crop cycle, suggesting that the positive effect of the combined addition of biochar and compost on yield extended into the second crop cycle. Earlier suggestions by Kammann *et al.* (2015) indicated that biochar can adsorb plant nutrients on its surface for gradual release and that its effect persisted in increasing yield in the second season. The findings corroborate with Kheir *et al*. (2023), who demonstrated that after three years of combined corn waste compost and biochar application, faba bean yield increased by 13.6% compared to their single applications. Similarly, Frimpong *et al*. (2021) reported that without any fresh application of amendments in the second cropping season, the sole compost and combined biochar and compost increased okra pod yield by 135% and 81.4%, respectively, relative to the unamended plot. The higher effect of the combined biochar and compost could be related to the presence of biochar in the treatment. The present results suggest that the differences in the quantity, availability and rate of nutrient release in the amendments may lead to their differential influence on crop yields.

4.3.2 Effect of soil amendments on okra nitrogen and phosphorus content, uptakes and use efficiency across the two cropping cycles

Nitrogen and phosphorus contents of okra did not significantly ($P > 0.05$) vary between the two cropping cycles but were significantly ($P < 0.05$) impacted by the amendments (Figure 32a). Sole biochar application at 10 t ha^{-1} significantly ($P < 0.05$) decreased okra nitrogen content by 25.6% compared to B0 in the first cycle. However, in the second cycle, the B10CP20 and B20CP20 decreased the okra nitrogen content significantly ($P < 0.05$) compared to B0. Okra phosphorus content was significantly ($P < 0.05$) increased by B20CP20 by 118% higher than B0 in the first cycle. In the second cycle, CP20, B10CP20 and B20CP20 significantly ($P < 0.001$) increased okra phosphorus content higher than the NPK and B0 treatments (Figure 32b).

Nitrogen uptake of plant amended with NPK, CP20, B10CP20 and B20CP20 in the first cycle was significantly ($P < 0.01$) higher than in the second cycle. Nitrogen uptake in both cropping cycles was significantly ($P < 0.01$) increased by CP20 and B20CP20 relative to the B0 (Figure 32c). However, only the P uptake of CP20 treatment differed significantly ($P < 0.05$) between the two cycles, with the second cycle recording a significantly lower P uptake than the first cycle (Figure 32d). In the first cycle, the P uptake of the EFB treatments was similar to the NPK treatment. However, in the second cycle, the P uptake in CP20, B10CP20 and B20CP20 was significantly ($P < 0.01$) higher than the NPK treatment.

The apparent nitrogen recovery efficiency (N ARE) of NPK, CP20, B10CP20 and B20CP20 treated plants declined significantly ($P < 0.01$) in the second crop cycle (Figure 32e). The CP20 and B20CP20 significantly ($P < 0.01$) increased N ARE in both crop cycles, compared to B0. In contrast, only the plants in the CP20 treatment recorded a significant ($P < 0.05$) decline in apparent phosphorus recovery efficiency (P ARE) in the second cycle (Figure 32f). Moreover, B20CP20 recorded the highest P ARE, which was significantly ($P < 0.001$) higher than NPK and B0 treatments in both cycles. Also, CP20 significantly ($P < 0.001$) increased P ARE relative to B0 in the first cycle and NPK and B0 treatments in the second cycle. Notably, B20 marginally decreased the uptake and recovery of N and P compared to B10 in the first crop cycle. However, the trend was reversed in the second cycle, where B20 increased N and P uptake and recovery efficiency compared to B10 (Figure 32e, f).

Low nitrogen contents in the okra shoots observed in both crop growth cycles were accompanied by increased okra growth and pod yields. According to Jarrell and Beverly (1981), this inverse relationship between growth and mineral concentration is termed the 'dilution effect', which occurs when dry-weight accumulation increases faster than mineral-nutrient accumulation. Conversely, phosphorus exhibited a 'synergistic effect', which Riedell (2010) attributed to nutrient accumulation increasing faster than dry matter. Yang *et al*. (2020) observed that potato nitrogen (N) content significantly decreased in biochar-amended soils compared to treatments without biochar. In contrast, Anwar *et al*. (2021) reported that the sole application of wheat straw biochar increased N and P content in okra shoots compared to compost and unamended treatments due to biochar’s ability to increase soil pH and CEC, enhancing nutrient availability and retention for use by plants.

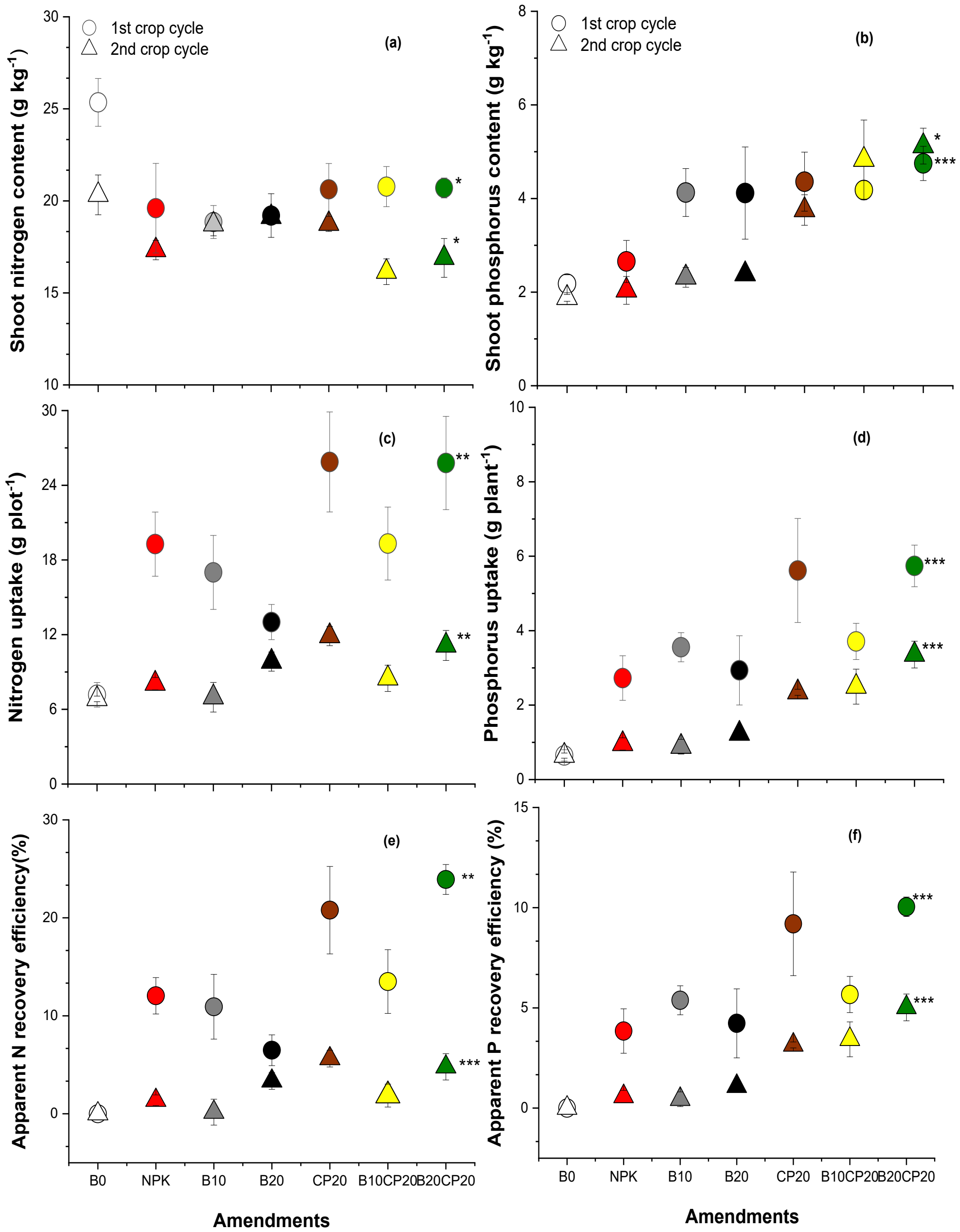

Figure 32: (a) Okra shoot nitrogen (N) content, (b) shoot phosphorus (P) content, (c) N uptake, (d) P uptake, (e) apparent nitrogen recovery efficiency and (f) apparent phosphorus recovery efficiency, as affected by crop cycles and soil amendments. Error bars show the standard error of means from four replicates. Asterisks *, **, ***, denote significant difference among the treatments at $P < 0.05$, $P < 0.01$, and $P < 0.001$, respectively. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Also, Frimpong *et al*. (2021) reported an increased okra N and P concentration from residual sole corncob biochar and biochar plus maize straw compost treatments, compared to sole compost and unamended treatments. Furthermore, the sole compost and combined biochar and compost applications (each applied at 20 t ha^{-1}) increased the N and P uptake and apparent recovery efficiency in both crop cycles.

However, N and P recovery efficiencies in the sole biochar treatments were not significantly different from those in the NPK and unamended treatments. Awodun *et al*. (2007) also showed that EFB biochar application increased crop growth and yield and enhanced phosphorus uptake by increasing the pH of acidic soils. Also, Amlinger *et al*. (2007) explained that compost application affects plant yield and nutrient use efficiency by intensifying essential interactions between root hairs, soil fauna and microorganisms due to an enhancement of specific surface area. The study showed that different organic amendments exhibit diverse effects on plant growth, yield, and nutrient use efficiency. EFB biochar and compost may display varying impacts based on the preparation condition, application rates, soil types, crop species and experimental duration. Understanding these influences is crucial for sustainable soil management and crop production.

4.3.3 Influence of treatments on soil chemical and microbial properties

Soil pH did not change between the two cropping cycles (Table 15). However, a significant *(P* < 0.001) pH increase of 1.0 and 1.4 units was recorded for soil amended with B10CP20 and B20CP20, respectively, compared to NPK in the first cycle. In the second cycle, all EFB amendments significantly (P < 0.001) raised the soil pH higher than the pH of the B0, but only the pHs of B10CP20 and B20CP20 treatments were significantly ($P <$ 0.001) higher than the NPK treatment (Table 15).

Moreover, only the cation exchange capacity (CEC) of B20CP20 was significantly ($P < 0.05$) higher in the first cycle when the two cropping cycles were compared (Table 15). Again, the B20CP20 significantly ($P < 0.001$) increased the CEC of the soil by 47.7, 46.8%, 54.8% and 85% relative to B10, B20, CP20, NPK and B0, respectively, in the first cycle, but in the second crop cycle, all the treatments did not significantly ($P > 0.05$) affect the CEC of the soil.

A significant ($P < 0.001$) increase in the soils' total organic carbon (TOC) content was observed across both cropping cycles (Table 15) after single and combined applications of EFB biochar and compost. Specifically, the soils' TOC content declined significantly ($P < 0.01$) by 33.5% from the first to second crop cycle (Table 15) and the most significant ($P < 0.05$) decline was seen in B20CP20.

Table 15: Changes in soil chemical properties in response to soil amendments across the croping cycles

Crop cycle	Treatment	pH	Cation exchange capacity (cmol $_{(+)}$ kg^{-1})	Total organic carbon (g kg^{-1})
First cycle	B0	5.8 ± 0.1d	6.1 ± 0.4b	8.7 ± 1.1d
	NPK	6.0 ± 0.2cd	7.3 ± 0.8b	10.8 ± 1.5d
	B10	6.4 ± 0.0cd	6.9 ± 0.2b	13.9 ± 2.2bcd
	B20	6.4 ± 0.3bcd	8.4 ± 1.3b	19.5 ± 1.3bc
	CP20	6.6 ± 0.2bc	7.7 ± 0.9b	12.4 ± 1.4cd
	B10CP20	7.0 ± 0.1ab	8.5 ± 1.2b	18.3 ± 0.8bc
	B20CP20	7.4 ± 0.8a	11.3 ± 0.3a	27.7 ± 0.1a
P value		< 0.001	< 0.001	< 0.001
Second cycle	B0	5.7 ± 0.3c	5.8 ± 0.1^{ns}	7.7 ± 1.0c^{ns}
	NPK	6.1 ± 0.2bc	6.6 ± 0.5^{ns}	9.1 ± 1.1c^{ns}
	B10	6.4 ± 0.2ab	6.7 ± 0.6^{ns}	10.6 ± 0.7abc^{ns}
	B20	6.3 ± 0.1ab	7.1 ± 0.6^{ns}	16.6 ± 2.6a^{ns}
	CP20	6.6 ± 0.1ab	6.7 ± 0.1^{ns}	9.5 ± 0.4bc^{ns}
	B10CP20	7.0 ± 0.0a	6.6 ± 0.2^{ns}	13.7 ± 1.7abc^{ns}
	B20CP20	7.01 ± 0.0a	7.9 ± 0.3*	16.1 ± 1.5ab*
P value		< 0.001	ns	< 0.001
Crop cycle × treatments		ns	0.006	< 0.001

Each value is the mean (± standard error) from four replicates. For the second crop cycle, * indicate a significant difference from the first crop cycle, "ns" indicates no difference (P > 0.05). Different letters in a column indicate significant differences among treatments within the same crop cycle and same letters mean there are no differences (P < 0.05). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

In the first cycle, the highest TOC content of 27.65 g kg^{-1} recorded by B20CP20 was 2.2 and 1.55 folds higher than the B0 and NPK treatments, respectively, while TOC contents of B20 and B10CP20 were significantly (P < 0.001) higher than B0 by 1.3 and 1.1 folds respectively.

Table 16: Soil macro and micronutrient contents as affected by the soil amendments at the end of the second crop cycle

Treatment	Total N ($g\ kg^{-1}$)	Available P ($mg\ kg^{-1}$)	Exchangeable K ($mg\ kg^{-1}$)	Exchangeable Mg ($mg\ kg^{-1}$)	Available Zn ($mg\ kg^{-1}$)	Available Fe ($mg\ kg^{-1}$)
B0	0.73 ± 0.09b	< 4b	112 ± 6.6b	90.8 ± 14.6b	1.24 ± 0.18c	33.8 ± 1.7bc
NPK	0.85 ± 0.1ab	< 4b	107.8 ± 16.6b	131 ± 34ab	1.78 ± 0.26bc	39.8 ± 4.6abc
B10	0.75 ± 0.05ab	< 4b	160 ± 4ab	115 ± 2.8ab	1.5 ± 0.11c	32 ± 0.9c
B20	0.98 ± 0.18ab	< 4b	175 ± 15.5ab	122.8 ± 14ab	1.57 ± 0.12c	38.1 ± 3.1bc
CP20	0.83 ± 0.05ab	10.3 ± 1.2ab	157.5 ± 14ab	117.5 ± 7.5ab	2.3 ± 0.18bc	40.9 ± 3.2abc
B10CP20	1.0 ± 0.17ab	20.8 ± 4.5a	152.5 ± 18ab	140 ± 12ab	2.9 ± 0.14ab	46 ± 2.5ab
B20CP20	1.03 ± 0.05a	18.8 ± 6.4a	212.5 ± 35.9a	162.5 ± 14a	3.8 ± 0.63a	51.8 ± 3.2a
P value	0.043	0.007	0.02	0.044	< 0.001	0.002

Each value is the mean (± standard error) from four replicates ($P < 0.05$). Different letters in a column indicate significant differences among treatments and same letters mean there are no differences ($P < 0.05$). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Meanwhile, in the second cycle, the highest TOC content (16.65 g kg^{-1}) was recorded in the B20 and was 1.15 and 0.84 folds higher than the B0 and NPK treatments, respectively.

The extent to which EFB biochar and compost application affected the various soil micro and macro nutrients varied (Table 16). Specifically, the available phosphorus content of the soil amended with B0, NPK, B10 and B20 was lower than 4 mg kg^{-1}, which was significantly ($P < 0.05$) less than the available phosphorus content of B10CP20 and B20CP20 (Table 16). Also, only the B20CP20 significantly increased the soil's total nitrogen, exchangeable potassium and magnesium contents compared to B0. Again, B20CP20 significantly ($P < 0.05$) increased the soil zinc content compared to the other treatments. Likewise, the B20CP20 had significantly ($P < 0.05$) higher soil iron content than the single biochar treatments and the unamended control (Table 16).

The result of the effect of the treatments on soil microbial biomass showed that CP20 significantly ($P < 0.05$) increased the biomass of arbuscular mycorrhizal fungi (AMF) and gram-positive bacteria compared to the B20 and NPK treatments (Table 17). Meanwhile, AMF and gram-positive bacterial biomass in the CP20 treatment did not differ from the B0, B10CP20 and B20CP20. There was a tendency for B20 to decrease soil microbial abundance, thus recording 5% and 23% less biomass of AMF and gram-positive bacterial biomass, respectively, relative to the B0 treatment (Table 17). However, the saprophytic fungal biomass was not significantly ($P > 0.05$) affected by the EFB amendments

Table 17: Soil microbial properties as affected by the soil amendments at the end of the second crop cycle

Treatment	AMF (nmol g^{-1} soil)	Saprophytic fungal (nmol g^{-1} soil)	Gram-positive bacteria (nmol g^{-1} soil)
B0	17.3 ± 1.8ab	77.1 ± 4.4	100.4 ± 4.5ab
NPK	15.9 ± 1.1b	54.9 ± 4.9	75.6 ± 0.1b
B10	20.2 ± 1.5ab	58.8 ± 6.3	87.2 ± 0 .0ab
B20	16.4 ± 1.2b	43.8 ± 6.2	76.9 ± 9.1b
CP20	29.2 ± 1.6a	63.1 ± 2.9	109.2 ± 9.9a
B10CP20	22.3 ± 2.1ab	60.0 ± 5.1	81.4 ± 7. 1ab
B20CP20	22.9 ± 5.5ab	67.2 ± 21.1	81.2 ± 9.1ab
P value	0.027	ns	0.019

Different letters in a column indicate significant differences among treatments and same letters mean there are no differences ($P < 0.05$). Each value is the mean (± standard error) from three replicates. AMF, arbuscular mycorrhizal fungi; B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Soil pH is an important indicator of soil properties influencing nutrient availability and crop growth. In the current study, soil pH was significantly increased by the single and combined applications of EFB biochar and compost by 0.6 to 1.6 points, compared to the unamended soil. Frimpong *et al.* (2021) observed an increased in soil pH from sole corn cob biochar, maize straw compost, and biochar plus compost in this same soil type after three cropping cycles. Cheah *et al.* (2014) attributed the liming effect of biochar to the surface organic functional groups (as conjugate bases), soluble organic compounds (also conjugate bases of weak acids), carbonates (salts of bicarbonate and carbonate), and other inorganic alkalis. Agegnehu *et al.* (2014) attributed the liming effect of compost application to its richness in alkaline cations such as Ca, Mg and K, which are liberated from organic matter due to mineralisation. Haynes and Mokolobate (2001) explained that the soil pH ameliorating effect of EFB compost can be attributed to either the oxidation of organic-acid anions from the decomposition, ammonification of organic nitrogen, or the specific adsorption of humic materials and/or organic acids onto Al and Fe hydrous oxides or the combination of these mechanisms. The high pH, calcium, carbonate, and ash contents of the EFB compost and biochar used in this study could explain the increase in soil pH observed in the current study. In disagreement with our findings, a one-time single and combined application of olive mill waste compost (applied at 20 t ha^{-1}) and beech wood biochar applied at (30 t ha^{-1}) did not significantly increase soil pH in the first year, and soil pH decreased significantly after the second year in all treatments (Iacomino *et al*., 2022). Zhang *et al*. (2016) explained that adding biochar can lead to a decrease in pH due to the adsorption of ammonia, thus preventing the dissolution of ammonia in compost solutions that release hydroxide ions. Busch and Glaser (2015) also reported that compost-biochar mixtures had no impact on soil pH, even at a rate of application of 25 t ha^{-1}. These contrasting reports suggest that the varying effects of organic amendments on soil pH depend on the type of amendment, sole or combined application of different amendment types, rate of application, and duration in the soil.

Co-application of EFB biochar and compost (each applied at 20 t ha^{-1}) significantly increased the CEC of the soil by 85 % and 54.8 %, compared to the NPK and B0 treatments, respectively, in the current study. The higher CEC in soil amended with B20CP20 may partly be explained by Gunarathne *et al*. (2017), who found that the large surface area, high negative surface charge, and greater charge density of biochar increased soil CEC. Also, Krull *et al*. (2004) explained that the production of carboxylic acids from easily decomposable organic matter of the compost increased soil CEC. Liu *et al*. (2012) found that the soil's CEC was significantly increased when compost applied at 32.5 t ha^{-1} was combined with biochar at 20 t ha^{-1}. Contrary to the findings from the present study, in a two-year field experiment by Nobile *et al*. (2022), compost-biochar mixtures failed to increase the CEC of soils in three different experimental fields.

The increased TOC of the soil after amending with sole biochar at 20 t ha^{-1} and combined biochar and compost was because of the enrichment of the soil with materials with high carbon content (EFB biochar and compost with total carbon of 44.8% and 14.2%, respectively) (Supplementary Table S1). Also, the higher TOC content of the B20 in the second cycle could result from higher aromaticity of the carbon content of EFB biochar, making it highly stable and resistant to oxidation and microbial degradation. Findings from this study agree with Amoakwah *et al*. (2020), who found a significant increase in TOC in the soil type used in this study following the application of 30 t ha^{-1} of corn cob biochar. Iacomino *et al.* (2022) also observed significantly higher TOC in biochar alone in the second year compared to other treatments and this corroborated with this study.

In this study, a significant increase in total N, available P, exchangeable K and Mg and Zn and Fe content of the soil was observed in B10CP20 and B20CP0 treatments, which could primarily be related to the high nutrient content of the compost component. While biochar contained negligible N and P amounts, it's high C: N ratio (54:1), aromaticity and surface area likely increased nutrient retention and reduced nutrient loss from the compost. The increase in Mg, Ca, and K is potentially due to the high ash content of the biochar (Zama *et al*., 2017). In line with the findings from Frimpong *et al*. (2021), an increased soil N and K were observed from sole corn cob biochar, maize straw compost and biochar plus maize compost after three cropping cycles, compared to the unamended soil.

Conversely, Nobile *et al*. (2022) noted that three diverse compost-biochar mixtures led to no significant changes in soil nutrient stock compared to the mineral treatment and compost alone. They attributed the lack of effect of compost-biochar mixtures on available nutrient concentrations to the fact that nutrient availability depends on soil fertility status and the source of biochar and its application rate. Even though organic amendments, especially compost and biochar, are frequently considered to improve soil fertility, their effects on the chemical properties of soils might vary, depending on the source and condition of processing of amendment, rate and frequency of application, the type and initial fertility status of soil and even the climatic conditions, being it temperate or tropical under which they are used.

The sole EFB compost amendment significantly increased the arbuscular mycorrhizal fungi and gram-positive bacterial biomass compared to the B20 and NPK treatments. The significant increase in the gram-positive bacteria after the sole compost application may be attributed to readily degradable carbon in the compost as an energy source for soil microbes. In contrast, AMF is an obligate biotroph, so the increase in soil AMF in response to compost application must be due to other qualities of the compost, such as the amount of humic acid and nitrogen, which were reported to stimulate AM hyphal growth and sporulation (Gryndler *et al*., 2009). Similar to the present findings, Biasi *et al*. (2005) found that labile soil organic matter produced significant shifts in soil AMF, saprophytic fungi, and gram-positive and gram-negative bacteria composition. Contrary to the current study, Azeem *et al*. (2020) reported that biochar addition increased the abundance of bacteria (16S rRNA) and gram-positive bacteria in soils planted with mash beans. Moreover, Xin *et al*. (2022) demonstrated that one-time rice straw biochar addition to black soil significantly enhanced AMF alpha-diversity and modified AMF community composition, while one-time cow dung and maize straw compost addition did not show any significant effect on AMF after two years of application.

In the current study, the EFB amendments did not significantly increase saprophytic fungal biomass compared to the unamended control, and there was a potential inhibitory effect of EFB biochar and compost application on soil saprophytic fungal biomass. Birk *et al*. (2009) observed that on a highly weathered Xanthic Ferralsol soil, the single addition of compost and mineral fertiliser caused an elevated proportion of saprophytic fungal biomass compared to the unamended control. However, in the soil where compost was applied in combination with charcoal, there were a reduced proportion of saprophytic fungi compared to the soil, which differed only from the lack of compost addition. The sampling time (at the end of the second crop cycle in this case), together with the method of microbial analyses (signature fatty acids that detect living organisms as fatty acids that decompose quite fast when the organisms die), might have contributed to the negligible effect of combined biochar and compost application on saprotrophic fungal biomass obtained current in the study. However, Wang *et al*. (2020) suggested that the negative effect of biochar application on soil microbial communities may be due to the toxic effect of heavy metals, chlorinated hydrocarbons and polycyclic aromatic hydrocarbons biochar on soil microorganisms when applied at a high rate. In the current study, the sum of 16 Environmental Protection Agency (EPA)-PAH of the EFB biochar was 20.3 mg kg^{-1} (dry matter), which is above the International Regulatory Maximum Allowed Thresholds of 6 mg kg^{-1} (dry matter), and this can be a potential cause of the inhibitory effect of a higher rate of application of the biochar on the saprophytic fungal biomass. The findings from the current study show that the synergistic or summative effect of combined biochar and compost on soil microbial properties is negligible. Nevertheless, the differences in the response of the soil microbial community to biochar amendment may be explained both by different soil types and soil microbial compositions but also by the differences in quality and quantity of biochar in the different studies (Yang *et al*., 2020), so it may not be possible to draw general conclusions.

4.3.4 Soil chemical and microbial properties drive okra nutrient use efficiency and yield

Pearson's correlation analysis was performed to determine whether there was a linear relationship between okra growth and yield parameters and soil properties (Figure 33). In both cropping cycles, soil pH and zinc content correlated positively with the okra growth and yield parameters, except shoot nitrogen content that correlated negatively with both soil pH and zinc content (Figure 33a). Moreover, the TOC and CEC correlated significantly ($P < 0.05$) with okra pod yield, N ARE and P ARE in the first cropping cycle. In this same cycle, soil AMF biomass correlated positively with okra pod yield, N and P uptakes, and apparent N and P recovery efficiency.

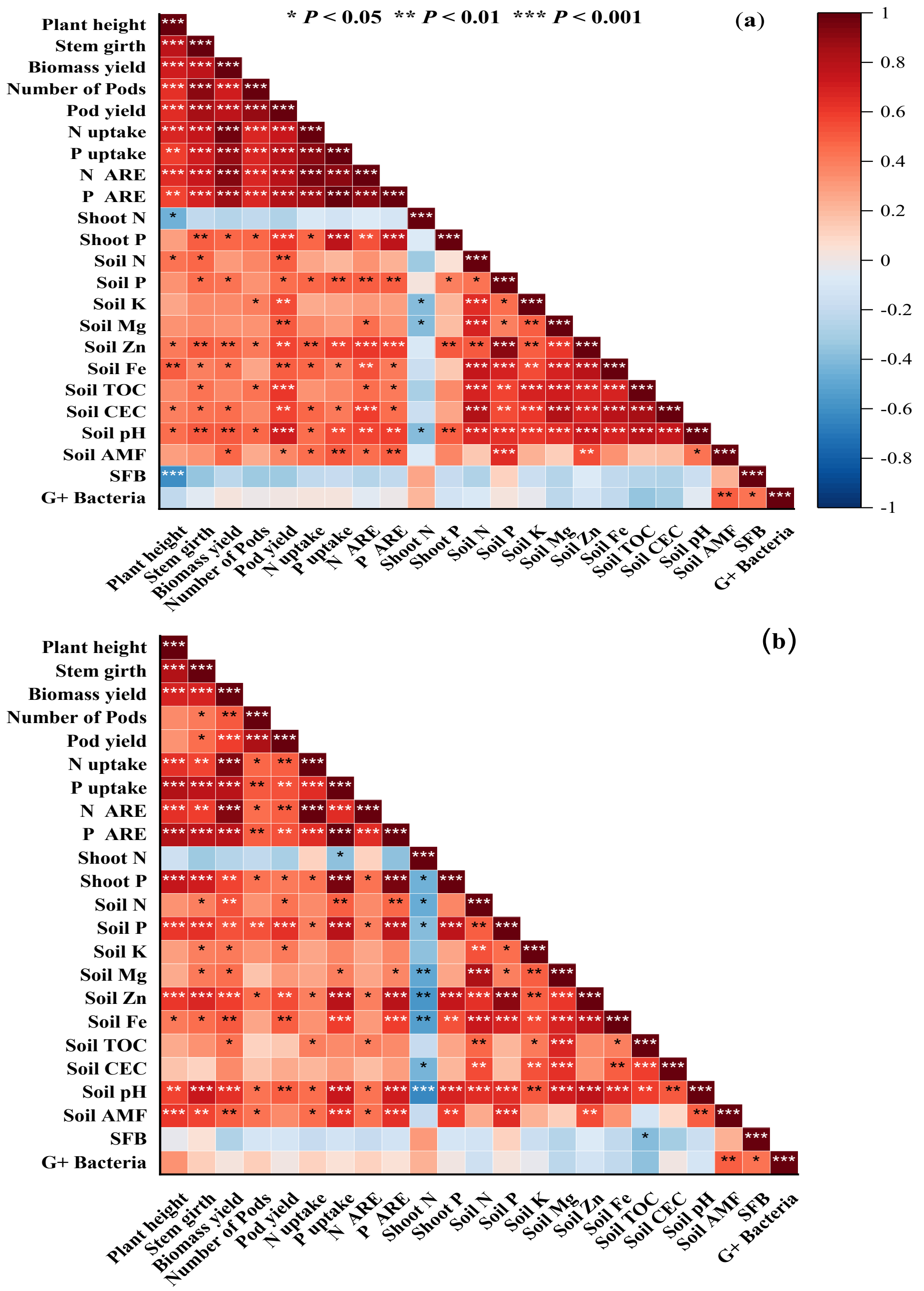

Figure 33: Pearson correlation matrix for soil chemical and microbial properties with okra growth, yield and nutrient uptakes and use efficiency in the (a) first and (b) second crop cycle. n = 28; Statistical significance is denoted as *** ($P < 0.001$), ** ($P < 0.01$) and * ($P < 0.05$). N ARE, nitrogen apparent recovery efficiency; P ARE, phosphorus apparent recovery efficiency; TN, total nitrogen; TOC, total organic carbon; CEC, cation exchange capacity; AM fungi, arbuscular mycorrhizal fungi; SFB, saprophytic fungi biomass.

Again, AMF biomass showed a significant ($P < 0.05$) positive correlation with soil pH, available P and zinc contents (Figure 33a). Similar to the first crop cycle, in the second crop cycle, soil pH, available P and zinc contents correlated positively with growth, yield parameters and N ARE and P ARE (Figure 33b). However, soil TOC and CEC did not correlate with okra pod yield in the second crop cycle. Soil AMF biomass had a significant ($P < 0.05$) positive correlation with soil available P and okra P content, uptake and apparent recovery efficiency in the second cycle (Figure 33b).

Results of the principal component analysis showed that, in the first cycle, soil pH, CEC, total N, Zn, N ARE and P ARE were positively associated with PC1, which explained 77.6% of the variance in the data set and these properties were regulated by B20CP20 (Figure 34a). The PC2 explained 10.0% of the variance in the data and was negatively associated with soil TOC and Mg and was positively associated with AMF biomass, which CP20 regulated. On the other hand, the number and yield of pods and soil available P, K and Fe were associated with PC3, which explained 6.6% of the variance in the data set. However, in the second crop cycle, PC1 captured 71.3% of the total variation among treatments; this component includes N and P ARE, soil pH, P, Mg, Fe and Zn content and these properties were regulated by B20CP20 (Figure 34b).

The PC2 and PC3 accounted for 14.6% and 5.5% of the total variation in soil properties and yield components. The second PC consisted of pod number and yield, soil N, TOC and AMF biomass, while the third PC consisted of CEC and K and were regulated by CP20 and NPK treatments, respectively.

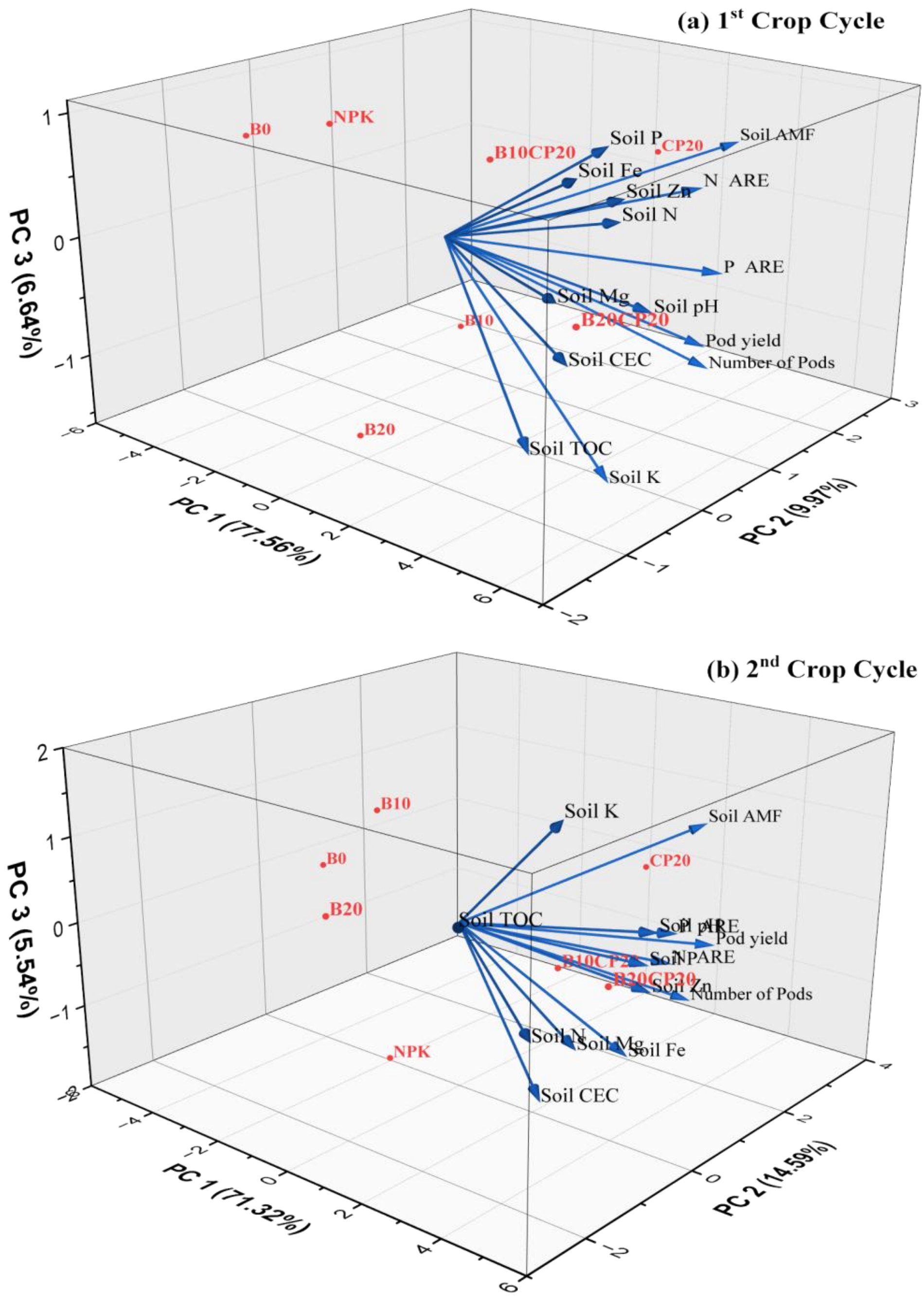

Figure 34: Biplot of principal component analysis (PCA) of pod yield, N and P apparent recovery efficiency and selected soil chemical and microbial properties for the (a) first crop cycle and the (b) second crop cycle. The loadings were pod number, pod yield, apparent nitrogen recovery efficiency (N ARE), apparent phosphorus recovery efficiency (P ARE), total organic carbon (TOC), total nitrogen (N), cation exchange capacity (CEC), pH, available phosphorus (P), potassium (K), iron (Fe), zinc (Zn), and arbuscular mycorrhizal fungi (AMF), with the soil amendments as scores. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Soil pH emerged as a major driver for the increased growth, pod yield, nutrient uptake and recovery across the two cropping cycles; this is reflected in the significant positive correlation between pH and okra growth, yield, nutrient uptake, recovery efficiency and soil chemical and microbial parameters in both cropping cycles. Furthermore, in the second crop cycle, soil pH, P, Mg Fe and Zn content and okra N and P ARE were the components of PC1, which captured 71.3% of the total variation among treatments. According to Blackwell *et al*. (2010), modifying soil pH after biochar application gives feedback on nutrient release, immobilisation and availability and alters microbial community structure. Also, Miller (2016) further elaborated that an increase in the pH of acidic soils decreases the sorption of P on Fe and Al oxides and thus makes P available for plant uptake or indirectly affects the availability of nitrogen, potassium, calcium and magnesium as well as the activities of soil microbes in the soil to drive crop yield.

Moreover, the zinc content of the soil was another driver of the increased okra yield and apparent nutrient recovery efficiency because zinc is known to play key roles in plant growth and productivity by enhancing the production of auxins, increasing the number of inflorescences and accelerating the rate of seed maturity as enunciated by Rudani *et al*. (2018). Despite this, the presence of AMF coincided with increased P uptake and a noticeable recovery in the second cycle. The indication of a positive contribution of AMF on P uptake and okra yield under phosphorus-deficient conditions can be explained by the integrated role played by AMF in plant nutrient uptake, increased resistance against abiotic and biotic stresses indicated by Ravnskov *et al*. (2020). Smith and Smith (2011) further elaborated that the most effective pathway by which AMF improves plant P nutrition and growth is to reduce the impact of P depletion in the rhizosphere by scavenging P from large volumes of soil and rapidly delivering to the plant root. Furthermore, Karimi *et al.* (2011) reported that phosphorus content in the shoots and roots of AM plants was significantly higher than in non-mycorrhizal plants grown under soil phosphorus deficiency. Together, these results suggest that differences in the properties of the amendments were the major drivers of the change in soil chemical and microbial properties in the amended soils that affected okra growth, yields and nutrient use efficiency across the two cropping cycles. The optimisation of soil pH, zinc availability and increased AMF biomass were the major drivers of okra yield and phosphorus use efficiency after soil amendment with EFB biochar and or compost. Nevertheless, these chemical and biological drivers of yield may change when the quantity and quality of the organic amendments and the soil type change.

4.3.5 Summary

Soil amendment with compost or biochar has proven to stabilise and increase crop yields due to their multiple positive effects on the physical, chemical and biological soil properties. However, the longevity of the effect of combined biochar and compost application on crop yields is yet to be adequately examined. The effects of one-time EFB biochar and/or compost application on soil chemical and microbial properties, nutrient use efficiency, and okra yields in two cropping cycles were examined. Sole application of EFB biochar at 20 t ha^{-1} increased soil total organic carbon and pH but did not affect soil nutrient content, okra nutrients uptake, and use efficiency, and it decreased soil microbial biomass. Sole EFB compost application increased soil microbial properties, especially AMF biomass, which drove phosphorus uptake and use efficiency when soil phosphorus content was deficient but did not increase soil total organic carbon and okra pod yield in the second crop cycle. The study demonstrated that the combined biochar and compost application to acidic and low-nutrient tropical soil can sustain okra productivity beyond one cropping cycle. This was manifested in the increased okra pod yields, nutrient uptakes and recovery efficiency by optimising soil pH and soil nutrient availability and balance, especially phosphorus and zinc content. A one-time combined application of EFB biochar and compost can positively enhance soil chemistry and microbial properties to drive okra yield and nutrient use efficiency across two cropping cycles, especially in tropical low-nutrient soils. However, long-term studies involving different soil types, varying application rates of EFB compost, and economic analyses are recommended.

4.4 Empty Oil Palm Fruit Bunch Biochar and Compost Application Affect Soil Carbon Storage and Management Index to Drive Soil Physical Fertility

Soil organic carbon is vital for sustaining yields because it influences all soil properties including water and nutrients retention and availability, soil structure and aeration and biota (Krull *et al.*, 2004). In tropical soils, which are often characterized by high temperatures, high rainfall, and intense weathering, soil organic carbon plays a vital role in nutrient cycling, water retention, and overall ecosystem productivity. The carbon management index (CMI) proposed by Blair *et al.* (1995) involves both quantity and quality factors of SOM in its estimation responds to seasonal variations more than total SOC (Laishram *et al*., 2012) and can thus be considered more effective in studying short-term seasonal changes brought about by the applications of organic amendments on soil quality. In addition to low organic carbon contents, another constraint to productivity in tropical poor soils particularly, sandy soil, are weak structure, poor water retention properties, and high sensitivity to compaction (Adekiya *et al.*, 2020) The amount of plant-available water relative to air-filled porosity at field capacity is often used to assess soil physical fertility (Kuo *et al*., 2004), and soil organic carbon has been shown to have a significant effect of these soil properties.

Consequently, this current objective is to evaluate the effect of sole and combined applications of EFB biochar and compost on soil carbon quantity, quality, stock, and management index and how they drive soil physical fertility. I hypothesized that; sole or combined applications of EFB biochar and compost would increase the quantity and quality of soil organic carbon, carbon stock and carbon lability and management indices of EFB amended soils compared to a twenty-year old orchard reference soil. The soil carbon would have a positive effect on soil physical fertility. The aforementioned hypotheses were tested in a two-season field experiment after one-time sole and combined applications of EFB biochar and compost at varying rates to a tropical sandy clay loam soil.

4.4.1 Effect of soil amendments on soil carbon components across the cropping cycles

The effect of sole and combined applications of EFB biochar and compost on soil permanganate oxidizable carbon (POXC) (labile carbon), total organic carbon and total organic carbon stock, across the two okra cropping cycles are presented on Figure 35. In the first crop cycle, the POXC content in the different EFB amended soils were not significantly ($P > 0.5$) from the chemical fertiliser (NPK) treated soil and the unamended control soil (Figure 35a). However, in the second cycle, the POXC content of the combined biochar and compost (10 t ha^{-1} and compost 20 t ha^{-1}) (B10CP20) was 79% higher than the unamended treatment (B0) but was not significantly different from all the EFB treatments and NPK treatment. The POXC contents of the EFB treatments did not differ significantly ($P > 0.05$) between the cropping cycles.

The EFB amendments application significantly ($P < 0.001$) increased soil total organic carbon contents in both cropping cycles (Figure 35b). Specifically, in the cropping first cycle, the highest TOC content (27.65 g kg^{-1}) was recorded in the B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost), and was 2 and 1.6 folds higher than the B0 and NPK treatments, respectively. Also, the TOC contents of B20 and B10CP20 were about 1.3 and 1.1 folds, respectively, higher than the B0, and were 0.8 and 0.7 folds higher than the NPK treatment. Moreover, in the second crop cycle, the highest TOC content was recorded in B20 (16.65 g kg^{-1}) which was 1.1 and 0.8 folds higher than B0 and NPK treatments. Again, the TOC content in the B20CP20 declined by 0.7-fold in the second crop cycle but differences in TOC between the two cropping cycles were not significant for the other treatments. The TOC content declined significantly ($P < 0.01$) on average by 33.5% from the first to the second cycle (Figure 35b).

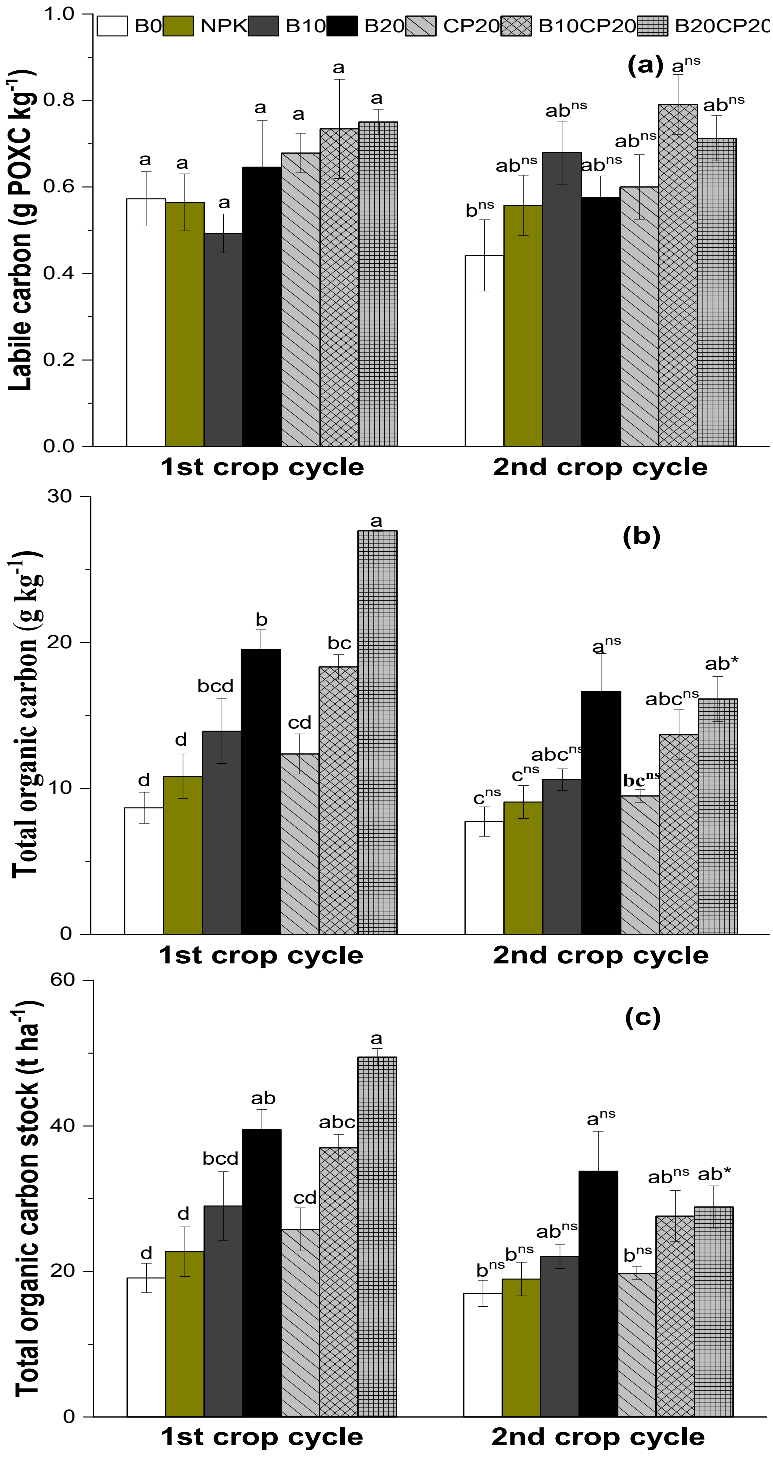

Figure 35: Changes in (a) labile carbon, (b) total organic carbon and (c) carbon stock, as affected by crop cycles and soil amendments. Error bars show the standard error of means from four replicates. For the second cropping cycle, bars with * indicate a significant difference from the first cropping cycle. Bars with "ns" indicate no difference (P < 0.05); those with different lowercase letters mean differences among amendments over the same crop cycle (P < 0.05). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Compared to the B0 and NPK, B20CP20 significantly ($P < 0.01$) increased the total organic carbon stock by 159% and 118% respectively. Moreover, B20 increased total organic carbon stock by 106% and 74% relative to the B0 and NPK treatments, respectively, in the first cycle (Figure 35c). However, in the second cycle, only B20 significantly increased significantly ($P < 0.01$) the total carbon stock by 99% and 78% relative to the B0 and NPK treatments, respectively. Again, the total carbon stock of the B20 was significantly ($P < 0.001$) higher than CP20 in both cropping cycles. Compared to the first cropping cycle total carbon stock dropped by an average of 32% in the second cycle.

The effect of sole and co-applications of EFB biochar and compost on carbon lability index and carbon management index (CMI) across the two cropping cycles are presented in Figure 36. In the first crop cycle, B0 recorded the highest (139%) carbon lability index which was twice and thrice that of the B20 and B20CP20, respectively (Figure 36a). Conversely, in the second cropping cycle, B10 recorded the highest carbon lability index (132 %) which was significantly ($P < 0.05$) higher than B20 (70.5%) which recorded the lowest carbon lability index. Again, the NPK treatment significantly ($P < 0.05$) increased the carbon lability index compared to B20. However, the EFB amendments did not significantly ($P > 0.05$) affect the lability index of carbon compared to the reference soil. Between the two crop cycles, the lability index was generally higher in the second crop cycle compared to the first crop cycle, though not significantly different ($P > 0.05$).

The carbon management index (CMI) did not significantly ($P > 0.05$) differ between the two cropping cycles (Figure 36b). Moreover, in the first cycle, B20CP0 significantly (P < 0.05) increased CMI by 70% compared to the reference soil but was not significantly different from the other treatments, including the B0.

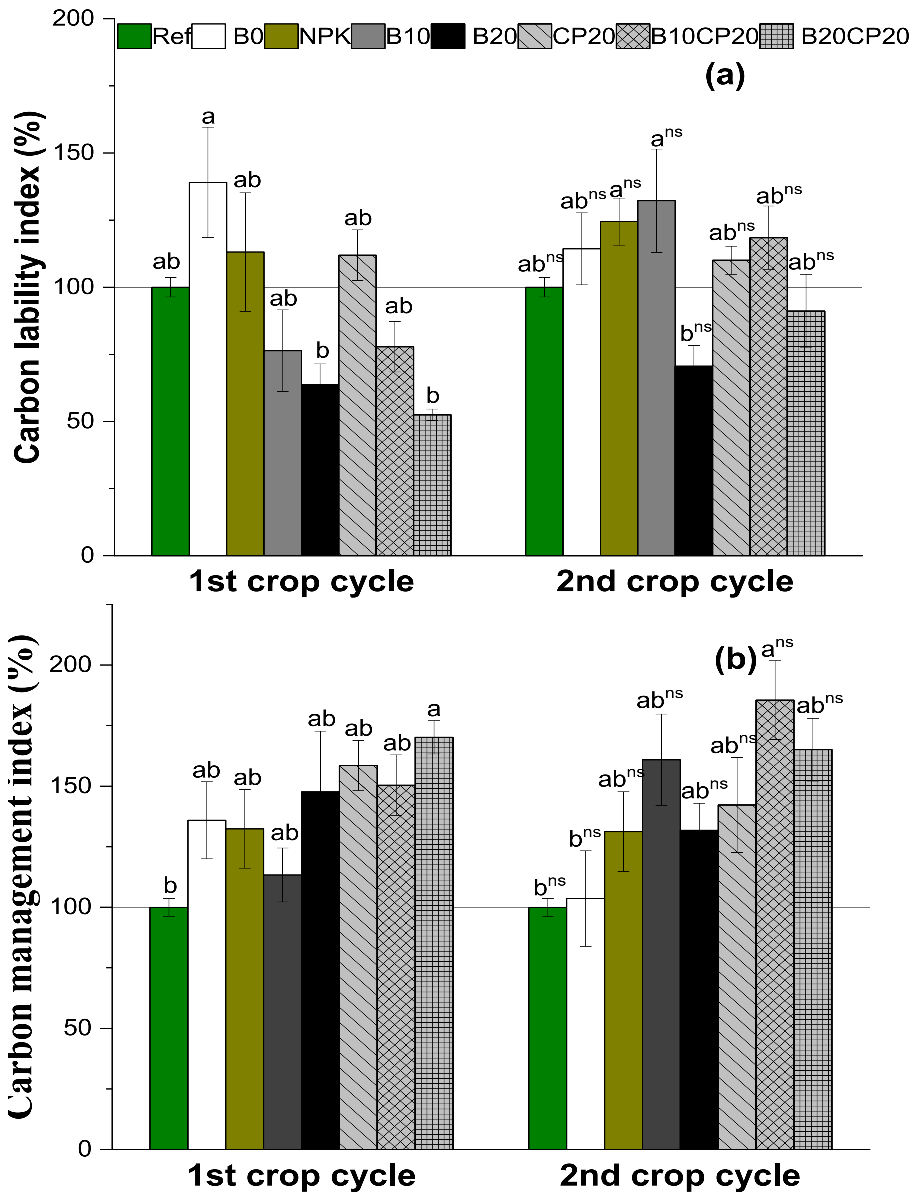

Figure 36: Effect of soil amendments on (a) carbon lability index and (b) carbon management index across two cropping cycles. Error bars show the standard error of means from four replicates. For the second cropping cycle, bars with "ns" indicate no difference (P < 0.05) from the first cropping cycle, those with different lowercase letters mean differences among amendments over the same cropping cycle (P < 0.05). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20, (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

However, the highest CMI (185.5%) was recorded by the B10CP20 in the second crop cycle, and was 86% and 82% significantly ($P < 0.05$) higher than the reference soil and B0, respectively (Figure 36b).

The current study examined the effect of sole and combined applications of EFB biochar and compost on soil labile carbon (POXC), total organic carbon (TOC), total carbon stock, carbon lability index and carbon management index across two cropping cycles. The result showed B10CP20 increased the POXC content compared to the other B0 in the second cropping cycle. This indicates that as the easily degradable carbon pools became depleted, a negative priming effect of biochar was observed with increasing biochar application rate as reflected in higher POXC of B10CP20 compared to B20CP20 in the second crop cycle. According to Agyarko-Mintah *et al*. (2011) and Fatima *et al*. (2021), the biochar-induced negative priming effects by the higher rate of biochar on the native carbon compared to compost, could be attributed to a combination of mechanisms including the adsorption of CO_2 to biochar surface, reduction in carbon mineralizing enzymes, increase in microbial carbon use efficiency, and immobilization of mineral nitrogen by the biochar. Conversely, Huang *et al*. (2022) found that colza straw biochar application increased soil readily oxidizable carbon by 2.4–46.4% and microbial biomass carbon by 10.4–41.7% compared to control, after one year of application.

Moreover, the increased TOC and carbon stock of the EFB amendments, relative to the unamended and NPK treatments could be related to the enrichment of the soil with materials that were highly rich in carbon (EFB biochar and compost with TOC of 44.8 ± 3.2% and 10.5 ± 0.6%, respectively). However, multiple reasons such as the quantity of stable carbon and C/N ratios of the different EFB amendments might have accounted for the higher TOC content and carbon stock of the B20 treatment relative to the B10 and CP20 treatments. Similarly, Amoakwah *et al*. (2020) reported a significant increase in TOC in this same soil type that was amended with 30 t ha^{-1} of corn cob biochar.

Again, the significant decline in TOC and carbon stock in the B20CP20 after the second cropping cycle showed that the B20CP20 still contained some active fractions of organic carbon in the first crop cycle, which was only mineralised in the second okra crop cycle. Agreeably, Hans *et al*. (2020) reasoned that up to 20 t ha^{-1} of biochar addition to the soil might have decreased SOC losses by increasing the formation of macro-aggregates which protected the internal organic carbon from decomposition. This implies that time plays a key role in the decomposition and mineralization of organic carbon in the soil because of an extension of the disturbance time during plant growth as irrigation, soil cultivation, and mechanical weeding led to more SOC losses. Surprisingly, at the end of the second crop cycle, the highest total organic carbon stock and total organic carbon content were recorded in the B20 instead of the B20CP20.

This may be due to the higher aromatic character of the EFB biochar as reflected in low H/C and H/C_{org} atomic ratios of 0.28 as indices for aromaticity and stability, respectively, which makes biochar highly stable and resistant to oxidation and microbial degradation. However, the addition of EFB compost to the biochar significantly increased its mineralization rate in the long run due to the high labile carbon and nitrogen content of the compost. Consistent with our studies, Kuzyakov *et al*. (2009) observed that slurry addition (as a source of labile C) to soil significantly increased biochar mineralization by a factor of 0.5 to 2 in the short term.

Incidentally, the highest lability index was obtained in the unamended treatment (B0) in the first cropping cycle. Gross *et al*. (2021) and Verheijen *et al*. (2010) demonstrated that disturbance to the soil that breaks up soil aggregates and exposes previously protected soil organic matter to microbial decomposition and mineralisation has a strong priming effect on SOC. Therefore, the soil cultivation, mixing, irrigation, and plant growth during the two okra cropping cycles might have accelerated the oxidation and decomposition of the native SOC and increased the lability of the native soil carbon of the B0 and NPK treatments relative to the reference soil and the biochar treatments.

However, B10 and NPK recorded significantly higher lability indices in the second cropping cycle compared to the B20. Contrary to our findings, Amoakwah *et al*. (2021) reported that the significant increase in the lability index of active carbon observed in 30 t ha^{-1} of corn cob biochar application did not differ significantly from that observed in 15 t ha^{-1}. Meanwhile, studies by Fan *et al*. (2020) gave contradictory findings that the positive priming effect of biochar is normally in response to fresh carbon inputs from biochar addition, while others suggest that the interaction of biochar with native soil reduces SOC decomposition at the soil-plant interphase (Vaccari *et al*., 2011).

The carbon management index (CMI) has been proposed as an indicator of soil carbon rehabilitation; greater values indicate that soil carbon is being rehabilitated and smaller values suggest the system is degrading (Blair *et al*., 1995). At the end of the second cropping cycle, B10CP20 increased the soil CMI compared to the soils of 20-year old orchard and the unamended soil. The plausible explanation may be that the highly stable carbon content of EFB biochar was complemented by the less stable carbon content of compost to increase the contents of labile organic fractions, enhance soil microbial activity and co-metabolism that stimulated the turnover rate of organic carbon in the short run of its application. The biochar application might have enhanced carbon stabilization from the compost to the soil through the mechanism of adsorption and complexion of organic carbon with mineral complexes and aggregates and the regulation of activities of soil microbes involved in organic carbon decomposition and synthesis (Fatima *et al*., 2021; Fischer & Glaser, 2012). Contrary to the current findings, Amoakwah *et al*. (2021) reported a CMI increase in this same soil when only corn cob biochar was applied at 30 t ha^{-1}.

Also, Qiu *et al*. (2023) reported that applications of high amounts of peanut shell biochar improved the CMI of sandy loam over time. Again, Sodhi *et al*. (2009) found that the application of rice stock compost at 16 t ha^{-1} resulted in a greater rate of soil carbon rehabilitation (46.0 to 48.2%) as compared with the lower rate of compost and inorganic fertiliser applications. The results from the current study suggest that EFB biochar has the potential to increase and stabilize carbon in tropical sandy soils and that its application at a lower rate (10 t ha^{-1}) together with EFB compost can provide a greater rate of soil carbon rehabilitation compared to a higher biochar rate with or without compost application.

4.4.2 Soil physical fertility response to amendment application

The effect of EFB amendment applications on soil water at field capacity (θ) (−300 hPa), permanent wilting point (PWP, −15000 hPa), plant available water (PAW), air-filled porosity (ε_a), air permeability (k_a) and relative gas diffusivity (Dp/Do) at−300 hPa, together with, bulk density (ρ_b), total porosity (Ø) and specific surface area (SSA) were examined, after two cropping cycles of EFB amendments application. The result showed that only the combined application of biochar and compost at a higher rate (B20CP20) significantly ($P < 0.01$) reduced the soil bulk density from 1.48 g cm^{-3} in the unamended soil to 1.18 g cm^{-3} (Table 18).

Table 18: Effect of soil amendments on soil physical properties

Treatment	**Bulk density (g cm^{-3})**	**Total porosity (cm^3 cm^{-3})**	**Specific Surface area (m^2 g^{-1})**
B0	1.48 ± 0.04b	0.442 ± 0.015b	33.24 ± 1.55
NPK	1.40 ± 0.03b	0.473 ± 0.013b	35.54 ± 1.26
B10	1.39 ± 0.04b	0.477 ± 0.015b	35.81 ± 1.79
B20	1.35 ± 0.03b	0.491± 0.012b	37.00 ± 1.60
CP20	1.42 ± 0.01b	0.462 ± 0.005b	35.88 ± 0.32
B10CP20	1.35 ± 0.03b	0.492 ± 0.010b	36.91 ± 1.99
B20CP20	1.18 ± 0.03a	0.556 ± 0.013a	39.05 ± 0.96
P value	***	***	ns

Each value is the mean (± standard error) from nine replicates. Different letters in a column indicate significant differences among treatments and same letters mean there are no differences ($P < 0.05$). "ns" indicates no difference. *** denote statistical significance at $P < 0.001$, with the Tukey HSD test. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

In contrast, the other EFB amendments (B10, B20, CP20, and B10CP20) resulted in marginal lower soil bulk density of 1.39, 1.35, 1.42, and 1.35 g cm^{-3}, respectively, compared to the unamended soil.

The total porosity was also significantly increased by B20CP20, compared to the other treatments including the B0 (Table 18). The CP20 treatment had a relatively lower total porosity among the EFB treatments. The specific surface area of the soil was not significantly ($P > 0.01$) significantly increased by the EFB amendments though B20CP20 still recorded the highest of 39.05 m^2 g^{-1}.

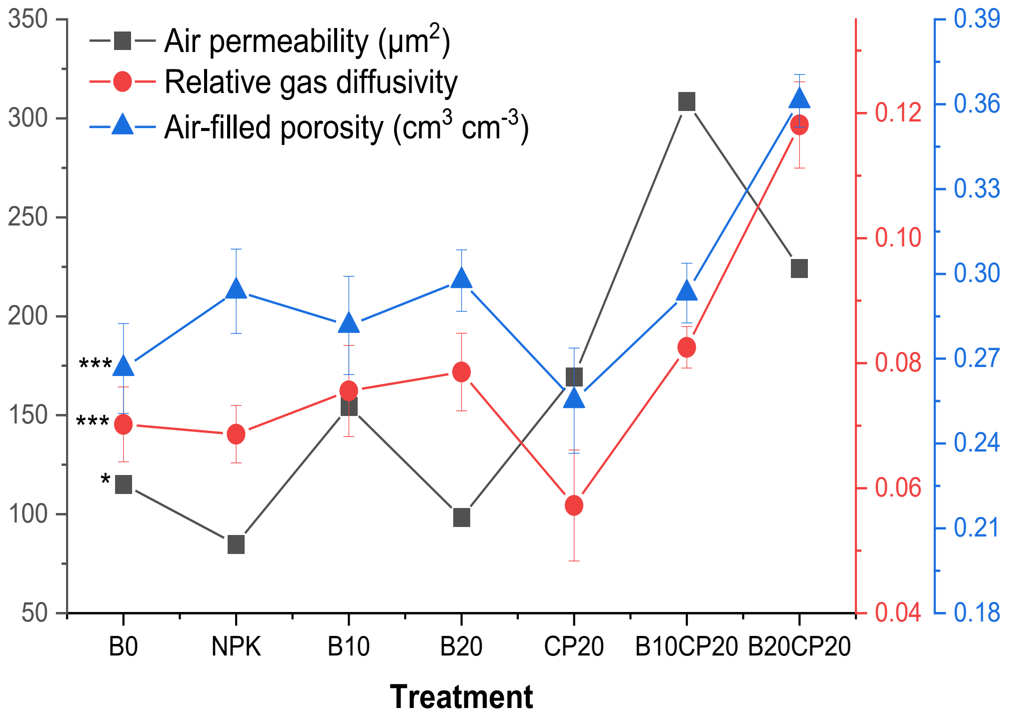

Figure 37: Effect of amendments on soil air-filled porosity, air permeability and relative gas diffusivity at−300 hPa. Error bars show the standard error of means from nine replicates. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Moreover, the B20CP20 significantly ($P < 0.05$) increased air-filled porosity and relative gas diffusivity compared to other EFB treatments as well as the

NPK and B0 treatments (Figure 37). Relative to the unamended treatment, CP20 marginally lowered air-filled porosity by 4.3% and relative gas diffusion of the soil by 18.5%. Contrarily, B10CP20 significantly ($P < 0.01$) increased by air permeability by 189% compared to the NPK treatments. Meanwhile, the permeability of air in B0 was not significantly different from all the EFB treatments as well as the NPK treatment.

The effect of EFB amendments application on soil water retention at field capacity, permanent wilting point and plant available water are presented in Figure 38. The soil EFB compost application at 20 t ha^{-1} CP20 significantly ($P < 0.01$) increased soil water retention at field capacity by 13.9% and 21% relative to the NPK and B0 treatments, respectively (Figure 38a). Notably, the B10CP20 marginally increased soil water retention compared to B20CP20. Meanwhile, plant available water content and the water retention at permanent wilting point were not significantly ($P > 0.05$) affected by EFB amendments compared to the B0 treatment. Also, the CP20 treatment recorded the highest plant available water content of 0.126 cm^3 cm^{-3} among the EFB amendments (Figure 38b). However, lowest water retention at permanent wilting point was recorded in B20CP20 treatment, with the highest retention recording in B10 (Figure 38c).

The reduction in the soil's bulk density after amending with combined biochar and compost (at 20 t ha^{-1} each) can be ascribed to the physical characteristics of these organic amendments, such as lower bulk density, higher porosity, and variations in particle size, shape, and surface area (Zhang *et al*., 2020). Another plausible explanation is that the organic matter supplied by biochar and compost might have improved soil aggregation by providing a food source and habitat for soil organisms, which contribute to the formation of micro, meso, and macro-pores (Carter *et al*., 2004). Converse to our findings, Li *et al*. (2018) observed that the sole application of maize straw biochar reduced soil bulk density and improved soil porosity in a semi-arid region.

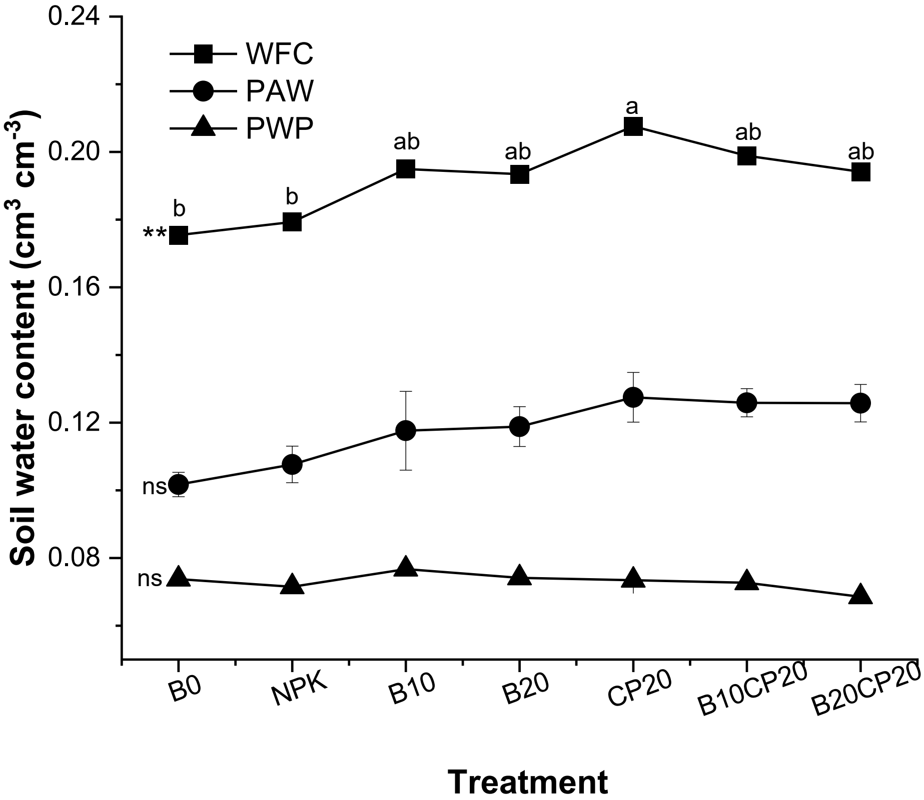

Figure 38: Soil water at field capacity, available water and water at permanent wilting point, as affected by the amendments. Different letters mean the treatments are significantly different and same letters mean there are no differences ($P < 0.05$). Error bars show the standard error of means from nine replicates. WFC, Water at field capacity; PAW, plant available water; PWP, permanent wilting point B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Again, Laghari *et al*. (2016) reported that application of rice husk biochar alone at 10 t ha^{-1} decreased soil bulk density and increased available water of a sandy clay loam soil. Agreeable to the findings from the current study, Reynolds *et al*. (2015) observed a negligible effect of low rate (< 75 t ha^{-1}) of food waste compost on soil bulk density after a single application. Amoakwah (2017) observed no significant effects on bulk density and porosity in this same soil type (Haplic Acrisol) when corn cob biochar was applied at 15 and 30 t ha^{-1} which is consistent with our current result. However, Abdulrazzaq *et al*. (2015) reported a significant decrease in bulk density and an increase in total porosity after just six months of EFB biochar application (at 15 and 30 t ha^{-1}) to loamy soil.

The lower impact of EFB compost on bulk density compared to biochar may be ascribed to the differences in the chemical and physical characteristics such as particle sizes, bulk density, porosity, aromaticity as well as carbon content and stability in the soil. For example, the EFB biochar was lighter (with bulk density = 0.19 g cm^{-3}) with powder-like particle sizes (> 50% biochar fraction < 2 mm) as well as higher carbon content (44.8%) and highly fused aromatic rings and stability (H/C atomic ratio of 0.28) compared to the EFB compost. Consistent with this current study, Şeker and Manirakiza (2020) reported that biochar amendment stood out from compost (both prepared from Oleaster tree pruning residue) in the context of lowering bulk density in a sandy clay loam soil, and a similar result was also reported by Agegnehu *et al*. (2015).

The increased air exchange in the B20CP20 observed in the current can partly be attributed to the significant ameliorating effect of organic carbon content from both compost and biochar, which stimulate the formation of soil pores (Amlinger *et al*., 2007). The higher rate of single biochar application (20 t ha^{-1}) marginally decreased air permeability and this finding is consistent with Cai *et al*. (2020), who observed that in poorly graded sand, biochar amendment reduced the soil gas permeability coefficient, which decreased continuously with an increase in biochar rate. Also, Kumari *et al*. (2014) reported a decrease in the air permeability of a sandy loam after seven months of application of 20 t ha^{-1} birch wood biochar and attributed it to localized high water contents in biochar-rich soil. Contrary to Obour *et al*. (2019) who reported an increased k_a at −300 hPa after three years of rice straw biochar addition at 30 t ha^{-1}.

Also, Sun *et al*. (2013) reported that 20 t ha^{-1} birch wood biochar application increased Dp/Do by 161% and k_a up to 223%. Wang *et al*. (2023) ascribed the observation to the smaller biochar particle size, which has a better filling effect within the soil's internal structure, increases soil density and thus decreasing gas permeability. However, Garg *et al*. (2019) reasoned that additional water that occupies the intra-pores of biochar could have caused the reduction in air conductive channels for gas migration, causing a decrease in air permeability in biochar-amended soils, especially with higher biochar content. That said, B20 should have increased bulk density or decreased air-filled porosity and increased water retention but the opposite effects were observed in the current study.

The amount of plant-available water relative to air-filled porosity at field capacity is often used to assess soil physical fertility (Kuo *et al*., 2004). Soil water retention is a key soil hydraulic property that governs soil functions and it is critical for crop growth, and strongly influences soil productivity as well as biological soil health (Obour *et al*., 2019). This soil property greatly depends on soil texture, structure, and organic matter (Panagea *et al*., 2021). The incidental increase in water content at field capacity and marginal increase in plant available water in single compost may be attributed to the elevated organic carbon content of the soil. Agreeably, Şeker and Manirakiza (2020) found increased soil water content in compost-amended sandy clay loam soil. In addition, Aşkın and Aygünand found that application of 10 t ha^{-1} hazel husk compost increased the water at field capacity and permanent wilting point compared to control.

Fischer and Glaser (2012) posited that compost might have supplied a well humified organic matter to promote the formation and stabilization of micro-aggregates responsible for storage of water in the soil. Again, Carter *et al*. (2004) reasoned that the organic matter applied by compost improves water conductivity of soils by providing a food source for soil organisms as well as the production of associated by-products that acts as glue between soil particles (Six *et al*., 2004), which facilitates aggregate formation and micro pore formation for water storage. In addition, Hudson (1994) explained that organic matter (OM) is able to take up 3 to 20 times more water compared to its own weight. The increase in soil's total organic carbon (TOC) content from compost application can result in the rise of soil water content.

Also, the increased specific surface area of the combined biochar and compost amended soils might have increased the potential of the soil to retain more water. On this basis, the performance of higher rates of biochar (B20 or B20CP20) on water retention should have surpassed that of lower rates (B10 or B10CP20) treatments. However, the opposite effect was observed in the current experiment and this might be due to the hydrophobic nature of biochar that might have lowered water infiltration into the biochar-amended soil. This explanation can be supported by the lower atomic ratio O/C (< 0.2

representing high hydrophobicity) which is an index of hydrophobicity of the biochar (0.16) compared to the compost (1.41).

According to Li *et al*. (2021), the hydrophobic surface of the biochar is known to have a low surface energy and therefore allows a fast water-drop penetration throughout macropores. In a study by Amoakwah et al. (2017) corn cob biochar applied at 20 t ha^{-1} to a sandy loam significantly increased the soil water retention at field capacity compared to application at 10 t ha^{-1} and the unamended soil. The results suggest that apart from the nature and contents of carbon, other intrinsic properties of biochar or compost can also affect the physical properties and most especially the flow and retention of water in tropical sandy soil.

4.4.3 Soil carbon storage and management index regulate soil physical fertility

Pearson correlation of soil carbon dynamics and soil physical fertility properties revealed that POXC had a significant positive correlation with plant available water content (r = 0.85; $P < 0.05$) and air permeability (r = 0.79; $P < 0.05$) (Figure 39). On the other hand, TOC correlated positively and significantly with total porosity (r = 0.81; $P < 0.05$) and SSA (r = 0.86; $P < 0.05$) but negatively with soil bulk density (r = -0.81 $P < 0.05$). Similar to the POXC, CMI showed a significant positive relationship with air permeability (r = 0.85; $P < 0.05$) and plant available water content (r = 0.78; $P < 0.05$), but TOC stock only correlated significantly with SSA (r = 0.77; $P < 0.05$). Conversely, the lability index did not have any significant correlation (r > 0.3; $P > 0.05$) with any of the soil physical fertility parameters (Figure 39).

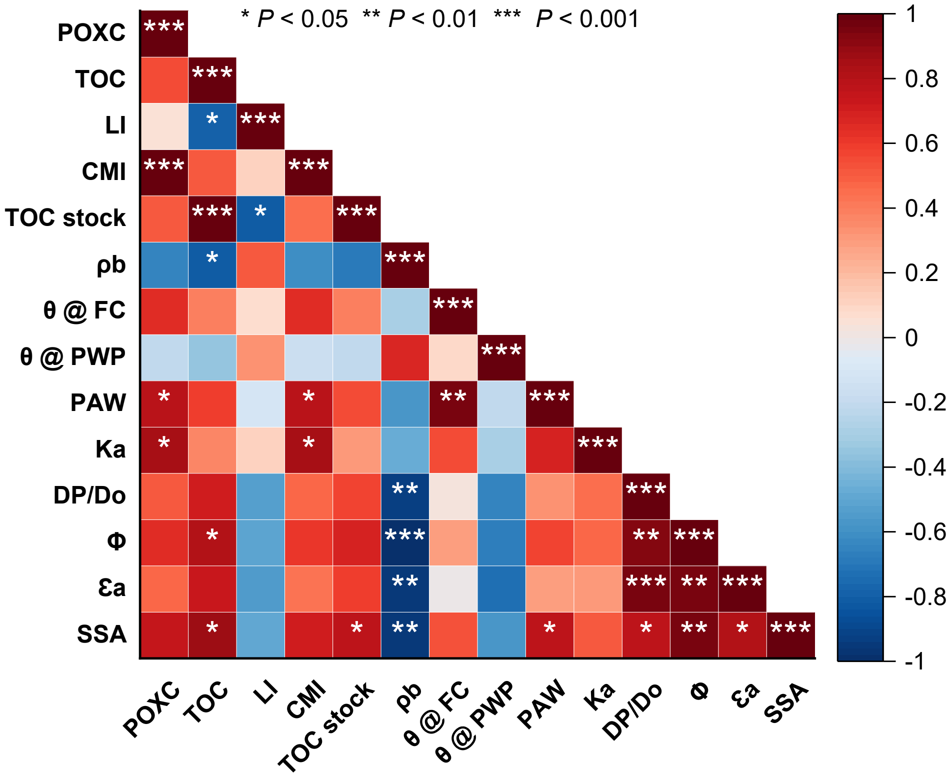

Figure 39: Correlation of soil carbon components with soil physical properties. Statistical significance is denoted as *** (P < 0.001), ** (P < 0.01) and * (P < 0.05). POXC, permanganate oxidizable carbon; TOC, total organic carbon; LI, lability index; CMI, carbon management index; TOC stock, total carbon stock; ρb, bulk density; Ø, total porosity; θ @ FC, volumetric water at field capacity; θ @ PWP, volumetric water at permanent wilting point; PAW, plant available water; Ka, air permeability; DP/DO relative gas diffusivity; Ɛa, air-filled porosity; SSA, specific surface area.

Again, the results of the principal component analysis showed that, the first two PCs explained 81.5% of the variations in the data sets (Figure 40). The first PC accounted for 60.89% of the variation in the data set. The PCI was positively associated TOC, TOC stock, SSA, Ø, ε_a and Dp/Do, negatively associated with ρ_b, LI, and θ @ PWP and was regulated by B20 and B20CP20. The PC2 explained 20.61% of the variance in the data and was positively associated with soil POXC, CMI, k_a and θ @ FC and PAW, which were regulated by B10CP20.

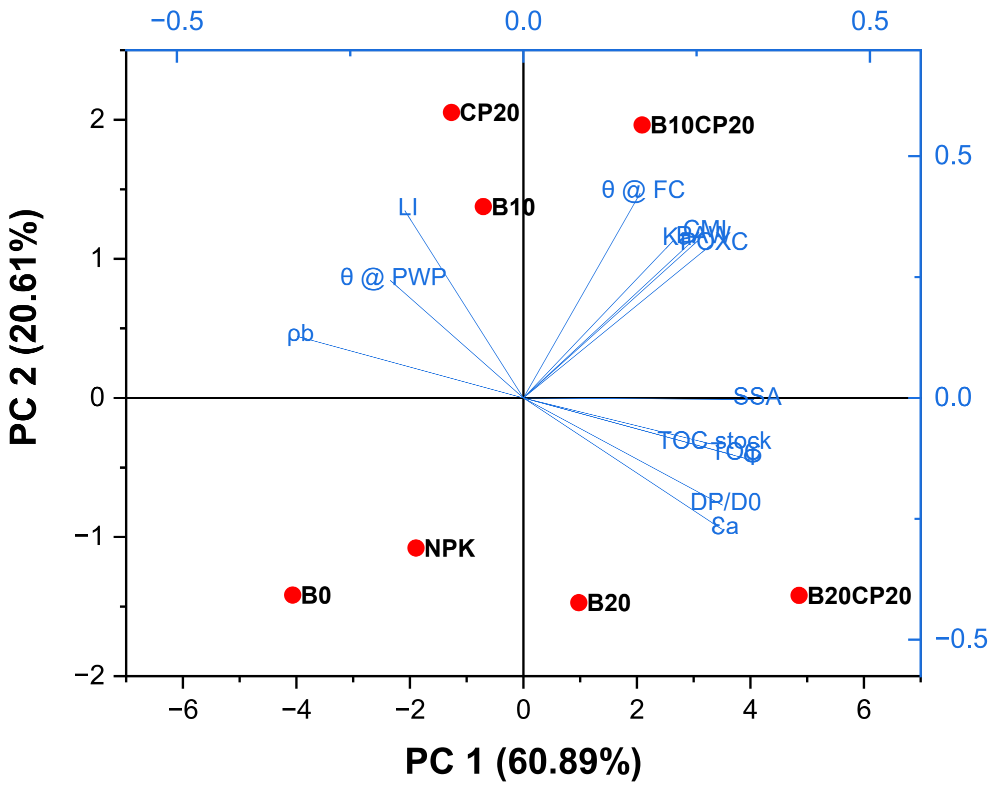

Figure 40: A biplot of soil physical properties and carbon rehabilitation components; the loadings are POXC, permanganate oxidizable carbon; TOC, total organic carbon; LI, lability index; CMI, carbon management index; TOC stock, total carbon stock; ρ_b, bulk density; θ @ FC, volumetric water at field capacity; θ @ PWP, water at permanent wilting point; PAW, plant available water; k_a, air permeability; DP/DO relative gas diffusivity; $\hat{e}_a$, air-filled porosity. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

The result from correlation and PCA analyses between soil carbon quality and dynamics and soil physical properties showed that POXC was the major regulator of soil water storage and availability for plant use while total organic carbon drove overall soil porosity for gas flow in the soil. These relationships can be ascribed to the modification of soil structure by organic matter quantities and qualities which together affected soil physical parameters such as bulk density, total porosity, air-filled porosity, water retention and air permeability and also indirectly affected relative oxygen diffusivity in the soil. Similar to the findings of Colombi *et al*. (2019), soil organic carbon content positively related with gas diffusion and water-holding capacity but not air-filled porosity. Also, Chaudhari *et al*. (2013) found a strong negative relationship between SOM and bulk density. In the current study, POXC and CMI correlated with air permeability and water retention but not air-filled porosity and relative oxygen diffusivity because the later parameters are likely to be associated with soil macro-porosity which is impacted by other fractions of soil carbon other than the POXC. Agreeably, Neira *et al*. (2015) explained that soil labile organic carbon serves as a substrate for microbial activity and results in the production of microbial bonding materials for the formation of micro-aggregates and pores that are responsible for air and water transport in the soil.

Moreover, an increased lability index was negatively associated with decreased air permeability, relative gas diffusivity and water retention. The plausible explanation will be that due to the transient nature of the labile organic carbon, it is rapidly decomposed by microorganisms and its effectiveness on soil physical parameters related to soil aggregation lasts only for a period of a few weeks or months, after which their effect diminishes as indicated by Krull *et al*. (2004).

The significant relationship of POXC and TOC with SSA implies that the effect of EFB compost and biochar on SSA was strongly dependent on their carbon addition to soil. In agreement with the findings of Hans *et al*. (2020), they observed that increased SSA of soil from biochar addition decreased soil carbon losses by protecting the organic carbon from microbial decomposition. The results from the current study confirm that different types of organic carbon perform various functions at different times that affect the flow and storage of water and distribution of gases in tropical sandy soil as speculated by Krull *et al*. (2004). Again, the carbon management index (CMI) proves to be a sensitive measure of the rate of soil carbon seguestration from land management systems in tropical soils because it was a significant driver of soil water retention which is one of the major challenges of crop production in highly weathered tropical sandy soils.

4.4.4 Summary

The current objective examined the sole and combined effects of EFB biochar and compost application on soil carbon quantity, quality, stock, and management index and how they drive soil physical fertility. The results showed that sole biochar application at 20 t ha^{-1} increased the TOC and TOC stock of carbon of the soil because of its high carbon storage value, aromatized carbon, and C/N ratio that made it resistant to oxidation and microbial degradation but its effect on soil physical properties was negligible. Soil carbon management index was significantly increased by the combined application of 10 t ha^{-1} biochar and 20 t ha^{-1} of compost with positive ripple effects on air permeability and soil water retention. Due to the hydrophobic nature of the EFB biochar, soil water retention in the soil amended with a higher rate (20 t ha^{-1}) of biochar was decreased. Moreover, the co-application of 20 t ha^{-1} each of biochar and compost significantly improved soil physical fertility by lowering the bulk density and increasing specific surface area, air-filled porosity, and relative gas diffusivity of the soil. The findings from this study suggest that the co-application of EFB biochar and compost can rehabilitate soil carbon in tropical sandy soils with positive ripple effects on soil physical fertility in the short run.

4.5 The Potential and Drivers of Greenhouse Gas Emissions from Empty Oil Palm Fruit Bunch Biochar and Compost-Amended Soil

Agro-ecological landscapes across Ghana are reported to be severely degraded by both climate drivers and non-climate drivers (Global Mechanism of the UNCCD, 2018). Multiple problems including the increasing emission of greenhouse gases such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) from the soil have become a growing concern in this region (Global Mechanism of the UNCCD, 2018). It will be prudent to assess the extent to which the utilization of EFB as either biochar or compost will impact climate change. Therefore, the fifth and last objective of the current study sought to examine the effects of single and co- application of EFB biochar and compost on soil gas permeability, water retention, pore characteristics, and their regulatory impact on soil CO_2, N_2O and CH_4 emissions. It was hypothesized that the changes in soil physical, chemical and microbial properties brought about by EFB amendment application will drive greenhouse gas emissions from the soil.

4.5.1 Air flow and water retention characteristics of the soil at -100 hPa as affected by the soil amendments

The effect of sole and combined applications of EFB biochar and compost on air flow and water retention in the soil at−100 hPa is presented in Figure 41. The combined application of EFB biochar and compost at 20 t ha^{-1} each (B20CP20) significantly ($P < 0.001$) increased air-filled porosity (ε_a) and relative gas diffusivity (Dp/Do), while marginally increasing air permeability

(k_a) compared to the other EFB treatments, as well as the unamended and NPK treatments (Figures 41). In contrast, the compost application alone (CP20) marginally decreased ε_a and Dp/Do by 4.4% and 15.5%, respectively, compared to the B0 treatment. Additionally, the CP20 and B10CP20 treatments significantly ($P < 0.05$) increased soil water content at −100 hPa by 17% compared to the B0 treatment but did not differ significantly ($P > 0.05$) from the other EFB treatments or the NPK treatment (Figure 41b). The air flow and water retention characteristics of the soil at −100 hPa followed trend as that at −300 hPa and these stems from the differences in the physical and chemical properties of the EFB biochar and compost, as enunciated in section 4.4.2.

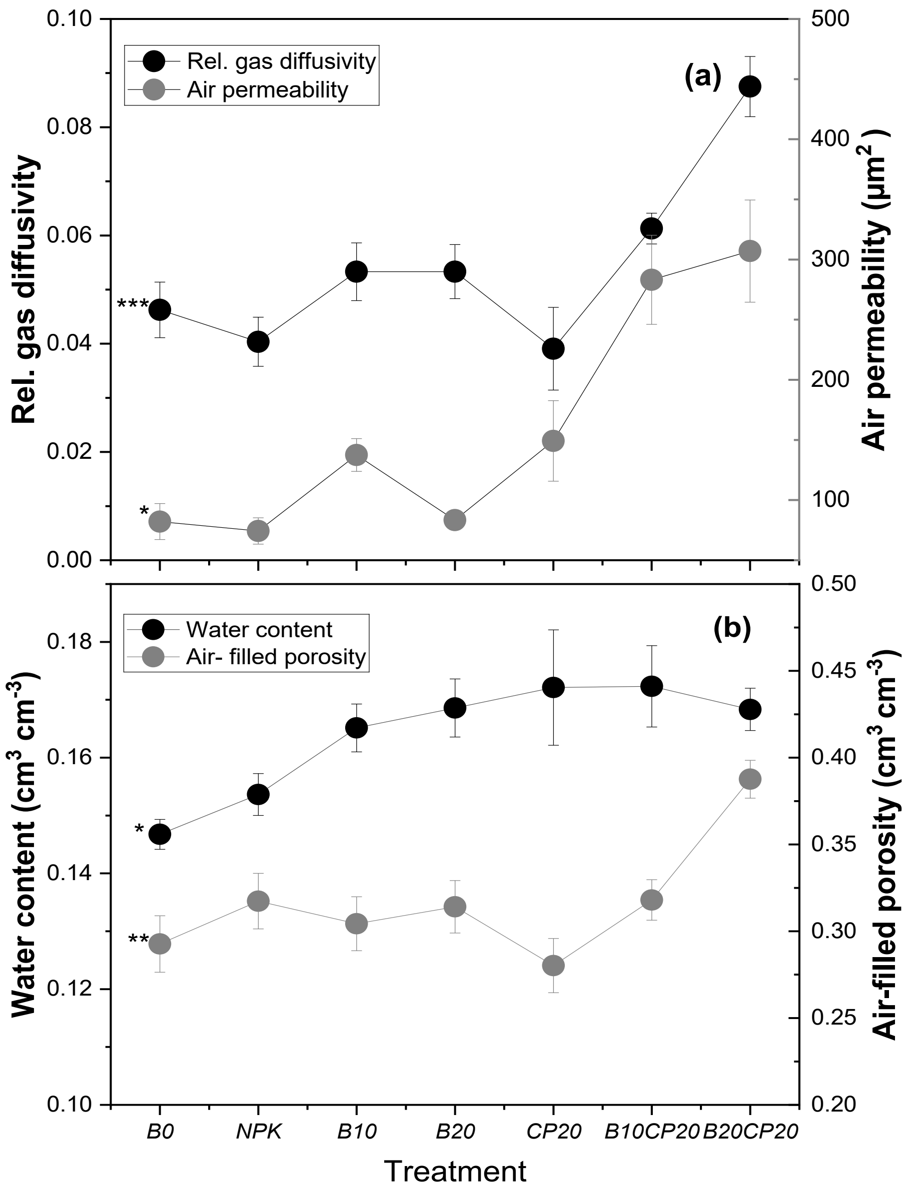

Figure 41: (a) Air flow and (b) water retention and air-filled porosity in the soil at − 100 hPa as affected by the soil amendments. Error bars show the standard error of means from nine replicates. ***, **, and * denote statistical significance at $P < 0.001$, 0.01, and 0.05, respectively, with the Tukey HSD test. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

4.5.2 Soil pore structure characteristic as affected by soil amendments

The effect of sole and combined application of EFB biochar and compost on soil pore tortuosity, organization, effective pore diameter and number of air-filled pores is presented in Figure 42. The CP20 treatment significantly ($P < 0.01$) increased soil pore tortuosity (τ) by an average of 42% compared to the combined applications of biochar and compost (B10CP20 and B20CP20), whereas the pore tortuosity in all EFB-amended soils did not differ significantly ($P > 0.05$) from the B0 treatments (Figure 42a). The B10, B20, B10CP20, and B20CP20 treatments marginally decreased τ relative to the B0 treatment. Soil pore organization (PO) and effective pore diameter (d_B) in the unamended treatment were not significantly different from the biochar and compost treatments (Figure 42a, b). However, CP20, B10CP20, and B20CP20 treatments marginally increased soil PO and dB relative to the B0 treatment. The B20CP20 treatment significantly ($P < 0.01$) increased the number of air-filled pores in a given soil cross-section relative to B0, NPK, and B20 treatments (Figure 42b).

The single application of biochar (20 t ha^{-1}) slightly decreased pore tortuosity, organization, effective pore diameter, and the number of air-filled pores, leading to a subsequent reduction in air permeability compared to the unamended treatment in the current study. Similarly, Keller *et al*. (2019) noted an insignificant increase in air-filled porosity following biochar application, attributing this to a less tortuous, more continuous, and better-connected pore system in biochar-treated soil. Conversely, Obour *et al*. (2019) observed an insignificant increase in pore organization indices and the number of air-filled pores in rice biochar-amended Acrisol compared to untreated soil.

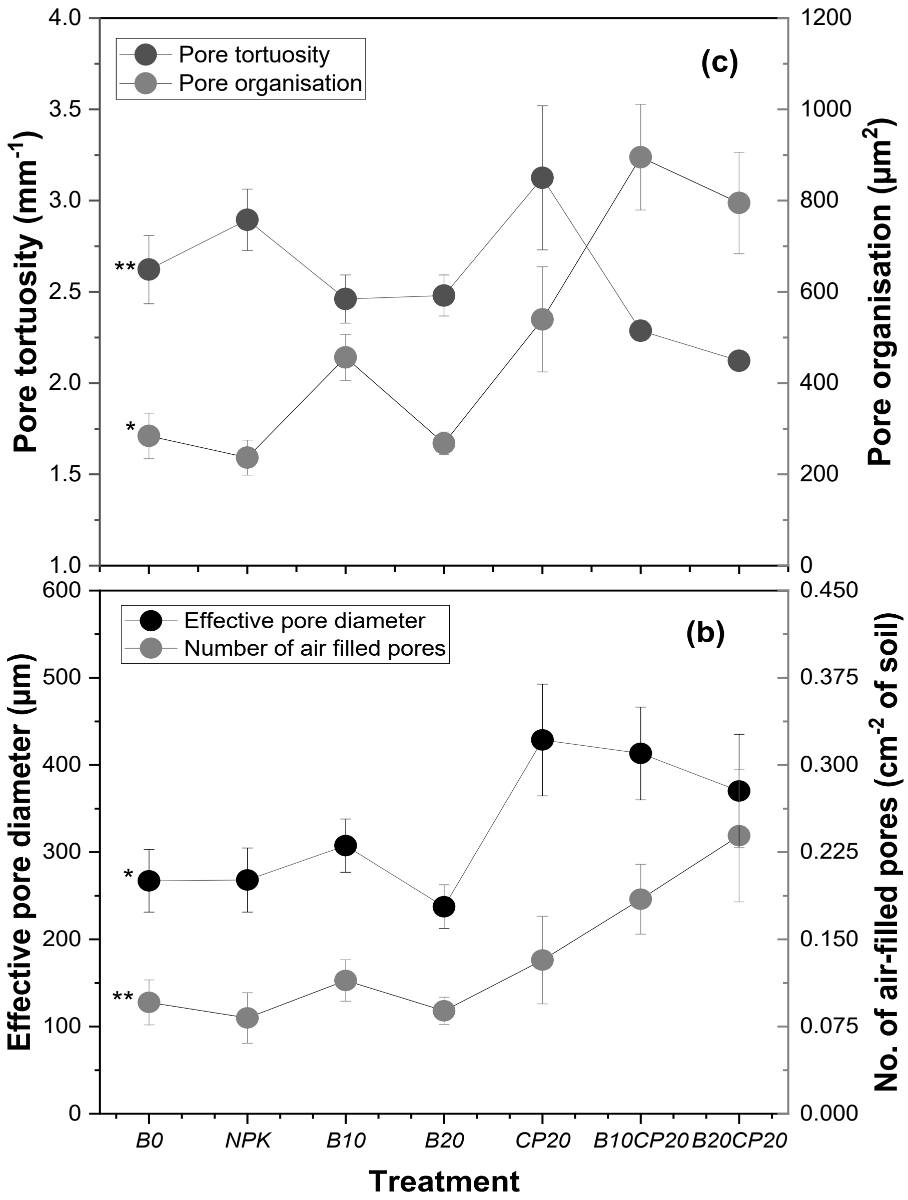

Figure 42: Model-derived (a) pore tortuosity and organization and (b) effective pore diameter and number of air-filled pores as affected by the soil amendments. Error bars show the standard error of means from nine replicates. ***, **, and * denote statistical significance at $P < 0.001$, 0.01, and 0.05, respectively, with the Tukey HSD test. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

McCarthy and Brown (1992) observed that structured soils typically have higher air permeability than poorly structured soils due to the larger inter-aggregate flow channels characteristic of structured soils. Similarly, Arthur *et al*. (2012) linked the increase in relative gas diffusivity to a rise in the volume of effective pore space, as inactive pores decrease more drastically with increasing bulk density. This may be due to the smaller biochar particle size, which has an internal structure-sealing effect on the soil, reducing gas permeability as suggested by Wang *et al*. (2023).

The increase in pore tortuosity, organization, and effective pore diameter in compost-treated soil aligns with observations by Fischer and Glaser (2012), who found that compost incorporation enhances aggregate stability in sandy soils by supplying well-humified organic matter (promoting micro-aggregates) and fresh, low-molecular organic matter (promoting macro aggregates). Similarly, Arthur *et al*. (2013) reported significant improvement in structural complexity in Glossic Phaeozems soil following amendment with ground rape (7.5 t ha^{-1}). Co-application of biochar and compost significantly decreased pore tortuosity and slightly reduced effective pore diameter compared to compost alone. This could be linked with the filling effect of the smaller biochar particle size within the soil's internal structure, which can clog soil pores and thus decreasing the effective pore diameter as suggested by Wang *et al*. (2023). The results suggest that apart from the nature and contents of carbon, other intrinsic properties of biochar or compost can also affect the pore structure, air flow and retention of water in tropical soils.

4.5.3 Effect of soil amendments on greenhouse gas emissions

The effect of sole and combined application of EFB biochar and compost on nitrous oxide (N_2O), carbon dioxide (CO_2) and methane (CH_4) emissions is presented in Figure 43. The fluxes of N_2O, CO_2, and CH_4 were significantly affected by the EFB amendments (Figure 43). Specifically, a single application of compost (CP20) decreased N_2O emission by 87% relative to the unamended treatment (B0) ($P < 0.001$). The co-applications of biochar and compost (B10CP20 and B20CP20) also significantly reduced N_2O emission by 70% and 78% respectively, compared to B0 treatment. The sole biochar application at 20 t ha^{-1} (B20) recorded the highest N_2O emission (4.12 ha^{-1} day^{-1}g), that was marginally higher than 4.018 ha^{-1} day^{-1} recorded for B0 (Figure 43a). The B10 and B20 treatments significantly ($P < 0.001$) increased N_2O fluxes compared to the CP20, B10CP20, and B20CP20.

The CO_2 emission followed a similar trend as N_2O emissions, with the highest CO_2 flux (8.12 g ha^{-1} day^{-1}) recorded in the NPK treatment and the lowest (3. 07 g ha^{-1} day^{-1}) in the CP20 treatment (Figure 43b). The CP20, B10CP20, and B20CP20 treatments significantly ($P < 0.001$) decreased CO_2 emission by~53% and~61% relative to B0 and NPK treatments. The compost application alone (CP20) significantly decreased CO_2 emission by 51% compared to the sole biochar application at 20 t ha^{-1} (B20).

Contrarily, the effects of the EFB amendments on CH_4 emission were inconsistent (Figure 43c). In general, biochar and compost application marginally increased CH_4 emissions compared to the unamended treatment (B0). For instance, the B10CP20 treatment significantly ($P < 0.05$) increased CH_4 emission by 684% relative to the B0 treatment but did not significantly differ from the NPK or the other EFB treatments.

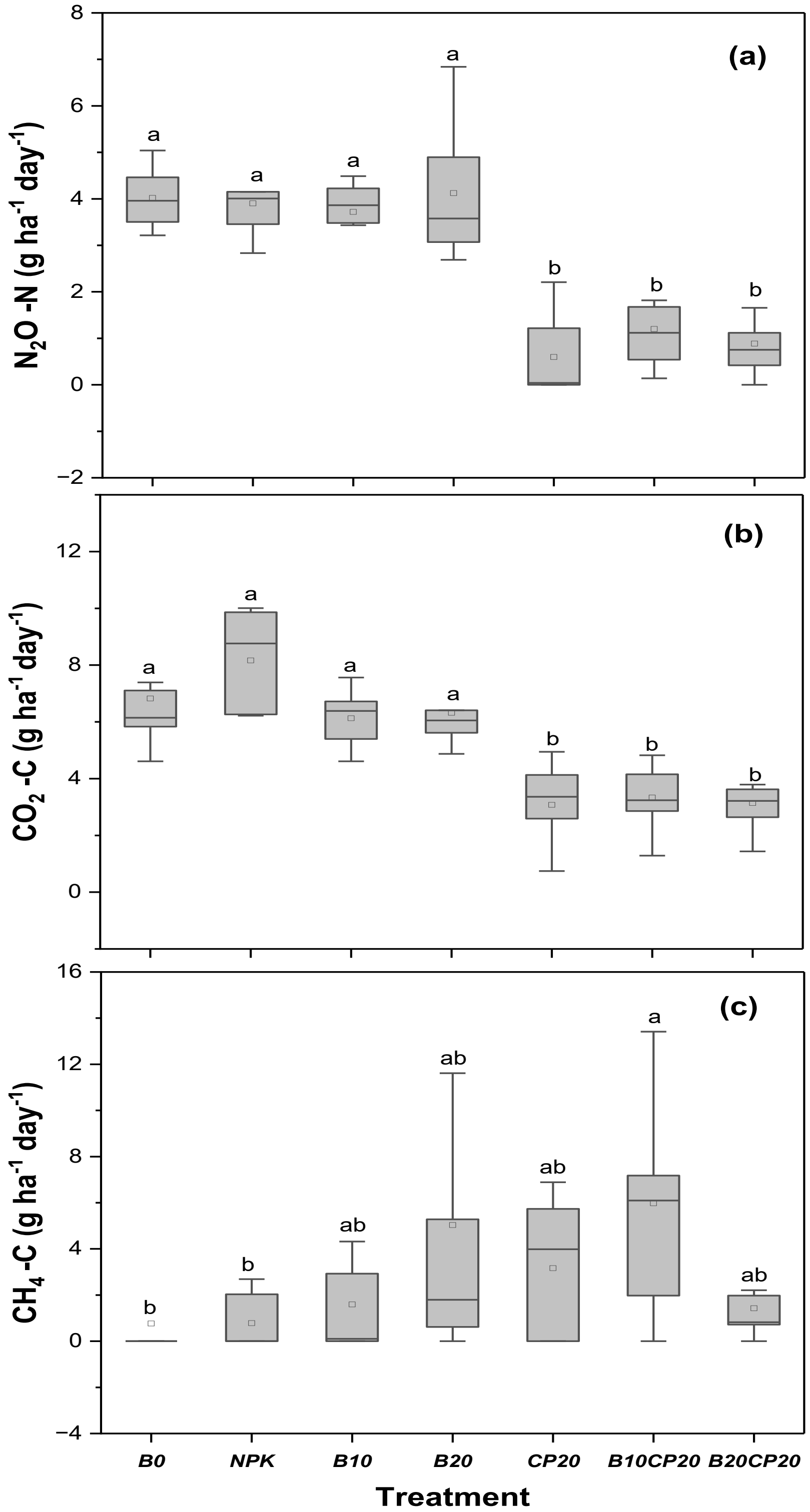

Figure 43: Fluxes of (a) nitrous oxide (N_2O), (b) carbon dioxide (CO_2), and (c) methane (CH_4) in the amended soil. Error bars show the standard error of means from nine replicates. Different letters mean differences among treatments (Tukey HSD test at $P < 0.05$). B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

The present study showed 12 months after the application of amendments, both sole compost and the combined application of biochar and compost reduced N_2O and CO_2 emissions in the soil. The result of the current finding contradicts earlier findings by Farooqi *et al*. (2024), who reported the highest N_2O and CO_2 emissions from compost and vermicompost applications, with increases of 57% and 46%, respectively, compared to the control. Again, Chau *et al*. (2024) observed that an incubation experiment with kitchen compost resulted in CO_2 emissions averaging 3.8 times higher than those from control soil. Additionally, a meta-analysis by Fu *et al*. (2023) revealed that organic amendments, including compost, increased CO_2 and N_2O emissions by 133.7% and 14.0%, respectively. Verhoeven *et al*. (2017) ascribed the increase in CO_2 and nitrous oxide (N_2O) emissions after compost application to the initial boosts in microbial activity due to the introduction of labile organic carbon to the soil.

Interestingly, in the current study, compost recorded the lowest emissions of both N_2O and CO_2 among all treatments, including the unamended control. Factors such as local climate, soil type, carbon quality and quantity, and nutrient availability are known to influence greenhouse gas emissions from agricultural soils, as noted by Basheer *et al*. (2024). In tropical conditions like those in this study, high radiation and soil temperatures may have caused the rapid decomposition of compost carbon, leaving behind only mineral nutrients or stabilizing the recalcitrant carbon into humus, as suggested by Fischer and Glaser (2012). The low content of degradable carbon in the compost-treated soil, as reflected in the insignificant POXC and TOC content after 12 months of its application (Table 2), likely accounted for the lower CO_2 and N_2O emissions compared to the unamended soil.

The slight increase in CH_4 emissions from the compost treatment, compared to B0 agrees with findings of Chau *et al*. (2024) who reported a significant increase in CH_4 emissions from soil treated with 1% kitchen compost. Fu *et al*. (2023) also found increased CH_4 emissions from soil amended with organic materials, including compost. This could be attributed to the provision of electron donors to the microbial community and the fermentation of compost, which increased CH_4 emissions and reduced soil redox potential under low-oxygen and near-saturated conditions (Fu *et al*., 2023). This was evidenced by the higher water content, low relative gas diffusivity, and reduced air-filled porosity at −100 hPa observed in the compost treatment.

The effect of sole biochar amendments, regardless of the application rate, on soil CO_2, N_2O, and CH_4 emissions were insignificant compared to the unamended and NPK treatments, although the CO_2 and N_2O emissions were significantly higher than in compost and co-compost and biochar treatments. This may stem from the low but increased mineralization of SOC and EFB biochar when nutrient sources were introduced before gas measurement (Wang *et al*., 2015). Spokas (2012) suggested that biochar's impact on CO_2 and N_2O emissions may change over time as biochar ages, and this supported the observation that biochar's effect on these emissions are predominantly short-term (< 12 months). According to Cross and Sohi (2011), short-term greenhouse gas emissions (< 1 year) are influenced by both inorganic C and labile organic C in biochar. Conversely, Liu *et al*. (2016) explained that for long-term impacts on soil CO_2 and N_2O emissions, biochar alters the microbial community through changes in habitat structure, chemical and physical protection of native soil organic matter, and modification of micro-scale soil redox status due to biochar's electrochemical properties.

When the single compost and biochar treatments are compared, the higher CO_2 and N_2O emissions from biochar can also be attributed to the positive priming effect of EFB biochar on existing soil carbon (Bass *et al*., 2016) or the continued mineralization of carbon within the biochar, especially after 12 months. This is evidenced by the higher TOC in biochar treated soil compared to compost or their co-application. Similarly, Walkiewicz *et al*. (2023) observed that sawdust biochar applied at 20 t ha^{-1} in a laboratory experiment resulted in insignificant CO_2, N_2O, and CH_4 fluxes compared to the control. Conversely, meta-analyses by Fu *et al*. (2023) revealed that biochar application significantly reduced soil N_2O and CH_4 emissions, but it had no effect on CO_2 emissions. In both laboratory-incubated soils and field-scale cropping systems, Fidel *et al.* (2019) found no significant effect on cumulative soil CO_2 emissions but observed reduced N_2O emissions in a no-till continuous corn cropping system after the application of soft and hardwood mixture biochar. Bass *et al*. (2016) also reported a reduction in N_2O emissions under biochar application, attributing it to increased water content in microsites, which likely decreased NO_3 availability and promoted complete denitrification to N_2.

The current study revealed a significant reduction in CO_2 and N_2O emissions, along with an increase in CH_4 emissions, from co-applied biochar and compost treatments compared to biochar, chemical fertiliser, and unamended treatments. The co-biochar and compost treatment contained higher levels of both easily degradable and recalcitrant carbon, as well as a considerable amount of nitrogen, which facilitated efficient microbial activity. According to Gao *et al*. (2023) and Zhang *et al*. (2020), this suggests high microbial carbon use efficiency, potentially leading to increased carbon storage and persistence through microbial necromass contribution, and resulting in lower emissions from the co-biochar and compost treatment. The significant increase in CH_4 emissions from the B10CP20 treatment can be attributed to the higher volumetric water content and lower air-filled porosity observed in this treatment. It is plausible that this led to water saturation and reduced oxygen availability, increasing the number of anaerobic soil microsites (Gao *et al*., 2023), which subsequently resulted in higher CH_4 fluxes. Literature on greenhouse gas emissions presents divergent findings. For example, Gao *et al*. (2023) reported that biochar co-compost significantly decreased CO_2 emissions but had an insignificant impact on N_2O and CH_4 emissions over one growing season of winter wheat. Additionally, a meta-analysis by Fu *et al*. (2023) found that the co-application of biochar and organic amendments increased soil greenhouse gas emissions, with an antagonistic effect on CO_2 but an additive effect on N_2O and CH_4 emissions. Bass *et al*. (2016) observed that soil CO_2 efflux increased with the addition of compost and co-applied biochar and compost treatments, but they reported no effect on N_2O emissions. All in all, the current research suggests that the co-application of EFB biochar and compost has a potential to reduce greenhouse gas emissions from tropical soils through efficient carbon sequestration.

However, further studies, including gas measurements after fresh application and long-term evaluations, are warranted.

4.5.4 Drivers of greenhouse gas emission from the EFB amended soil

The N_2O and CO_2 emissions under EFB amendment were significantly and negatively correlated with air permeability (r =−0.80, $P < 0.05$), effective pore diameter (dB) (r =−0.90, $P < 0.001$), and the number of air-filled pores in a soil across-section (nB) (r =−0.80, $P < 0.05$) (Figure 44).

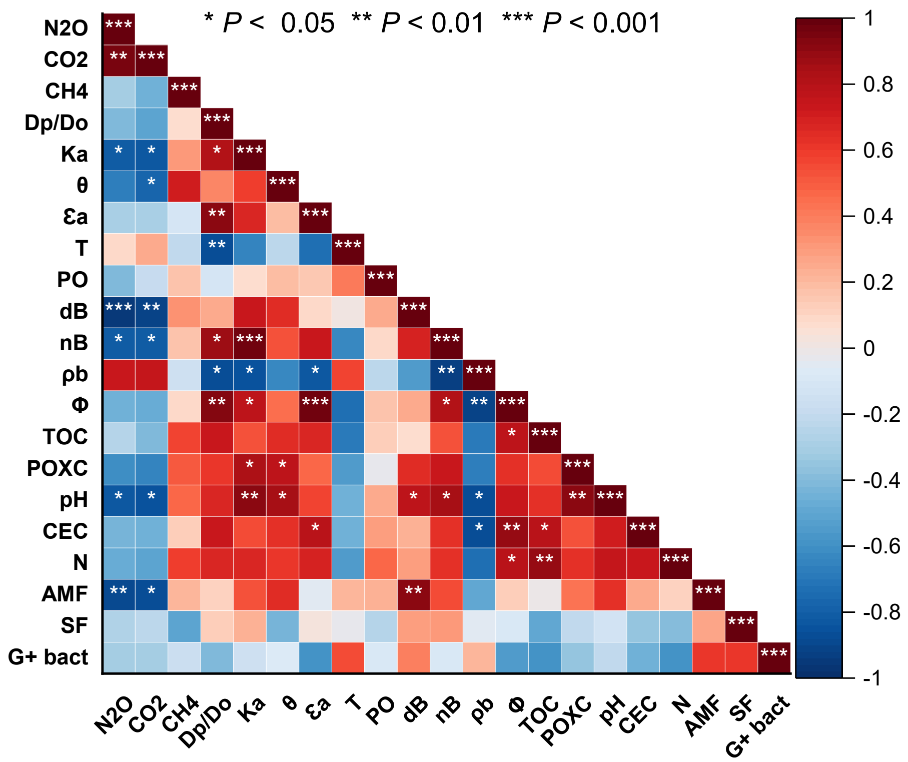

Figure 44: Correlation of soil physical, chemical, and microbial properties with greenhouse gas fluxes. Statistical significance is denoted as *** ($P <$ 0.001), ** ($P < 0.01$) and * ($P < 0.05$). N_2O, nitrous oxide; CO_2, carbon dioxide; CH_4, methane; ρ_b, bulk density; Ø, total porosity; θ volumetric water content; k_a, air permeability; DP/DO relative gas diffusivity; ε_a ,air-filled porosity; τ, pore tortuosity; PO, pore organization; d_B effective pore diameter; nB, number air-filled pores in a given soil cross-section; POXC, permanganate oxidizable carbon; TOC, total organic carbon; TOC stock, total carbon stock; CEC, cation exchange capacity; AMF, arbuscular mycorrhizal fungi; SF, saprophytic fungi; Gram+ bact., gram-positive bacteria.

Among the soil chemical properties, soil pH showed a strong negative correlation with N_2O ($r = -0.83$; $P < 0.05$) and CO_2 ($r = -0.84$, $P < 0.05$) emissions, while AMF was the only soil microbial property that negatively correlated with N_2O ($r = -0.89$, $P < 0.01$) and CO_2 ($r = -0.86$, $P < 0.01$) emissions (Figure 44). For CH_4 emissions, no significant correlations ($P > 0.05$) were observed with the soil's physical, chemical, and microbial properties.

The principal component analysis (PCA) revealed that the first three principal components (PCs) explained 86.75% of the total variance in the dataset (Figure 45). The PC1, explaining 53.68% of the variance, was positively correlated with pH, CEC, total N, and nB, while being negatively correlated with ρ_b, and this PC1 was regulated by B20CP20. The PC2 which explained 20.95% of the variance, was negatively linked with the CO_2, and N_2O emissions, TOC, and ε_a, but positively associated with τ, dB, AMF, and gram-positive bacteria. The PC2 was mainly regulated by the single compost treatment. The PC3 contributed 12.4% of the variance and was associated with CH_4 emissions, Dp/Do, volumetric water content, PO, and saprophytic fungi biomass, under the influence of the single biochar treatment at 20 t ha^{-1}.

When organic amendments are applied to soils, changes in the rates of greenhouse emissions can occur as a consequence of several factors, including the extent to which the amendments impact soil chemical, physical and biological properties Basheer *et al.* (2024). In the current study, air permeability, water retention, effective pore diameter and number of air-filled pores in soil across section were the physical drivers of N_2O, CO_2 and CH_4 emissions in the single compost and co-biochar and compost treatments.

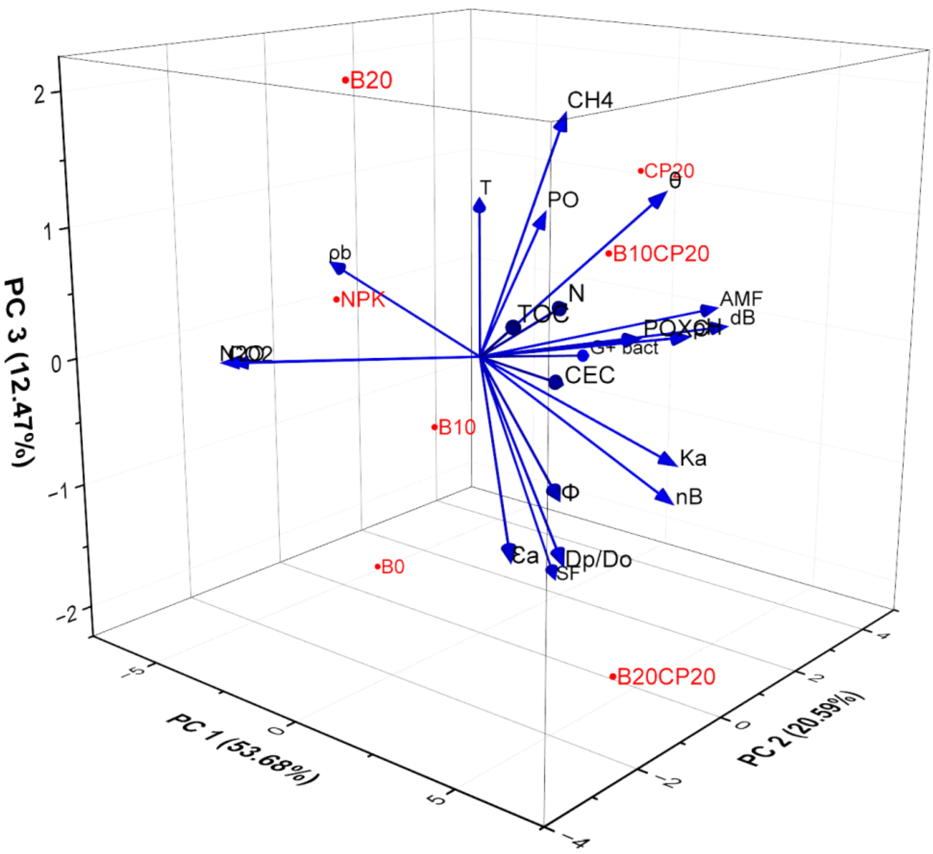

Figure 45: Biplot of Principal component analysis (PCA) of greenhouse gas emissions and soil properties. The loadings include the following variables, N_2O, nitrous oxide; CO_2, carbon dioxide; CH_4, methane; ρ_b, bulk density; Ø, total porosity; θ volumetric water content; k_a, air permeability; D_P/D_O relative gas diffusivity; ε_a, air-filled porosity; τ, pore tortuosity; PO, pore organization; d_B effective pore diameter; nB, number air-filled pores in a given soil cross-section; POXC, permanganate oxidizable carbon; TOC, total organic carbon; CEC, cation exchange capacity; AMF, arbuscular mycorrhizal fungi; SF, saprophytic fungi; Gram+ bact., gram-positive bacteria. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Specifically, higher gas permeability and increased air-filled pores observed in the co-biochar and compost treatments were associated with low N_2O emission. Similarly, Mukherjee *et al*. (2014) reported higher gas permeability and enhanced soil aeration under biochar amendment contributed to decreased N_2O emissions. Also, according to Hu *et al*. (2023), enhanced soil aeration decreased N_2O production by changing denitrification, which can be promoted in anaerobic soil conditions.

However, the mechanism behind the negative correlation between CO_2 and air permeability, number of air-filled pores and effective pore diameter seem difficult to explain. The reason is that earlier researchers found that increased air permeability and aeration is associated with increased CO_2 emissions especially under biochar addition because higher soil aeration accelerates soil organic matter mineralisation rate. Also, Jarecke (2015) reported that CO_2 concentration dynamics had the strongest relationship with soil O_2 dynamics, which suggests that soil O_2 availability under short-term soil saturation creates conditions for carbon mineralization under the increased labile substrate. Conversely, positive correlations between soil CO_2 emissions and total porosity, air-filled porosity, and gas diffusivity were reported by Li *et al*. (2019) as against the insignificant negative correlation between CO_2 and these same parameters observed in the current study. They highlighted that diffusion pathways and oxygen availability is key for aerobic microbial metabolism and gas transport in the soil. Meanwhile, it is plausible to infer that due to the enhanced specific surface area of the soil after amendment with co-biochar and compost, the CO_2 produced in the soil might have been physically absorbed to inhibit their emission as indicated by (Yang *et al*. 2022).

The negative correlation between CO_2 emissions and volumetric water content observed in the current study is similar to the findings of Gui *et al*. (2023) who observed lower fluxes of CO_2 under wet conditions and recorded a negative correlation between CO_2 production and volumetric water content.

They attributed it to saturated soil condition leading to constrained diffusion and decreased oxygen availability for aerobic respiration that might have limited microbial activity and soil CO_2 production. Meanwhile, Buragiene *et al*. (2019) reported a positive correlation between CO_2 emissions and soil moisture.

Soil pH was the major soil chemical property that was negatively associated with N_2O and CO_2 emissions in the co-biochar and compost-amended treatments. Agreeably, Hénault *et al*. (2019) observed that soil pH was the major factor that explained most of the variability of a soil's capacity to reduce N_2O, explaining 59% of index variability. They further elaborated that N_2O reduction appeared inefficient at pH < 6.4 but was very efficient at pH > 6.8. Also, a pH value of 6.8 has been highlighted by Wang *et al*. (2018) as the default emission factor value of 1.0% for agricultural soil proposed by the Intergovernmental Panel on Climate Change (IPCC) is relevant. Conversely, a study by Lui *et al*. (2016) did not find a significant correlation between soil pH value and N_2O emissions. Studies have shown that microbial processes of denitrification in anaerobic conditions and nitrification in the presence of oxygen are accepted to be the main sources of N_2O in agricultural soils (Butterbach-Bahl *et al*., 2013).

According to Hénault *et al*. (2019) the last step of the denitrification process which is the reduction of N_2O to N_2, is identified as the only known pathway for the terrestrial removal of N_2O. In the current study, the greenhouse measurement was done at near soil saturation (−10 hPa) and these treatments were associated with the highest water contents, thus, it is plausible to infer that denitrification dominated the N_2O emission under these treatments. However, the increased soil pH of the co-biochar and compost-amended treatments might have facilitated complete denitrification that reduced N_2O to N_2 as enunciated by Butterbach-Bahl *et al*. (2013). The finding from the current study is further supported by the high soil pH ~7.0 of the co-biochar and compost treatments as against the low pH range of 5.7–6.3 of the other treatments where N_2O emission was enhanced.

The biomass of arbuscular mycorrhizal fungi (AMF) was identified as the primary regulator of reduced N_2O and CO_2 emissions in the single compost treatment. In agreement with the current study, Li *et al*. (2023) observed that AMF significantly reduced N_2O emissions under variable precipitation in a temperate grassland. This reduction was attributed to AMF's ability to lower the soil's available nitrogen concentration and alter the composition of N_2O-producing bacteria. In the same study, Li *et al*. (2022) found that AMF hyphal length density was negatively correlated with the key N_2O production genes nirK and nirS and positively correlated with the N_2O consumption gene nosZ. Furthermore, Li *et al*. (2023) indicated that AMF suppressed N_2O producers, particularly nirS-type denitrifiers, by slowing the release of carbon and nitrogen from degraded residues, resulting in a significant reduction of N_2O emissions.

Additionally, Hawkins *et al*. (2023) explained that AMF reduces CO_2 released into the atmosphere through several mechanisms, including the use of carbon to build and support an active mycelial network, the conversion of carbon to fungal necromass that acts as a scaffold for soils, and the release, immobilisation, and stabilisation of soil carbon. Boyno *et al*. (2023) also found that AMF-treated soil reduced CO_2 release by an average of 9.4% compared to non-mycorrhiza-treated soil. In this study, greenhouse gas emissions were taken from soil with near-saturation water content, 12 months after amendment application. AMF was positively correlated with soil pore diameter (r = 0.92), the number of air-filled pores (r = 0.54), and soil water content (r = 0.65), but negatively correlated with soil bulk density (r = -0.5), all of which are influenced by soil aggregation. Taken together, it is plausible to deduce that AMF affected the decomposition, release, and availability of substrates through soil aggregate formation, and also influenced CO_2 fluxes or denitrification to reduce N_2O emissions in the compost-amended soil. However, the specific microbial mechanisms by which AMF regulate N_2O emissions during residue decomposition are beyond the scope of this study.

4.5.5 Summary

This study assessed the drivers of potential greenhouse gas emissions of a tropical Acrisol amended with oil palm empty fruit bunch compost and biochar, using intact soil core samples. The findings showed that both single and combined applications increased soil pH but did not affect cation exchange capacity. Sole biochar improved total organic carbon (TOC) but reduced microbial biomass, whereas combined biochar and compost application enhanced air-filled porosity, gas diffusivity, and water content. Compost alone improved water retention but reduced soil aeration. In terms of greenhouse gas emissions, sole biochar had a minimal impact, while compost and combined treatments significantly reduced CO_2 and N_2O emissions but increased CH_4 emissions, likely due to enhanced microbial activity from the carbon and nitrogen content in amendments. The reduction in CO_2 and N_2O emissions was linked to improvements in soil pH, air permeability, and AMF fungal biomass. Overall, EFB amendments showed potential to improve soil health and reduce greenhouse gas emissions in tropical soils, though further in-situ studies are needed to confirm long-term effects.

CHAPTER FIVE

SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

5.1 Summary

The first objective of the study examined the elemental composition, molar ratios, nutrient content, heavy metal and polycyclic aromatic hydrocarbon (PAH) concentrations in Empty Oil Palm Fruit Bunch (EFB) feedstock, two types of EFB biochar from two different reactors; a rotary reactor (B1) and locally designed kiln (B2) and three EFB compost produced from co-composting of EFB with poultry manure at mixing ratios of 1:1.5 (C1); 1:2 (C2) and 1:3 (C3) respectively. The result revealed that pyrolysis lowered the atomic ratios of H/C and O/C_{org} from 1.56 and 1.55, respectively, in the EFB feedstock to < 0.7 and < 0.4, respectively, in the biochars, inferring high aromatisation, polarity and hydrophobicity and increased the nutrient content except for K_2O and SO_3 in B1 compared to B2 and EFB feedstock. Incidentally, the sum of 16 PAHs in both biochars were above the European Biochar Commission (EBC) thresholds of 6 mg kg^{-1} (d.m.), with B2 having the highest of 23.7 mg kg^{-1}.

Also, the sum of EFSA 8 PAHs of 1.2 and 1.8 mg kg^{-1} were obtained in the B1 and B2 respectively which were above the allowable threshold of 1 mg kg^{-1} and this was possibly due to contamination of the biochar with pyrolysis vapours that contained PAHs during the cooling stage. The EFB composts had a relatively low C/N ratio and a high N and ash content compared to the original EFB feedstock. The C1 had the highest nutrient content and lowest C/N ratio compared to C2 and C3. The PAHs in all three composts were below the detection limit of < 0.1 mg kg^{-1} and the heavy metal contents were below the threshold limit of European Biochar Commission.

The second objective of the study determined the loading capacity of EFB biochar for four different tropical soils and assessed the short-term chemical response, potential phytotoxicity, and nutrient uptake at varying application rates. EFB biochar application increased the pH of all the soil types except the Vertisol. At a 2% w/w application rate, available phosphorus increased by 175%, 68.9% and 49% and organic carbon contents also increased by 49%, 22%, and 19% in Acrisol, Brown Ferralsol and Vertisol, respectively. The study observed no phytotoxic effect of EFB biochar on maize germination in all soil types. Additionally, the co-application of EFB biochar with inorganic fertiliser significantly increased N and P uptakes in the Acrisol but the positive effect was insignificant in the other soil types. Above all, based on the comprehensive amelioration score, the highest application rate of 2.0% emerged as the ideal rate for Acrisol, Red Ferralsol and Vertisol soil types, whilst rates of 1% was best suited for the Brown Ferralsol.

Objective three evaluated the impact of EFB biochar and compost on soil properties, okra nutrient use efficiency and yield over two cropping cycles in nutrient-poor tropical soil. Treatments included EFB biochar (10 and 20 t ha^{-1}) [B10, B20], compost (20 t ha^{-1}) [CP20], biochar-compost combinations [B10CP20, B20CP20], an unamended control [B0], and inorganic fertiliser (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}) [NPK]. The study examined the soil chemical properties using standard laboratory procedures and microbial biomass using the Whole-Cell Fatty Acid (WCFA) approach. In the first cycle, EFB amendments boosted okra pod yield by up to 283% over the unamended control. CP20 and B20CP20 increased yields by 58% and 100%, respectively, compared to NPK. However, only B20CP20 maintained higher yields in the second cycle. All EFB treatments increased the soil's pH by 0.6 to 1.6 points, enhanced phosphorus uptake and recovery efficiency, while B20CP20 notably increased soil cation exchange capacity and available phosphorus and zinc content, relative to the unamended control. CP20 also increased arbuscular mycorrhizal fungi and gram-positive bacterial biomass than NPK and B20. Overall, the one-time co-application of EFB biochar and compost improved soil fertility, microbial properties, and phosphorus use efficiency, and sustained okra yields better than sole biochar, compost, or NPK application.

Under the same field experiment, the effect of EFB amendments on soil carbon quality, storage and management index and how they affect soil physical fertility was examined. The results showed that the sole biochar application at 20 t ha^{-1} (B20) increased soil total organic carbon (TOC) and TOC stock by 115% and 98.8% respectively but decreased air permeability by 37.5%. Incidentally, the sole EFB compost application (CP20) significantly increased soil water at field capacity by 21% relative to the unamended control. The combined applications of biochar and compost significantly increased permanganate oxidisable carbon, and carbon management index, lowered the soil's bulk density and increased gas flow by convection and diffusion. Notably, the carbon management index correlated positively with all physical parameters except air-filled porosity and relative gas diffusivity, showcasing the positive effect of soil carbon rehabilitation on soil physical fertility.

Finally, the last objective of the study assessed the impact of EFB biochar and compost, applied singly and in combination, on gas flow, pore characteristics and greenhouse gas (GHG) emissions. After 12 months of amendment application, bulk and intact 100 cm^3 samples were extracted and water retention, relative gas diffusivity, and air permeability at a matric potential of−100 hPa were measured. Subsequently, GHG emissions were also measured on the soil cores at a soil water potential of−10 hPa after the addition of KNO_3 and glucose as nitrogen and carbon sources, respectively. The result revealed that the volumetric soil water content increased uniformly by 15% across all treatments. Gas flow by diffusion and convection significantly increased in the combined applicatios (32−89% and 28–30%, respectively), indicating enhanced soil pore system organization and complexity. Despite the improved soil properties, the sole compost and biochar-compost co-application led to significantly lower emissions of N_2O (~74% decrease) and CO_2 (~50% decrease) compared to the unamended treatment. However, CH_4 emissions were notably higher (87− 685%) for all biochar and compost treatments compared to the unamended treatment. The soil pH, air permeability and number of air-filled pores in a soil across section were negatively associated of N_2O and CO_2 emissions in the combined biochar and compost treatments. However, biomass of arbuscular mycorrhizal fungi (AMF) was the main regulator of reduced N_2O and CO_2 emissions in the sole compost treatment.

5.2 Conclusions

Firstly, the pyrolysis of EFB lowered the atomic ratios of the biochars inferring high aromatisation, polarity and hydrophobicity and increased the plant nutrient content compared to the EFB feedstock. Incidentally, the sums of 16 PAHs and EFSA 8 PAHs in both biochars were above the EBC thresholds of 6 mg kg^{-1} and 1 mg kg^{-1} (d.m.), respectively. These imply that pyrolysis conditions can alter the toxicant content of EFB which can limit the EFB biochar use for soil application. Thus, EFB pyrolysis under improved conditions such as a kiln design that separates the pyrolysis gas from biochar can reduce possible PAH accumulation on the biochar. On the other hand, the co-composting of EFB with poultry manure at different mixing ratios significantly varied but reduced the C/N ratio to < 20 and increased the nitrogen and ash content compared to the EFB feedstock. Also, the PAHs and heavy metal concentration in all three EFB co-composts were below the threshold limit, suggesting the safe use of these EFB composts as a source of nutrient for crop production. Based on the findings of this research, the first hypothesis was accepted.

The second hypothesis asserted that different tropical soils exhibit different short-term chemical response and phtytotoxicity to EFB biochar application. Notably, the application of EFB biochar significantly increased pH across all soil types except the Vertisol, implying that Vertisol may require biochar application at a rate higher than 2 % w/w due to its high buffering capacity. Again, the biochar application led to available phosphorus retention in the Red Ferralsol and this is attributed to phosphorus adsorption to calcium and magnesium ions when the pH of the soil was elevated beyond 7. Acrisol showed a significant increase in nutrient uptake from the co-application of empty fruit bunch biochar and mineral fertiliser, while the other soil types exhibited no effect, leading to the acceptance of the hypothesis that co-application of empty fruit bunch biochar and inorganic fertiliser will affect nitrogen and phosphorus uptake differently across soil types. Collectively, the highest application rate of 2.0% emerged as the ideal rate for Acrisol, Red Ferralsol and Vertisol soil types, while rate of 1% was best suited for the Brown Ferralsol, giving a clue on the empty fruit bunch biochar loading capacity of these dominant and most cultivated soil types in the tropics. Consequently, empty fruit bunch biochar shows promise as a soil conditioner in low-nutrient tropical soils, with effects dependent on rate and soil type. My findings provide insights on the dynamics of empty fruit bunch mineralisation and the release of mineral nutrients and toxic compounds on germination, growth and yield of crops depending on the soil type and the rate of application. This knowledge is vital for scaling up empty fruit bunch biochar application as a soil conditioner in tropical soils.

The third hypothesis underpinning this study was that EFB biochar and compost application enhance soil chemical and microbial properties which will drive nitrogen and phosphorus use efficiency and yield. EFB amendments (Sole biochar (10 and 20 t ha^{-1}), sole compost (20 t ha^{-1}) and biochar-compost combinations) boosted okra pod yield by up to 283% over the unamended control in the first cropping cycle. However, only the combined application of EFB biochar and compost increased soil cation exchange capacity, available phosphorus and zinc content and maintained higher yields in the second cycle implying that only the combined application of EFB biochar and compost can improve soil fertility to drive okra phosphorus use efficiency and sustain okra yields beyond one cropping cycle. The findings highlight the potential of EFB amendments for long-term soil productivity and crop performance in tropical agroecosystems. Based on the results of the study, the third hypothesis was accepted.

Moreover, the fourth hypothesis of the study stated that soil carbon storage increases from EFB biochar and compost application will impact soil physical fertility. Concurrently, the sole biochar (20 t ha^{-1}) application significantly increased soil total organic carbon content and stock by 115%, 98.8% respectively, but it's effect on soil physical fertility was negligible. However, the sole application of EFB compost did not significantly increase carbon management index but it increased soil water retention at field capacity. Moreover, the combined applications of EFB biochar and compost significantly increased permanganate oxidizable carbon and carbon management index that led to decrease in the soil's bulk density, increased specific surface area, and gas flow by convection and diffusion and increased soil water retention, thus the fourth hypothesis underpinning this study was accepted. The practical implication of these results is that sole EFB biochar or compost application to tropical sandy soils for improved carbon sequestration and physical fertility will have short comings and these can be resolved by applying these two amendments together at the same time.

Lastly, the hypothesis that soil amendment with EFB biochar and compost triggers a shift in intrinsic soil physical and biochemical properties that will regulate greenhouse gas emissions was tested after 12 months of their applications in an incubation studies. The results showed enhanced soil pore system organization and complexity which led to improved gas flow in the soil when biochar and compost were applied together. Despite this, sole biochar applications significantly increased CO_2 and N_2O emissions compared to the sole compost and combined biochar and compost applications. However, CH_4 emissions was significantly higher in the sole compost and combined biochar and compost applications compared to the sole biochar applications, chemical fertiliser, and unamended treatments. Consequently, soil pH optimisation and increase in the biomass of arbuscular mycorrhizal fungi and improved air permeability and number of air-filled pores drove the lower emissions of N_2O and CO_2 in the sole EFB compost and combined biochar and compost amended soils. These findings led to the acceptance of the fifth hypothesis of the study. The study suggests that single compost and combined application of biochar and compost can positively influence various soil physical, chemical and microbial properties and these together can lower N_2O and CO_2 emissions. However, it is crucial to monitor methane emissions, which increased with the application of biochar and compost.

Above all, the work concludes that when empty oil palm fruit bunch compost and biochar are applied together, it has the potential to improve the *production function* of tropical soils (as reflected by high nitrogen and phosphorus uptake and use efficiency and increased in crop yields) without compromising the c*limate-regulative function* of the soil as manifested by the increased carbon stock and management index, high soil water retention and gas flow and lower emissions of greenhouse gases.

5.3 Recommendations

Firstly, empty oil palm fruit bunch (EFB) was characterized by high carbon content, low levels of toxicants, and a substantial array of essential macro and micronutrients, making them a valuable and cost-effective organic feedstock for soil restoration in available areas. For the co-composting of EFB and poultry manure, an optimal mixing ratio of 1:1.5 (w/w) is recommended for starters (Farmers) for increased nutrient value of EFB for a soil amendment. Furthermore, it is recommended to EFB biochar producers and farmers that EFB pyrolysis should be conducted using improved kiln designs that separate pyrolysis gases from the biochar, thereby reducing the potential accumulation of polycyclic aromatic hydrocarbons (PAHs) in the final biochar.

Again, the use of Chain saw in the shredding of the empty fruit bunches before composting was expensive and time consuming, thus further studies is suggested to identify an easy, efficient and cost-effective approach forreducing the particle size of the bunches before composting. Again, further studies into the use of compost accelerators to enhance the decomposition and maturation time for EFB compost is recommended.

Also, EFB biochar promises to be a potential soil conditioner that can be used to resolve constraints of crop production in tropical soils, especially those with low organic matter content, phosphorus sorption and acidity problems. However, higher application rate beyond 2.0% is recommended for soils with high clay, cation exchange capacity and organic carbon for the positive effect of EFB biochar to be fully realized.

Finally, the findings from this study recommends the combined application of EFB biochar and compost instead of their single applications in soil degradation restoration for increased and sustained crop production, carbon rehabilitation and greenhouse gas mitigation especially in tropical sandy soils. However, the current field study lasted for only a year and so long-term field scale studies in different agro-ecological zones and cropping conditions are recommended to confirm these observations and to examine the lasting impacts of these EFB amendments. Also, it is crucial to monitor methane emissions, which can increase with the application of biochar and compost.

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APPENDICES

Appendix A: Supplementary Information

Table S1: Chemical properties of EFB biochar and compost applied to the experimental plots

Property	EFB Biochar	EFB Compost
Ash (%)	41.6	66.7
Volatile matter (%)	8.9	22.1
Total carbon (%)	44.8	14.2
Carbonate (%)	2.95	2.15
pH ($CaCl_2$)	9.9	8.1

Nitrogen (%)	0.85	1.05
Phosphorus as P_2O_5 (%)	0.66	3.7
Potassium as K_2O (%)	3.54	3.4
Calcium as CaO (%)	2.49	9.7
Magnesium as MgO (%)	1.59	1.4
Sodium as Na_2O (%)	0.15	0.6
Iron (%	1.02	1.7
Sulfur (%)	0.10	0.21
Manganese (mg kg^{-1})	267	243
Zinc (mg kg^{-1})	170	152
Boron (mg kg^{-1})	41	270
Total 16 EPA-PAH (mg kg^{-1})	20.3	n.c

EPA, Environmental Protection Agency; PAHs, polycyclic aromatic hydrocarbons; nc, non-calculable.

Table S2: Characteristics of soil (Acrisol) before amendments application

Property	**Value**
pH (water 1: 2.5 w/v)	5.53 ± 0.12
Electrical Conductivity (µs cm^{-1})	30.0 ± 0.00
Organic Carbon (%)	0.99 ± 0.04
Organic Matter (%)	1.70 ± 0.07
Total Nitrogen (%)	0.62 ± 0.06
Available Phosphorus (mg kg^{-1})	4.15 ± 1.48
Potassium (cmol kg^{-1})	0.36 ± 0.01
Calcium (cmol kg^{-1})	1.20 ± 0.10

Magnesium (cmol kg^{-1})	0.46 ± 0.02
Sodium (cmol kg^{-1})	0.11 ± 0.01
Sum of bases (cmol kg^{-1})	2.12 ± 0.09
Exchangeable Acidity (cmol kg^{-1})	0.31 ± 0.065
ECEC ($cmol_+$ kg^{-1})	6.87 ± 1.36
Clay (%)	25.0 ± 1.0
Silt (%)	17.0 ± 1.0
Sand (%)	58.0 ± 1.0
Textural class	Sandy clay loam
Bulk density (g cm^{-3})	1.49

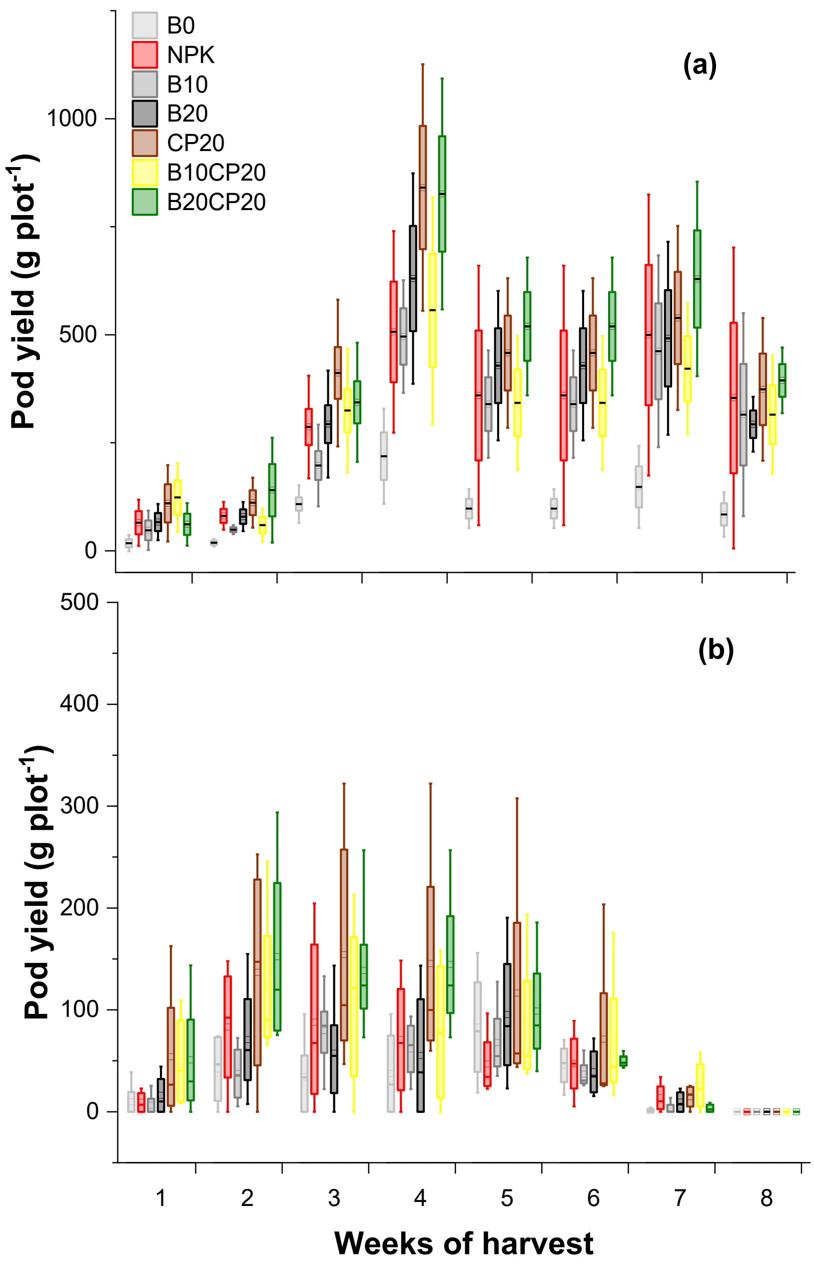

Figure S1: Weekly trends of pod yields in response to one-time application of amendments across two cropping cycles. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

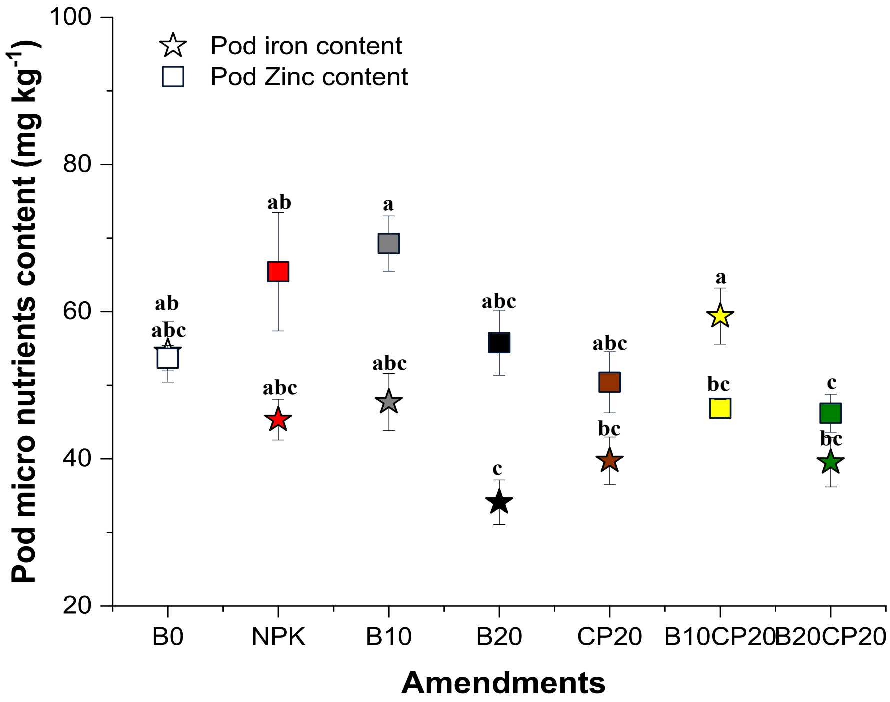

Figure S2*:* Zinc and iron content of okra pod as affected by treatments in the first crop cycle. Bars with different letters mean differences among treatments (P < 0.05). Error bars show the standard error of means from nine replicates. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

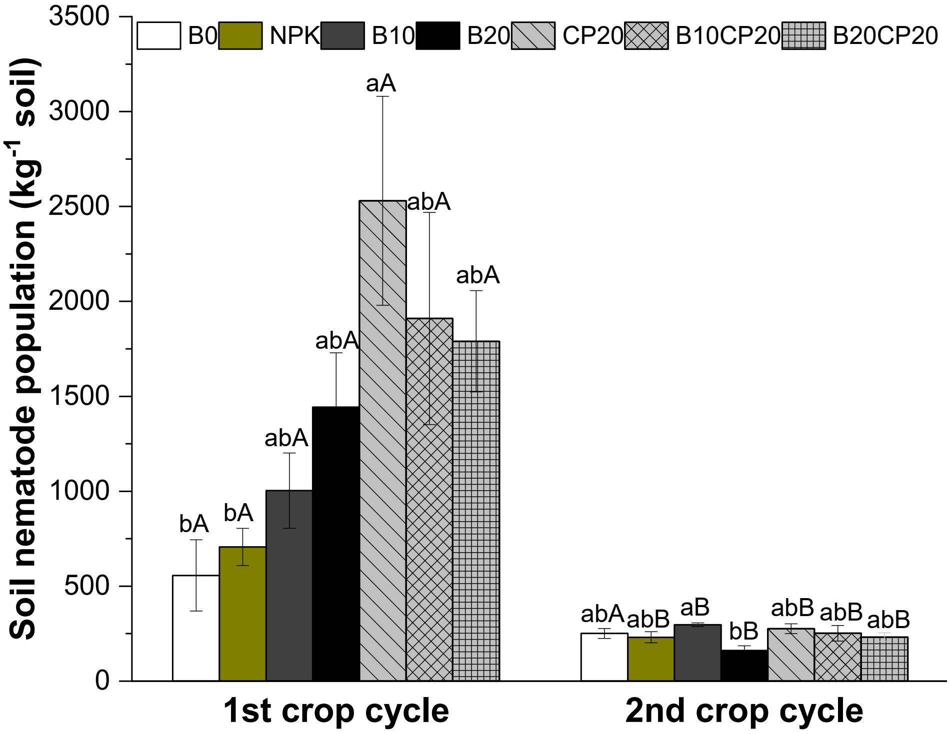

Figure S3: Soil nematode population as affected by the amendments across the two cropping cycles. Bars with different capital letters mean the differences between the two cropping cycles under the same amendment ($P < 0.05$). Bars with different lowercase letters mean differences among amendments over the same crop cycle ($P < 0.05$). Error bars show the standard error of means from nine replicates. B0 (unamended); NPK (100 kg N ha^{-1} 60 kg P_2O_5 ha^{-1} and 60 kg K_2O ha^{-1}); B10 (10 t ha^{-1} biochar); B20 (20 t ha^{-1} biochar) CP20 (compost 20 t ha^{-1}); B10CP20 (10 t ha^{-1} biochar + 20 t ha^{-1} compost); B20CP20 (20 t ha^{-1} biochar + 20 t ha^{-1} compost).

Appendix B: Lists of Manuscripts and Publications from the Thesis

Article	Title	Authors	Journal	Status
1	Immediate Chemical Response and Potential Phytotoxicity of Different Tropical Soils to Empty Oil Palm Fruit Bunch Biochar Application	Dorcas Blankson, Kofi Atiah, Emmanuel Arthur* and Kwame Agyei Frimpong	Journal of Soil Science and Plant Nutrition	Published
2	Empty Oil Palm Fruit Bunch Biochar and Compost Application Improve Soil Properties that Drive Okra (*Abelmoschus esculentus*) Nutrient Use Efficiency and Yield	Dorcas Blankson, Emmanuel Arthur, Kofi Atiah, Kwame Agyei Frimpong*, Patrick Marfo and Sabine Ravnskov	Vadose Zone Journal	Under review
3	Drivers of Greenhouse Gas Emissions from a Tropical Acidic Soil Amended with Empty Fruit Bunch Biochar and Compost	Dorcas Blankson*, Kwame Agyei Frimpong, Kofi Atiah, Maliheh Fouladidorhani, John Bright Amoah Nyasapoh and Emmanuel Arthur	Soil Science Society of America Journal	Under review
4	Effects of Empty Oil Palm Fruit Bunch Biochar and Compost Application on Soil Carbon storage and Soil Physical Fertility	Dorcas Blankson, Emmanuel Arthur, Kwame Agyei Frimpong and Kofi Atiah*	---	Under review with co-authors

* indicates correspondence