



School of Earth Sciences and Engineering
Department of Sustainable Natural Resources

**SOIL AGGREGATE STABILITY AND ORGANIC CARBON STOCK
AS INFLUENCED BY LAND USE IN THE PANDAMANTENGA
ALLUVIAL PLAINS, BOTSWANA**

by

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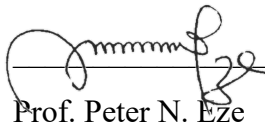
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CERTIFICATION

The undersigned certifies that he has read and hereby recommends for acceptance by the School of Earth Sciences and Engineering a dissertation titled: *Soil aggregate stability and organic carbon stocks as influenced by land use in the Pandamatenga alluvial Plains, Botswana*, in fulfilment of the requirements for the degree of Master of Science in Environmental Science of the BIUST.

 07. 07. 2025
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ABSTRACT

Soil aggregate stability and organic carbon (OC) storage are key biogeochemical indicators of soil health, and their ability to function within an ecosystem includes providing sustainable and just media for the transformation of agri-food systems. Soil aggregation protects OC, and its disruption through natural or human activities can lead to OC loss. While land use impacts soil quality, its effect on aggregate stability and OC stocks in Botswana's agriculturally important Pandamantenga alluvial plains is not well understood. This study assessed the influence of four distinct land uses—arable land (subject to tillage and mechanical disturbance), grassland exclosures (areas protected from grazing to allow vegetation recovery), natural exclosures (undisturbed areas with minimal human intervention), and pastures (grazed rangeland) on soil structure and carbon dynamics. Soil samples were collected at three depths (0–15, 15–30, and 30–45 cm) and sieved into six aggregate size classes (>4.75 , 4.75–2.00, 2.00–1.00, 1.00–0.50, 0.50–0.25 and <0.25 mm). Routine laboratory methods, including particle size, bulk density, pH, electrical conductivity, exchangeable bases, OC content, and aggregate-associated carbon, were used to analyse the soil properties. Aggregate stability was assessed via wet sieving, and indices such as water-stable aggregate (WSA), mean weight diameter (MWD), geometric mean diameter (GMD), and $R_{0.25}$ were calculated. The results revealed that land use significantly affected the soil structure and carbon dynamics. The natural exclosures presented the highest MWD and GMD, followed by the grassland exclosures, pastures, and arable land. The WSA varied by land use and aggregate size, with arable land having more WSA in smaller fractions and less WSA in larger fractions. Pastures had more mid-sized aggregates, whereas natural exclosures were dominated by larger aggregates. $R_{0.25}$ remained consistent across land uses because of the high clay content. Larger aggregates (>0.25 mm) contained more OC and showed a strong positive correlation between stability and OC, highlighting the importance of

aggregate stability for carbon sequestration and soil health by minimizing carbon loss and enhancing SOC storage. The natural exclosures had the highest OC stocks, particularly in the larger aggregates, followed by the arable land. Grassland exclosures and pastures had lower OC stocks. These findings will inform strategies to improve sustainable agricultural productivity, soil conservation, and carbon management in the face of climate change.

Keywords: Carbon sequestration; Soil conservation; Tillage; Water stable aggregates; Vertisols

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CHAPTER ONE

INTRODUCTION

1.1. Background

The interaction between land use and soil health has become a pressing concern in contemporary agricultural and environmental contexts. Land use and management not only influence food production and economic development but also play vital roles in maintaining ecosystem health and sustainability (Yost et al., 2016). Therefore, achieving a balance between human needs and environmental conservation relies on our understanding of how different land use practices influence soil properties. In this context, soil structure stands out as a critical indicator of soil health. It significantly impacts soil functionality, supports plant and animal ecosystems, and influences environmental quality, particularly through its effects on soil carbon storage and nutrient dynamics (Manna et al., 2007; Udeigwe et al., 2025). Environmental factors affecting soil structure are often evaluated by examining soil aggregates, the fundamental structural units of soil. Aggregate stability, which measures the resilience of soil aggregates to different stresses, such as tillage, water erosion, and wind erosion, serves as a key metric for examining alterations in soil structure.

The stability of soil aggregates influences the ability of soil to carry out ecosystem functions, including food production, climate change mitigation, aeration, erosion, and biological activity (Okolo et al., 2020). Soil organic carbon (SOC) serves as both a binding agent and as a nucleus in the development of soil aggregates. Specifically, SOC contributes to soil aggregation by binding with soil mineral particles, which in turn leads to a reduction in aggregate wettability and improved mechanical strength of the aggregates (Georgiou et al., 2022). Moreover, the increase in soil organic matter partially mitigate carbon emissions and climate change. One important aspect that demands attention is the relationship between soil aggregate stability and

soil carbon sequestration, a link that holds profound implications for climate change mitigation. The stabilization of soil aggregates not only safeguards essential ecosystem functions but also serves as a key indicator of soil carbon dynamics (Ebouel et al., 2024; Kumar et al., 2023). Consequently, understanding the links between aggregate stability and soil organic carbon stocks is crucial for developing sustainable soil management strategies and mitigating the adverse impacts of climate change.

Land use changes, specifically deforestation, transition from natural environments to agricultural uses, and intense cultivation practices, can lead to major variations in soil processes and characteristics, and hence soil functioning (Zajícová and Chuman, 2019). Many studies have highlighted how changes in land use impact indicators of soil fertility and quality due to changes in abiotic factors, such as moisture and temperature (Dong et al., 2022; Srivastava et al., 2016; Xu et al., 2017), as well as biotic factors, like vegetation and biodiversity (Angers and Eriksen-Hamel, 2008; Blankinship et al., 2016). It also influences the stabilization of soil organic matter (Mataix-solera, 2016; Schwendenmann and Pendall, 2006). Land use changes primarily lead to significant modifications in the inputs and dynamics of organic carbon, which govern soil physical, chemical and microbial processes (Peng et al., 2023; Shrestha et al., 2007; Tobiašová, 2011). In general, the decline in soil organic matter following changes in land use is attributed primarily to decreased rates of organic matter input, increased decomposability of both crop residues and native soil organic matter (SOM), and a reduction in the physical protection of SOM (Feng et al., 2021; Liu et al., 2014). The conversion of forest to cropland often leads to the disruption of the soil structure (Hoogsteen et al., 2015; Wen et al., 2023). This disruption enhances the mineralization of organic carbon, as organic matter is no longer physically protected from microbial decomposition. Moreover, cultivation can alter aggregation and aggregate size distributions (Gelaw et al., 2015).

Although the literature provides valuable insights into the effects of land use on soil organic carbon stocks and aggregate stability, significant gaps remain. However, no research has explored the effects of land use changes on soil aggregate stability and soil organic carbon stocks, particularly in semi-arid Vertisols such as those in the Pandamatenga Plains.

1.2. Statement of the problem

The Pandamatenga alluvial Plains have experienced major land use changes over time, transitioning from natural landscapes to areas dominated by agricultural activities and other anthropogenic operations. This alteration may have a negative influence on soil health and ecosystem services, specifically soil aggregate stability and soil organic stocks, which serve as crucial indicators of soil quality and fertility. Despite the widely acknowledged importance of soil health in maintaining agricultural production and environmental quality, there is little evidence of a relationship between land use, organic carbon, and soil aggregate stability in the Plains. Previous research in the area focused on the potential for no-tillage (Tapela et al., 2007) and agricultural drainage water reuse (Kakooza, 2023). The lack of this knowledge impedes the development of long-term land management plans that could improve soil health and resilience. Therefore, the challenge lies in investigating organic carbon stocks and aggregate stability in the agricultural hotspot soils of the Pandamatenga Plains.

1.3. Justification of the study

The evolving landscape of the Pandamatenga alluvial Plains, resulting from fluctuating land use over time, needs an in-depth investigation on the impacts of these practices on soil aggregate stability and organic carbon stocks. Land use activities have the potential to alter the long-term sustainability of soil properties, impair the functioning of the ecosystems and contribute to global challenges such as land degradation and climate change. Although the Pandamatenga Plains are highly important, there has been no research into how different land use practices affect soil aggregate stability and organic carbon stocks. This study seeks to fill a major knowledge gap by investigating the impacts of different land use practices on soil structure and organic carbon stocks. Understanding these interactions is critical for creating tailored, site-specific management practices that improve soil resilience, preserve soil productivity and contribute to sustainable land management. The results of the research provide valuable insights into the effects of land use changes on soil quality, as well as evidence-based recommendations for optimizing land use practices in the area. This study contributes to educated decision-making processes, which helps to protect the Pandamatenga Plains soil resources, assuring their long-term productivity and role in food security.

1.4. Aim

The overarching aim of this study is to investigate the effects of different land use practices on soil aggregate stability and organic carbon stocks in the Pandamatenga alluvial Plain.

1.5. Specific objectives

1. To characterize the physical, chemical and mineralogical properties of the soils of the Pandamatenga Plains.
2. To assess the distribution of water-stable aggregates in various size classes and determine aggregate stability parameters across different land uses.
3. To quantify and examine the distribution of organic carbon associated with different aggregate sizes and determine the contribution rate of each size class to the bulk SOC
4. To evaluate soil organic carbon stocks in bulk soils and within different aggregate size classes across various land use regimes.

1.6. Thesis layout

This thesis consists of six chapters.

Chapter One: General introduction

Chapter Two: Review of Contemporary Literature

Chapter Three: Methodology

Chapter Four: Results

Chapter Five: Discussion of the Findings

Chapter Six: Synthesis and Conclusion

CHAPTER TWO

REVIEW OF CONTEMPORARY AND RELEVANT LITERATURE

2.1. Soil aggregation pathways in the soil environment

A comprehensive review of formation pathways, land use change effects, and roles of soil aggregates in ecosystem function delivery has been critically done by Ponyane et al., (under review). The process of soil aggregation comprises joining and holding together aggregates of different sizes with the help of different organic and inorganic materials (Tisdall and Oades, 1982). The inorganic binding agents include mainly clays; polyvalent metal cations such as Ca^{2+} , Fe^{3+} , and Al^{3+} ; oxides and hydroxides of Fe and Al; calcium and magnesium carbonates; and gypsum. The organic binding agents are categorized into three categories on the basis of the age and degradation of organic matter: the transient type, which comprises polysaccharides of plant and microbial origin; the temporary type, which includes roots and fungal hyphae; and the persistent type, which comprises resistant aromatic humic substances associated with polyvalent metal cations (Strickland et al., 1988). These humic substances are physically protected within clay and silt-sized aggregates. The formation of soil aggregates has been presented through various conceptual models. Emerson (1959) proposed a soil crumb model comprising oriented clay and quartz particle domains, with soil organic matter enhancing stability by connecting these domains. In this model, it is proposed that SOM increases the stability of a soil crumb by connecting oriented clay and quartz particle domains. Emerson (1959) suggested that the quantity of protected SOM is directly related to the surface area of these domains, with only a single layer of SOM being linked to the domain surface and stabilized through bonds involving aluminum, iron, and hydrogen. Edwards and Bremner (1967) introduced the microaggregate theory, which proposes that microaggregates form through solid-phase interactions involving organic matter, polyvalent metals, and electrically

neutral clays. This model emphasizes that microaggregates are created through bonding of clay-sized units involving clay particles and polyvalent metals (P), such as Fe, Al, Ca, and organometallic complexes (Singh et al., 2014). Additionally, the organic matter incorporated into these microaggregates is physically protected and inaccessible to microorganisms. The hierarchical model of Tisdall & Oades (1982) proposed that different binding agents act as distinct hierarchical levels of aggregation. This suggests that persistent agents (such as humified organic matter, polyvalent metal cation complexes, and oxides) bind free the primary particles and silt-sized aggregates (<20 μm) to form microaggregates. The implication is that microaggregates initially form independently and subsequently act as foundation units in the formation of macroaggregates. Two years after the aggregate hierarchy theory was published, Oades (1984) proposed a modification to how aggregates are built hierarchically. The original hierarchical aggregate model suggested that aggregates are formed sequentially, with microaggregates initially forming independently and then becoming the foundation for macroaggregate formation. However, Oades (1984) introduced the idea that roots and hyphae, which bind macroaggregates together, serve as the central core for microaggregate formation within macroaggregates. Because roots and hyphae are transient binding agents that break down into fragments, the mucilage-covered fragments become coated with clay, resulting in the formation of a microaggregate within a macroaggregate. According to this model, macroaggregates are first formed and then burst into microaggregates. Many contemporary models and research efforts have been built upon Oades (1984) work where unstable macroaggregates burst into medium microaggregates and then into small microaggregates, but the recently proposed models have incorporated additional insights, considering the dynamic interactions among particulate organic matter (POM) and soil aggregates. Six et al. (2004) proposed a model that highlights that after new organic residue is introduced into the soil, soil fungus and soil microbes use readily available carbon to produce mucilage, leading to the

formation of larger aggregates around coarse (250 μm) interaggregate particulate organic matter (that is, coarse iPOM). These coarse iPOMs further decompose and transform into fine (<250 μm) iPOMs, which are physically protected from microbial decomposition. According to Six et al. (2004), this process is halted if disturbance (tillage) increases macroaggregate turnover, which lowers carbon sequestration. That is, tillage practices increase the rate at which macroaggregates form and breakdown; hence, tillage practices inhibit the formation of organic carbon-enriched microaggregates within macroaggregates. Blanco-Canqui and Lal (2004) stated that existing models designed to assess aggregate hierarchy, particularly in contemporary contexts, fail to universally explain the hierarchical organization of aggregates across diverse soil types. These models predominantly hold validity in soils where soil organic matter (SOM) serves as the primary binding agent. They highlighted a significant research gap concerning the lack of comprehensive models capable of elucidating aggregate hierarchy across various soil types, particularly those where SOM might not be the dominant binding agent. Oades and Waters (1991) investigated how Alfisols, Mollisols, and Oxisols were fractionated into different particle sizes following a variety of disaggregating methods. Only the Alfisols and Mollisols seemed to break down in stages: macroaggregates with a diameter of >250 μm fragmented into microaggregates with a diameter of 20–250 μm , and then 20 μm were released. However, with the Oxisols, when aggregate disruption was initiated, particles <20 μm were released, and there was no stepwise disintegration of the particles; hence, they concluded that there was no evidence of aggregate hierarchy.

Table 2.1 Structured summary of the key soil aggregation models and pathways

Conceptual model	Binding Agents	Aggregate formation process
Emerson (1959)	Soil organic matter, clay, quartz particles.	SOM enhances stability by connecting clay and quartz domains, stabilized by metal cations (Al, Fe, H).
Edwards & Bremner (1967)	Clay-sized units, polyvalent metals (Fe, Al, Ca), organo-metal complexes, organic matter.	Microaggregates are created through bonding of clay particles and are protected from microbial activity.
Tisdall & Oades (1982)	- Inorganic: Clays, polyvalent cations (Ca^{2+} , Fe^{3+} , Al^{3+}), oxides/hydroxides of Fe and Al, carbonates, gypsum. - Organic: Transient (polysaccharides), temporary (roots, fungal hyphae), persistent (humic substances with metal cations).	Microaggregates form first, then act as building blocks for macroaggregates.
Oades (1984)	Roots, fungal hyphae, clay, mucilage from decomposing roots/hyphae.	Roots and hyphae initiate macroaggregate formation, which later bursts into microaggregates.
Six et al. (2004)	Soil fungi, soil microorganisms, coarse iPOM, fine iPOM, mucilage.	Organic residue is used by fungi/microbes to form macroaggregates, with iPOM protected within. Tillage disrupts this process.

2.2. Methods of measuring soil aggregate stability

There are several different ways to measure soil aggregation; some methods focus on aggregate size distribution (Fér et al., 2016; Goebel et al., 2005; Wang et al., 2014), whereas others focus on aggregate stability (Amézketa, 1999; Du et al., 2013; Siddique et al., 2017). The soil aggregate distribution is a fixed measurement that needs to be repeated over time to reflect dynamic changes in the soil. Soil aggregate stability refers to the resistance of size ranges of aggregates to stress. On the other hand, aggregate stability reflects the ability of soil aggregates to retain their structure of solids and voids during rainfall or irrigation. Raindrop impact can disrupt these aggregates, leading to crust formation (Le Bissonnais, 2016), soil compaction (Lin et al., 2022), and a decrease in hydraulic conductivity (Ben-Hur & Lado, 2008). A wide range of techniques have been proposed to assess aggregate stability. The particular breakdown mechanism under investigation is the primary determinant of the approach choice. The four primary breakdown mechanisms are raindrop impact, differential swelling, dispersion, and slaking (Cosentino et al., 2006; Hu et al., 2018; Xu et al., 2017). Researchers have studied aggregate breakdown by wetting, whereas others have focused on mechanical breakdown, and still others have combined the two actions. Wet sieving is the most commonly used method for measuring aggregate stability. Wet sieving tests aggregate resilience against water erosion. Other techniques, including turbidimetric methods, rainfall simulation, and ultrasonic vibration, can also be used to measure the water stability of soil aggregates. Bronick and Lal (2005) reported that the aggregate stability results varied significantly between the methods since each method mimicked a different process and the energy applied by the methods was either arbitrary or nonexistent in the field. For example, ultrasonic vibration does not correlate with field soil behavior. Wet sieving amplifies slaking effects (Collis-George, 1979), whereas rainfall simulation focuses on mechanical raindrop impact forces (An et al., 2021). The wet sieving method is a viable option because of its ability to provide detailed information on the

size and quantity of fragments released from soil aggregates. In addition, the results of the other methods are highly dependent on the aggregate size used. Soil is made up of many aggregates of varying sizes, which are subjected to erosion and mechanical pressures. When describing soil structure, it is important to consider both aggregate size and stability. Determining aggregate stability on the basis of a certain size does not accurately reflect the overall stability of the soil. To determine aggregate stability, all aggregate sizes and soil quantities under investigation should be considered. The soil aggregate size distribution is typically assessed via dry or wet sieving methods. Dry sieving methods are employed to assess the stability of soil aggregates in relation to their resistance to wind erosion and mechanical disturbances, such as those caused by tillage operations (Weidhuner et al., 2021). Kemper and Rosenau (1986) created a rotary sieve that could measure differences in the size distribution of dry aggregates by separating samples of broken soil into different-sized dry aggregates (Figure 2.1A). The aggregate size fractions are separated via sieves of various sizes. The mesh sizes of the sieves are placed vertically, with the largest sieves at the top and the lowest at the bottom. The setup was placed on a rotary sieve device to sieve the dried samples for approximately three minutes. Following sieving, the soil in each sieve is weighed, and the proportion of each aggregate size class is calculated. Wet sieving involves repeatedly immersing and sieving soil (Figure 2.1B), simulating the natural pressures involved in water absorption into soil aggregates (Ikenna Ekwue et al., 2018). Wetting soil can be performed by subjecting it to various treatments, including slow wetting, fast wetting, and stirring after prewetting (Lado et al., 2004; Ruiz-Vera and Wu, 2006). The results from wet sieving are usually expressed as the mean weight diameter (MWD), the percentage of stable water aggregates (WSA), or the geometric mean diameter (GMD). These metrics are commonly known as stability parameters or indices.

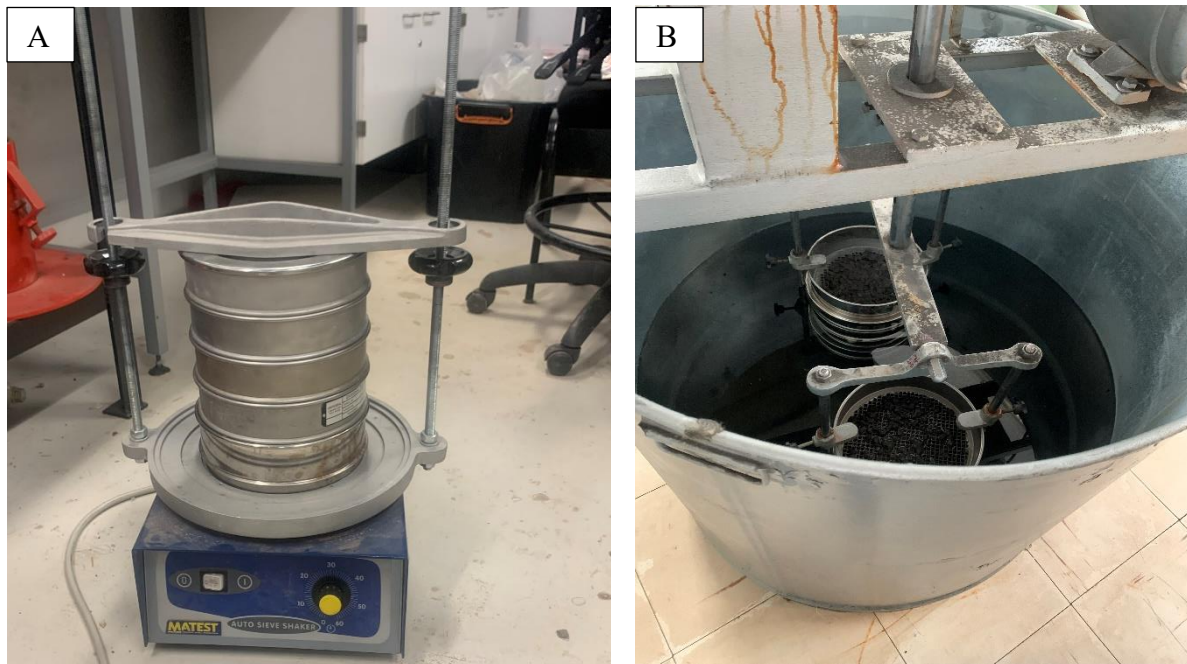


Figure 2.1 A; Dry aggregate stability apparatus, B; Wet aggregate apparatus

2.3. Ecological importance of soil aggregate stability

Soil aggregate stability is a critical feature that influences soil production and resistance to erosion and degradation (Mikha et al., 2015). It is a highly complicated parameter that influences a wide range of soil qualities, including carbon sequestration, soil porosity, water infiltration, aeration, compatibility, water retention, hydraulic conductivity, and resistance to erosion by water (An et al., 2021; Papuga et al., 2022; Rawls et al., 2004). As a result, retaining high aggregate stability is critical for preserving soil productivity, preventing soil erosion and degradation, and, ultimately, reducing environmental pollution. Aggregate stability plays a crucial role in the physical protection of organic matter from microbial degradation, thereby facilitating the accumulation of organic carbon within aggregates and ultimately enhancing carbon sequestration (Noormets et al., 2014). The level of physical protection of the SOM within the aggregates typically determines SOC sequestration in the soil. Physical protection refers to the entrapment of organic particles inside macro- and microaggregates. The

encapsulation of soil organic matter in the center of microaggregates is the key mechanism for sequestering SOC (Kumar et al., 2013). This process protects organic materials against both physical and chemical breakdown by microbial activities, while also sequestering soil organic carbon. However, the disruption of aggregation leads to faster mineralization of SOM and preferential loss of readily available carbon. Thus, increased protection of SOM by aggregates in less disturbed soils leads to more labile carbon (C) accumulation than in disturbed soils. Moreover, aggregate stability also plays a significant role in nutrient cycling and biological activity within soil ecosystems (Paul et al., 2013). Soil aggregates ensure efficient nutrient availability for plants. The soil organic matter that is stored in stable aggregates acts as a reservoir for essential nutrients such as nitrogen, phosphorus, and potassium. These nutrients that are bound within the organic matter are gradually released through microbial decomposition, ensuring a steady supply for plant uptake (Gupta and Germida, 2015). In addition, stable aggregates provide an excellent habitat for microorganisms, hence supporting extremely active biological and metabolic activities, principally supported by extracellular enzymes. In general, beneficial microorganisms contribute to the decomposition of organic matter and release essential nutrients essential for plant growth. Aggregate stability is also important for water retention (Otalvaro et al., 2016). The soil's ability to hold water is usually determined by its pore structure, which is influenced by the arrangement and size distribution of soil aggregates. Consequently, understanding the morphological characteristics of soil structure is crucial for assessing soil hydraulic properties. The distribution of soil aggregates plays a crucial role in governing soil water retention. Bonsu and Tuffour (2014) conducted a study on the effect of soil aggregate size distribution on water retention and reported that aggregates in the sizes of 1-2 mm and 0.5-1 mm had lower water retention rates than larger fractions did, and that water retention decreased with decreasing aggregate size. This suggests

that larger aggregates have greater water retention capacity because of their ability to create larger pores and provide more space for water storage within the soil structure.

2.4. Soil aggregate stability–soil carbon sequestration nexus

The relationship between aggregate stability and soil carbon sequestration is well documented, with aggregate stability playing a critical role in the ability of soil to sequester carbon. Numerous studies have established the connection between these two phenomena, demonstrating that soil aggregates are fundamental for carbon sequestration through various mechanisms (González-Rosado et al., 2023; Villasica et al., 2018). Soil aggregates are classified into various size categories, such as macroaggregates, microaggregates, and silt–clay aggregates. Each aggregate type plays a distinct role in carbon sequestration due to differences in their stability and the protection they offer to soil organic carbon (SOC) from decomposition. Soil organic carbon (SOC) sequestration is influenced by the physical, chemical, and physiochemical stabilization of soil organic matter (SOM) within aggregates (Lal, 2010). Physical protection involves the entrapment of organic particles in both macro- and microaggregates, whereas chemical stabilization results from the binding of SOM to soil minerals such as clay and silt. Physiochemical interactions define the maximal SOC sequestration capability of soil. Stable soil aggregates encapsulate SOM, providing physical protection that slows decomposition by reducing exposure to microorganisms and environmental factors (Chen et al., 2022). This encapsulation slows decomposition, allowing carbon to persist in the soil longer.

The turnover rate of SOC increases from microaggregates to macroaggregates, suggesting that microaggregates provide greater physical protection for SOC and thus facilitate carbon sequestration more effectively (Šimanský and Bajčan, 2014). Macroaggregates, while promoting SOC storage, are more transient than microaggregates because of their faster

turnover rate. The dynamics of macroaggregates are crucial, as they influence the formation of microaggregates and the stabilization of carbon within these microaggregates. Therefore, rapid turnover of macroaggregates may hinder the formation of microaggregates and the stabilization of carbon within them. Many studies have reported a positive impact of soil aggregation on SOM accumulation (An et al., 2010a, 2010b; Shein & Milanovskii, 2003). However, land use practices such as cultivation disrupt these aggregate structures, leading to the release of carbon and an increase in the availability of carbon. Specifically, cultivation reduces carbon-rich macroaggregates while increasing carbon-depleted microaggregates.

CHAPTER THREE

MATERIALS AND METHODS

3.1. Geographical setting of the study area

The specific study area is the Pandamatenga region, which is located in the northeast fringe of Botswana between latitudes 18.30° S and 18.69° S and longitudes 25.45° E to 25.61° E at elevations between 1066 m and 1088 m (Figure 3.1).

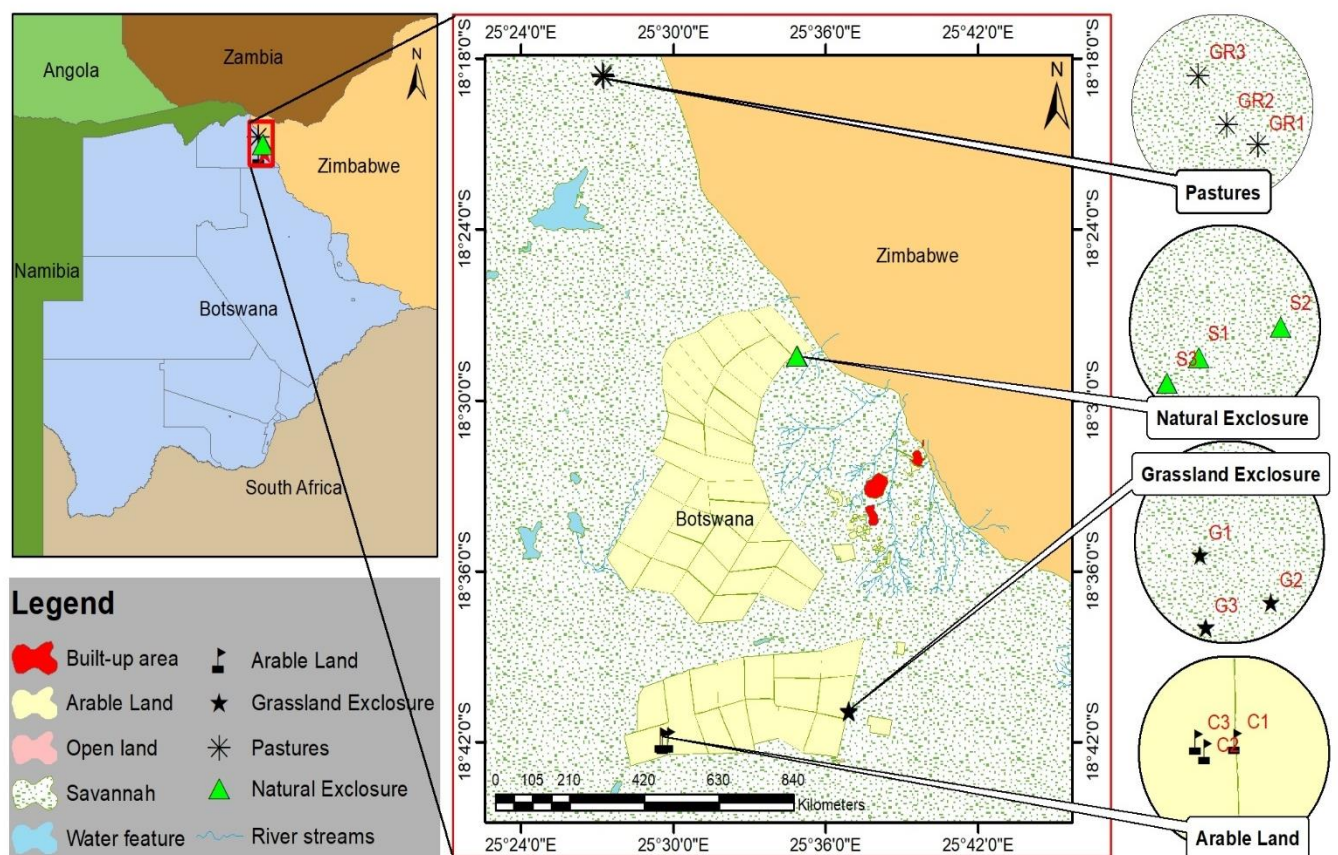


Figure 3.1 Map of the study location

3.2. Site description

3.2.1 Climate

The Pandamatenga region experiences a semiarid climate characterized by hot, humid summers and mild, dry winters. The study area receives an average annual rainfall of approximately 600 mm, predominantly from convective storms. However, rainfall totals vary considerably across short distances, ranging from 394 mm to 1,050 mm annually. Most precipitation occurs from October to April, with the peak months being December, January, and February. This rainfall often results in short, intense storms that cause rapid flooding on some farms due to high run-off. The maximum temperatures range from 26°C to 34°C between October and March, whereas the minimum temperatures range from 11°C to 20°C from November to July. The soil climate of Pandamatenga is characterized by a ustic moisture regime and an iso-hyperthermic temperature regime, with an aqueous moisture regime present in the poorly drained parts of the alluvial plain.

3.2.2 Soil and geology

The soils that predominate in the Pandamantenga alluvial plains are classified as Eutric Vertisols according to the FAO World Reference Base for Soil Resources (WRB, 2015). These soils exhibit textures ranging from sandy clay loam to clay, with particle size analysis revealing average contents of 31–71% clay, 2–18% silt, and 20–64% sand across all land uses (Table 4.5). Vertisols are characterized by the presence of vertic horizons—clayey subsurface layers with polished and grooved ped surfaces (“slickensides”), or wedge-shaped/parallelepiped aggregates—resulting from significant shrink–swell behavior.

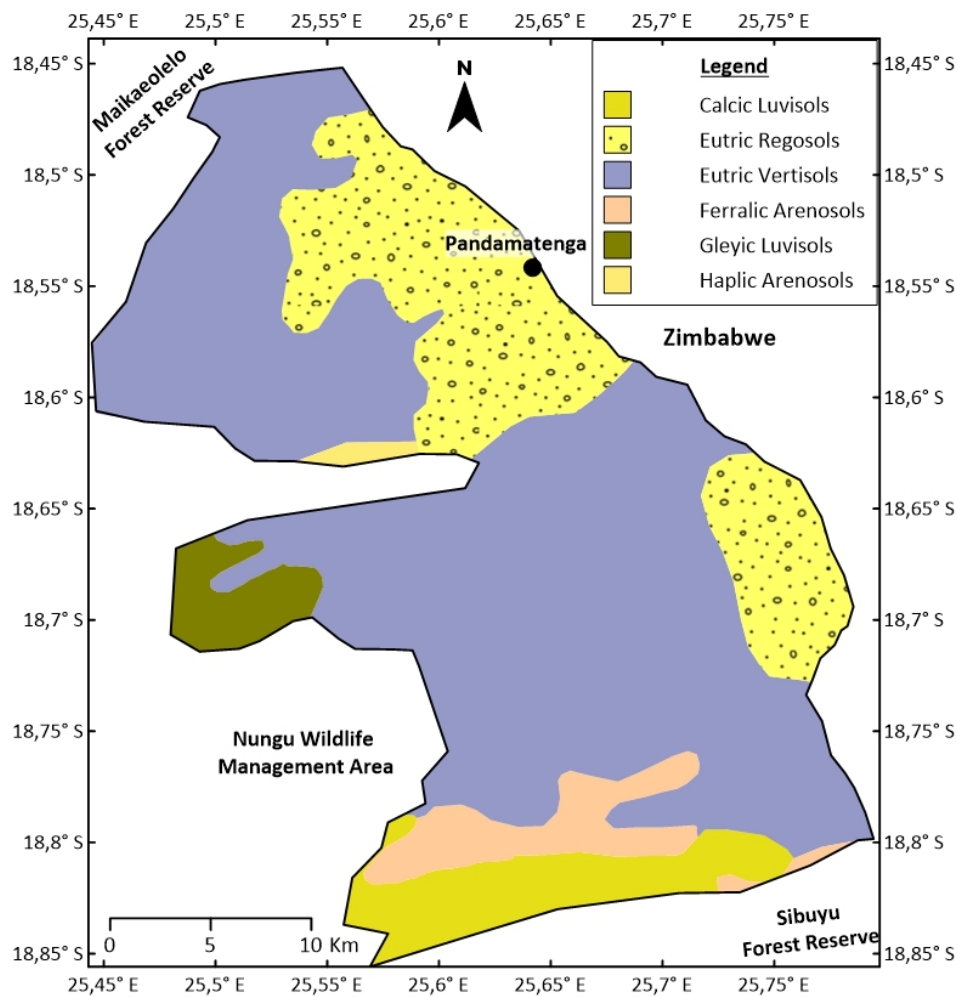


Figure 3.2. Soil units in the Pandamatenga alluvial Plain, adapted from the Food er agriculture organization of the United Nations soil mapping and advisory services Botswana the soils of the Mpandamatenga Plains united nations development programme field document12 republic of Botswana, (1990)

The Pandamatenga alluvial Plains are underlain by basalt from the Stormberg lava stage of the Karoo Age, which is occasionally interspersed with sandstone of the same age. These basalts are exposed primarily around Pandamatenga village and extend eastwards into Zimbabwe. The plains are bordered by superficial Kalahari sand deposits, some of which form east–west oriented longitudinal dunes. These dunes appeared to have been deposited before the formation of the lake and were later submerged and eroded. The area is very flat with a gentle slope, and rainwater flows following natural drainage routes.

3.2.3 Vegetation

The vegetation in the study area is characterized by extensive grassland savanna interspersed with mopane (*Colophospermum mopane sp*) and various acacia species.

3.2.4 Land use

The study area encompasses a range of land uses that reflect its ecological and socioeconomic significance. The dominant land uses include agriculture, particularly pastoral and arable farming, which are integral to the livelihoods of the local population. In addition to agricultural activities, the landscape features natural exclosures and extensive grasslands, which contribute to the region's biodiversity.

3.3. Field sampling

The soil organic carbon (SOC) stocks and aggregate stability induced by different land use types in the Vertisols of Pandamatenga were investigated by collecting soil samples from four land use types (Figure 3.3). To ensure that the land use types were in the same soil forms, a reconnaissance soil survey was conducted prior to sampling. Land use types were also surveyed, and field management practices were assessed through interviews with the farmers.

Soil samples were collected at three depths (0–15, 15–30 and 30–45 cm) via an auger. The samples were stored in polythene sampling bags. For each land use type, the first sampling point was randomly selected, and the subsequent two points were systematically placed 1 meter apart. At each point, soil samples were collected at three depths (Table 3.1). At each depth, the soil samples were collected three times and mixed to create composite samples. This resulted in nine soil samples per land use type and a total of 36 samples for all land use types among the disturbed bulk soil samples. Core samples were also collected at each land-use type for determination of bulk density. For these measurements, three core samples were collected at

each sampling point, yielding a total of 27 core samples per land use type and a total of 108 core samples across all land use types.

In total, 36 bulk soil samples and 108 core samples were collected.

For this study, four specific land uses were selected: arable land (AL), pasture (PA), natural enclosure (NE), and grassland enclosure (GE).

Table 3.1 Key characteristics of the selected land uses.

Sampling point	Land use	Coordinates	Elevation (m)	Crop type	Land management
AL1	AL	S 18.69845 E 25.49689	1071	Sun Hemp	30 years of crop rotation; sun hemp, maize, and millet
AL2	AL	S 18.69945 E 25.49353	1071	Sun hemp	30 years of crop rotation; sun hemp, maize, and millet
AL3	AL	S 18.69851 E 25.49249	1075	Millet	30 years of crop rotation; sun hemp, maize, and millet
GE1	GE	S 18.68153 E 25.61514	1082	None	None
GE2	GE	S 18.68190 E 25.61576	1088	None	None
GE3	GE	S 18.68210 E 25.61519	1080	None	None
PA1	PA	S 18.31065 E 25.45451	1067	None	Grazing
PA2	PA	S 18.31039 E 25.45405	1066	None	Grazing
PA3	PA	S 18.30975 E 25.45363	1070	None	Grazing
NE1	NE	S 18.47415 E 25.58123	1072	None	Natural succession
NE2	NE	S 18.47403 E 25.58159	1071	None	Natural succession
NE3	NE	S 18.47425 E 25.58109	1071	None	Natural succession

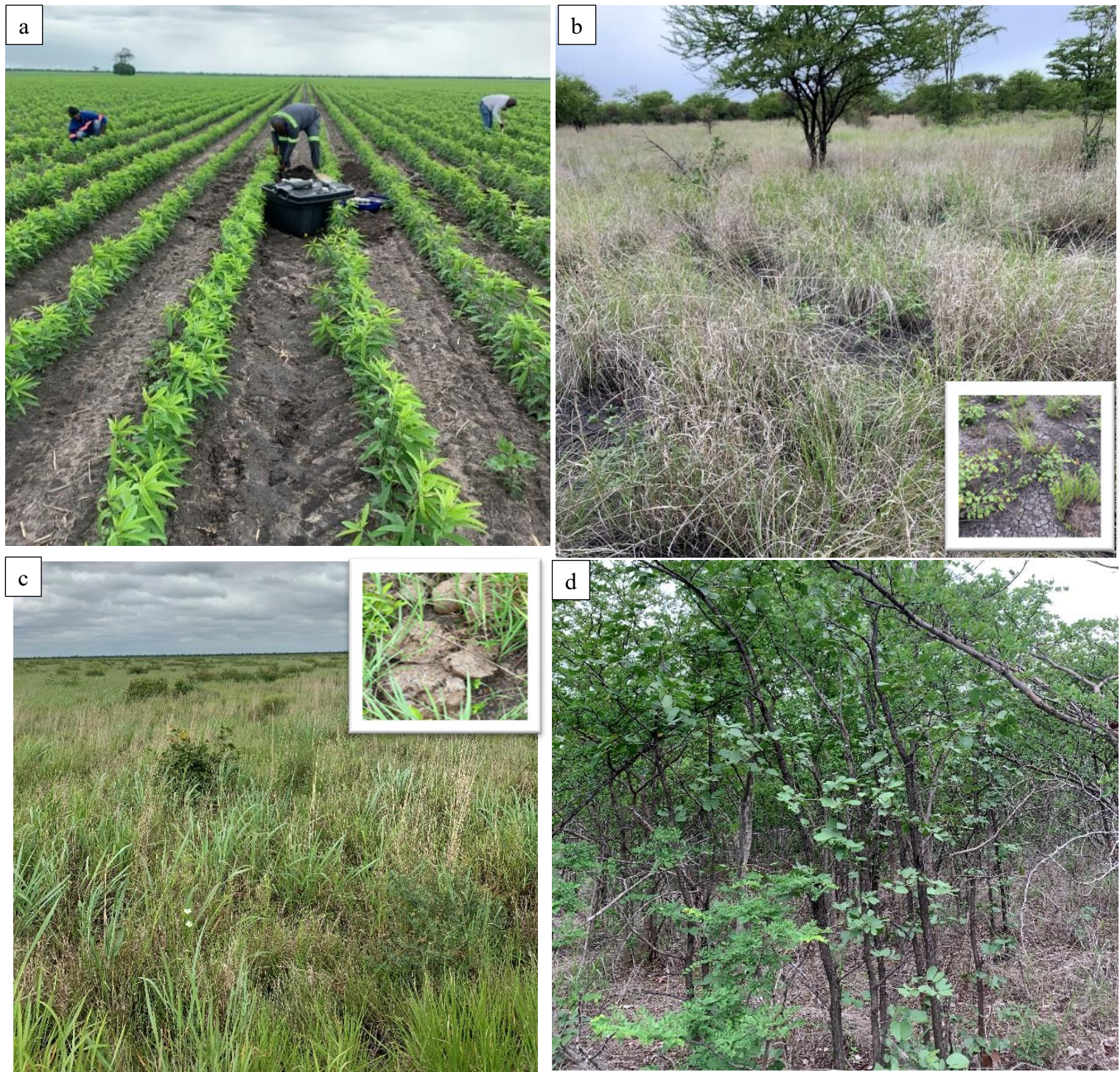


Figure 3.3 The four land use types: a) Arable land (AL); prepared using tillage practices) grassland enclosure (GE); These ecosystems are fenced and they experience naturally occurring wildfires almost every year, c) Pastures (PA); Areas are designated for livestock and wild animals to graze on the vegetation, d) Natural enclosure (NE); These areas are protected from wild animals and anthropogenic activities, the ecosystem are cauterized by the prevalence of shrubs.

3.4. Laboratory methods

Prior to conducting the soil physicochemical analysis, excluding bulk density, the soil samples were ground and sieved through a 2 mm mesh. Given that the samples were collected during rainy conditions, the moist soil was air-dried.

3.4.1 Bulk density

The bulk density of the soil was determined via the core sampling method (Shammary et al., 2018). A core sampler facilitated by a wooden block was employed to extract the soil samples. Care was taken during the extraction process to minimize compaction or loss of soil from the core. After removal, any excess soil was carefully trimmed away via a flat bladed knife. The extracted sample was then placed into a labelled polyethylene sampling bag. To ascertain the precise volume of the core, three evenly spaced measurements were taken and averaged. The exact height of the core was determined for accurate measurement of the soil volume. These samples were then dried at 105 degrees Celsius for 24 hours, and the mass of the dry soil was recorded. The total soil volume was estimated as the internal volume of the cylinder, and the bulk density was calculated by dividing the weight of the dry soil by the total soil volume.

3.4.2 Particle size distribution

The particle size distribution was determined via the hydrometer method, where no treatments were applied. Dispersing solution, 5%; 50 g of sodium hexametaphosphate $\text{Na}_6(\text{PO}_3)_6$ was dissolved in distilled water and diluted to 1 L. Since the number of soil samples was 36, 250 g of sodium hexametaphosphate was weighed and transferred to a 100 ml beaker, it was then dissolved in distilled water, transferred to a 5 L conical flask and filled to the mark.

For the blank solution, a 100 mL aliquot of a 5% dispersing solution was mixed with 880 mL of distilled water in a 1000 mL sedimentation cylinder. Notably, the blank was not diluted to

1000 mL because of the 20 mL displacement caused by 50 g of soil. First, 50 g of previously sieved soil was precisely weighed via an analytical balance and transferred into a beaker. Subsequently, 100 mL of a 5% dispersing solution was added to the soil, which was allowed to settle overnight. The resulting mixture was then transferred to a dispersing cup, affixed to the mixture, and agitated for 2 minutes. The suspension was quantitatively transferred from the dispersing cup into a 1000 mL cylinder, and distilled water was added to reach the 1000 mL mark. Following thorough mixing, a hydrometer was gently inserted into the suspension, with readings taken at 40 second intervals along with temperature measurements, which included the amount of silt plus clay. After a 2-hour interval, the hydrometer readings along with the temperature were recorded again. This parameter represents the amount of clay in a suspension. The same procedure was applied to all 72 samples. A textural triangle was used to determine the soil texture of the soil sample on the basis of the percentages of clay, silt and sand.

$$1) \% (\text{silt} + \text{clay}) = (4 \text{ s reading/mass of soil sample}) \times 100$$

$$2) \% (\text{clay}) = (6 \text{ hours reading/mass of the soil sample}) \times 100$$

$$\% (\text{silt}) = \text{equation (1)} - \text{equation (2)}$$

$$3) \% (\text{sand}) = 100 - (\% \text{ silt} + \% \text{ clay})$$

3.4.3 Wet aggregate stability

The size distribution of water-stable aggregates (WSA) was determined via the wet sieving method, as described by Yoder (1936). A set of sieves with mesh sizes of 4.75, 2.0, 1.0, 0.5 and 0.25 were employed. A 100 g sample of aggregates, between 5–10 mm in size, was placed in the topmost sieve (4.75 mm). The water level in the container was carefully adjusted to reach the base of the sieve. To prevent the aggregates from breaking down due to sudden air release when immersed in water, they were allowed to saturate slowly by capillary action. After the

soil was allowed to saturate for 5 minutes, the sieve stack with the pre-saturated soil sample in the top sieve was shaken in water for 10 minutes at a rate of 30 vertical oscillations per minute. The soil retained on each sieve was then corrected for sand content and dried at 105°C for 36 hours, and the proportion of aggregates in each size class was calculated as a percentage. The percentage ratio of aggregates in each sieve represents the WSA in the following size classes: < 4.75 mm, 4.75–2.00 mm, 2.00–1.00 mm, 1.00–0.50 mm, 0.50–0.25 mm, and < 0.25 mm.

Water-stable aggregates (WSAs) larger than 0.25 mm were classified as macroaggregates, whereas those smaller than 0.25 mm were identified as microaggregates. The weights of the different sizes of water-stable aggregates were used to compute parameters such as the mean weight diameter (MWD), geometric mean diameter (GMD), proportion of aggregates larger than 0.25 mm ($R_{0.25}$). The WSA is computed as follows:

$$WSA = [(W_a - W_c)/W_o] \times 100$$

where WSA is the water-stable aggregate for each size class, W_a is the weight of the material on the sieve after wet sieving, W_c is the weight of the coarse material in size, and W_o is the weight of the aggregates placed on the sieve prior to wet sieving.

$$MWD = \sum_{i=1}^n X_i \cdot W_i$$

where X_i is the mean diameter of the i th sieve size and W_i is the proportion of the total aggregates in the i th fraction. If the value of the MWD is high, the proportions of macroaggregate and microaggregate fractions are greater in the soil.

$$R_{0.25} = m_{a>0.25}/M$$

where $m_{a>0.25}$ is the mass of aggregates with particle sizes greater than 0.25 mm and where M is the total weight of the sample used for the analysis.

$$GMD = \exp\left(\frac{\sum_{i=1}^n w_i \cdot \log x_i}{\sum_{i=1}^n W_i}\right)$$

where W_i is the weight of the aggregates present in a diameter class, X_i and $\sum_{i=1}^n$, and W_i is the total weight of the experimental soil sample.

3.4.4 Soil pH and electrical conductivity (EC)

Calibration buffer preparation

Thirty milliliters of pH 4.0, 7.0 buffer solution and electrical conductivity standard solution of 1400 $\mu\text{S}/\text{cm}$ were poured separately into 50 mL beakers and covered with parafilm prior to calibration. The beakers were then labelled. Another 30 mL of each pH buffer solution was poured into three separate 50 mL beakers, the beakers were then labelled, and these beakers were used to rinse the electrode during calibration. The buffers were then allowed to reach the same temperature.

Calibration procedure

After the buffers were allowed to reach the same temperature, which was normally 25°C, the electrode with multiple parameters (pH and EC) was rinsed with distilled water and then rinsed in a pH 4.0 buffer solution rinse beaker. The multiparameter electrode tip was fully immersed in a pH 4.0 buffer solution. Calibration on the meter was started, and a stable reading of the pH 4.0 buffer solution was obtained for two minutes. After a stable reading was obtained, the electrode was removed, rinsed with distilled water and then rinsed in a pH 7.0 buffer solution rinse beaker. The electrode tip was immersed in a pH 7.0 buffer solution. A stable reading was allowed for two minutes. After the stable reading appeared, the electrode was removed, rinsed

with distilled water and then rinsed with an electrical conductivity standard solution of 1400 $\mu\text{S}/\text{cm}$ in the rinse beaker. The electrode tip was fully dipped into a standard electrical conductivity solution of 1400 $\mu\text{S}/\text{cm}$, and after a stable reading appeared, it was removed and rinsed with distilled water.

The soil pH and EC measurements followed a standardized procedure (FAO, 2021), whereby 10 grams of sieved soil sample were carefully weighed and transferred into 50 mL polyethylene bottles. Subsequently, 25 mL of water was added to the soil sample to achieve a ratio of 1:2.5 for the soil–water suspension. The mixture was agitated for 30 minutes on an orbital shaker, followed by a 30-minute settling period. The multiparameter electrode was immersed in the soil–water suspension, and the pH and EC readings were recorded once stabilization occurred. Prior to analysing subsequent soil samples, the electrode was rinsed with distilled water to prevent contamination. This process was repeated for all 72 plastic bottles containing soil samples.

3.4.5 Carbonate content

The carbonate content was determined via a calcimeter. The calcimeter works in accordance with the method of Scheibler (M-0853E, 2021), which involves determining the carbonate content in the soil via a volumetric method.

Blank determinations and calibration

For every series of measurements, two blank determinations and two determinations of 0.2 g and 0.4 g CaCO_3 for calibration were executed. For blank determinations, two reaction vessels were filled with 20 mL of water each. For calibration, two samples of 0.2 g and 0.4 g of CaCO_3 were weighed, the quantities were transferred to the reaction vessels, and 20 mL of distilled water was added to each sample. Subsequently, 7 mL of hydrochloric acid was dispensed into small test tubes, which were then placed into the reaction vessels of the blank determinations

and those containing calcium carbonate using tweezers. The determination of the blanks commenced with starting values of 20 and 80 mL, whereas the determination for calibration commenced with the starting level of 3 mL. This is set using the buffer vessels. The cock was turned down to the measuring position. The reaction vessels were then tilted so that the hydrochloric acid could flow over and initiate the reaction. The reaction vessels were shaken to allow the reaction to proceed to near completion as much as possible. The gas generated during the reaction caused fluctuations in the fluid levels within the burette and buffer vessel. Therefore, the buffer vessels were adjusted to align with the level indicated on the burette. The readings were then recorded once stability was achieved.

For the analysis of the soil samples, an initial step involved treating 2 g of each sample on Petri dishes with approximately 1 mL of hydrochloric acid. The carbonate content was estimated on the basis of the duration of bubbling observed during this process, which in turn guided the determination of sample quantities required for analysis. To determine the carbonate content of the soil samples, the appropriate quantity of prepared samples was weighed and transferred into reaction vessels, followed by the addition of 20 mL of water. These reaction vessels were subsequently positioned on the calcimeter. Small test tubes were filled with 7 mL of hydrochloric acid and carefully inserted into the reaction vessels via tweezers.

Tilting the reaction vessels facilitated the flow of hydrochloric acid into the soil samples. Concurrently, adjustments were made to the buffer vessels to align them with the level indicated on the burette. The readings were then recorded once the volume had stabilized.

3.4.6 Exchangeable bases

Exchangeable bases (Ca, Mg, Na and K) were determined following a standard extraction procedure (FAO,2022). Five grams of soil was weighed into 50 mL extraction tubes. For each of the 72 samples, a blank and a reference sample were included to ensure quality control. The

soil samples were treated with 33 mL of 1 N ammonium acetate solution adjusted to pH 7.0. The mixture was shaken for 15 minutes using a reciprocal shaker to facilitate the exchange reaction. The samples were subsequently centrifuged, and the supernatant was added to a clean 100 mL volumetric flask via quantitative fine porous filter paper.

Steps involving the addition of ammonium acetate, shaking, centrifuging and filtering were repeated twice to ensure complete extraction of the exchangeable bases, resulting in a total volume of approximately 100 mL. The volumetric flask was then filled to the 100 mL mark with ammonium acetate, and the contents were mixed thoroughly. The final extract was analysed for calcium (Ca), magnesium (Mg), sodium (Na) and potassium (K) concentrations via inductively coupled plasma optical emission spectroscopy (ICP–OES).

3.4.7 Soil organic carbon

The content of soil organic carbon was determined via the Walkley black method. First, the reagent and titrant solutions were prepared as follows:

- Potassium dichromate solution (1 N): Forty-nine grams of dry potassium dichromate solution was dissolved in distilled water via a magnetic stirrer. The solution was then transferred to a 1.5 L volumetric flask and diluted to a final volume of 1 L with distilled water.
- Ferrous ammonium sulfate solution (0.5 M): A total of 196 g of ferrous ammonium sulfate solution was dissolved in distilled water. Subsequently, 20 mL of concentrated sulfuric acid was added to the solution, which was then mixed thoroughly. The mixture was transferred to a 1.5 L volumetric flask and diluted to a final volume of 1 L with distilled water.

- O-Phenanthroline-Ferrous Indicator: O-phenanthroline monohydrate (1.485 g, analytical grade) and ferrous sulfate heptahydrate (0.695 g, analytical grade) were dissolved in distilled water. The solution was then diluted to a final volume of 100 mL.

In the experimental procedure (FAO, 2020), one gram of air-dried and sieved soil sample was added to a 250 mL conical flask, and then 10 mL of 1 N potassium dichromate was pipetted and added to the flask containing the soil sample, followed by gentle swirling. Twenty milliliters of concentrated sulfuric acid was carefully added to the flask, followed by gentle swirling to homogenize the mixture. The mixture was allowed to cool for 30 minutes. Afterward, 100 mL of distilled water was added to the solution and thoroughly mixed. Three drops of o-phenanthroline-ferrous indicator were then added to the solution. The soil samples were titrated with a 0.5 M ferrous ammonium sulfate solution. The endpoint was determined by the transition of color, from a greenish cast to dark green, then to blue, and finally to red. To standardize the 0.5 M ferrous ammonium sulfate solution, two blanks (without a soil sample) were subjected to the same procedure.

3.4.8 Clay mineralogy

Clay mineralogy analysis was conducted via X-ray diffraction (XRD) with a Bruker-AXS D8 Advance powder X-ray Diffractometer. A 5 g soil sample was dispersed in sodium hexametaphosphate. The organic matter was removed via hydrogen peroxide, followed by the removal of carbonates via hydrochloric acid. The samples were then thoroughly washed, and the clay fraction (<0.002 mm) was isolated through repeated centrifugation and decantation. XRD patterns were obtained for smeared slides prepared from the clay fraction and scanned from 5° to 70° 2θ with an increment of 0.0205°. The clay mineralogical patterns were obtained for air-dried samples only. The peaks were used to identify the specific minerals present in each sample.

3.5. Estimation of soil organic carbon stocks

The following equations were used to compute SOC stocks of bulk soil samples and aggregate size fractions (FAO, 2020)

SOC stocks in bulk soils;

$$SOC_{stocks} = SOC\% \times BD \times D \times 10$$

Stocks of organic carbon in aggregate size fractions

$$M_i = \frac{D \times BD \times W_i}{10}$$

$$Stock_{OC_i} = M_i \times OC_i$$

where:

D=depth

BD= Bulk density

M_i = Amount of soil in the ith size fraction (kg/m²)

OC_i= OC content of the ith size fraction (%)

W_i = proportion of the total soil in the ith size fraction (%)

3.6. Estimation of the contribution rate of different aggregate sizes

The contributions of different factors to total SOC under different land uses were evaluated via the following equation;

$$C_{a_i} = \frac{w_{ai} \times SOC_{a_i}}{w_b \times SOC_b} \times 100\%$$

where:

W_{ai} = mass weight of aggregate i

SOC_{ai} = organic carbon in aggregate i

W_b = mass weight of the bulk soil

SOC_b = soil organic carbon in bulk soil

C_{ai} = contribution of aggregate i to soil organic carbon.

3.7. Statistical analysis

Microsoft Excel computer software was used to organize the data. Descriptive statistics, including the calculation of means and standard deviations for the soil physicochemical properties, were performed and visualized via RStudio 4.4.1 (R Core Team, 2021).. The distributions of soil aggregates of different sizes; distributions of organic carbon within different aggregate sizes; MWD, GMD, and R0.25; soil organic carbon stocks in both bulk soil samples and aggregate sizes; and contribution rates of different aggregate sizes to soil organic carbon across various land uses and soil depths were compared through two-way variance analysis (ANOVA). When ANOVA indicated a significant effect, Tukey's post hoc test at the $P < 0.05$ significance level was applied for pairwise comparisons. Simple linear regression was used to explore the relationships between organic carbon stocks within different aggregate size classes and total (bulk) soil organic carbon stocks. Both ANOVA and regression analyses were

conducted and plotted via RStudio 4.4.1. Multivariate statistical analyses were performed with OriginPro 2024 software, which was also used to examine the relationships among the studied parameters. To evaluate potential sources of variation in the study, factor analysis was also carried out via OriginPro 2024 software. (Origin (Pro) (Version 2022b)).

CHAPTER FOUR

RESULTS

4.1. Morphological characteristics

Soil morphological properties, including color, structure, consistence, and reactions with HCL vary significantly across land uses and depths. The color matrix is primarily in the 10YR hue, with very dark hues (10YR 2/1) found in the natural enclosure (NE) under both dry and wet conditions (Table 4.4), as well as in the topsoil layers across different land uses. In contrast, light brown colors (10YR 5/1 to 10YR 6/1) are visible in deeper layers of arable land (AL2 and AL3) (Table 4.1). Furthermore, pasture soils (PA3) (Table 4.3) and grassland enclosure (GE3) (Table 4.2) soils generally have dark brown hues (10YR 4/1) in the deeper layers when wet.

The consistence of the soil changes with land usage and depth. Natural enclosure (NE) soils have extremely hard consistence when dry and very firm consistence when wet (Table 4.4). Similarly, grassland enclosures (GE) soils are very hard when dry yet firm when moist (Table 4.2). In arable land (AL) and pasture (PA), the soil was generally hard when dry and friable when moist at depths of 0–15 cm and 15–30 cm, respectively. However, at depths of approximately 30–45 cm, the soil was extremely hard when dry and firm when wet.

Massive structures were prominent in natural enclosures (NE), pastures (PA), and grassland enclosures (GE), particularly at depths of 30–45 cm. On the other hand, granular structures were more abundant on arable land and grassland enclosure, particularly at depths ranging from 0 to 15 cm. The hydrochloric acid (HCL) reaction intensities vary greatly, with arable land having a relatively strong HCL reaction (Table 4.1). Pasture and grassland enclosures had fewer weak reactions, whereas natural enclosures had no reactions with HCL (Table 4.4).

Table 4.1: Morphology of arable land soils (HA; hard, VHA; very hard, EHA; extremely hard, FR; friable, FI; firm, VFI; very friable)

Sampling point	Depth	Color matrix		Consistence		Structure	HCL content %
		Dry	Wet	Dry	Moist		
1	0-15	10YR 3/1 (Very dark gray)	10YR 3/1 (Very dark gray)	HA	FR	Granular	Moderate
	15-30	10YR 2/1 (Black)	10 YR/4/1 (Dark gray)	HA	FI	Granular	Weak
	30-45	10 YR 3/1 (Very dark gray)	10 YR 3/1 (Very dark gray)	VHA	FI	Granular	Weak
2	0-15	10YR 2/1 (Black)	10YR 2/1 (Black)	HA	FR	Granular	Moderate
	15-30	10YR 3/1 (Very dark gray)	10 YR 3/1 (Very dark gray)	HA	FI	Granular	Moderate
	30-45	10YR 4/1 (Dark gray)	10 YR/3/1 (Very dark gray)	VHA	FI	Granular	Strong
3	0-15	10YR 3/1 (Very dark gray)	10YR 3/1 (Very dark gray)	HA	FR	Granular	Strong
	15-30	10YR 5/1 (Gray)	10 YR/4/1 (Dark gray)	HA	FR	Granular	Strong
	30-45	10YR 6/1(Gray)	10YR 6/1(Gray)	VHA	FI	Granular	Strong

Table 4.2: Morphology of grassland enclosure soils (HA; hard, VHA; very hard, EHA; extremely hard, FR; friable, FI; firm, VFI; very friable)

Sampling point	Depth	Color matrix		Consistence		Structure	HCL content %
		Dry	Wet	Dry	Moist		
1	0-15	10YR 2/1 (Black)	10YR 2/1 (Black)	VHA	VFI	Granular	Weak
	15-30	10YR 3/1 (Very dark gray)	10YR 3/1 (Very dark gray)	VHA	VFI	Massive	Weak
	30-45	10 YR 3/1 (Very dark gray)	10 YR 3/1 (Very dark gray)	VHA	VFI	Massive	Weak
2	0-15	10YR 2/1 (Black)	10YR 2/1 (Black)	VHA	VFI	Granular	Weak
	15-30	10 YR 3/1 (Very dark gray)	10 YR 3/1 (Very dark gray)	VHA	VFI	Massive	Weak
	30-45	10 YR/3/1 (Very dark gray)	10 YR/3/1 (Very dark gray)	VHA	VFI	Massive	Weak
3	0-15	10YR 2/1 (Black)	10YR 2/1 (Black)	VHA	VFI	Granular	Weak
	15-30	10YR 3/1 (Very dark gray)	10YR 3/1 (Very dark gray)	VHA	VFI	Massive	Weak
	30-45	10 YR/4/1 (Dark gray)	10 YR/4/1 (Dark gray))	EHA	VFI	Massive	Weak

Table 4.3: Morphology of pasture soils (HA; hard, VHA; very hard, EHA; extremely hard, FR; friable, FI; firm, VFI; very friable)

Sampling point	Depth	Color matrix		Consistence		Structure	HCL content %
		Dry	Wet	Dry	Moist		
1	0-15	10YR 2/1	10YR 2/1	HA	FR	Granular	Weak
		(Black)	(Black)				
	15-30	10YR 3/1	10YR 3/1	HA	FI	Massive	Weak
		(Very dark gray)	(Very dark gray)				
	30-45	10 YR 3/1	10 YR 3/1	VHA	VFI	Massive	Weak
		(Very dark gray)	(Very dark gray)				
2	0-15	10YR 2/1	10YR 2/1	HA	FR	Granular	Weak
		(Black)	(Black)				
	15-30	10 YR 3/1	10 YR 3/1	HA	FR	Massive	Weak
		(Very dark gray)	(Very dark gray)				
	30-45	10 YR/3/1	10 YR/3/1	VHA	VFI	Massive	Weak
		(Very dark gray)	(Very dark gray)				
3	0-15	10YR 2/1	10YR 2/1	HA	FR	Granular	Weak
		(Black)	(Black)				
	15-30	10YR 3/1	10YR 3/1	HA	FR	Massive	Weak
		(Very dark gray)	(Very dark gray)				
	30-45	10YR 3/1	10 YR/4/1	VHA	VFI	Massive	Weak
		(Very dark gray)	(Dark gray)				

Table 4.4: Morphology of natural exclosure soils (HA; hard, VHA; very hard, EHA; extremely hard, FR; friable, FI; firm, VFI; very friable)

Sampling point	Depth	Color matrix		Consistence		Structure	HCL content %
		Dry	Wet	Dry	Moist		
1	0-15	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	15-30	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	30-45	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	2	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(black)				
2	15-30	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	30-45	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	3	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(black)				
3	15-30	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	30-45	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(Black)				
	3	10YR 2/1	10YR 2/1	EHA	VFI	Massive	None
		(Black)	(black)				

4.2. Soil physicochemical properties

The soil bulk density (BD) in pastures revealed an uneven tendency, with some land uses (such as grassland and natural exclosures) increasing with depth and arable land declining with depth. The highest BD value (1.3 g/cm^3) was found at the 0–15 cm depth in arable land, whereas the lowest value (0.8 g/cm^3) was found in grassland exclosures (Table 4.5).

The soil particle size distribution at three depths (0–15 cm, 15–30 cm, and 30–45 cm) are shown in Table 4.5. Arable land soils were sandy clay loam, whereas pastures, grassland exclosures, and natural exclosures were categorized as clay.

The electrical conductivity (EC) varied from 30 to 350 $\mu\text{S/cm}$. Across all land uses, EC increases with soil depth, except for natural exclosures (NE), where it decreases. The grassland exclosures (GE) presented the greatest EC value of 350 $\mu\text{S/cm}$ at the 30–45 cm depth, whereas NE presented the lowest EC values, ranging from 30 $\mu\text{S/cm}$ to 70 $\mu\text{S/cm}$ across all depths (Figure 4.1)

The pH varied greatly with land use and soil depth (Figure 4.2). The pH of arable land (AL) ranged from 7.2 to 7.8, with little variation among depths. The GE soils were slightly acidic, with pH values ranging from 6.8–7.5 and increasing with depth. Pastures (PAs) consistently had the highest pH values, ranging from 7.5 to 8.3, with the pH gradually increasing as depth increased. In contrast, the NE soils had the lowest pH levels, ranging from 5.6–6.2, with the greatest variation found in the topsoil layer (0–15 cm).

The calcium carbonate (CaCO_3) levels varied between land use types and soil depths, ranging from 0 g/kg in NE to 40 g/kg in AL. In AL, CaCO_3 concentrations increased with soil depth, from 20 g/kg at 0–15 cm to 40 g/kg at 30–45 cm. The GE soils presented consistently low

CaCO₃ values at all depths. The CaCO₃ levels in PAs increased gradually with depth, from 0 to 4 g/kg. NE soils presented no detectable CaCO₃ at any depth (Figure 4.3).

The soil organic carbon content varies depending on land use and soil depth. The organic carbon concentration in arable land decreased with increasing depth, from 0.9% to 0.6% (Figure 4.4). However, the grassland exclosures maintained a steady organic carbon concentration of 0.9% throughout all the soil depths. The soil organic carbon levels in pastures ranged between 0.7% and 0.9%. The natural exclosures presented the highest organic carbon concentrations, ranging from 1.00% at 30–45 cm to 1.7% at 0–15 cm.

Table 4.5 Soil physical properties across different land uses per soil depth

Land uses	depth	Silt%	Clay%	Sand%	Texture	BD (g/cm ³)
AL	0-15	4.67±1.03	31.01±0.89	64.32±1.37	Sandy clay loam	1.31
	15-30	2.67±1.03	33.55±3.23	63.79±4.18	Sandy clay loam	1.23
	30-45	4.00±0.00	38.85±6.87	57.15±6.87	Sandy clay	1.19
GE	0-15	10.67±2.07	67.95±2.42	21.39±1.63	Clay	0.69
	15-30	8.67±2.07	71.28±3.10	20.05±1.03	Clay	1.02
	30-45	11.33±1.03	70.95±3.44	23.72±7.97	Clay	1.25
PA	0-15	6.00±0.00	59.94±1.04	34.05±1.03	Clay	0.79
	15-30	6.00±1.79	61.95±1.03	32.05±2.73	Clay	0.73
	30-45	4.00±0.00	65.28±0.00	30.72±0.00	Clay	1.00
NE	0-15	18.00±3.58	43.28±8.94	38.72±7.80	Clay	0.93
	15-30	12.67±2.73	53.28±0.00	34.06±2.73	Clay	1.01
	30-45	14.00±1.79	53.95±1.03	32.06±2.08	Clay	1.14

BD; Bulk density. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

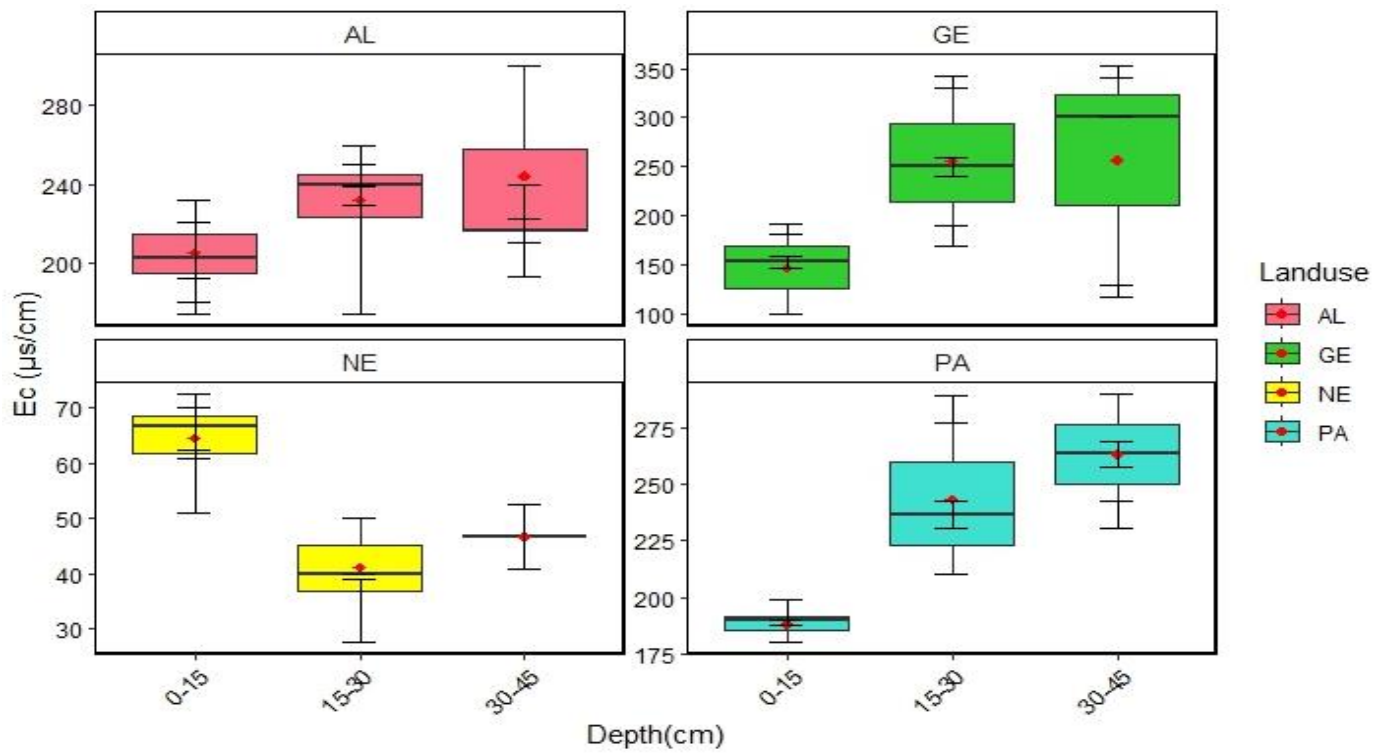


Figure 4.1 Soil electrical conductivity (EC) per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

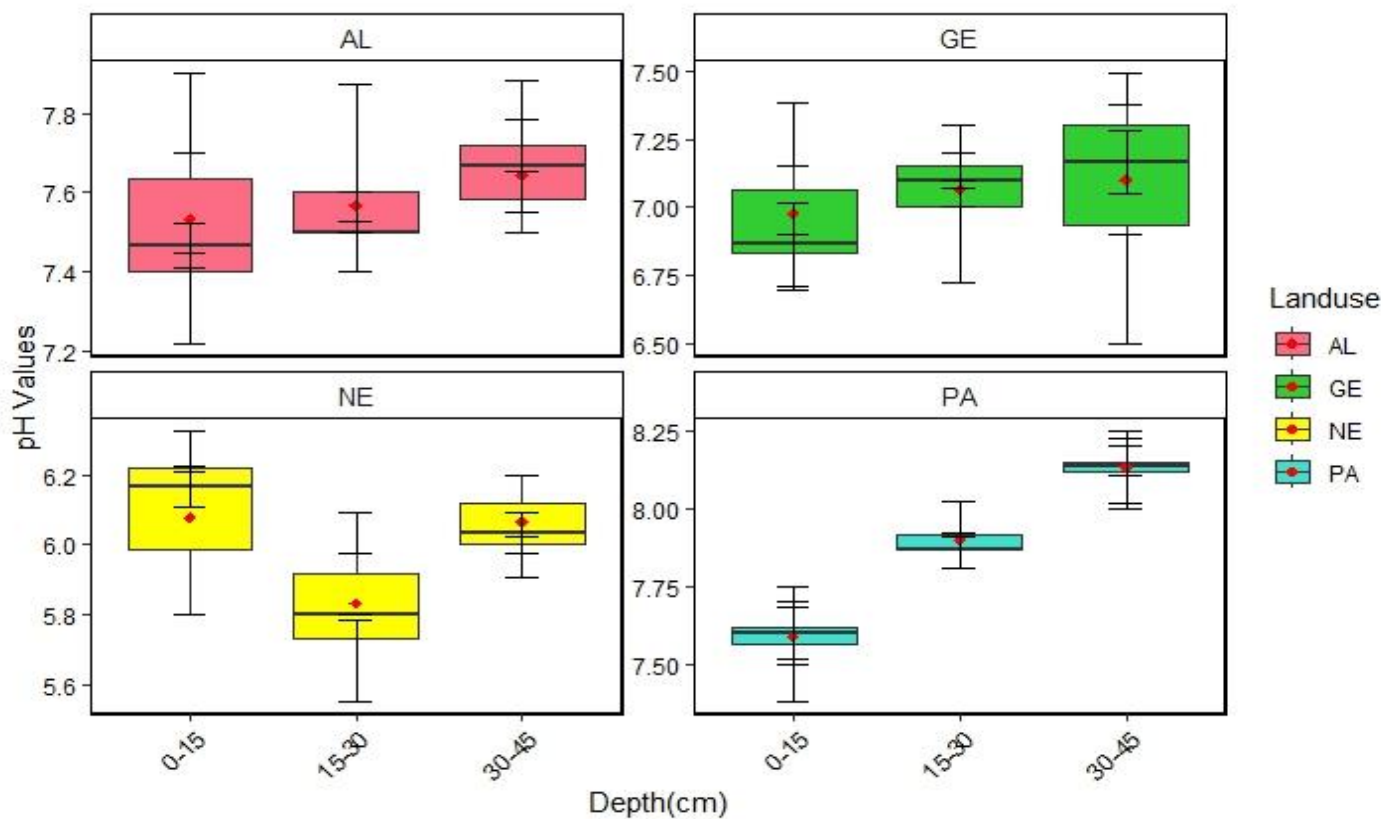


Figure 4.2 Soil pH per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

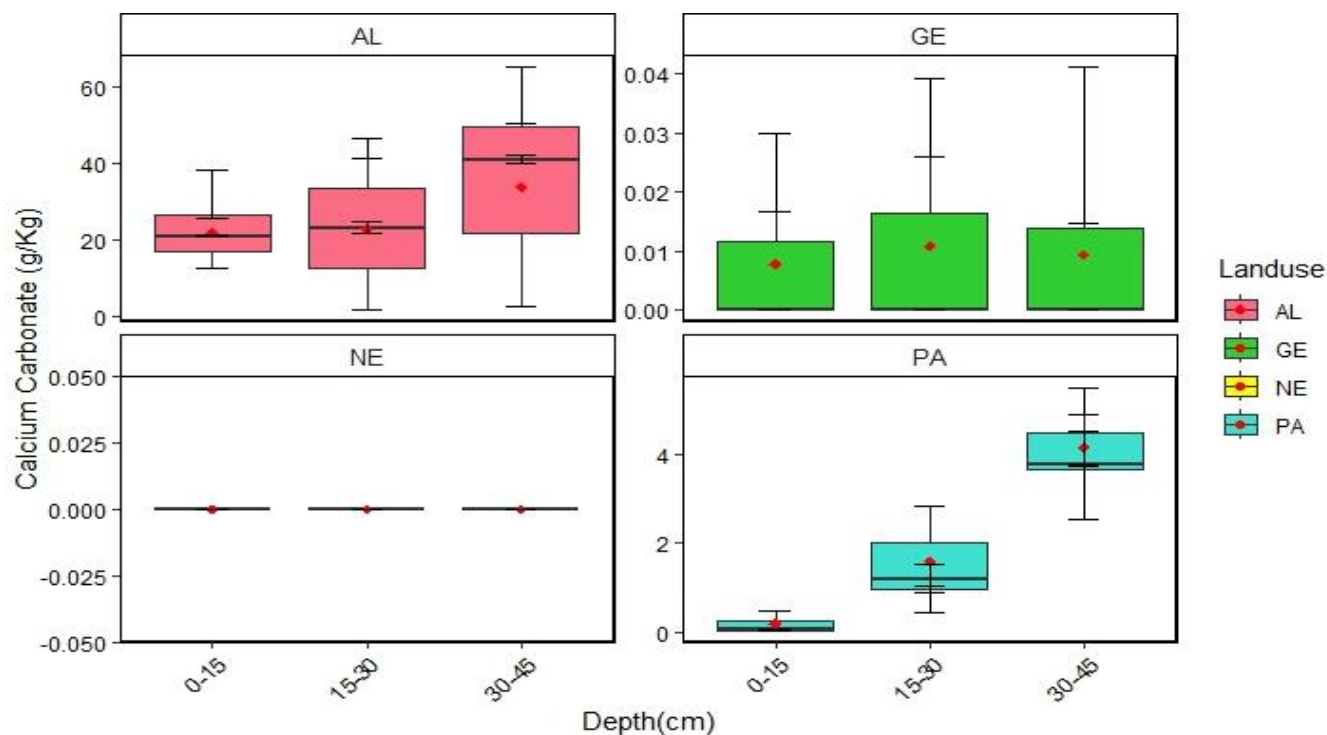


Figure 4.3 Calcium carbonate content per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

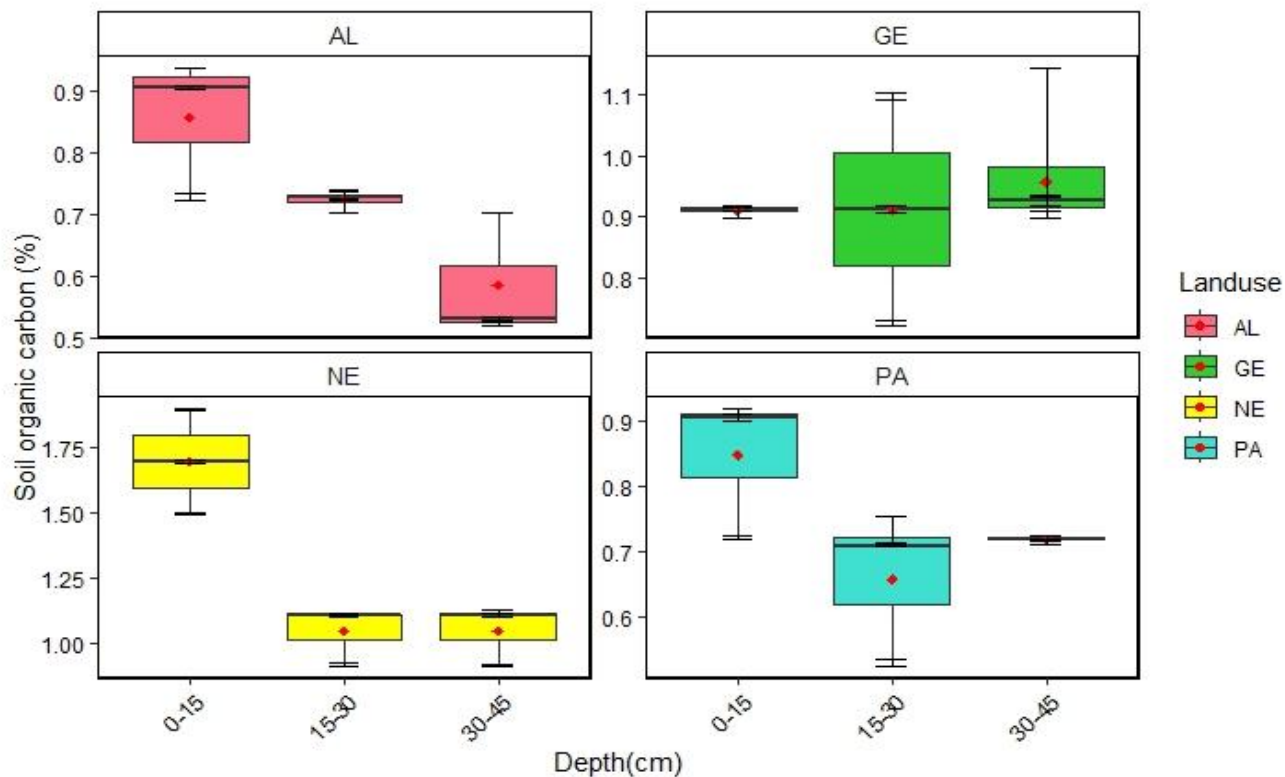


Figure 4.4 Soil organic carbon per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

4.3. Exchangeable Bases (Nutrient analysis)

The soil depth and land use type strongly affected the exchangeable calcium (Ca^{2+}), magnesium (Mg^{2+}), and potassium (K^{+}) and sodium (Na^{+}) levels. The trends for exchangeable calcium differ in response to land use (Figure 4.5). Calcium content in GE slightly decreased with depth, whereas in AL soils, generally increased with soil depth. Conversely, NE showed a fluctuating trend across the soil profile. The highest concentration of exchangeable calcium (35.0 $\text{cmol}(+)/\text{kg}$) was observed in GE, whereas the lowest concentration (20.0 $\text{cmol}(+)/\text{kg}$) was recorded in NE.

The exchangeable magnesium concentrations across the land use types ranged from 2.5 $\text{cmol}(+)/\text{kg}$ to 10.2 $\text{cmol}(+)/\text{kg}$ at the 0–15 cm depth, 2.3 $\text{cmol}(+)/\text{kg}$ to 10.3 $\text{cmol}(+)/\text{kg}$ at the 15–30 cm depth, and 2.7 $\text{cmol}(+)/\text{kg}$ to 10.3 $\text{cmol}(+)/\text{kg}$ at the 30–45 cm depth. At every depth, the lowest concentrations of exchangeable magnesium were found in AL, whereas the highest values were consistently observed in GE (Figure 4.6).

All land use types presented decreased concentrations of exchangeable potassium (K^{+}) and sodium (Na^{+}), ranging from 0.2 $\text{cmol}(+)/\text{kg}$ to 1.3 $\text{cmol}(+)/\text{kg}$ and 0.05 $\text{cmol}(+)/\text{kg}$ to 0.9 $\text{cmol}(+)/\text{kg}$, respectively. Potassium showed a decreasing trend with soil depth across all land uses (Figure 4.7). Sodium presented the highest concentration (1.3 $\text{cmol}(+)/\text{kg}$) in PA at the 30–45 cm depth, whereas the lowest concentration (0.05 $\text{cmol}(+)/\text{kg}$) was recorded in AL at the 15–30 cm depth (Figure 4.8).

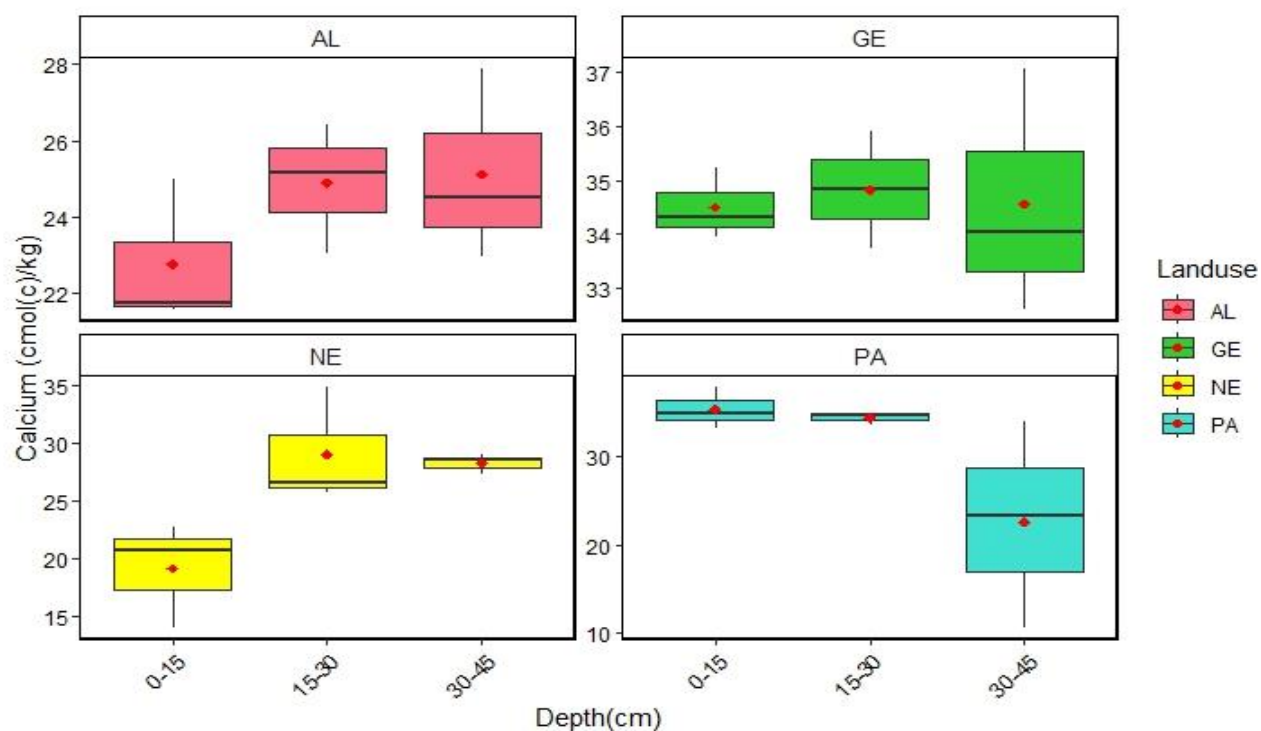


Figure 4.5 Calcium ion content per land use. Arable land (AL), Grassland enclosure (GE), Natural exclosures (NE), Pasture (PA).

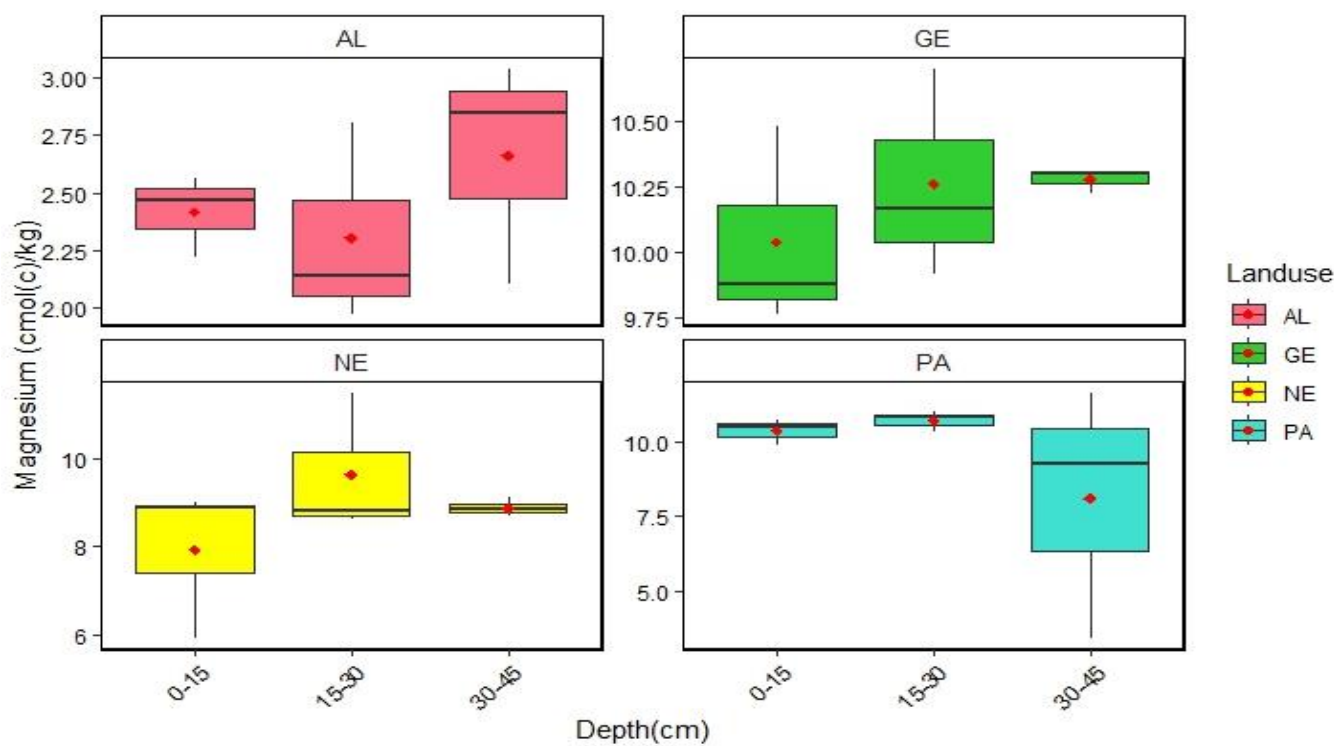


Figure 4.6 Magnesium ion content per land use. Arable land (AL), Grassland enclosure (GE), Natural exclosures (NE), Pasture (PA).

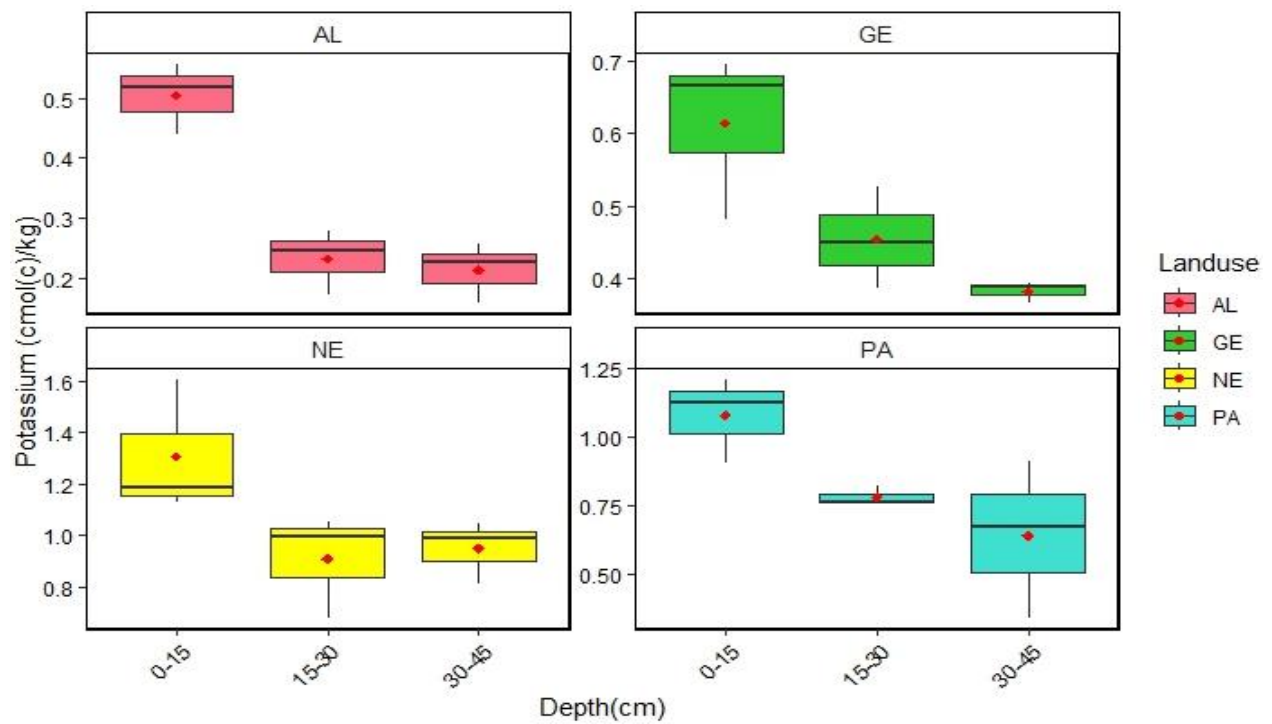


Figure 4.7 Potassium ion content per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

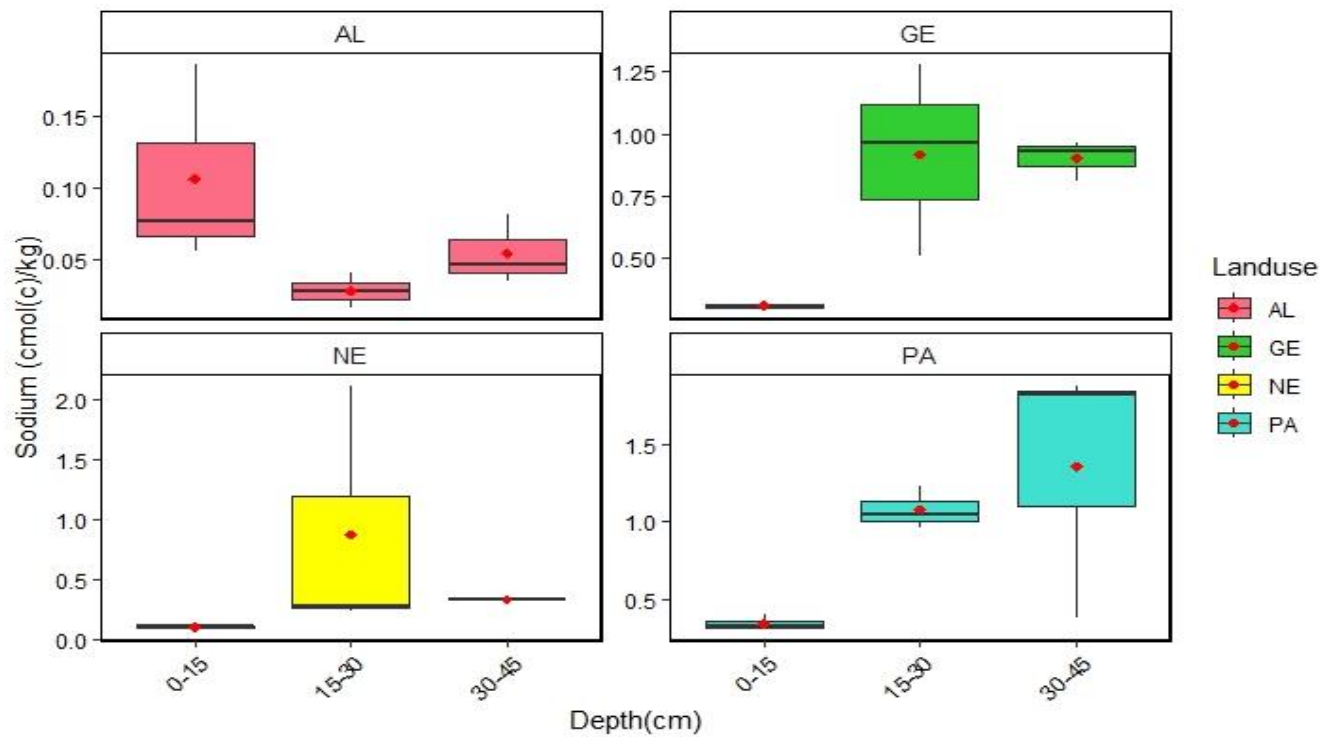


Figure 4.8 Sodium ion content per land use. Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

4.4. Clay mineralogy

The XRD analysis showed that montmorillonite was the most prevalent mineral across all 36 samples, with identical diffraction patterns observed in each sample, confirming uniformity in mineral composition across the study area. In this study, montmorillonite and quartz were the dominant minerals observed in all soil samples, with montmorillonite exhibiting a characteristic diffraction peak at $5.9^{\circ} 2\theta$ and quartz being identified at $20.9^{\circ} 2\theta$.

4.5. The distribution of water-stable aggregate classes

Land use had a significant effect on the distribution of water-stable aggregate (WSA) sizes ($P < 0.05$). Among all land use types, AL soils had a greater proportion of small macroaggregates (0.5–0.25 mm) than larger macroaggregates (>4.75 –1.00 mm), whereas NE soils had a greater proportion of larger macroaggregates. Depth-wise, the distributions of microaggregates, smaller macroaggregates and larger macroaggregates varied significantly across land uses. At 0–15 cm, the proportion of > 4.75 mm WSAs was highest in NE (46.62%), followed by GE (12.36%), whereas PA and AL soils had the lowest proportions of aggregates ≥ 4.75 mm, with values of 5.82% and 0%, respectively. Conversely, the proportion of smaller aggregates (0.5–0.25 mm) increased significantly in PA and AL soils, while the highest percentage of 0.25 mm water-stable aggregates (47.36%) was found in AL soils. There was no significant difference in the 4.75–2.0 mm WSAs between land use types (NE, GE and PA) ($P > 0.05$). Similarly, no significant differences were detected in the distribution of <0.25 mm WSAs across depths ($P > 0.05$). Notably, microaggregates (<0.25 mm) accounted for the lowest percentage of WSAs across all land use types.

The percentages of >4.75 mm water-stable aggregates decreased dramatically at the 15–30 cm depth, ranging from PA (46.62%) $>$ GE (31.28%) $>$ NE (7.76%) $>$ AL (0%). PA and GE had much greater proportions of >4.75 mm, 4.75–2.00 mm, 2.00–1.00 mm, and 1.00–0.50 mm

aggregates, with the largest percentages observed in the 2.00–1.00 mm and 1.00–0.50 mm size classes. In contrast to those at the 0–15 cm depth, the proportions of >4.75 mm, 4.75–2.00 mm, and 2.00–1.00 mm aggregates in the NE decreased. In AL, there was a significant increase in the 2.00–1.00 mm and 1.00–0.50 mm aggregates, accompanied by a slight decrease in the 0.50–0.25 mm size class.

At the 30–45 cm depth, GE and NE presented the highest proportions of > 4.75 mm at 53.18% and 50.67%, respectively. Arable land (AL), on the other hand, presented an increase in intermediate-sized aggregates (2.00–1.00 mm and 1.00–0.5 mm) but a decrease in smaller aggregate fractions (0.50–0.25 mm).

Table 4.6 The distribution of water-stable aggregate classes at a soil depth of 0-45 cm across various land uses

Land uses	----- (%) -----					
	> 4.75 mm	4.75-2.00 mm	2.00-1.00 mm	1.0-0.50 mm	0.5-0.25 mm	<0.25 mm
<u>0-15 cm</u>						
AL	-	-	11.39±3.23(a)	24.55±15.50(a)	47.36±18.50(b)	8.07±3.67(a)
PA	5.82±1.41(bc)	6.29±2.30(a)	17.23±8.71(a)	24.99±0.31(a)	24.35±7.77(ab)	7.18±1.84(a)
GE	12.36±2.51(a)	9.20±10.52(a)	29.34±8.73(a)	12.77±7.30(a)	17.41±6.65(a)	4.65±2.60(a)
NE	46.62±9.04(d)	8.28±2.17(a)	6.29±3.52(a)	17.94±3.88(a)	8.27±1.43(b)	5.72±0.63(a)
<u>15-30 cm</u>						
AL	-	-	12.33±1.87(a)	26.54±14.53(a)	46.27±18.88(b)	8.37±3.20(a)
PA	46.62±9.04(cd)	5.78±3.13(a)	22.00±11.71(a)	34.44±18.88(a)	13.61±5.20(b)	6.69±0.94(a)
GE	31.28±14.05(ab)	16.48±3.64(a)	22.10±12.49(a)	7.71±3.05(a)	12.89±4.36(a)	4.01±1.33(a)
NE	7.76±2.52(d)	6.00±1.00(a)	12.33±1.87(a)	37.59±8.26(a)	17.32±6.85(b)	7.05±1.60(a)
<u>30-45 cm</u>						
AL	-	-	28.62±8.65(a)	32.43±19.15(a)	21.60±6.54(b)	7.21±1.48(a)
PA	13.00±4.58(cd)	14.33±4.04(a)	18.28±11.84(a)	21.11±8.83(a)	19.16±5.68(b)	5.78±1.36(a)
GE	53.18±7.77(ab)	12.52±6.95(a)	10.40±2.46(a)	7.54±3.00(a)	7.00±4.82(ab)	3.62±1.36(a)
NE	50.67±9.24(d)	11.88±5.59(a)	9.51±5.10(a)	11.58±3.35(a)	9.00±2.26(b)	4.48±0.92(a)

Values are mean±standard error. Values with different letters indicate significant differences (analysis of variance; P<0.05) within the same aggregate. Arable land (AL), Grassland exclosure (GE), Natural exclosures (NE), Pasture (PA).

4.6. Stability parameters of water- stable aggregates

To assess the aggregate stability characteristics, three different indicators were selected (Figure 4.9). When assessing the stability and distribution of soil aggregates, the MWD and GMD are crucial indicators that consider not only the R0.25 concentration but also other particle sizes. Greater agglomeration and stability of soil aggregates are indicated by higher R0.25, MWD, and GMD values. The soil aggregate MWD and GMD varied significantly depending on land use and soil depth ($P < 0.05$). A trend was observed for the MWD and GMD across the three soil depths (0–15 cm, 15–30 cm and 30–45 cm). The recorded MWD and GMD values followed a similar pattern and were strongly correlated ($r = 1.0$) (Figure 4.11). Among the three soil layers, the average MWD and GMD values under arable land (AL) were significantly lower than those under natural exclosures (NE), grassland exclosure (GE) and pastures (PA).

The MWD and GMD values at the 0–15 cm depth were in the following order: natural exclosures > grassland exclosures > pastures > arable land. Similarly, the MWD and GMD values at this depth were much greater under natural and grassland exclosures than under arable land. At the 15–30 cm depth, the MWD and GMD decreased in the following order: grassland exclosures > pastures > natural exclosures > arable land. At the 30–45 cm depth, the MWD and GMD followed the trend of grassland exclosures > natural exclosures > pastures > arable land. Notably, at all the soil depths, the MWD and GMD values in the 30–45 cm soil layer were significantly greater than those recorded in the upper layers ($P < 0.05$). These findings highlight that land use significantly influences soil aggregate structure.

The R0.25 values did not differ significantly across land uses ($P > 0.05$) but did vary significantly across soil depths ($P < 0.05$). The R0.25 content was unaffected by land use, since the values remained consistent across the various management approaches. While the R0.25 content increased with soil depth, the differences were quite small. The R0.25 content was

significantly greater in deeper soil layers (15–45 cm) than in topsoil layers (0–15 cm) ($P < 0.05$). Although the R0.25 content significantly differed across the soil depths ($P < 0.05$), the actual numerical changes were relatively small. This indicates that soil depth has a minor influence on the R0.25 content.

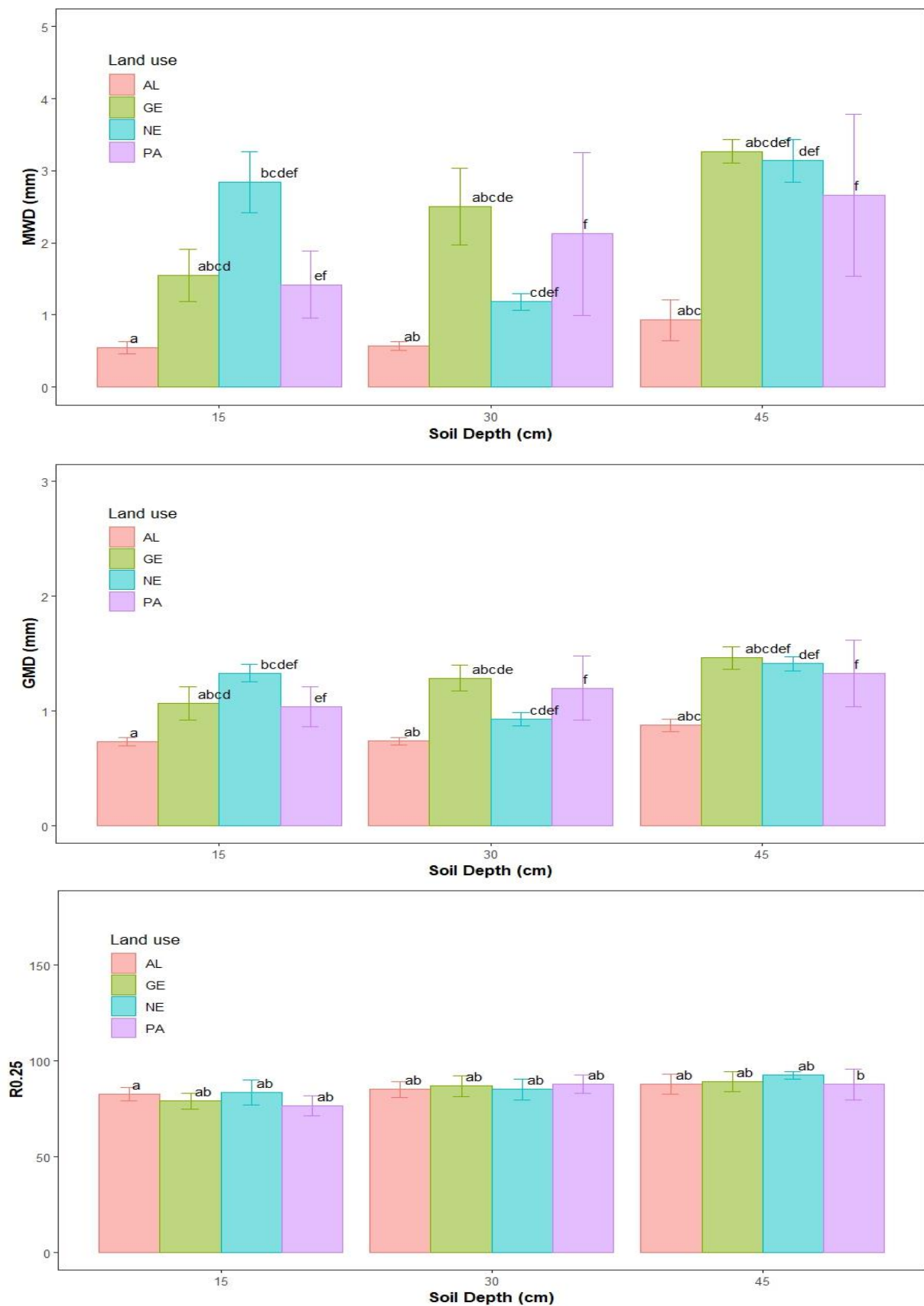


Figure 4.9. Aggregate stability parameters. Mean weight diameter (MWD), GMD (Geometric mean diameter), R0.25; aggregates >0.25 mm. Different letters indicate significant difference between land use types and soil depths (analysis of variance; $P < 0.05$). Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

4.7. The distribution of aggregate associated organic carbon under different land uses

Variations were observed in the various soil aggregate size fractions associated with SOC concentrations across land use systems and depths. The land use system had a significant ($P < 0.05$) effect on the SOC concentrations recorded in various aggregate size classes. The SOC concentration of the macroaggregates (> 0.25 mm) was greater than that of the microaggregates (< 0.25 mm) in all the land use types, except for the AL, where the SOC was concentrated in the microaggregates (< 0.25 mm). However, no significant variation ($P > 0.05$) was detected in macroaggregate (> 4.75 mm and 4.75 – 2.00 mm) organic carbon storage with depth.

For the 4.75 mm aggregates, the SOC did not significantly ($P > 0.05$) differ among the land use types (NE, GE and PA). At 4.75 – 2.00 mm, SOC was significantly greater in NE, whereas GE and PA presented inconsistent trends with fluctuating SOC concentrations with depth. For 2.00 – 1.00 mm, both land use and soil depth significantly affected the SOC ($P < 0.05$). Among all the soil layers, the SOC content was highest in the NE soil, and the lowest was recorded in the AL soil. The SOC content was relatively high in the upper layers (0 – 15 cm) for NE, GE and PA, but the highest concentrations were observed at 30 – 45 cm. In contrast, the lowest concentrations were observed in AL.

Significant differences ($P < 0.05$) were detected among the land use types in the 1.00 – 0.5 mm water-stable aggregates. The highest SOC percentage and the lowest SOC percentage were recorded in the AL, ranging from 0.95 – 1.44% and 0.48 – 0.70% , respectively. No significant differences ($P > 0.05$) were detected in the soil depths from 1.00 – 0.5 mm. Similar observations were made at 0.5 – 0.25 mm, where the highest contents were also reported under NE and the lowest under AL. Microaggregates (< 0.25) did not significantly differ with land use ($P > 0.05$).

Table 4.7 The distribution of aggregate associated soil organic carbon at a soil depth of 0-45 cm across various land uses

Land uses	----- (%) -----					
	> 4.75 mm	4.75-2.00 mm	2.00-1.00 mm	1.0-0.50 mm	0.5-0.25 mm	<0.25 mm
<u>0-15 cm</u>						
AL	-	-	0.40±0.35(cd)	0.48±0.30(bc)	0.47±0.23(cde)	0.73±0.45(a)
PA	1.11±0.16(a)	1.04±0.13(ab)	1.00±0.07(bc)	0.90±0.14(abc)	0.71±0.11(ab)	0.60±0.20(a)
GE	1.24±0.32(a)	1.31±0.08(a)	1.09±0.00(a)	1.04±0.23(a)	0.91±0.04(a)	0.89±0.07(a)
NE	1.74±0.43(a)	2.37±0.50(b)	1.61±0.16(bcd)	1.44±0.23(bc)	1.14±0.10(bcd)	1.04±0.06(a)
<u>15-30 cm</u>						
AL	-	-	0.67±0.13(d)	0.50±0.26(c)	0.23±0.01(de)	0.46±0.62(a)
PA	1.50±0.64(a)	1.56±0.54(b)	0.76±0.05(bc)	0.77±0.04(abc)	0.65±0.20(ab)	0.44±0.30(a)
GE	1.24±0.29(a)	0.96±0.14(ab)	1.03±0.08(ab)	0.96±0.06(ab)	0.89±0.07(ab)	0.39±0.35(a)
NE	1.81±0.14(a)	1.83±0.11(b)	1.24±0.21(bcd)	0.96±0.29(bc)	0.90±0.07(bcd)	0.74±0.12(a)
<u>30-45 cm</u>						
AL	-	-	0.38±0.35(d)	0.70±0.14(c)	0.38±0.18(e)	0.77±0.20(a)
PA	1.62±0.24(a)	1.91±0.04(b)	0.82±0.14(bc)	0.84±0.06(abc)	0.58±0.16(abc)	0.60±0.26(a)
GE	1.31±0.39(a)	1.17±0.26(ab)	0.91±0.06(abc)	0.91±0.06(abc)	0.82±0.00(ab)	0.72±0.06(a)
NE	1.97±0.80(a)	1.85±0.91(b)	1.03±0.16(bcd)	0.95±0.08(bc)	0.87±0.10(bcde)	0.79±0.11(a)

Values are mean±standard error. Values with different letters indicate significant differences (analysis of variance; P<0.05) within the same aggregate.

4.8 Contribution rates of different aggregates to SOC under different land use types

The contribution rate of aggregates of each particle size to SOC varied across land uses (Figure 4.10). Compared with small aggregates, larger particles (>4.75 mm and $4.75\text{--}2.00$ mm) contributed more to the SOC. The contribution rates of the 4.75 mm aggregates did not change significantly ($P < 0.05$) across land use types (NE, GE, and PA) but did vary significantly with soil depth. At the $30\text{--}45$ cm depth, the 4.75 mm aggregates from land uses (NE, GE, and PA) made the greatest contribution to the SOC pool. However, aggregates larger than 4.75 mm at the $0\text{--}15$ cm depth declined significantly in NE, GE, and PA, with contributions of 47.35%, 23.15%, and 17.18%, respectively. Soil depth and land use had insignificant impacts ($P > 0.05$) on the rates of contribution of the $4.75\text{--}2.00$ mm particles.

Additionally, as aggregate size decreases, so does the contribution rate to SOC. Land use systems were shown to have a substantial ($P < 0.05$) impact on the particle size range of $2.00\text{--}1.00$ mm. GE (40%) and PA (24%) contributed more to SOC at a depth of $0\text{--}15$ cm, whereas AL had the lowest contribution rate (5%). The contribution rates decreased for the $1.00\text{--}0.5$ mm aggregates, with no significant variations observed across land uses or soil depths ($P < 0.05$). In AL, the $0.5\text{--}0.25$ mm aggregates contributed the most to the SOC at all soil levels, whereas the <0.25 mm aggregates contributed the least (1.5–8.9%). There were no significant differences found across land uses or soil depths in the <0.25 mm particles.

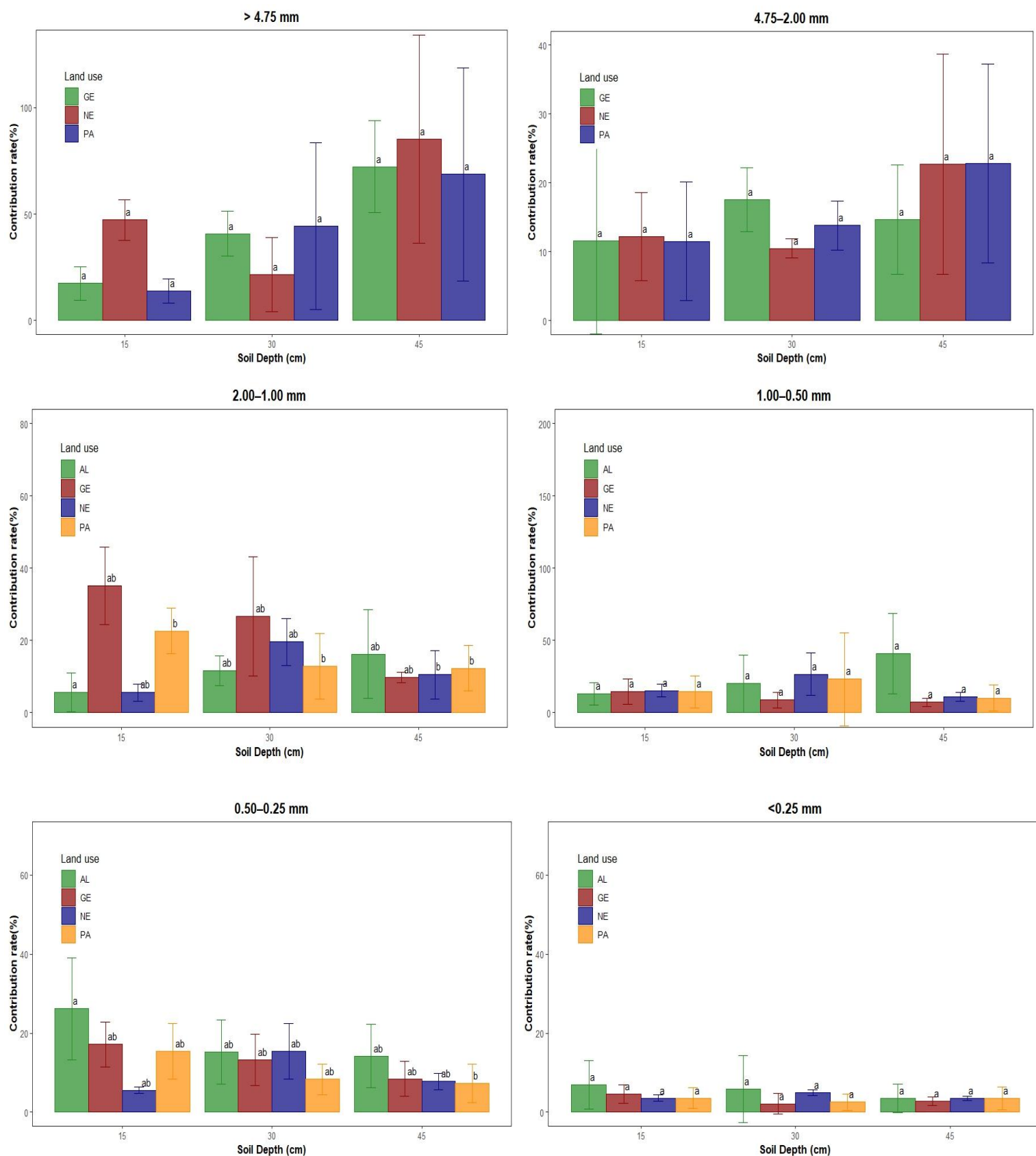
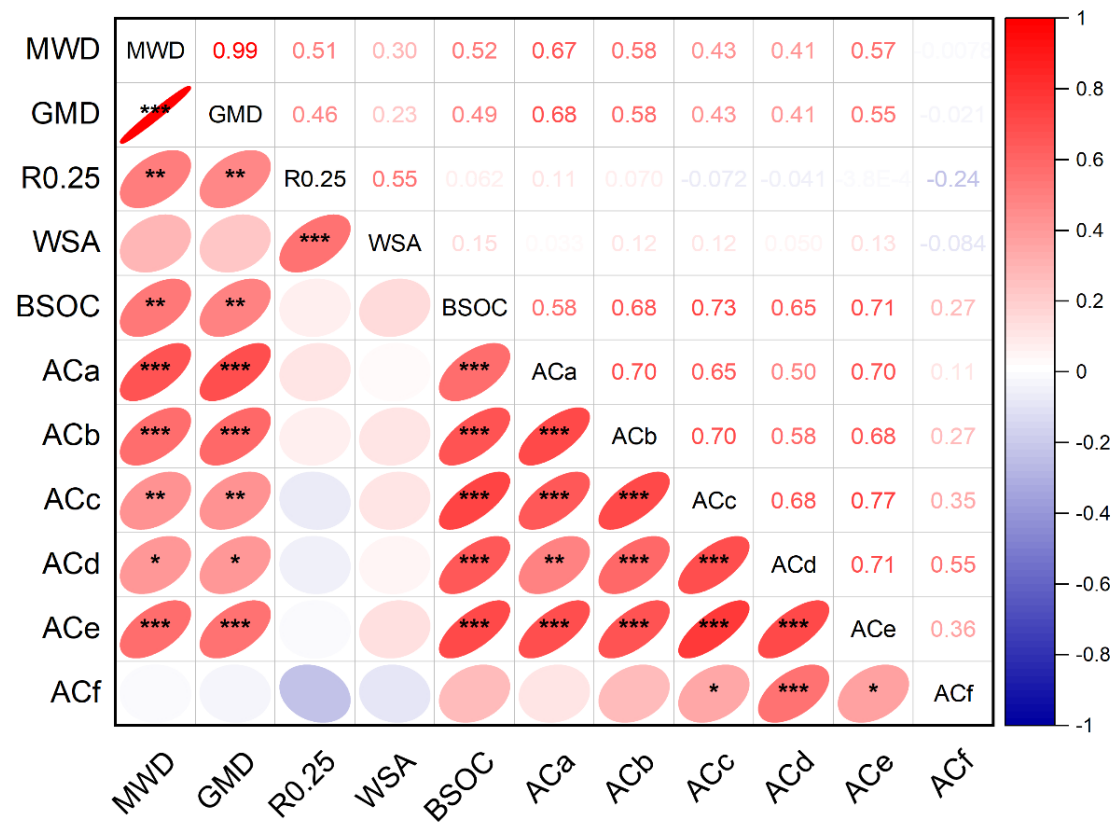


Figure 4.10. Contribution rates of aggregate classes to bulk soil organic carbon. Different letters indicate significant difference between land use types and soil depths (analysis of variance; $P < 0.05$) within each aggregate size.

4.9 Correlation between the stability parameters of aggregates, aggregate-associated C and total SOC in 0–45 cm depth soils at all land uses combined

The correlation matrix elucidates the relationships between aggregate-associated carbon fractions and soil stability indicators (Figure 4.11). Moderate to strong positive correlations ($r = 0.5$ – 1.0) between the mean weight diameter (MWD), geometric mean diameter (GMD), and R0.25 highlight how interrelated these factors affect the stability of soil aggregates. Similarly, there was a positive correlation ($r = 0.5$) between the bulk soil organic carbon (BSOC) content and the MWD and GMD. Strong positive relationships with MWD and GMD are also observed for aggregate-associated carbon fractions in aggregates greater than 0.25, indicating that larger, more stable aggregates are essential for carbon retention. Conversely, finer fractions (<0.25 mm), exhibited negative correlations with the stability metrics, suggesting that <0.25 mm plays a lesser role in carbon storage and overall stability. R0.25 has a weak to negative relationship with aggregate-associated organic carbon but a positive correlation with water-stable aggregates ($r = 0.6$).

Additionally, in aggregates larger than 0.25 mm, BSOC is positively associated with aggregate-associated carbon fractions, whereas in smaller fractions (less than 0.25 mm), BSOC is negatively correlated. This implies that aggregates larger than 0.25 mm are important indices of soil carbon pool saturation and the main sites for SOC storage. These results highlight the importance of maintaining macroaggregates to improve soil health in Vertisols and the crucial role that soil aggregation plays in carbon sequestration.



* $p \leq 0.05$ ** $p \leq 0.01$ *** $p \leq 0.001$

Figure 4.11 mean weight diameter (MWD), geometric mean weight diameter(GMD),water stable aggregates (WSA),Bulk soil organic carbon (BSOC), Mean weight diameter (MWD), geometric mean weight diameter (GMD), water stable aggregate (WSA), bulk soil organic carbon (BSOC), aggregate-associated OC (>4.75 mm(ACa), aggregate-associated OC (4.745-2.00 mm (ACb), aggregate-associated OC (2.00-1.00 mm (ACc), aggregate-associated OC (1.00-0.50 mm (ACd), aggregate-associated OC (0.50-0.25 mm (ACe), aggregate-associated OC <0.25 mm (ACf))

4.10. Soil organic carbon stocks in bulk soils and in different aggregate size classes across various land use regimes

The SOC stocks among the aggregate size fractions were significantly impacted by both soil depth and land use ($P < 0.05$). In comparison with the other land uses, natural exclusion (NE) resulted in greater SOC stocks in larger aggregates (>4.75 mm and $4.75\text{--}2.00$ mm) at the 0–15 cm and 30–45 cm soil depths (Figure 4.12). Compared with the other land uses, grassland enclosure (GE) resulted in the highest SOC stocks in the $2.00\text{--}1.00$ mm aggregate size fraction at 0–15 cm, with the SOC stocks decreasing with depth, and the lowest values (1965 kg C m^{-2}) were recorded at 30–45 cm. The SOC stocks in the $1.00\text{--}0.50$ mm aggregate size class did not differ significantly ($P > 0.05$) among the soil depths and land uses. However, in the $0.50\text{--}0.25$ mm aggregate fraction, land use had a substantial effect on SOC levels, with arable land (AL) resulting in greater SOC stocks. Nevertheless, no significant differences ($P > 0.05$) were detected among the soil depths. The SOC stocks in the <0.25 mm fraction were the lowest among all aggregate size classes, and neither soil depth nor land use had a significant effect on their concentrations ($P > 0.05$).

Land use had a significant ($P < 0.001$) effect on the SOC stocks in the bulk soil, according to two-way ANOVA. Across all depths, NE had the highest SOC stores overall. The SOC stocks at 0–15 cm were greatest in NE China (250 kg C m^{-2}), followed by AL China (80 kg C m^{-2}). On the other hand, the SOC stocks in AL tended to decrease with depth. The impact of land use on SOC stocks varied with depth. At 0–15 cm, the SOC stocks followed the order $\text{NE} > \text{AL} > \text{PA} > \text{GE}$, whereas the following decreasing pattern of SOC stocks was observed ($\text{NE} > \text{GE} > \text{AL} > \text{PA}$) at 15–30 cm. At 30–45 cm, the SOC stocks followed the sequence $\text{NE} > \text{GE} > \text{PA} > \text{AL}$.

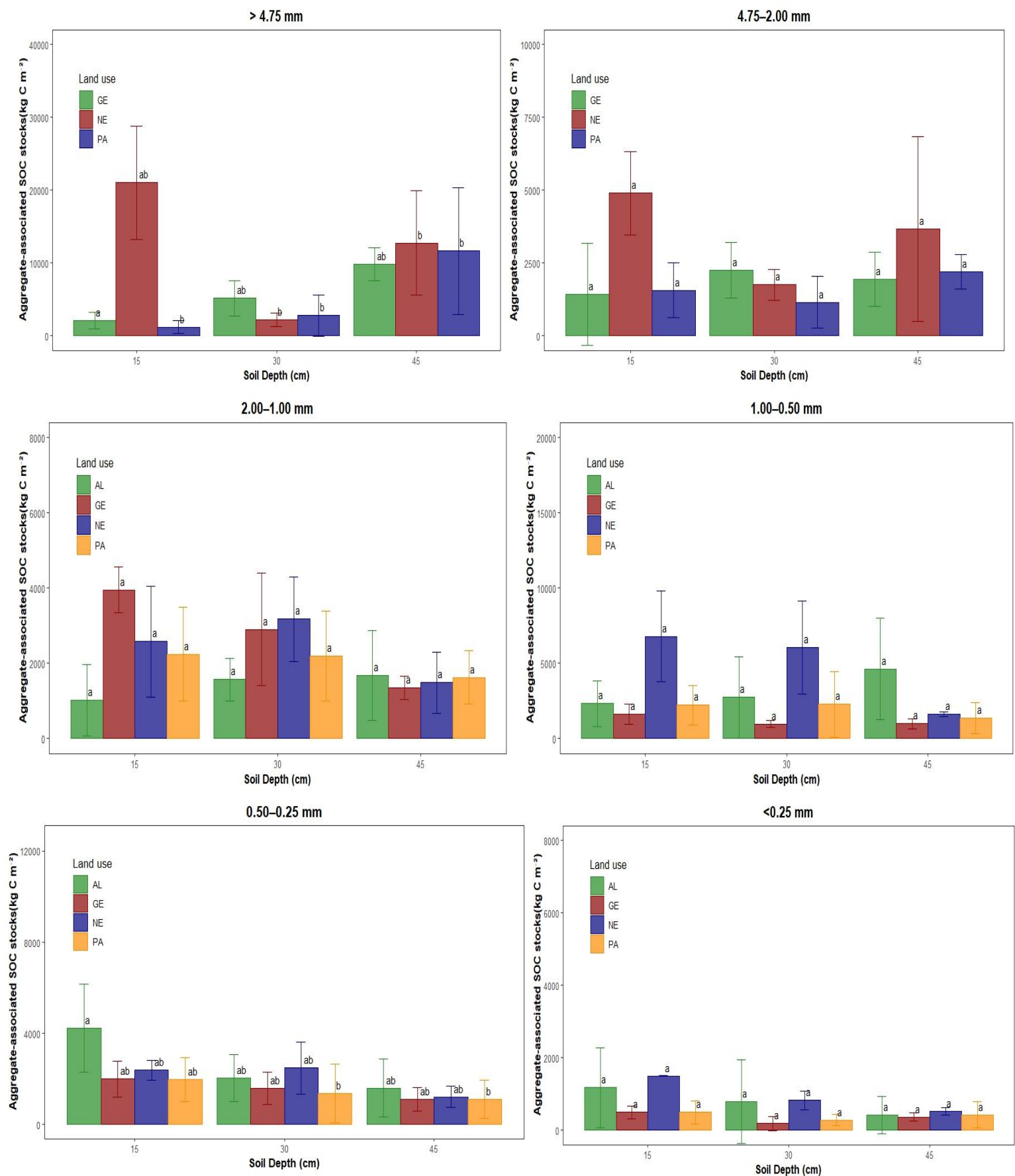


Figure 4.12 Aggregate-associated organic carbon stocks per land use. Different letters indicate significant difference between land use types and soil depths (analysis of variance; $P < 0.05$) within each aggregate size.

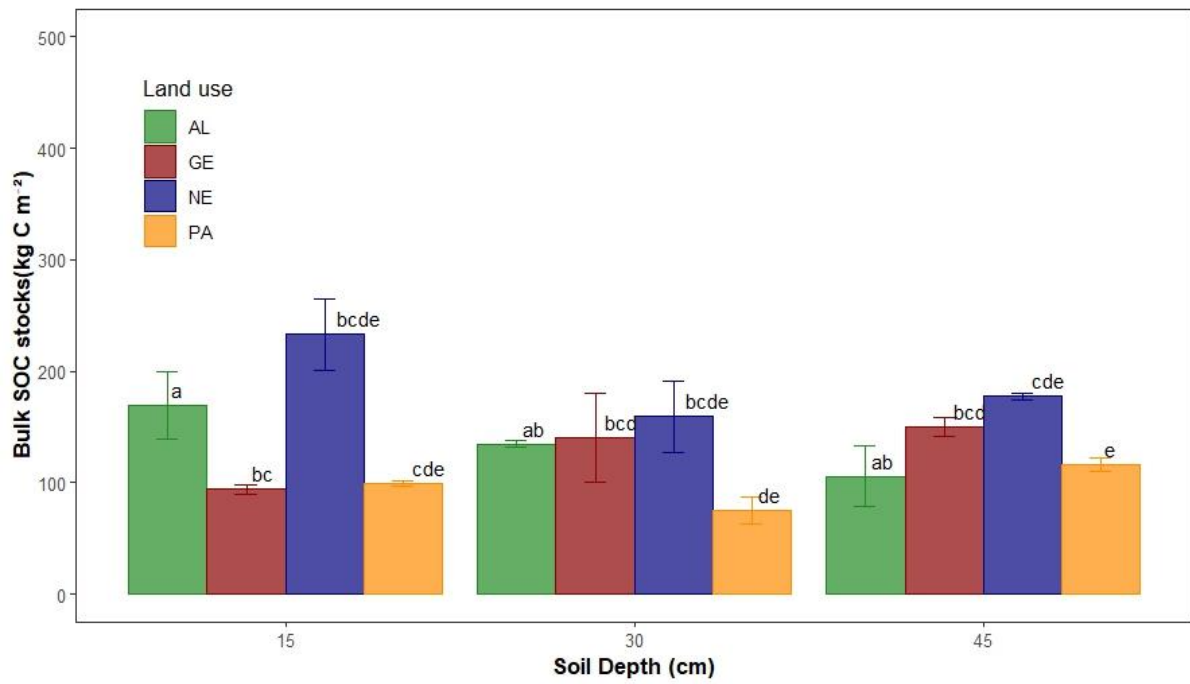


Figure 4.13 Bulk soil organic carbon stocks per land use. Different letters indicate significant difference between land use types and soil depths (analysis of variance; $P<0.05$) within each aggregate size.

4.11. Relationships between organic carbon stocks in aggregates and bulk soils

A simple linear regression analysis was used to investigate the relationships between the organic carbon (OC) stocks in the aggregates and those in the bulk soils (Figure 4.14). The linear model was statistically significant, although there was a weak to no correlation between the OC stocks in the aggregates and bulk soils. The bulk soil OC stocks were more significantly related to the larger aggregates (>4.75 mm and 4.75–2.00 mm), with coefficients of determination (R) of 0.34 and 0.2, respectively, than to the smaller aggregates (0.5–0.25 mm and <0.25 mm). However, this finding indicates that only a small proportion of the variation in large aggregate-associated SOCSs can be explained by bulk SOCSs.

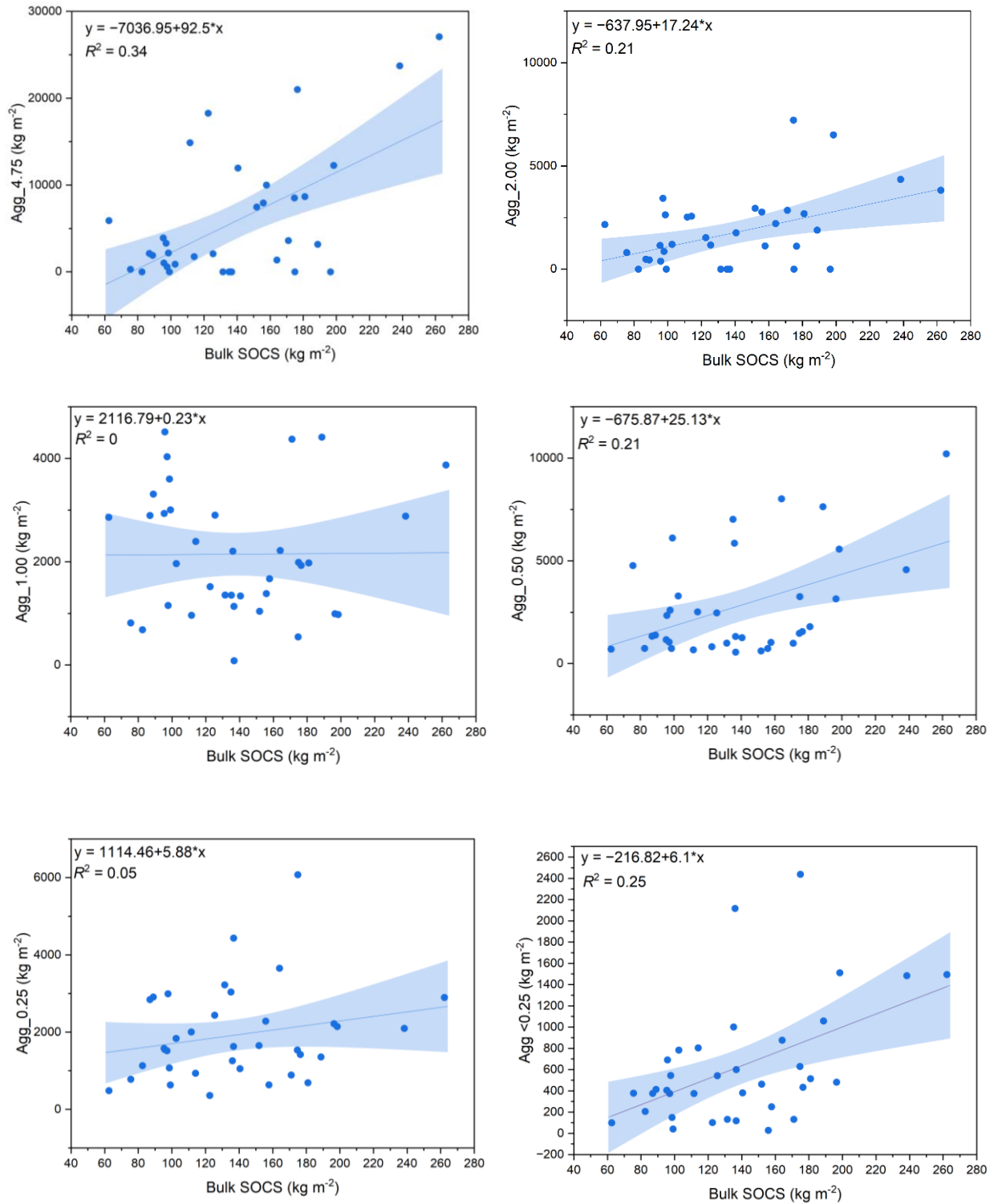
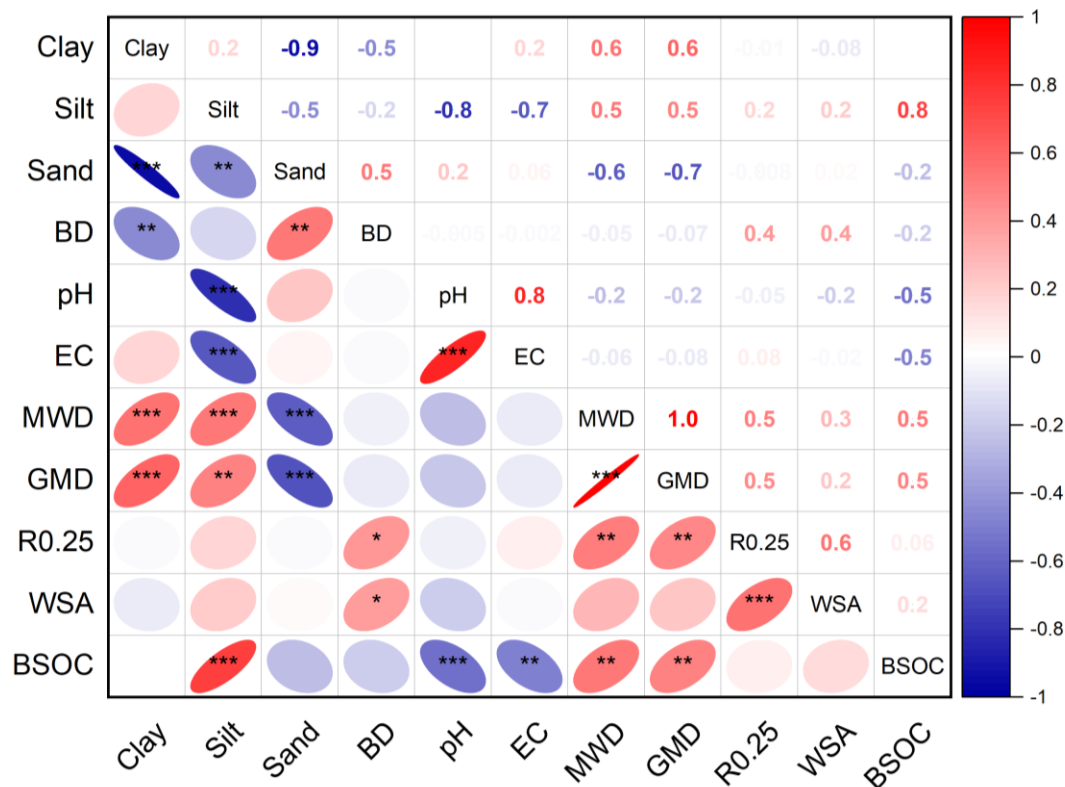


Figure 4.14 Linear relationships between organic carbon stocks within different aggregate size classes and total (bulk) soil organic carbon stocks. Different letters indicate significant difference between land use types and soil depths (analysis of variance; $P < 0.05$). Aggregate sizes are in (mm).

4.12. Pearson correlation between soil properties

Pearson correlation analysis was used to investigate the correlations between different soil properties and their effects on soil aggregate stability (Figure 4.15).



* $p \leq 0.05$ ** $p \leq 0.01$ *** $p \leq 0.001$

Figure 4.15 Pearson correlation between soil properties. Bulk density (BD), Electrical conductivity (EC), mean weight diameter (MWD), Geometric mean diameter (GMD), Water stable aggregates (WSA), R0.25; aggregates >0.25 mm, Bulk soil organic carbon (BSOC).

The findings demonstrated a substantial and significant positive connection between the mean weight diameter (MWD) and geometric mean diameter (GMD), indicating a close relationship in determining soil structural stability. GMD and MWD had a moderate positive correlation with R0.25 ($r = 0.5$), although their associations with WSA were weak but significant. However, R0.25 exhibited a moderate positive correlation with WSA, indicating its greater contribution to aggregate stability.

The clay content was positively related to the MWD ($r = 0.6$) and GMD ($r = 0.6$), implying that an increased clay concentration improved aggregate stability. The sand content, on the other hand, had a negative correlation with these stability measures, implying that it plays a role in lowering soil structural integrity. Significant negative relationships were found between pH and stability metrics such as WSA, GMD, MWD, and R0.25. Similarly, electrical conductivity (EC) has a negative connection with these stability measures. Bulk density had a positive relationship with R0.25 and WSA but a negative correlation with GMD and MWD, indicating an inconsistent influence on soil structure. Furthermore, bulk density was negatively correlated with clay content ($r = -0.5$), indicating that a higher clay concentration corresponds to less soil compaction (Figure 4.15). Additionally, the silt content was strongly correlated ($r=0.8$) with bulk soil organic carbon (BSOC), which was moderately positively correlated with the GMD and MWD but had weak to no correlations with R0.25 and WSA, indicating its variable influence on soil aggregation.

CHAPTER FIVE

DISCUSSION

5.1 Soil properties and land use

The physical, chemical, and mineralogical properties of the soils were characterized to establish baseline conditions, which provide a foundation for understanding subsequent changes in carbon storage and aggregate stability. These baseline properties are critical in assessing how land use influences soil structure and function over time. The electrical conductivity (EC) levels are less than 2.0 dS/m in all soils, according to (FAO, 2021) guidelines, indicates that the soils were not saline. The soil pH varies significantly across land uses and depths. Notable factors which are known to influence soil pH include vegetation cover (Igwe et al., 1999), organic matter decomposition (Milne & Haynes, 2004) , and mineral weathering (Martens et al., 2023). Arable land had relatively stable pH values (7.2-7.8) across depths. This consistency in pH may be due to lime or fertilizer inputs, which are known to help regulate soil acidity (Kezlya et al., 2022). The natural exclosures had the lowest pH values (5.6–6.2), with the greatest variation occurring in the topsoil layers. The relatively lower pH in NE soils is likely influenced by significant organic matter inputs, which, through microbial decomposition, release organic acids such as humic and fulvic acids. These acids contribute to soil acidification by donating protons (H^+). Additionally, the prolonged uptake of basic cations (e.g., Ca^{2+} , Mg^{2+} , K^+) by deep-rooted perennial vegetation may lead to cation depletion and further pH reduction in the surface layers (Ampong et al., 2022).

Moreover, we also observed a negative correlation between soil pH and EC with aggregate stability parameters, including mean weight diameter (MWD), geometric mean diameter (GMD), the proportion of stable macroaggregates ($R_{0.25}$), and water stable aggregates (WSA). (Figure 4.11). This implies that soil salinity and acidity may play a role in aggregate

destabilization. According to (Marchuk et al., 2013) findings, this is also possible since the pH of the soil influences other clay minerals, which in turn alter the structure of soil aggregates.

Our study showed that the amount of organic carbon of all soils was low, which is characteristic of arid and semi-arid regions with little to no vegetation cover. However, NE soils showed the highest soil organic carbon levels in the surface soil layers. This results are consistent with the undisturbed nature of NE soils, which support higher plant biomass turnover and lower oxidation rates, resulting in increased carbon sequestration in the soil (Wasis, 2012). In contrast, SOC levels decreased in AL and PA land uses, reflecting the effects of tillage, crop residue clearance, microbial decomposition, and animal trampling, which all contribute to lower carbon inputs in subsoils (Mhete et al., 2020). Moreover, we observed a moderately positive correlation ($r = 0.5$) between bulk soil organic carbon (BSOC) and GMD and MWD, indicating that higher organic carbon levels contribute to greater aggregate stability which was in line with the results of other studies. For instance, (Idowu, 2003) discovered a linear relationship between soil organic carbon and MWD in water-stable soil aggregates. (Kv et al., 2019) discovered that SOC increased the agglomeration of microaggregates into macroaggregates. (Bogunovic et al., 2020) demonstrated that adding organic matter to the soil improves its hydrophobicity and reduces damage from air particles. However, there are contradictory findings. For instance, some research discovered that the association between soil organic carbon and MWD was neither significant nor negatively connected (Aziz and Islam, 2015; Aziz and Karim, 2016; Igwe and Stahr, 2004). This happens since the formation, stability, and turnover of aggregation are influenced not just by the organic matter in the soil, but also by various factors, such as the coagulation of root secretions (Gale and Cambardella, 2002), the activity of microorganisms, and inorganic compounds such as iron and aluminum oxides (Zhao et al., 2023). Bulk density (BD) values across different land uses remained within acceptable ranges for plant growth as opined by earlier studies (e.g., Obidike-Ugwu et al.,

2023). AL showed the highest BD, this is likely due to the surface compaction caused by the use of agricultural machinery. These findings are consistent with the study of (Özdemir et al., 2022) who reported that agricultural land experiences higher BD due to soil compaction and organic matter degradation. Moreover, (Figure 4.11) shows a negative correlation between BD with SOC and aggregate stability parameters. (Wagner et al., 2007) found a negative correlation between BD and SOM, indicating that SOM reduces bulk density. (Semenov et al., 2010) also discovered that BD has a negative relationship with SOM. The negative correlation between BD and aggregate stability is due to soil degradation over time, which affects components like SOM, which help build stable soil aggregates (Grigal and Vance, 2000).

All soils in the study area have more than 30% clay content, indicating a clay-rich texture. The high clay content contributes significantly to soil carbon stability by preventing organic matter breakdown and increasing aggregate formation. Furthermore, clay-rich soils are more resistant to erosion (Abu-Hamdeh et al., 2006; Liu et al., 2019; Tatarko, 2001). We also observed a positive correlation between silt and clay particles with MWD and GMD. (Oades and Waters, 1991) proposed an aggregate model, indicating that silt and clay particles are first bonded by organic matter to form microaggregates, which are referred to as the smallest unit of soil structure. (Fernández-Ugalde et al., 2013) founded that clay particles play a significant role in the flocculation and cementing processes in the soil. (Almajmaie et al., 2017) shown that an increase in the fraction of clay highly increased the water stability of aggregates.

This study shown that the highest CaCO_3 content was found in AL. This is likely due to the use of calcium-based amendments such as lime. However, NE soils exhibited the highest Ca^{2+} concentration than AL. These results are consistent with the statement of (Virto et al., 2017) who highlighted that at high pH values, calcium carbonate is relatively insoluble in water. Since calcium carbonate is less soluble in alkaline environments, a high pH may result in a low

concentration of dissolved calcium ions even in soils with high calcium carbonate contents (Guo, 2021). Factors that cause the variability of exchangeable bases (Ca^{2+} , Mg^{2+} , Na^{+} and K^{+}) in soils include leaching losses (Tang et al., 2023), variations in parent material composition (Kou et al., 2012), clay mineralogy, and land use conversion.

Our findings show that montmorillonite dominated all the soil samples, indicating that there are no substantial changes in clay mineralogy across land uses. Similarly, working on Vertisols in Serowe, central Botswana, Eze et al. (2025) also reported the abundance of montmorillonite in the soils. The impact of the smectitic clay content on the soil structure and stability is debatable in literature. While some studies indicate that soils containing even trace amounts of smectite are highly dispersible and prone to surface sealing (Kumar et al., 2023), others have reported that Vertisols have high aggregate stability due to their smectitic nature (Churchman et al., 2020). Conversely, some findings indicate that the shrink-swell behavior in Vertisols may reduce aggregate stability (Barzegar et al., 1997; Chenu et al., 2000), although this same process can also enhance the soil structure over time. Despite arguments that the self-mixing nature of shrink clays mitigates land use impacts on soil structure, our findings shown that the aggregate stability parameters such as the $R_{0.25}$ values were stable throughout land use and also the distribution of the aggregate fraction < 0.25 contained lower contents across all land uses possibly due to the influence of smectitic clay. These findings support the idea that Vertisols, owing to their smectitic clay composition, are more resistant to degradation.

5.2. Relationships between land use and soil aggregate parameters

Soil quality and fertility are directly influenced by the quantity (proportion of stable aggregates) and quality (size, cohesion, and resistance to disintegration) of soil aggregates, as they regulate soil structure, erosion resistance, water retention, and organic carbon protection (Mi et al., 2018). The stability of soil aggregates influences their ability to withstand exogenic action and

remain stable when subjected to changes in the surrounding environment (Rohila et al., 2017). The factor analyses (Figure 5.1) show that arable land was mostly associated with small aggregate fractions (1.00-0.50 mm, 0.50-0.25 mm, and <0.25 mm). This higher proportion of small aggregate fractions is most likely due to the mechanical disturbance induced by rotary tillage. Therefore, tillage operations promoted the breakdown of macroaggregates, resulting in a higher proportion of small aggregates. This led to a decrease in soil aggregation, making the soil more susceptible to erosion from water and wind, indicating reduced soil quality. These results are consistent with the findings of (Devine et al., 2014), who found that the large fraction of microaggregates in agriculture might be attributed to agricultural cultivation, which is a key component of deconstructing macroaggregates. Moreover, the large aggregate sizes (mostly >4.75 mm and 4.75—2.00 mm) were observed in both natural and grassland exclosures. Higher MWD, GMD, and R0.25 were also recorded under natural and grassland exclosures as compared to arable land. This can be associated with fact that soil aggregate stability is primarily regulated by binding agents between soil particles, with organic materials being the most prominent in these surface soils (Jat et al., 2019). Thus, MWD and GMD are impacted by changes in plant biomass among land uses and the crop residues in arable land. These factors impact organic material inputs and overall soil stability. Factor analyses show that BSOC was mostly linked to natural and grassland exclosures (Figure 5.1). Leaf litter and root exudates from plant biomass contribute to the soil organic matter (Amin and Adhikary, 2007). It is well established that soil organic matter (SOM) enhances microbial processes that promote soil aggregation. During decomposition, SOM fuels microbial growth and leads to the production of cementing agents such as extracellular polysaccharides and fungal hyphae, which bind soil particles into stable aggregates (Guimarães et al., 2013). Therefore, the greater aggregate stability observed in soils under natural and grassland exclosures may be partly attributed to higher SOM inputs that support these aggregation mechanisms. On the other hand, the lower

SOC in arable soils could be attributed to the usage of tillage, also known to introduce oxygen into the soil and expose organic matter from deeper soil layers, hastening SOC mineralization. (Nciizah and Wakindiki, 2014) also reported that forest soils presented higher MWD and GMD values and more resilient macroaggregates than cultivated soils did.

Previous studies (Aziz et al., 2013; Naresh et al., 2017; Norton et al., 2006) demonstrated that long-term conservation tillage strategies promote the production of water-stable macroaggregates, improve aggregate mean weight diameter (MWD), and increase SOC content, ratio, and storage. Our findings further support this viewpoint, underlining the need to use of conservation tillage strategies to increase soil structure and SOC sequestration in the Pandamatenga Plains. Furthermore, higher MWD and GWD can be attributed to the soils' recurring wetting and drying (WD) and higher clay contents. Large aggregates are therefore formed as a result of WD being encouraged by the high clay concentration in the soils under natural and grassland exclosures (Table 4.5). The findings highlight the effects of land use change and management approaches on soil structure and aggregate size distribution, highlighting the necessity of sustainable land management in preserving soil stability and function.

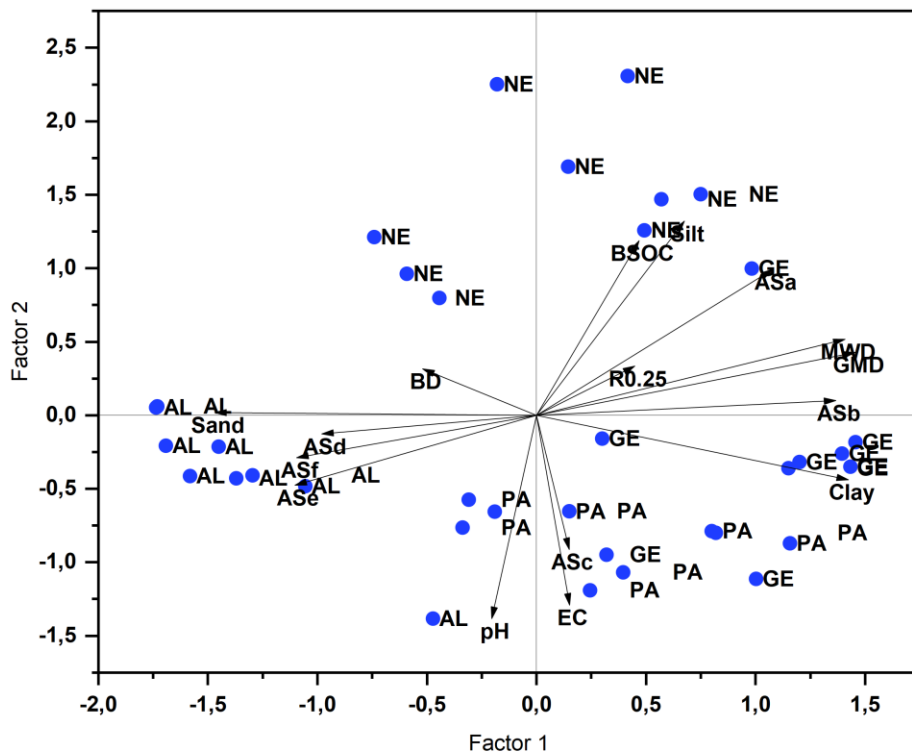


Figure 5.1. Factor analyses between soil properties and water stable aggregates between different land uses. Bulk soil organic carbon (BSOC), Bulk density (BD), Mean weigh diameter (MWD), Geometric mean diameter (GMD), Water stable aggregates :>4.75 mm (ASa), 4.745-2.00 (ASb), 2.00-1.00 mm (ASc), 1.00-0.50 mm (ASd), (0.50-0.25 mm (ASe), <0.25 mm (ASf). Arable land (AL), Grassland enclosure (GE), Natural exclosures (NE), Pasture (PA).

5.3. Relationships between land use and aggregate-associated SOC

Our findings align with earlier research, indicating that aggregate-associated SOC varies with respect to aggregate size and land use regimes (Bashir et al., 2016; Pinheiro et al., 2004; Sheng et al., 2020). All aggregate fractions in natural exclosures showed increases in OC concentrations (Figure 5.2). In addition, fast turnover aggregate fractions, such as the macroaggregates (> 4.75 mm, 4.75-2.00 mm and 2.00—1.00 mm), which are known to be more dynamic and decompose more rapidly than microaggregates, showed greater increases in OC concentrations. The greater SOC content in macroaggregates observed in this study can be

explained by the hierarchical soil aggregation theory (Oades, 1984). According to this theory, organic cementing agents such as root exudates, fungal hyphae, and polysaccharides bond together smaller soil particles (e.g., silt and clay) to produce macroaggregates. As the binding agent concentration increases, the SOC becomes more stable within bigger aggregates, reducing degradation by soil microorganisms. These results are consistent with the results of (Sekaran et al., 2021) study who found that the majority of SOC was deposited in macroaggregates, indicating that organic materials were the primary binding agent for aggregates in their soils. Additionally, SOC stored within macroaggregates had limited longevity. In nutshell, the increase in SOC was predominantly due to enrichment within macroaggregate fractions, indicating that these larger aggregates played a major role in storing organic carbon in the soil. Moreover, macroaggregates provide initial physical protection for organic materials by enclosing them and promoting microaggregate formation within, which limits microbial access. The reduced macroaggregate proportion in arable land likely limits this early-stage protection, resulting in faster microbial decomposition (Six et al., 2004). Due to the addition of significant amounts of plant debris to the soil, the no-tilled ecosystems (such as natural and grassland enclosures) caused fine soil particles to aggregate or increased root exudates, which is why the outcomes were anticipated. The less soil disturbance in natural enclosures aided in shielding SOM from microbial degradation in various aggregate-size classes, which in turn promoted the formation of soil macroaggregates that are physically stable (Mikha et al., 2021). This is similar with the previous research of (Zheng et al., 2018), which showed that land conversion to agriculture changes soil structure, leading to macroaggregate disintegration and the redistribution of SOC into smaller aggregates. (Okolo et al., 2020) also reported that in the forests of Desa'a and Hugumburda, macroaggregates had higher organic carbon levels than microaggregates, particularly in the surface layer, which suggested that macroaggregates play a crucial role in the retention of organic carbon in their study. These

findings imply that macroaggregate dynamics may have a bigger impact on the long-term carbon sequestration potential in these soils than direct land use effects on microaggregates, since microaggregates are known to shield SOC from microbial decomposition over longer timescales (Qiu et al., 2015). These results highlight the necessity of preserving stable soil aggregates in order to maximize soil quality and SOC sequestration. In semiarid alluvial ecosystems, land management techniques that promote the development of macroaggregates, such as reduced tillage, organic supplements, and vegetation restoration, may be crucial in reducing carbon losses and enhancing soil health.

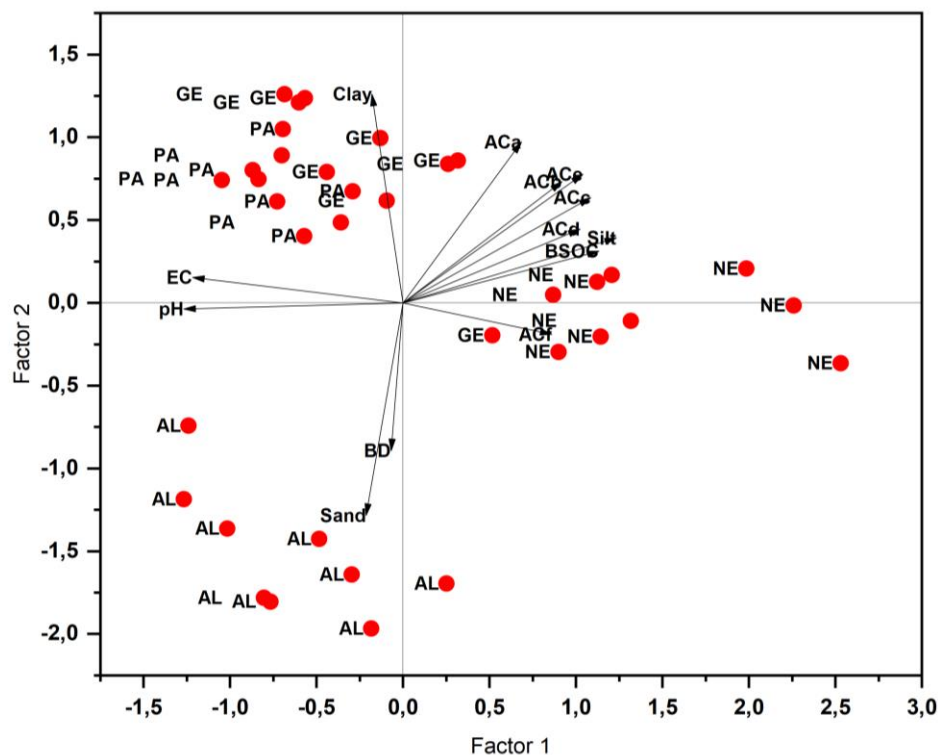


Figure 5.2. Factor analyses soil properties and aggregate-associated organic carbon across land uses. Bulk soil organic carbon (BSOC), Bulk density (BD), Mean weigh diameter (MWD), Geometric mean diameter (GMD), aggregate-associated OC (>4.75 mm (ACa), aggregate-associated OC (4.745-2.00 mm (ACb), aggregate-associated OC (2.00-1.00 mm (ACC), aggregate-associated OC (1.00-0.50 mm (ACd), aggregate-associated OC (0.50-0.25 mm (ACE), aggregate-associated OC <0.25 mm (ACf)), Arable land (AL), Grassland enclosure (GE), Natural enclosures (NE), Pasture (PA).

5.4. Depth distribution of soil organic carbon stocks

The SOC stocks in the soil were significantly impacted by soil depth. In general, SOC stocks were substantially greater in topsoil (0-15 cm) compared to other soil depths across different land uses (Figure 4.13). The reason for this discrepancy is the significant buildup of new organic matter in the surface soil which comes mostly from plant biomass, litter, and roots, as well as an improved fine root system. Although SOC stock decreases with depth, natural enclosure soils have greater stock in the 30-45 cm layer than the 15-30 cm layer. This could be due to the movement of dissolved organic carbon (Jayaraman et al., 2022), soil microbial (Cyle, 2021), or deep-rooting plant species (Cai et al., 2023). (Saha et al., 2011) found that SOC stock declined below 20 cm but slightly increased in the third layer, which is consistent with our findings. The observed tendency can be attributed to the mixing of upper soil layers through cultivation or soil fauna in natural vegetation, as well as leaching and illuviation in deeper levels (Hossain et al., 2017). Earthworms, ants, and termites can dig burrows up to several meters deep in the soil. Earthworm burrows made by anecic species transfer considerable amounts of SOM to the subsoil through the buildup of middens, which include raw and decomposed plant leftovers, casts, and dirt (Dominy and Haynes, 2002).

Moreover, (Ameer et al., 2023) reported higher SOC stocks in the upper soil layers in forest soils. The trees' deeper root systems accumulate more biomass, which is shielded from decomposition due to slower oxygen diffusion and substrate restriction (Liu et al., 2024). Grassland enclosure on the other hand exhibit higher SOC stocks with increasing soil depth, which differs from other land uses (Figure 4.13). This could be due to the grassland's history of frequent wildfires, which may have contributed to carbon loss. Root biomass may have led to greater SOC levels in deeper soil layers due to the recalcitrant nature of soil organic carbon. (Reinhart and Vermeire, 2016) found that grassland soils can absorb more SOC in surface soils than cultivated soils. Cultivation reduces soil organic carbon (SOC) through mineralization,

leaching, and erosion. Surprisingly, our findings revealed that arable land held more carbon in surface soils than grassland and pastures, probably due to crop cover and associated organic matter inputs, which resulted in SOC buildup in surface soils. Likewise, arable land exhibited higher SOC stocks in the upper surface (0–15 cm) in 0.5–0.25 mm fraction. This is attributed to frequent tillage which promoted aggregate turnover by fragmenting larger aggregates into small aggregates and transferring carbon into this finer aggregate size (Okolo et al., 2023). These findings highlight the importance of focused soil conservation methods, such as minimal soil disturbance, and vegetative cover, to increase SOC sequestration, improve soil fertility, and promote long-term crop production.

CHAPTER SIX

CONCLUSION

This study highlights the significant effects of land use practices on soil aggregate distribution, stability, and organic carbon content. The results show that natural exclosure (NE) increases macroaggregate formation (>4.75 mm) while decreasing the proportion of small macroaggregates and microaggregates. Higher GMD and MWD values were detected under NE, grazing exclosure (GE), and pasture (PA) compared to arable land (AL), indicating that land uses with higher organic inputs and less disturbance increase soil aggregate stability. The strong positive relationships between aggregate stability indices, macroaggregates (>0.25 mm), and SOC content in aggregates and bulk soil highlight the vital role of macroaggregates and SOC in soil structural integrity. Furthermore, soil aggregate stability had a positive correlation with clay and silt concentration but negatively correlated with sand content, emphasizing the significance of finer soil fractions in aggregation processes. Notably, arable land showed higher SOC stocks in the upper soil layers compared to grasslands and pastures. However, since SOC stocks were calculated using bulk density as a multiplying factor, the elevated values in arable soils may be more indicative of increased soil compaction resulting from repeated tillage and heavy machinery use than of higher organic carbon concentrations. These findings give a theoretical foundation for optimizing land use options to improve soil structure, increase carbon sequestration, and reduce land degradation. Furthermore, the findings provide scientific support for soil conservation, erosion control, and sustainable land management techniques, especially in semi-arid regions like Botswana's Pandamantenga Plains. Sustainable land use planning can be enhanced by incorporating land management practices that improve macroaggregate stability and SOC retention, resulting in increased soil resilience, agricultural productivity, and long-term climate change mitigation.

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