ASSESSMENT OF CARBON STOCK AND SELECTED SOIL FERTILITY INDICATORS IN A BUSH ENCROACHED SAVANNA AT ERICHSFELDE FARM, NAMIBIA

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Abstract

Bush encroachment is a form of land degradation common predominantly in semi-arid areas of the world. The resulting habitat degradation and loss of resource productivity make bush encroachment a serious environmental and economic problem in Namibia. Despite these negative effects, bush encroachment offers potential woody carbon stock storage, which can render Namibia a net carbon sink. This study was, therefore, aimed at quantifying potential woody carbon stock storage from different pools and assessing selected soil fertility indicators in low, medium and high bush-encroached sites at Erichsfelde farm in Otjozondjupa region. This was a quantitative study employing a stratified random sampling with systematic sampling along transects. For vegetation surveys in each site, five 20m x 10m plots were set up along two line transects, while established allometric equations were applied to related measured vegetation variables to estimate carbon stock. Additionally, five 1m x 1m soil pits were nested within the vegetation plots and soil samples collected at different soil depths for analysis. Across the three sites, the Kruskal-Wallis test showed no significant difference in the total ($P=0.294>0.05$) and above-ground ($P=0.718>0.05$) carbon stock. However, the below-ground carbon stock was significantly higher in the medium encroached site ($P<0.001$). This can be attributed to the fact that savanna vegetation accumulates most of its biomass underground as a protection against fire and herbivory. Additionally, the dense shrubs in the high encroached site do not invest in the below-ground biomass and carbon stock because of low herbivory as livestock will find it difficult to roam in there as oppose to the medium encroached site that feature higher below-ground carbon storage to protect the vegetation from herbivory and fire. The fact that the medium encroached site sequestered more below-ground carbon than the high encroached sites, suggest that bush encroachment does not positively contribute to carbon sequestration. Soil analyses showed no significant differences in the SOM, EC, pH, total N, Mn and ions (Cl^-, Ca^{2+}, Mg^{2+}, and Na^+) across the three sites. Contrariwise, the high bush encroached site was found to have greater amounts of soil P and K^+ (Kruskal-Wallis $P=0.009<0.05$ & $0.049<0.05$) and lower amounts of K, Ca, Mg, Cu and SAR. The difference in these major nutrients content indicated a soil nutrient deficiency due to the fact that savanna bushes have less biomass that returns low nutrients to the soil. It can be concluded that despite the known ecological importance of invader bushes, they do not sequester significant amounts of carbon nor do they positively contribute to soil nutrients. Thus, both farmers and decision makers need to put in place interventions to control invader bushes.

Key words: Bush encroachment, woody vegetation, carbon stock, soil nutrients

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Dedication

This thesis is dedicated to my son Dylan Ndaitavela Nuwuseb who graced us with his presence in the last year of my studies (6 February 2015). Your existence in my life has changed my whole being “Mubasen”.

Declarations

I, Rufina I. Shifa, declare hereby that this study is a true reflection of my own research, and that this work, or part thereof has not been submitted for a degree in any other institution of higher education.

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Acronyms and Abbreviations

AGB	Above-ground biomass
ANOVA	Analysis of variance
BECVOL	Biomass Estimation from Canopy Volume
BEF	Biomass Expansion Factor
BGB	Below-ground biomass
BIOTA	Biodiversity Monitoring Transect Analysis in Africa
BV	Biomass of inventoried volume
CEC	Cation exchange capacity
CO_2	Carbon dioxide
CRBD	Completely Radomise Block Design
DRFN	Desert Research Foundation of Namibia
EC	Electrical conductivity
FAO	Food and Agriculture Organisation
GDP	Gross Domestic Product
GHGs	Greenhouse Gases
GPS	Global Positioning System

IPCC	Intergovernmental Panel on Climate Change
LSU	Livestock Unit
MET	Ministry of Environment and Tourism
MTI	Ministry of Trade and Industry
NAD	Namibian dollar
NAU	Namibian Agricultural Union
NPC	National Planning Commission
OC	Organic Carbon
OM	Organic Matter
SAR	Sodium adsorption ratio
SASSCAL	Southern African Science Service Centre for Climate Change and Adaptive Land Management
SOC	Soil organic carbon
SOM	Soil organic matter
SPSS	Statistical Package for the Social Sciences
UNFCCC	United Nations Framework Convention on Climate Change
WD	Wood density
ZAR	South African rand

CHAPTER 1: INTRODUCTION

Bush encroachment is a form of land degradation described as an increase in the biomass, and abundance of woody species, and the suppression of perennial herbs. As a result, this causes an imbalance in the grass: bush ratio as well as a decrease in biodiversity and carrying capacity of rangelands (Mendelsohn, Jarvis, Roberts & Robertson, 2002; de Klerk, 2004; Wiegand, Ward & Saltz, 2005; von Oertzen 2009; Rohde & Hoffman, 2011). Bush encroachment has been documented for arid and semiarid rangelands much more frequently than for temperate regions (Oldeland, Dorigo, Wesuls & Jürgens, 2010; González-Roglich, Swenson, Jobbágy, & Jackson, 2014). African savannas that fall in the arid and semiarid region are known for high levels of grazing and browsing (de Klerk, 2004). Not only does the high level of herbivory modify vegetation structure and composition, it also reduces the growth and reproduction of individual plants thereby influencing the vegetation’s competitive abilities (de Klerk, 2004). The reduced competition from grass communities in overgrazed areas inevitably leads to an increase in the density of the woody species (Mukaru, 2009; de Klerk, 2004) causing bush encroachment. In most cases these changes are irreversible because of the effect of woody species on the grasses that compete strongly for moisture (Mukaru, 2009).

About 32% of Namibia’s arid to semiarid land area is encroached by invader bush resulting from suppressed bush fires, overgrazing, a decrease in grazers, and changing climatic conditions (de Klerk, 2004; de Wet, 2015). These factors lead to a reduction of livestock carrying capacity and economic losses due to a lower amount of pasture (Mendelsohn et al., 2002; de Klerk, 2004).

Bush encroachment affects Namibia on a massive scale, as it currently affects approximately 26 to 30 million hectares (32 %) of farmland in nine of the country's 14 regions (de Klerk, 2004; de Wet, 2015). It has also severely negatively impacted Namibia's beef production, which has decreased by about 50% of what it was in 1950s (NAU, 2010).

Bush encroachment is a result of land mismanagement that affects both the agricultural and the environmental sectors. Several scholarly works have shown that bush encroachment increases soil erosion and causes a disruption on the soil surface of some soils through the collection of raindrops in twigs; whereby big rain drops fall with a stronger spatter effect on soil (NAU, 2010; de Klerk, 2004). As a result of the disruption in the soil surface, soils experience nutrient leaching and moisture loss. This usually happen on sandy soils. With soils being one of the most important abiotic components of the ecosystem supporting biodiversity, changes in its structure and composition affect the distribution of plants and animals that depend on the soil for their existence (Esler & Cowling, 1993).

According to studies carried out by de Klerk (2004) and Hoffman, Schmiedel, & Jürgen (2010), soil moisture and nutrients are among the primary determinants of a savanna's functioning and dynamics. They have an overwhelming effect on vegetation structure, composition, and productivity (de Klerk, 2004; Whitford, 2002). Hence, prolonged denudation of soils caused by droughts and overgrazing, followed by above-average rainfall years with infrequent rainfall events favour mass bush recruitment at the expense of grasses, increasing bush encroachment (de Klerk, 2004). The alarming increase in bush encroachment makes it difficult to reverse without some external intervention (NAU, 2010).

Erichsfelde farm is a commercial enterprise found in the highly bush encroached area of the Otjozondjupa region, faced with various environmental challenges as a result of invader bushes. This study will determine the effect of bush encroachment on above- and below-ground biomass, standing woody carbon stock as well as its effect on the soil nutrients and fertility indicators on this farm. Knowledge obtained from this study will be used to answer some of the farmers' concerns and may assist land users in making informed decisions on sustainable rangeland management. Additionally, the study intends to provide baseline information on carbon stock bound in bushes and potentially enhance understanding of bush encroachment in a holistic context. Furthermore, the study intend to change the perspective that bush encroachment is a farmer's concern alone and advocate that interventions are needed from all sectors rather than the agricultural sector only.

1.1. Orientation of the study

For over a decade, bush encroachment has been considered a rangeland productivity problem and a concern for farmers (NAU, 2010). Although many farmers may see bush encroachment as an expensive nuisance, von Oertzen (2009) and Schultka & Cornelius (1997) argued that bushes are important for browsers, which constitute a significant source of income for the tourism industry, game farmers and conservancies. Furthermore, de Wet (2015) pointed out that the encroacher bush has developed into a huge biomass resource, estimated at about 200 to 300 million tonnes with opportunities for energy generation and value chain development in other sectors. Besides the economic benefits, bushes can sequester significant amounts of carbon dioxide, which can render Namibia a net carbon sink and generate revenues from carbon credits. Even though Namibia is a non-Annex I country under the Kyoto Protocol (von Oertzen, 2009) if bushes can sequester significant amounts of carbon dioxide, Namibia can generate revenues from these bushes. A non-annex I country is a developing country recognised by the United Nations Framework Convention on Climate Change (UNFCCC) as being especially vulnerable to the adverse impacts of climate change and prone to desertification and drought (MET, 2008).

Altered carbon sequestration is the transfer of CO_2 and secures storage of atmospheric CO_2 into long-lived pools (Takimoto, Nair & Nair et al., 2008). This CO_2 is then later re-emitted and is one of many potential biogeochemical consequences of bush encroachment (Lal, 2004; Archer, Boutton & Hibbard, 2001). In response to different types of land use changes, Henry, Valentini, & Bernoux (2009) state that the size and dynamics of the soil carbon pool of the world is still poorly known and that there are uncertainties concerning estimates of carbon in ecosystems. They further state that there is need to improve the understanding of the carbon cycle in Africa, especially with respect to the soil carbon pool. Thus, regional studies are necessary to refine global estimations obtained by the aggregation of regional estimates, mainly at country level scale (Henry et al., 2009).

Carbon trading is a relatively new concept, and is considered difficult and expensive to measure (Bruce, Sims, Walcott, Baldock, & Grace, 2009). A carbon trade is defined as an exchange of credits between nations designed to reduce overall carbon dioxide emissions in the atmosphere. Carbon trading which originated with the 1997 Koyoto protocol allows industrialized countries with a GHG reduction commitment to invest in mitigation projects in developing countries or countries with low carbon emissions to make up for their higher carbon dioxide release in the atmosphere (Takimoto et al., 2008; Oelbermann, Voroney, & Gordon, 2004). von Oertzen (2009) indicated that carbon sequestration and associated trading mechanisms from improved rangeland and soil management practices are being discussed at a variety of international fora. He also stated that there are considerable researches, development, and procedural gaps remaining before carbon revenues from biochar and biofuels from invading bushes can be generated. Biochar is charcoal created by burning organic material, such as woody vegetation, at high temperatures and can be used as a bush encroachment management strategy (von Oertzen, 2009; Mendelsohn et al., 2002; Bruce et al., 2009). This charcoal is a stable solid that is rich in carbon content, and can lock carbon into the soil (von Oertzen, 2009).

This study therefore intends to fill the knowledge gap that exists regarding the amount of carbon stock available in bush encroached areas and related nutrient availability using Erichsfelde farm a case study.

1.2. Statement of the problem

Bush encroachment has a negative effect on Namibia’s economy. From a government viewpoint, bush encroachment was considered an agricultural problem to be solved by farmers themselves (NAU, 2010). With many commercial farms in the Otjozondjupa and Khomas regions being highly bush encroached, it has become a national problem that needs government intervention (de Klerk, 2004; Coetzee, 2003;

Bester, 1999). This is because in the past bush encroachment was estimated to result in the instant loss of income of more than N$700 million per annum in beef production in Namibian commercial farms (de Klerk, 2004; NAU, 2010). Today, recent estimates put this lost income in beef production at about N$1.6 billion (NPC, 2012); while bush encroachment continues to cause extensive losses in land productivity resulting in lower food security and nutrition in communal farms (NAU, 2010).

According to Lal (2004) & Kuyah et al. (2012), carbon sequestration in vegetation and soil systems offers an opportunity for mitigating the greenhouse effect, but the relationship between soil carbon stocks and carbon fixation in natural vegetation and crops remains one of the least studied fields. To date, bush encroachment is still not well understood at a fundamental level, both by scientists and land owners who must deal with the problem in a practical manner (Smit, 2004).

Consequently, bush encroachment needs to be viewed within a holistic context. This would include economic losses to Namibia, environmental sustainability issues, climate change, and the negative affect on the provision of water supply beyond the farm boundary to support a range of economic activities including agriculture, mining, and urban supply (NAU, 2010).

Despite the fact that bush encroachment is known to decrease biodiversity and reduce livestock pasture, Namibia’s Greenhouse Gas (GHG) Inventory report to the UNFCCC underscored that there is a lack of knowledge as to how much carbon is bound in the bush encroached areas (MET, 2008). Well quantified data are lacking, hence more research is needed to significantly improve the understanding and speculation of bush impact on Namibia’s GHG profile. Woody vegetation represents a significant carbon pool and has been identified to play a role in carbon sequestration in some systems (Kuyah et al., 2012). According to San Jose, Montes & Farińas (1998), previous studies on bush encroachment focused on its effects on vegetation composition, biodiversity, and agricultural productivity rather than on the amount of carbon bound in the bush. Additionally, the effects of woody plant encroachment on carbon stocks in semi-arid rangelands are uncertain (González-Roglich et al., 2014). As such, little is understood on how bush encroachment affects carbon stock in vegetation and soil systems aswell as its affect on ecosystem functioning.

1.3. Aim of the study

The aim of this study is to quantify potential total woody carbon stock storage, above- and below-ground carbon stock, and assess selected soil fertility indicators in low, medium, and high bush encroached sites at Erichsfelde farm in the Otjozondjupa region.

1.4. Specific objectives

a) To determine and compare total woody carbon stocks in a low, medium, and high bush encroached site at Erichsfelde farm in the Otjozondjupa region.
b) To measure and compare above-ground (standing woody vegetation) and below-ground carbon stock in a low, medium, and high bush encroached site at Erichsfelde farm in the Otjozondjupa region.
c) To assess soil fertility indicators by measuring and comparing selected essential soil nutrients, organic matter, pH, electrical conductivity, and the sodium adsorption ratio in a low, medium, and high bush encroached site at Erichsfelde farm in the Otjozondjupa region.

1.5. Null hypotheses

a) There is no significant difference in the total woody carbon stock among the low, medium, and high bush encroached site.

b) There is no significant difference in the above- and below-ground carbon stock among the low, medium, and high bush encroached site.

c) There is no significant difference in the levels of essential soil nutrients, organic matter, pH, electrical conductivity, and sodium adsorption ratio among the low, medium, and high bush encroached site.

1.6. Significance of the study

While looking at carbon sequestration, this study will also focus on the entire ecosystem services, thereby answering more ecological questions related to bushes. Carbon stock results from this study will be used to evaluate the significance of encroaching bushes in storing carbon, and determine whether this can contribute to Namibia's carbon credits. Consequently, this will also extend the knowledge of bush encroachment in Namibia as measures to control and maintain bushes are explored. Likewise, a study of this nature will help determine and weigh the advantages of bush encroachment compared to the disadvantages. Moreover, outcomes of the study intend to contribute to the understanding of climate change mitigation by carbon sequestration, thereby providing baseline information for policy formulation and rangeland management.

1.7. Limitations of the study

Time limitation did not allow for seasonal comparisons and limited parameters that are dependent on seasonal changes to be measured and compared. Seasonal changes can have a potential effect on the vegetation biomass and carbon stock as vegetation is expected to have more biomass (leaves) during the rainy season than the dry season. Additionally, the current study did not examine the quantity of carbon stored in herbaceous vegetation and dead wood due to time and financial constraints. Herbaceous vegetation and dead wood contributes to above-ground carbon stock and plays a vital role in carbon storage.

CHAPTER 2: LITERATURE REVIEW

2.1. Introduction

Encroacher or invader bush is a term that refers to problematic woody species that are indigenous to Namibia that cause bush encroachment (NAU, 2010). Bush encroachment is a form of land degradation that is considered symptomatic of mismanaged savanna rangelands (de Klerk, 2004). This phenomenon has been observed across southern Africa where non-migratory cattle ranching on fenced-off commercial farms exist, causing significant financial loss in the commercial farming industry (DRFN, 2009). Namibian commercial farmers are reported to suffer a great deal from bush encroachment compared to farmers in other southern African countries such as Zambia (McGranahan, 2008).

According to de Wet (2015) and Kahumba (2010), current de-bushing methods used in Namibia can be categorised as:

- mechanical: highly mechanised, manual and semi-mechanised, and conventional medium to large-scale mechanised operations;
- chemical;
- biological; and
- other: this include fire-oriented methods such as flame throwers, controlled veld fires, and fire-herbivory interactions.

Chemical, biological, and mechanical methods used to control and manage bush encroachment have been studied for nearly a century (Bond et al., 2002; Kahumba, 2010; de Wet, 2015), yet there is neither clear agreement nor a standard method of controlling bush encroachment to date (Bond et al., 2002). Additionally, these proposed methods of controlling and managing bushes may cause other environmental problems such as a reduction in starch storage, de-nitrification, loss of soil fertility and introduction of other invader bushes (SASSCAL, 2012; Schutz, Bond & Cramer, 2009; Smit, 2004). Moreover, de-bushing methods can be costly and time-consuming (de Wet, 2015).

Despite the negative impacts on agricultural productivity and livelihoods, invader bushes present positive opportunities of energy generation as a result of improved ecosystem conservation, greater biodiversity, and poverty alleviation (DRFN, 2009; de Wet, 2015). Moreover, most are hardwoods that can be used for fencing, fire wood and carving, and represent social and economic importance to communal farmers. Ecologically, the encroachment of woody plants into grasslands, pastures, and croplands is generally thought to increase the carbon stored in these ecosystems even though there is evidence of a decrease in soil carbon stocks after land use change (Alberti et al., 2011). If substantial carbon stock is detected in bush encroached areas, it can potentially earn Namibia carbon credits that would provide farmers with incentives. Additionally, information on carbon stock could be used to develop viable land-use options for agricultural sustainability and rangeland management (Gnanavelrajah, Shrestha, Schmidt-Vogt, & Samarakoon, 2008).

Biochar production is a positive economic activity from bush encroachment that is not well studied and could be explored to help generate income (von Oertzen, 2009). Biochar is charcoal created by burning organic material such as woody vegetation at high temperatures, and can be used as a bush encroachment management strategy (von

Oertzen, 2009; Mendelsohn et al., 2002; Bruce et al., 2009). Not only can income be generated from the production of biochar, but it can prevent the rapid disintegration and release of carbon dioxide into the atmosphere when buried in the soil, thereby reducing the need for fertilisers and water. In turn, this improves arable land and agricultural yields (von Oertzen, 2009). Biochar can also be used for carbon sequestration due to its ability to lock carbon in the soil, which can increase soil organic carbon and increase soil fertility.

Organic carbon accumulation in soil reflects the balance of carbon inputs as both organic matter (returns of plant and root residues) and carbon losses from the soil (as carbon dioxide, dissolved organic carbon, and loss through erosion) (Lal, 2004; Alberti et al., 2012; Gnanavelrajah et al., 2008). Alberti et al. (2012) demonstrated that studies have been done on soil organic carbon on abandoned agricultural farms in some parts of the world such as Italy, and northern mid-latitudes. Studies conducted in Australia and Thailand also showed that increasing soil organic carbon has the potential to reduce atmospheric carbon and offer benefits for farmers through improved ecosystem services such as increased plant nutrient availability, improved physical and chemical soil condition, and increased productivity (Bruce et al., 2009; Gnanavelrajah et al., 2008). Namibia and other arid and semiarid areas in Africa with low soil fertility could greatly benefit from an increase in soil organic carbon, which could improve the livelihood of much of the population that directly depends on the land for survival. However, the aforementioned studies have shown that even if agricultural land abandonment is a widespread phenomenon, detailed studies on carbon sequestration in soils post abandonment are lacking, and few data are available to confirm or reject the contribution of carbon from soils and vegetation (Alberti et al., 2012). The same can be said for the contribution of bushes to carbon; literature is lacking to confirm whether bushes can sequester significant amounts of carbon to be considered for trading. Alberti et al.'s (2012) findings are supported by Bruce et al.'s (2009) overview on the role of soil in capturing and storing (sequestering) carbon emissions on Australian soils. Bruce et al. (2009) concluded that there are still elements of measuring and monitoring changes in soil organic carbon that need to be better understood if they are to have a role in a carbon trading scheme. With most of the studies focused on abandoned mines and agricultural land, studies on bush encroached commercial farm sites remain limited. Studies on soil organic carbon on commercial farm land use needs to be explored, particularly in southern Africa where commercial farming is common.

Global warming as a result of carbon emission by humans continues to threaten the world's environment today (Henry et al., 2009). However, sequestration of carbon could be one of the mitigation measures against the release of excessive CO_2 while simultaneously combating land degradation, which is a current environmental challenge as well (Lal, 2004). Vegetation plays an important role in sequestering atmospheric carbon and encroaching woody species are no exception (Lal, 2004, 2009). This is supported by Gnanavelrajah et al. (2008) who agreed that forests are best suited for carbon sequestration. Additionally, vegetation in woodlands and savanna ecosystems in arid and semiarid biomes also needs to be considered for carbon sequestration studies. Other agricultural land uses options need to be identified and carbon stock should be determined on this land. Commercial farms and bush encroached sites are examples of such land use that still need to be studied to determine whether they are capable of increasing carbon stock in soil and vegetation. The importance of CO_2 emission to the climate has provided the stimulus for research on the global carbon cycle with particular attention on carbon stocks in main terrestrial compartments, especially soils and phytomass (Henry et al., 2009).

2.2. Causes of bush encroachment

Bush encroachment is the the increase in density of woody plant species that results in a low grass to bush ratio that is associated with decreased grazing capacity of large areas (Smit, 2004; de Klerk, 2004; Mendelsohn et al., 2002; Throop, Lajtha, & Kramer, 2012). This usually happens to indigenous woody species occurring in their natural environment and is associated with the savanna biome of arid and semiarid areas (Smit, 2004; NAU, 2010; Oldeland et al. 2010). Causes of bush encroachment are still debated even though there is evidence to show that effects of poor land management, such as overgrazing by cattle, are the main factors that facilitate the spread of encroacher species (Oldeland et al., 2010; Bond et al., 2002).

According to Smit (2004), an increase in woody plant abundance is primarily brought about by two processes; the first is an increase in the biomass of already established plants (vegetative growth), and the second is an increase in tree density that generally comes from the establishment of seedlings (reproduction). Smit (2004) further explains that some influences may inhibit vegetative growth and/or reproduction, resulting in decreased biomass of woody plants. Hence, the reasons for an increase in the abundance of woody plants in any vegetation type are diverse and complex.

On the other hand, Eamus & Palmer (2007) reported the decreased stomatal conductance and increased rates of carbon fixation arising from an enriched atmospheric CO_2 concentration in conjunction with reduced rates of pan evaporation favours an increase in woody plant density. This is supported by Oelbermann et al. (2004) who mentioned that elevated atmospheric CO_2 concentrations could increase forest productivity. Therefore, a single factor cannot account for bush encroachment.

Like most environmental challenges, determinants of savanna systems are directly or indirectly modified by humans (Smit, 2004; de Klerk, 2004). These determinants may either be primary–such as climate and soil, or secondary–such as fire and the impact of herbivores (Smit, 2004; de Klerk, 2004). The latter are of particular interest since they can often be directly modified by management, although they act within the restrictions imposed by the primary determinants (Smit, 2004; de Klerk, 2004). These are modifications such as:

- the exclusion of occasional hot fires;
- the replacement of most indigenous browsers and grazers by domestic (largely grazing) livestock often at extremely high stocking rates;
- the restriction of movement of herbivores by the erection of fences;
- poor grazing management practices; and
- the provision of artificial watering points (Smit, 2004; DRFN, 2009; de Klerk, 2004).

African savannas have an evolutionary history of high levels of browsing by ungulates that are capable of significantly modifying the savanna structure and composition of woody plants if not well managed (Smit, 2004; de Klerk, 2004; DRFN, 2009). One way that humans indirectly modify savanna systems is by the removal of browsing ungulate game species and the introduction of livestock. This may contribute to the bush encroachment problem (Smit, 2004), and may explains the high bush encroachment situation on commercial farms such Erichsfelde in Otjozondjupa region, Namibia. According to Joubert, Rothauge, & Smit (2008) Namibian rangelands are encroached because of poor understanding of vegetation dynamics, phenology and physiology of encroaching species as well as inappropriate fire management of savanna ecosystem models. The aforesaid reasons explains why encroacher species need to be studied in-depth, and species-specific management and controlling methods need to be employed to minimise introduction of more problems in the efforts of eradicating another. As such, more studies like this one need to be carried out on bush and non-bush encroached sites to help understand the dynamics of woody encroaching species, and to treat the bush encroachment problem from its source. Moreover, this substantiates why bush encroachment is not only a farmer’s problem, but a challenge for ecologists, hydrologists, and decision-makers.

According to de Klerk (2004) and the NAU report (2010), if the grass layer is over utilised in an ecosystem, it loses its competitive advantage over trees and shrubs, and so can no longer use water and nutrients effectively. This results in high water and nutrient infiltration into the subsoil, further causing a high bush and tree density relative to the grass density. This is because grasses have fibrous root systems that cannot penetrate deep enough to reach the water table. Also, when the grass layer is over utilised, bare spaces are left for trees and shrubs to grow, and typically out-compete the grass species. However, Ward (2005) mentioned that bush encroachment is not entirely a result of overgrazing. This is because bush encroachment is widespread in areas where there is a single soil layer and where grazing is infrequent and light. On the other hand, de Klerk (2004) suggests that bush encroachment is a temporary event-driven phenomenon that depends on the state of environmental conditions such as rainfall and its variability. Instead, Ward (2005) indicated that single-factor explanations are not the answer to questions of bush encroachment, which is why challenges to control and manage bush encroachment still exist. If single-factors were the cause of bush encroachment, efforts to reverse land degradation would not be so challenging. Ward (2005) further explained that mechanistic models and multifactorial experiments are needed for guidance to tease out the interactions among causal factors. According to Wiegand et al. (2005) and Joubert et al. (2008), savannas can be interpreted as patch-dynamic systems where landscapes are composed of many patches in different states of transition, lying anywhere between grassy and woody dominance. However, within the paradigm of patch-dynamic savannas, bush encroachment is part of a cyclical succession between open savanna and woody dominance (Wiegand et al., 2005). Wiegand et al. (2005) and Oldeland (2010) attributed the conversion from a patch of open savanna to a bush-encroached area to being initiated by the spatial and temporal overlap of several localised rainfall events sufficient for the germination and establishment of dominating *Vachellia* and *Senegalia* (Kyalangalilwa et al., 2013) encroacher species. Again, localised rainfall events cannot fully account for the initiation of bush encroachment, because the

Otjozondjupa Region in the northern part of Namibia that receives relatively abundant rainfall also suffers from this phenomenon. This is true to a certain extent as climate variability can also account for bush encroachment. The reason for this is because invader bushes have long tap roots and can penetrate deep into the soil to reach the water table when rainfall is scare, unlike its grass and browser bush competitors. However, rainfall cannot be the only factor responsible for bush encroachment because areas with abundant rainfall are also heavily encroached.

In their paper, Wiegand et al. (2005) show support for factors; such as overgrazing conventionally claimed to causing bush encroachment. Whereas, factors such as rainfall amount and frequency, coupled with specific soil nutrient levels, may actually drive the bush encroachment phenomenon.

Joubert et al. (2008) indicated how bush encroachment is a natural process that is accelerated by anthropogenic activities. Bush encroachment consists of a patchy mosaic of all states, where thickets form patches, while the open, grassy states form the matrix. Thickets is a type of vegetation transistional between savanna and forest, that is almost impenetrable, not divided into strata and with variable herbaceous cover (Coetsee, Gray, Wakeling, Wigley & Bond, 2013). Thus, dense thickets often made up of thorny or unpalatable bushes negatively affect the carrying capacity of the land, which has a further negative affect on the economic value of rangelands (Oldeland et al., 2010).

In general, savanna ecosystems consist of a discontinuous crown of tree cover and shrubs, with an undergrowth of grasses (Archibold, 1995). Joubert et al. (2008) explained this phenomenon as follows: a mature tree or thicket of mature trees may constantly attract grazers, which reduces grass cover and the chances of fire, thus allowing seedlings to survive. Away from the trees however, grass cover may be sufficient to initiate an adequately hot fire. A fire may destroy some of the establishing seedlings, but may fail to affect others, thereby resulting in a shifting mosaic effect. It is when conditions are uniformly suitable for a certain transition on a landscape level, such as when grazing at high intensities is maintained throughout the year over a period of several years, and fire is completely excluded, that thicket patches form a matrix as the mosaic gradually deteriorates. This is often the case on commercial farms where large areas of rangeland are subjected to a rigid grazing plan, and fire is deliberately excluded, or where overgrazing has reduced the fuel load. Furthermore, the mosaic effect is a good example to explain why bush encroachment cannot be caused by rainfall only, but also by the exclusion of fire and overgrazing. Joubert et al. (2008) further pointed out how bush thickening may be self-perpetuating, since it forces farmers to overgraze existing open patches, thus increasing the likelihood of a transition from grassland to bush in these areas. The disagreement between authors on what causes bush encroachment is a clear indication of how this phenomenon still has many knowledge gaps and needs to be studied in depth. Despite these gaps, evidence shows that bush encroachment is caused by different factors that cannot be studied in isolation, and neither management nor control of this land degradation should be left to one sector only.

2.3. Extent of bush encroachment in Namibia

Over the past 150 years, bush encroachment has drastically changed land cover patterns in drylands in many parts of the world including Nambia (Throop, Lajtha & Kramer, 2012). According to Rohde & Hoffman (2011), various sources of evidence suggest that landscapes of central and southern Namibia have changed drastically since the 19th century. Bush encroachment is widely spread in Namibia and affects both commercial and communal farming areas (de Klerk, 2004). de Klerk (2004), de Wet (2015) , NAU (2010) and Buyer, Schmidt-Küntzel, Nghikembua, Maul & Marker (2016) showed how 32% of Namibia's surface area is encroached by invader bushes on approximately 26 million hectares of woodland savannas, which includes about 15,777,000 ha in communal and 10,482,000 ha in commercial land. Almost 64% of the country is covered by savanna range types with open to moderately dense *Vachellia* and *Senegalia* trees and shrubs (de Klerk, 2004). Six of Namibia's 14 major vegetation types are characterised as savanna biomes and include areas generally associated with bush thickening (de Klerk 2004; Mendelsohn et al., 2002). These areas are: Dwarf Shrub Savanna, Camelthorn Savanna, Highland Savanna, Mixed Tree-and-Shrub Savanna, Mountain Savanna, and Thornbush Savanna, where Erichsfelde is situated. Savanna ecosystems associated with high bush encroachment are characterised by rainfall of <600 mm per year (Mendelsohn et al., 2002), and are equally associated with low carrying capacities, making them vulnerable to bush encroachment (NAU, 2010; de Klerk, 2004).

According to Oldeland (2010), species of the genera *Vachellia* and *Senegalia* in Africa are often the dominant encroachers. In Zanzibar, *Vachellia auriculiformis* A.Cunn. ex Benth. has been identified as a heavy encroacher, while in Botswana, *V. tortilis* (Forssk) Galasso & Banfi, and *V. erubescens* Welw. ex Oliv. have been reported to increase in rangelands. On the other hand, Namibia is heavily encroached by *Senegalia mellifera* (Vahl) Seigler & Ebinger, *V. erubescens*, and *V. reficiens*

(Wawra) Kyal. & Boatwr. but this is not limited to the *Vachellia* and *Senegalia* genera as *Dichrostachys cineria* Wight et Arn., *Terminalia sericea* Burch. ex DC. , *T. prunioides* M.A.Lawson and *Colophospermum mopane* (Kirk ex Benth.) Kirk ex J.Léonard are amongst the other encroaching species in Namibia (de Klerk, 2004; NAU, 2010). From de Klerk's (2004) and Coetzee's (2003) maps, *S. mellifera* subsp. *dentinens* (Burch.) Kyal. & Boatwr. seems to be the most common encroaching species in the eastern, northern-central, as well as the central parts of the country where Erichsfelde is located. The southern region of the country is dominated by *Rhigozum trichotomum* Burch., while the far northern parts are encroached by *C. mopane* and *D. cineria.* As various species are associated with assorted soil types, encroaching species occur in distinct parts of the country as a result of their respective root systems' ability to extract and intercept water (NAU, 2010).

2.4. Impacts of bush encroachment

2.4.1. Impact on agriculture

Land degradation reduces rangeland productivity. Bush encroachment is a form of land degradation that is estimated to have an annual economic impact of about N$1.6 billion in meat production losses on freehold farming areas (Diop, 2012). This and climate variability, among other factors, make farming in Namibia a risky business (Mendelsohn et al., 2008; NAU, 2010). Indeed, nearly 50% of commercial ranching areas of Namibia are affected by bush thickening, mainly by *Senegalia mellifera* subsp. *detinens* (Joubert et al., 2008), which has serious economic implications on Namibia's agricultural sector (NAU, 2010). In such areas, rangeland productivity and capacity has declined spontaneously over the last century with the carrying capacity of rangelands decreasing from one Livestock Unit (LSU) per 10 hectares to one LSU per 20 or 30 hectares every year (de Klerk, 2004; Ward & Ngairorue, 2000). The severe impact of bush encroachment on land productivity and biodiversity poses a major socioeconomic threat, not only to the welfare of livestock farmers, but to all downstream industries (abattoirs, meat wholesalers, exporters) and their employees (NAU, 2010). Whenever bush encroachment occurs, arboricides are often used as a chemical bush control agent, with predominantly negative effects on the environment as well as on the quality of farm products such as the meat that supplies local markets (Oldeland, 2010). As such, an increase in bush encroachment means a decrease in quantity of products. In turn, this causes a decrease in the contribution of the agricultural sector to the Gross Domestic Product (GDP) as shown by the decline from 7.4 % in 1980 to 3.4 % in 2014 despite the agricultural sector being the backbone of Namibia's economy that sustains some 70% of the population (MTI, 2014). The GDP is the monetary value of all the finished goods and services produced within a country's borders in a specific time period (www.investopedia.com, 2016).

Ward & Ngairorue (2000) reported a 50% decline in grass biomass per unit annual rainfall from the biomass reported previously by Walter (1939) along a rainfall gradient in Namibia, in spite of a lack of change in rainfall in the interim. A decrease in grass biomass consequently results in a decrease in livestock forage. Most encroaching woody species of *Vachellia* and *Senegalia* are not favoured by domestic livestock, which therefore decreases the amount of forage in a given area (Katjiua & Ward, 2007). This is because most encroaching woody species are defended from herbivores by thorns and spines. Any decrease in forage area can affect Namibia's economy, as it is a beef export country (de Klerk, 2004). Certainly, this decrease in forage caused by bush encroachment has reduced the number of cattle on commercial farms by 47% over the past 30 years and results in a low carrying capacity of the land, affecting about 70% of the population that depends on land directly and indirectly (de Klerk, 2004; Smit, 2004). This is often to such an extent that many livestock enterprises can no longer remain economically viable.

In contrast, physically unprotected species such as *Grewia flava* DC. (= G. cana Sond., G. hermannioides Harv.), *Bauhinia petersiana* Peters, 1861 and *T. sericea* may provide valuable browse to livestock (Katjiua and Ward, 2006). Consequently, while bush encroachment may reduce range productivity on some rangelands, it may also provide valuable forage for livestock production during times of drought when grazing is scarce, thereby serving as a source of fodder, particularly on communal farms (de Klerk, 2004; Katjiua & Ward 2006). However, Namibia cannot fully benefit from these bushes because most of them belong to the *Senegalia* and *Vachellia* genera that have thorns and spines, which are not readily browsed. Furthermore, game farming that could make use of bushes as forage is not highly practiced in Namibia.

Increases in woody plant cover do not only cause losses of beef production, it also alters the distribution of carbon in the ecosystem and can affect water and nutrient cycling (González-Roglich et al., 2014). NAU (2010) found that bush encroachment can negatively affects 89% of Namibia's high groundwater potential areas and 52% of moderate groundwater potential areas, thereby limiting water supply for high value activities such as industry, mining, and urban development. This is because encroaching woody species has long tap roots that easily deplete the water table. With such a great loss by bush encroachment, reactive interventions are the norm. Thus, de-bushing programmes need to be put in place to help controlled and manage bushes. Despite such interventions, bush thickening remains a problem yet to be fully understood by researchers, farmers and decision-makers at large (Katjiua & Ward, 2007).

2.4.2. Impact on biodiversity

According to the NAU report (2010), bush encroachment is not only an economic problem, but also an environmental one, as dominant aggressive invader bushes reduce the country's biodiversity and expose soil to erosion by limiting grass growth beneath the bush canopy. However, if carbon is sequestered and soil organic carbon increased, soil erosion can be controlled because soil organic matter stabilises other parts of the soil, binding soil particles into aggregates that are more resistant to erosion (Bruce et al., 2009; Sundurmeier et al., nd).

Many semi-arid and arid savanna areas in southern Africa are water-limited ecosystems, and bush encroachment is considered a major factor contributing to the low occurrence or in severe cases even the total absence, of herbaceous plants (Smit, 2004). This is because bushes are able to out-compete herbaceous plants due to the herbaceous plants' extensive root system that can not reach for underground water compared to the bushes long tap root that draws water from the water table. Studies in southern Africa and around the world showed that trees and bushes with deep roots use large volumes of water as opposed to grasses (NAU, 2010; Eamus & Palmer 2007), thereby allowing them to displace other plant species such as herbs and cause a significant decline in plant diversity (NAU, 2010).

Tree roots have preferential access to soil water in the topsoil and subsoil while roots of grass species usually occur in the topsoil layer. Consequently, grass-tree competition is vastly influenced by the amount of water retained in the upper soil layer that moves to the lower soil where tree roots predominate and minimal grass roots are found (de Klerk, 2004). In cases where the grass layer is over utilised through poor management, it loses its competitive edge over woody species and leads to bush encroachment (de Klerk, 2004). Grasses are unable to compete effectively with bushes and trees when the woody component is too dense (NAU, 2010). Perennial grasses are thus gradually replaced by annual grasses, which eventually die out, leaving large bare areas with no ground cover beneath the bush canopy. If the grass layer is detrimentally affected by poor management or environmental factors, it loses its competitive advantage and can no longer utilise water and nutrients effectively (de Klerk, 2004). Once again, it is out-competed by the woody species and results in an ecological system that is dominated by bushes, causing low vegetation diversity. Moreover, denudation of land during prolonged drought, followed by a few wet years provides favourable conditions for bushes to establish (de Klerk, 2004). As a result, fewer non-woody species are found to take root in the bush encroached vegetation, which again reduces the diversity of both the vegetation in these areas and the fauna that depend on it.

The NAU report (2010) also indicated that heavily encroached areas are characterised by denuded soils with a sparse annual grass cover and few or no perennial grasses. As bush density increases, the cover of perennial grasses decreases significantly, as does the total grass cover. With the soil surface unprotected by herbs, rainfall runoff is rapid and infiltration is low with an even lower total volume of runoff. Together with the high evapotranspiration rate associated with dense bush, land productivity is thus adversely and severely affected (de Klerk, 2004). Furthermore, plant and animal diversity are negatively affected by the decrease in vegetation structural diversity, leading to a change of ecosystem functioning (Oldeland, 2010). Moreover, the dense woody bushes also have adverse impacts on water-use efficiency and underground water tables, therefore contributing to the process of desertification which can drastically change the floristically diverse status of a country (de Klerk, 2004). Thus, an increase in an abundance of bushes does not mean an increase in vegetation diversity, as mostly woody encroacher species are found in the area where these bushes occur. In addition to that, the low vegetation diversity negatively affects the fauna and microbial diversity that depend on the different plants thereby, decreasing the overall biodiversity of the ecosystem.

2.4.3. Bush encroachment and climate change

Climate change refers to a collection of large-scale, long-term changes to global weather conditions. Examples include increases in temperature, rainfall, and increased frequency of drought and flooding due to a significant departure of the earth's climate from average weather conditions (MET, 2008; IPCC, 2001). Climate change stands out as one of the major challenges of the 21st century, and threatens progress in the achievement of national and sustainable development goals of many countries (NPC, 2013). Since the beginning of the industrial revolution, carbon dioxide (CO_2) concentration in the atmosphere has been rising alarmingly from around 290ppm in 1850 to 385ppm in 2009 (Kumar et. al, 2006; Lal, 2009). The drastic increase in atmospheric concentration of CO_2 and other greenhouse gases (GHGs) that contribute to climate disruption are driven by the increase in the world's population (Lal, 2009). In 2014, the world's population was approximately 7.1 billion and is estimated to increase to 9.2 billion by 2050 (www.worldometers.info, 2015). Also, heavy reliance on the burning of fossil fuel, logging, along with biomass burning, are all sources of GHGs (Lal, 2009; San Jose et al., 1998; Oelbermann et al., 2004).

Alteration of the global climate characterised by rising temperatures (global warming) is thought to be the result of CO_2 emissions and the accumulation of greenhouse gases (GHGs) (Smit, 2004; Lal, 2004; Beets et al., 2012). Global warming is primarily the result of post-industrial revolution human activity (IPCC, 2001; Hairiah et al., 2001a). Sufficient scientific evidence shows that although climate may vary naturally, human activity, mainly through the burning of fossil fuels and deforestation due to rapid increase in global human population, has caused significant changes in global climate (MET, 2008; Smit, 2004 IPCC, 2001).

Burning of fossil fuels has contributed to accumulation of greenhouse gases such as methane, nitrous oxide, and carbon dioxide (MET, 2008; Oelbermann et al., 2004). This tremendous rate of increase in concentrations of greenhouse gases in the atmosphere led to the development in 1994, of the United Nations Framework Convention on Climate Change (UNFCCC) to help monitor and control these emissions (Smit, 2004).

As with many environmental challenges, bush encroachment is interlinked with climate change. Rangelands comprise about 40 % of the global land surface and changes in their vegetation cover can significantly influence global climate (González-

Roglich et al., 2014). Research has shown that increases in atmospheric CO_2 improve water efficiency and increase carbon uptake in *Vachellia* and *Senegalia* trees –the most prominent encroaching bush in the study area (Bond et al., 2002). This is because at low CO_2 levels, plants have to open up their stomata wider to take up CO_2 from the atmosphere. This results in a high evapotranspiration rate that causes drier conditions for plant growth (Bond et al., 2002). The opposite is true for high CO_2 levels that create moister conditions because the stomata does not have to open up widely to take up CO_2 from the atmosphere. Trees, unlike grasses, need to invest carbon into woody structures to attain height, and so this large carbon demand can be met more efficiently and quickly under elevated CO_2 conditions. Subsequently, changes in atmospheric CO_2 increase the probability of woody plants growing into fire-resistant sizes and alter the bush: grass ratio in favour of bushes. To summarise, greater fluctuations in increased temperatures as a result of climate change and increasing CO_2 levels reduce the transpiration rate of grasses and favours invader bush growth.

In variable and changing weather conditions, in some areas, climate change accelerates and prolongs the severity of droughts (Mendelsohn, 2002). When droughts are prolonged, encroaching woody species are favoured because they can still extract moisture far below the soil's top layer due to the presence of long tap roots and lateral root systems, unlike grass species. Rooting depth is an important functional trait in most savanna ecosystems because niche separation of soil water use at different depth is believed to be an important mechanism to tree-grass coexistence (Liu, Archer, Gelwick, Bai, Boutton, Wu, 2013). Grasses usually have fibrous root systems that are restricted to the top soil layer, so they suffer severely during extended periods of drought when moisture in the topsoil layer is severely diminished (NAU, 2010). Also, when encroaching woody species are developing, they can quickly grow roots deeper than the zone utilized by grasses, which enhanced their survival during the earlier stage of their life history (Liu et al., 2013). In addition to that, Ward (2010) indicated that C_3 trees are expected to have a higher net photosynthetic rate than C_4 grasses which predominate in savannas. Furthermore, trees may be better defended and, thus, lose less material to herbivory. This will cause trees to have higher growth rates than grasses and may explain why encroaching woody species are able to outcompete grasses during droughts and localised rainfall events that result in bush encroached areas in low and infrequent rainfall ecosystems such as savannas.

Similarly, in some areas climate change tends to reduce levels of soil moisture, soil carbon, nutrients, and microorganisms that fix nitrogen (NAU, 2010). Nitrogen fixing organisms are capable of transforming atmospheric nitrogen into fixed nitrogen that is usable by plants. In some areas, effects of climate change increase atmospheric temperatures, reduce rainfall infiltration, and cause drought that, in turn reduces soil moisture. Furthermore, when soils are dry, nutrient availability also decreases. This is because nutrients especially ions need moist environments to be taken up by plants and nitrogen-fixing organisms need moist environments to thrive. Accordingly, it is of vital importance to manage bushes to improve grass cover, especially perennial grasses that protect the soil and enhance infiltration of rainfall and discourages drought conditions (NAU, 2010).

2.4.4. Impact of bush encroachment on carbon stock and soil nutrients

Soil is one of the most important abiotic components of ecosystems that support biodiversity (Hoffman et al., 2010). Soil acts as both a source of greenhouse gases and a sink for carbon (Bruce, 2009; de Wit, Palosuo, Hylen & Liski, 2005). Studies indicate that the structure and process of terrestrial ecosystems, especially those of vegetation, are largely influenced by soil chemical and physical properties (Hoffman et al., 2010; Whitford, 2002).

While invader trees and shrubs in savanna ecosystems compete with grasses for moisture, trees also maintain soil fertility, which also benefits grasses (Hoffman et al., 2010). These trees and shrubs create fertility islands beneath their canopies thus increasing the organic matter and enriching the top soil with nutrients (Hoffman et al., 2010; Smit, 2004). This demonstrates why it is of great importance to strike an optimum balance between trees, shrubs and grasses in order to help maintain soil fertility, prevent soil erosion, and optimise grazing resources. Vegetation supports different life forms such as fauna and microbes. In savanna ecosystems, small mammals burrow near vegetation where soil is loose and easy to dig. Likewise, trees and shrubs provide shade for burrowing mammals who then contribute to soil fertility when they defecate, as well as when they later die and decompose. Birds equally contribute to soil fertility by building nests in trees and shrubs. When vegetation sheds leaves or dies, its biomass contributes to soil fertility thus creating fertility islands under trees –and-shrub canopies.

There are also human activities that can alter the soil. These activities disturb soil components such as soil horizons, soil microbes, the nutrient cycle, and structure

(Sheoran et al., 2010). A study conducted by Balesdent et al. (1998) indicated that human activities such as converting forests into cultivated land and grassland encourages changes in soil nutrients. The study further revealed that the conversion of lands from forest to agriculture generally leads to a decrease in the soil organic matter pool, which also contributes to the growth of invasion bushes. This can be attributed to the fact that grasslands and cultivated lands have less biomass and do not return high levels of nutrients into the soil compared to trees and shrubs. Consequently, a decrease in the soil organic matter pool as a result of low return of nutrients to the soil is an indication of minimal soil organic carbon. Soil organic carbon is a major component of the global carbon cycle and accounts for more carbon than the terrestrial biomass and atmospheric pools combined (Throop et al., 2012).

Bush encroachment from nitrogen-fixing trees such as *Vachellia* and *Senegalia* species is well-documented in some parts of the world (Creamer et al., 2012). However, for other parts such as Namibia, only the extent of bush encroachment is known but their effects on carbon stock and soil nutrients are less studied. In Creamer et al.'s (2012) study, it was indicated that in the Rio Grande Plains region of southern Texas, woody encroachment by leguminous *Prosopis glandulosa* Torr. var. *torreyana* (Benson) Johnson trees increased soil carbon and nitrogen. These leguminous trees decrease microbial nitrogen biomass relative to soil nitrogen and accelerate nitrogen mineralisation and nitrification when fixing nitrogen. This increased organic carbon pool from leguminous plants in terrestrial ecosystems can sequester large amounts of carbon for long time periods. Thus, invader bushes that fix nitrogen could play an important role in the sequestration of carbon. Alberti et al. (2012) and Bruce (2009) carried out detailed studies in which they quantified carbon in the soil in Italy and Australia. Their studies estimated that soils contain approximately twice or three times as much carbon as the atmosphere or terrestrial vegetation. This explains why a relatively small increase in the proportion of soil carbon could make a significant contribution to reducing atmospheric CO_2 (Bruce, 2009). If reductions in soil carbon in woody encroached ecosystems are found where high rainfall levels and significant nitrogen losses can occur, then bushes should be reduced.

Soil organic matter is described as any biological material that decomposes and becomes part of the soil (Bot & Benites, 2005). This organic matter is created by the cycling of organic compounds in plants, animals, and microorganisms into the soil (Sundurmeier et al., n.d.). The magnitude of soil organic matter and carbon stock results from an equilibrium between the inputs (mostly from biomass detritus) and outputs (mostly decomposition and transport) to the system that are driven by various parameters of natural or human origins (Lal, 2004; Henry et al., 2009; Alberti et al., 2011). According to de Klerk (2004) and Henry et al. (2009), organic matter is of vital importance in agriculture because it can reverse the effects of denudated soils. This takes place when organic matter increases water-holding capacity of the soil and thus the proportion of water available for plant growth (McCauley & Jacobsen, 2005). This is because low amounts of organic matter do not aggregate soils, allowing water to penetrate the soils deeper (Henry et al., 2009). This further promotes encroaching woody plant growth since they are able to extract water from the deep water table something grasses cannot do due to their shallow fibrous root systems. A decrease in soil organic matter can also negatively affect soil structure stability, compactness, nutrient storage and supply, and soil biological life such as mycorrhizae and nitrogen-fixing bacteria (Henry et al., 2009). This happens because the soil becomes less aggregated and causes its soil exchange capacity to decrease. Additionally, soil organic matter found on the soil's surface helps protect the soil from the effects of wind, rainfall, and sun and with low organic matter; soil erosion increases (Bot & Benites, 2005). Consequently, soil organic matter is a key component of any terrestrial ecosystem, such that any variation in its abundance and composition has important effects on many of the processes that occur within this system (Henry et al., 2009). It follows then, that a soil with high organic matter is more productive than a soil where much of its organic matter has been lost through poor management practices, soil surface runoff, and erosion (Sundurmeier et al., n.d.).

Nitrogen cycling in semiarid and arid grassland ecosystems becomes dramatically altered following the encroachment of woody leguminous trees and shrubs. This is due to nutrients that become rapidly concentrated under the developing canopy and amplify a number of biogeochemical feedbacks within the system that can accumulate or reduce soil organic matter (Creamer et al., 2012). Soil nutrients and organic matter have a propensity to be concentrated in the upper 2-5 cm of the soil, with the greatest amounts found beneath the canopies of individual shrubs (Ward, 2009). Additionaly, nitrogen-fixing invader bushes do not guarantee an increase in soil nutrients. The dry conditions of semiarid and arid grasslands with their low cation exchange capacity make soil nutrient accumulation even more difficult.

Smit (2004) suggested that nutrients such as nitrates, phosphorus, a series of anions and cations, and various trace elements, are all essential to plant nutrition, as they act as determinants of the composition, structure, and productivity of vegetation. While the base richness of the parent material is initially important in determining soil fertility, biological activities on base-poor substrates are important in the creation and maintenance of localised areas of enhanced soil fertility (Smit, 2004). Hence, studies on soil organic matter and how bush encroachment affects soil nutrients are of great importance.

According to Du Preez et al. (2011), most South African soils have low organic matter levels because of low rainfall that leads to poor plant growth. Similar to South Africa, Namibia´s climate has been generally dry for many millions of years and as a result, there is a lack of deep soils around the country and evidence of low levels of soil nutrients and organic matter in most of the soils (Mendelsohn et al., 2009). A study conducted by Heathcote (1983) found that the low available moisture in arid environments slows down both the chemical processes, and the breakdown of plant materials into organic matter and humus. Consequently, the depth and quantity of both soil organic matter and humus in arid soils is low and declines rapidly down the rainfall gradient. Organic matter is the major source of nutrients such as nitrogen, available phosphorus, and potassium in unfertilised soils such as desert soils (Smit, 2004). Thus, given that desert soils are low in organic matter due to their low litter supply and rapid mineralisation, amounts of these nutrients are expected to be low too. However, if soils in bush encroached areas are found to be rich in calcium, nitrogen and phosphorus, which occurs due to contributions by nitrogen-fixing invader bushes, they could potentially enhance microbial activity and nutrient availability in the soils, provided that temperatures are favourable. Yet, as woodland ecosystems are changed into agricultural lands, and systems are covered with invader bushes, the physical protection of soil organic matter within stable aggregates will be reduced by soil tillage and the presence of bushes (Balesdent et al. 1998).

Moreover, trees act as biological agents by creating islands that differ from the bare soils (Smit, 2004; Joubert et al., 2008), and provide a habitat for grass species to grow. However, any decrease in moisture content may cause these grasses to be out-competed by the trees and shrubs forming the islands. Sufficient evidence in support of soil enrichment under tree canopies exists with regard to total nitrogen, soil organic carbon, and exchangeable cations such as calcium, potassium, magnesium, and sodium (Smit, 2004). In his study, Smit (2004) indicated that many theories have been presented as sources and mechanisms of soil enrichment under tree canopies, such as litter from leaf fall. Other possible sources of nitrogen enrichment include the occurrence of nitrogen fixation from microbial association with leguminous trees, bird droppings, as well as dung and remains of large and small mammals that spend time under trees (Smit, 2004; Joubert et al., 2008; Balesdent et al., 1998). This shows that savanna woody plants are essential biological agents that contribute to areas of enhanced soil fertility, which will be lost over time by the complete removal of all encroaching woody species. This is especially important on nutrient-poor soils as occurring in semiarid and arid ecosystems. Thus, encroaching woody species need to be controlled and managed rather than completely eliminated due to their ecological role in ecosystems.

2.5. Importance of carbon stock

Carbon management is a serious concern that challenges the world today. A number of summits have been organised on this subject ranging from the Stockholm convention to the Kyoto protocol (Kumar, Pandey & Pandey, 2006; Oelbermann et al., 2004). Carbon sequestration which refers to the removal of CO_2 from the atmosphere through vegetation and storing the carbon in soil in the form of soil organic matter, to be reemitted later is one carbon management challenge faced today (Kumar et al., 2006; Lal, 2004; Bruce, 2009; Sundurmeier, et al., n.d.). If the CO_2 in the atmosphere is not sequestered, it may have a detrimental effect on net agricultural productivity, human health, property damage due to increased flood risk, and on the value of ecosystem services (Kumar et al., 2006).

As with any other continent, Africa plays a growing role in the carbon cycle. However, Africa is one of the weakest links when it comes to understanding the global carbon cycle, particularly when considering the soil component (Henry et al., 2009). This is evident in the low number of studies done on soil carbon in Africa as compared to Asia and Australia (Lal, 2004; Alberti, et al., 2011).

Soil carbon consists of organic and inorganic forms (Bruce, 2009; Schumacher, 2002; Hairiah, et al., 2001a). Inorganic forms of soil carbon exist as calcite and dolomite, which are relatively stable and less effected by land management (Bruce, 2009; Schumacher, 2002). On the other hand, organic carbon is more manageable as a carbon store, and accounts for roughly half the soil organic matter mass. It is the most studied form of carbon because of its effects on agricultural production and carbon sequestration (Bruce, 2009). For the purposes of this study, the soil organic carbon will be the focal point.

According to Henry et al. (2009), estimates of global soil carbon storage over the past 70 years have ranged from 400 PgC to 9,120 PgC. PgC been is a Petagram of carbon, also known as a Gigaton (Gt), and is equal to 10^15 grams or one billion tonnes. Although more recent studies estimate this range to be between 1,115 PgC and 2,200 PgC in the first metre of soil, they generally converge on a value of about 1,500 PgC, while plant biomass is estimated to range between 560 PgC and 835 PgC (Henry et al., 2009; Janzen 2004). Additionally, Hairiah et al. (2001a), the IPCC (2007) and Lal (2009) showed the main CO_2 sinks to be the atmosphere that takes up about 34%, followed by land-based sinks or soils at 27%, oceans at 23%, and vegetation at 16% yearly. Soils are found to store more carbon than plant biomass. This increase in global soil carbon over the years can be attributed to the increase in CO_2 that has resulted from the high reliance on burning of fossil fuels over the past century. Furthermore, the world's mineral soils represent a large carbon reservoir that accounts for about two-thirds of global terrestrial carbon stocks, that is why soils store more carbon than vegetation (Henry et al., 2009; Janzen 2004; Oelbermann et al., 2004). Similarly, soils contain nearly as much carbon as vegetation in a rainforest, but considerably exceed the biomass in other ecosystems by a factor of 10 to 2 (Henry et al., 2009; Bruce, 2009).

The flows of carbon between pools and their feedbacks have kept atmospheric CO_2 reasonably constant for millennia (Janzen, 2004; Lal, 2009). However, humans have increasingly distorted this balance by changing land use and by injecting fossil carbon back into the cycle (Janzen, 2004). Consequently, atmospheric CO_2 has increased recently by more than 3 PgC per year. Indeed, its concentration may rise to twice that of pre-industrial levels by the end of century (Janzen, 2004; Lal, 2009). In the case of Africa, land use has changed dramatically over a century due to industrialisation (Lal, 2009). African human populations are more dependent on land today compared to when they were nomads. In fact, more rural areas are being turned into urban areas. This increases deforestation and the burning of fossil fuels and in turn increases CO_2 in the atmosphere.

Carbon is taken up by plants in the growth process and stored in above- and below-ground plant biomass (Lal, 2004).

In savanna ecosystem, above-ground plant biomass is expected to have less amounts of carbon stock as opposed to below-ground biomass, since the vegetation of this ecosystem accumulate most of their biomass underground to protect them from fire and herbivory (Grace, Jośe, Meir, Miranda, & Montes, 2006). In addition to carbon uptake, leaf litter production and biological processes may lead to the accumulation of carbon in the soil, which explains why vegetation is considered to create fertility islands (Turpie et al., 2010; Smit, 2004). As plants are the main source of soil organic carbon, the amount of carbon stored in plant biomass is a relatively constant fraction of total mass or demand for structural allocation. Whereas the rate of carbon uptake from the atmosphere depends on the growth and photosynthetic rates of these plants (Turpie et al., 2010; Bond et al., 2002; Kumar et al., 2006). On the other hand, the amount of carbon stored in soils varies according to vegetation cover, existing levels of carbon, soil type, temperature, rainfall, land use, and land management (Turpie et al., 2010; Bruce, 2009).

Studies carried out in neighbouring South Africa have shown how carbon sequestration in ecosystems has a positive economic value. Conservative estimates indicate that climate change and the increase in CO_2 in South Africa will carry a cost of about 1% to 2%, and possibly as much as 6%, of Gross Domestic Product (GDP) by 2050 (Turpie et al., 2010). This cost is attributed to changes in ecosystem productivity, ecotourism opportunities, disease vectors, and agricultural production due to infrastructural damage. Turpie et al. (2010) further estimated the damages of increase atmospheric CO_2 to be equivalent to about ZAR80 (NAD 80) per ton of carbon emitted, taking into account the fact that carbon contributes about 60% of total greenhouse gas emissions in South Africa.

Unlike South Africa however, carbon sequestration in Namibia are inadequately studied (Turpie et al., 2010). Based on studies in other arid and semi-arid regions, carbon sequestration in Africa, may be of some importance. However most of these studies are based on research in agricultural areas and not on woody encroached savannas (Lal 2004; Bruce et al., 2009). Arid and semiarid regions could greatly benefit from carbon sequestration as it would improve the fertility of the nutrient deficient soils there. Though studies have been made on cultivated lands, ecosystem services benefits such as increase buffering capacity of soils, increase soil moisture, and higher crop yields and improves food security are just the same for other land uses (Bruce et al., 2009). Research also suggested that conserved natural systems within arid regions would have higher value as carbon sinks than degraded or heavily grazed areas outside protected areas (Turpie et al., 2010).

2.6. Effects of clearing woody encroaching plants

According to Janzen (2004) and the IPCC (2001), the main pools of actively cycling carbon are atmospheric CO_2, biota (mostly vegetation), soil organic matter (including detritus), and the ocean. The ocean contains the largest reserves of carbon though most of it is not in active circulation (Janzen, 2004; Lal, 2009). For communal farmers and those who directly depend on natural resources, woody plants are of great importance.

As the human population increases, more of the global net primary productivity is being used for human consumption (Janzen, 2004). Woody plants in southern African savannas are used for firewood, rough construction timber, charcoal production, and wood carvings (Smit, 2004; Mendelsohn et al., 2002). For many rural communities, wood is still the only source of fuel for cooking and heating, and the wood of several savanna tree species is known for its excellent fuel properties (Smit, 2004; de Wet, 2015).

Biochar manufacturing could be one area that could be explored to help control and manage woody encroaching bushes. In this way, countries could earn revenue from the sale of charcoal, and the environment would not become polluted from harmful gases since biochar manufacturing is achieved through pyrolysis. Additionally, branches from spiny woody species such as *V. tortilis* and *V. erubescens* that are common in Namibia are used for the construction of kraals where livestock can be protected from predators overnight (Smit, 2004). Moreover, expansion of Namibia's tourism industry has popularised the market for woodcarvings of indigenous tree species at a rapid rate. Woody plants are also an important source of browse for both domestic stock and game. Game ranching is currently one of the fastest growing sectors in the agricultural industry of southern Africa. The recent expansion of game ranching in southern Africa makes woody plants important to the economy (Smit, 2004). Woody plants creates unique habitats that can support a great diversity of species including herbivore and browser game species far better than other ecosystems without woody plants. In addition, cattle may consume significant amounts of browse during the dry season (Smit, 2004; Katjiua & Ward, 2007). As such, many communal farmers collect pods of encroaching *Vachellia* and *Senegalia* trees as fodder for their livestock during dry seasons (de Wet, 2015). Variable rainfall patterns have led to an increase of this practice. In fact, some low income households in urban areas depend on the sale of these pods for income. Finally, contrary to the common belief that bush encroachment is detrimental to grazers but not browsers, there are indications that bush encroachment may also be detrimental to some browsers, as some these tree species can harm browsers with their spines and hooks (Smit, 2004).

Though socially and economically important to some inhabitants, encroaching woody plants pose a nuisance to many commercial farmers, where efforts are being made to clear, control, or otherwise manage them. Woody vegetation management can be divided into preventative and restorative measures, where the latter includes chemical, mechanical, or biological control programmes in encroached areas that focus on tree thinning rather than clearing (Smit, 2004; NAU, 2010; Kahumba, 2011; de Wet, 2015). The restoration approach is considered to be more viable because the thinning of woody vegetation takes in account their ecological roles and does not completely disrupt the ecological function and dynamics of the ecosystems where this vegetation occurs.

Much is to be considered in making decisions on clearing woody species, as more ecological problems could be introduced in the efforts of eradicating encroaching woody species. It is important for any land manager to realise that there is no quick solution to the problem of bush encroachment. Smit (2004) explained that effective bush encroachment management should not be considered as a once-off event; rather it is a long-term commitment with various alternative approaches that are not necessarily the simplest or cheapest. As bush encroachment is caused by multiple factors, its management and control should also be studied in depth and over a long period of time. Thus, the least expensive method of killing trees may not be the most economical approach in the long term, so careful planning need to be in order. It is also possible that once invader bushes are lost from the ecosystem, land owners may discover that they now have to manage a much more unstable and altered system that requires frequent and repeated efforts in dealing with a high rate of re-encroachment–often from other, more threatening woody species. Consequently, it is also important to avoid or minimise other causes of bush encroachment such as grazing management practices, especially during the wet season which will ensure a vigorous and competitive herbaceous layer (Smit, 2004).

Before clearing of woody encroaching species, one needs to understand different ecosystem dynamics such as carbon sequestration. While it is relatively straightforward to determine the standing stock of carbon in a landscape, the rate of carbon sequestration is a more complex issue as it not only relate to the rate of carbon storage, but also to how permanently the carbon is stored (Turpie et al., 2010). Similarly, soil organic matter is the component of interest for managers of soil quality and production, but it is the actual soil organic carbon content that is relevant to carbon sequestration (Bruce, 2009). Though long-lived indigenous trees are typically considered as good carbon sinks, faster growing vegetation may result in high levels of soil carbon sequestration, even if carbon biomass is not stored for long (Turpie et al., 2010). This is why invader bushes and their ability to outgrow other trees should not be overlooked when it comes to carbon sequestration.

Lal (2004) indicated that sub-Saharan Africa's increasing demand for food can encourage farmers to reduce the length of fallow periods and cultivate continuously, overgraze fields and remove much of the above-ground biomass through fuel collection and as building materials. Such practices can potentially result in the reductions of soil organic carbon (Lal, 2004). In a predominately cattle country like Namibia, overgrazing is expected to be one of the major practices contributing to the reduction of soil organic carbon. In turn, overgrazing results in bush encroachment, which can have a positive effect on the amount of carbon stock stored in that ecosystem as the bushes can potentially sequester carbon. Likewise, soil erosion can be worsened by the barren soils found beneath the dense bushes. However, the same soil can be held together in areas where there are bushes thus helping in the accumulation of soil nutrients in these soils and potentially increasing the soil fertility of encroached areas.

2.7. Summary

To summarise, this chapter discusses how bush encroachment is a major challenge that results in land degradation in the savanna biome. The causes of bush encroachment are still sketchy although they can be attributed to mismanagement such as overgrazing of rangelands. This is supported by the evidence presented in this chapter that shows bush encroachment to be more common in commercial farms where browsing ungulate game species were removed and replaced by livestock. However, as pointed out by some authors, causes of bush encroachment are not single-factor driven, and so gaps exist in the understanding of the causes of this phenomenon. Consequently, it is challenging to manage bush encroachment when its causes are not fully understood.

Studies that suggest mechanical, biological, and chemical methods to control and maintain bush encroachment could introduce even greater problems by making the ecosystem vulnerable to more serious invader species such as alien species. Rather than considering bushes as a nuisance, studies on their positive contribution should also be considered. Despite negative impacts on agricultural productivity and livelihoods, native invader bush present positive opportunities of energy generation and their resultant impact of improved ecosystem conservation and greater biodiversity. Similarly, bushes have ecological importance such as creating fertility islands beneath their canopies that increase soil organic matter and enrich top soils with nutrients. Additionally, bushes can be used for biochar production to increase soil productivity by increasing soil organic carbon, while earning carbon credits. Still, elements of measuring and monitoring changes in soil organic carbon need to be understood if they are to have a role in carbon sequestration and trading schemes. This information on carbon stock could also be used to develop viable land-use options for agricultural sustainability. In short, a balance between trees, shrubs and grasses should be reached for optimum ecosystem service and functioning. Studies on control and maintenance of bushes need to be carried out for rangeland management and policy formulation.

CHAPTER 3: MATERIAL AND METHODS

3.1. Study Area

3.1.1. Location of the study area

The study area lies in the Thornbush savanna biome as described by Giess (1998). It is located in central Namibia on the commercial cattle farm Erichsfelde or Otjiamongombe as it is locally known (Figure 1). Erichsfelde is situated approximately 40 km north of Okahandja in the Otjozondjupa region, an area known to be highly affected by bush encroachment (de Klerk, 2004). This farm lies at 21.591852°S, 16.868216°E at an altitude of 1, 495 m above sea level.

The 13,000 ha farm is privately-owned and forms part of the Biodiversity Monitoring Transect Analysis in Africa (BIOTA) Observatory under the Southern African Science Service Centre for Climate Change and Adaptive Land Management (SASSCAL) project (Jürgens et al., 2010). The farm is mainly used for cattle ranching (Figure 2) although a small fraction of its income is from game hunting (Jürgens et al., 2010). The BIOTA Observatory was established to monitor changes in the vegetation structure brought about by intensive cattle farming. This farm has areas that are heavily (> 1,000 bushes per hectare) bush encroached, intermediate-to-medium (about 500 to 1,000 bushes per hectare) bush encroached from clearing, and low (<500 bushes per hectare) bush encroached or cleared land that has turned into a grassland, making it suitable for this study.

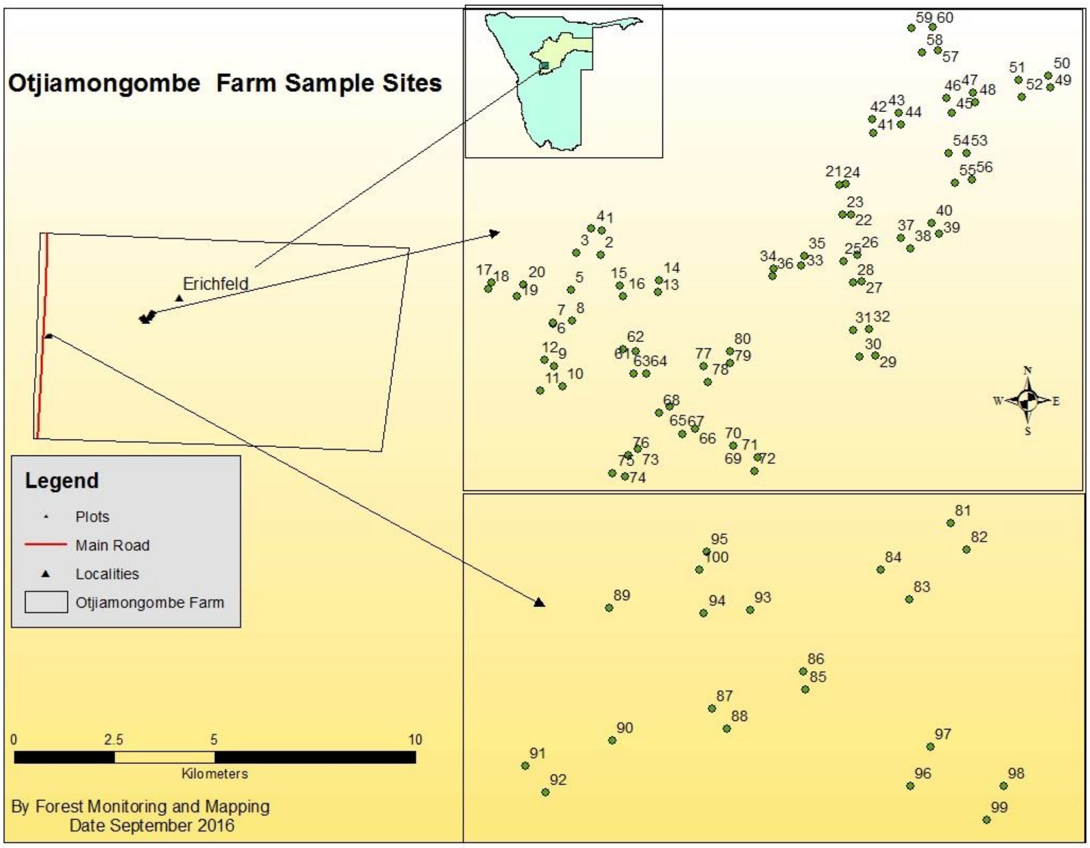

Figure 1: Location of the study area. Adapted from the database of the Ministry of Agriculture, Water and Forestry (2015)

Figure 2: Erichsfelde (Otjiamongombe) notice board

3.1.2. Climate

Erichsfelde is characterised by semiarid climate conditions. Rainfall in this area shows a strong seasonality with almost all rain falling during the summer season, with the highest rainfall being experienced in February (Jürgens et al., 2010). The area receives mean annual rainfall of 350 mm during the summer and mean annual potential evapotranspiration of 1,732 mm, which is not far off from the region's average annual evaporation of 1,960 mm/year (Jürgens et al., 2010; Mendelsohn et al., 2002). During the summer months (October to March), the mean maximum temperature is about 32°C, whereas in the winter months (May to August) mean minimum temperature is 4°C (Mendelsohn et al., 2002). May to August can bring frost, averaging between 10 to 20 days annually (Mendelsohn et al., 2002; Jürgens et al., 2010).

3.1.3. Geomorphology, and soils

The Otjozondjupa region can be divided into two main landscapes and landforms, namely the central-western plains, and the Khomas Hochland Plateau (Mendelsohn et al., 2002). It is in the Otjozondjupa region that the country's Waterberg Plateau with its rich biodiversity is located. Soil types at Erichsfelde are ferralic cambisols and arenosols, as well as chromatic luvisols (Figure 3); presenting topsoil colours that are reddish or bright yellow (Oldeland, Dorigo, Wesuls, & Jürgens et al., 2010).

According to Jürgens et al. (2010), the soils of Erichsfelde can be subdivided into three areas: the eastern part, characterised by deeply developed reddish clayey luvisols; the western part, featuring loamier calcisols and cambisols with a dark brown colour; and the shallow calcisols that are found within the vicinity of the dry river and are restricted in their rooting depth. Luvisol soils are slightly acidic with very low nutrient content, while the calcisols and cambisols have high pH values and a higher content of organic carbon, whereas the shallow calcisol soils are characterised by neutral to alkaline soils with high electrical conductivity and organic carbon content.

The carbonate-rich soils of Erichsfelde also have well-developed biological soil crusts, although they are either weakly developed or completely absent in the carbonate-free soils of the eastern part of the farm (Jürgens et al., 2010). However, the well-established cyanobacterial crusts in this area are a sign of significant amounts of carbon from dead plant material (Jürgens et al., 2010).

3.1.4. Vegetation

Erichsfelde's typical Thornbush Savanna biome is characterised by patches of dense trees and thorny shrub cover. This alternates with open grasslands that are dominated by *Stipagrostis uniplumis* and scattered trees of Vachellia and Senegalia species (Jürgens et al., 2010; Mendelsohn et al. 2002). The patchiness in arid Savanna biome is usually driven by rainfall (Wiegand et al., 2005). Two main habitat types can be identified on the farm: the plains and the river, which are quite distinct in vegetation structure and in species composition (Jürgens et al., 2010). The plain where this study was carried out is associated with occurrence of a calcrete layer in the soil that may be responsible for changes in vegetation structure. Accordingly, dwarf shrubs occur on this soil (Jürgens et al. 2010). Bush encroachment by *Vachellia mellifera* was also evident on some areas on the plain. The farm is mainly covered in loamy substrates that are overlain by *Vachellia* and *Senegalia* species, and *Stipagrostis* grasses. However, depending on the habitat and the intensity of disturbances by grazing and trampling animals, annuals can dominate over perennial climax grasses in the herbaceous layer (Jürgens et al., 2010).

As reported in Jürgens et al. (2010), three distinct vegetation units occur on Erichsfelde. The first unit is the *Bothriochloa radicans-Ziziphus mucronata* community, featuring the riverine vegetation of the river habitat type. The dry riverbed with its sandy, deep soil layer act as a water reservoir and allow for the enhanced growth of trees such as *Z. mucronata, V. reficiens,* and *S. mellifera*. Jürgens et al. (2010) showed that the perennial grasses such as *B. radicans* and *E. rotifer* occur in the moist understorey of thickets and form dense stands along and within the riverbed.

The second vegetation unit is the *Seddera suffruticosa-Melhania virescens* shrub community and is found in parts of the plain habitat where the soil has a calcrete layer characterised by elevated concentrations of calcium carbonate (Jürgens et al., 2010). The shrub *Catophractes alexandrii* is dominant throughout the plain area, a reasonably palatable perennial grass, *Monelytrum luederitzianum,* and a shallow rooted grass, *Enneapogon desvauxii,* are found in the *Seddera suffruticosa-Melhania virescens* shrub habitat (Jürgens et al., 2010).

The third vegetation unit is the *Eragrostis rigidior-Gisekia africana* community that covers the rest of the farm area. *E. rigidior* is a perennial grass that is typical of sandy soils of the northern Kalahari but is also found on loamy soils such as those on Ericshfelde (Jürgens et al., 2010). This community, also has annual herbs such as *G. africana, Crotalaria heidmannii,* and *Tephrosia burchelli,* that occur only after good rains, and are absent during poor rainfall years. According to Jürgens et al. (2010), bush encroachment by *V. mellifera* and high abundances of grasses from the genus *Aristida* in some parts of this unit are a sign of poor land management from the past. Jürgens et al. (2010) also stated that many parts of this vegetation unit show a high density of non-encroacher woody species such as *Senegalia cinerea* (Schinz) Kyal. & Boatwr., *V. tortilis*, *V. hebeclada*, and *Boscia albitrunca*.

3.1.5. Fauna

Although Erichsfelde is mainly a cattle farm, large herds of game species such as Oryx, Kudu, Eland, Springbok, Damara Dik-dik, Common Duiker, Red Hartebeest, Steenbok, and Warthog occur on the farm. Predators such as Leopards, Cheetahs, Jakals and African Wild Cats roam freely on the farm too (Jürgens et al., 2010). Small mammals such as Antbears, Pangolins, Rock Dassies, Porcupines, Scrub Hares, Ground Squirrels, Chacmas, and Baboons can also be found at Erichsfelde. The farm is home to a wide variety of birds such as the Grey Heron, Hammerkop, Stork, Marabu Stork, South-African Shelduck, Secretary Bird, Whitebacked Vulture, Lappetfaced Vulture, Kite, Black Eagle amongst others (Jürgens et al., 2010). Typical to Otjozondjupa region is the widespread of termite *Macrotermes michaelseni* mounds at this farm. According to Jürgens et al. (2010), from the 45 BIOTA Observatories, Erichsfelde possessed the highest millipede species richness with species such as *Spirostreptus heros*, *Doratogonous rugifrons* and *Chaleponcus limbatus* occurring on the fam.

3.2. Selection of the sampling sites and demarcation of plots

Fieldwork was conducted in July 2014. The study was quantitative and followed a stratified random sampling with systematic sampling along transects. Based on de Klerk (2004) and a modification of Coetzee's (2003) bush encroachment definition, a satellite image was used to identify three sampling sites on the farm. A non-encroached site (<500 bushes per hectare) that will be reffered to as a low bush encroached site in this study, medium (about 500 to 1,000 bushes per hectare), and high (> 1,000 bushes per hectare) bush encroached site (Figure 3 & Appendix 4).

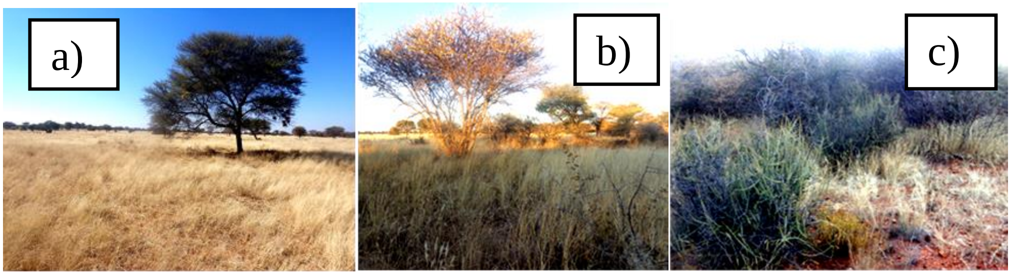

Figure 3: The three bush encroached sites, a) low encroached site, b) medium encroached and photo c) high encroached site

In each site, two 50 m line transects were randomly laid. The position of the line transects were determined by randomly throwing a stone. Along each transect, following a systematic sampling method, five 20 m x 10 m plots were set up 25 m apart for vegetation survey. Five 1 m x 1 m soil pits were nested within the plots for one of the line transects at each site. In total, 30 plots were sampled for vegetation survey whereas 15 soil pits were dug for soil sampling (Figure 4).

Distance between plots: 25m

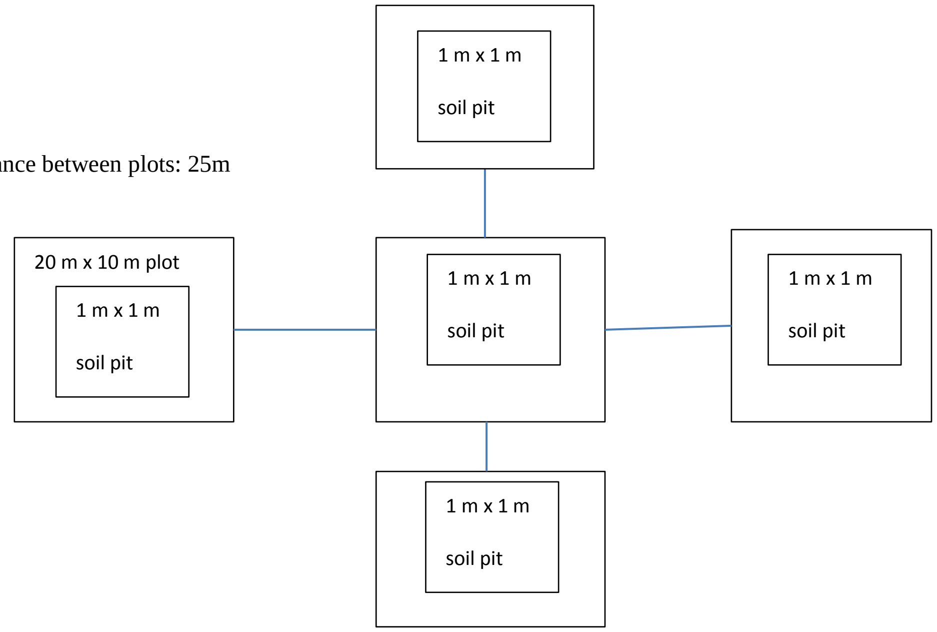

Figure 4: Plots layout of the vegetation and soil sampling quadrats. Not drawn to scale

3.3. Soil Samples

3.3.1. Soil sampling

Before soil pits were dug (Figure 7), the sampling plots were cleared of plant litter. About 200 g of soil was collected at each horizon, defined by the soil profile (e.g. 0 cm-10 cm, 10 cm-20 cm, 20 cm-30 cm) and depending on substrate's hardness. Locations of each soil pit were recorded using a GPS (Garmin III Plus). A photograph of each soil pit was taken in the field for better visualisation at the lab. A total of 54 soil samples were collected, consisting of 13 from the highly encroached site, 20 from the medium encroached site and 21 from the low encroached site. Soil samples were placed in soil sampling plastic bags, labelled, and stored at room temperature for three days (Figure 5) and transported to the Ministry of Agriculture, Water and Forestry Analytical Laboratory for chemical analysis.

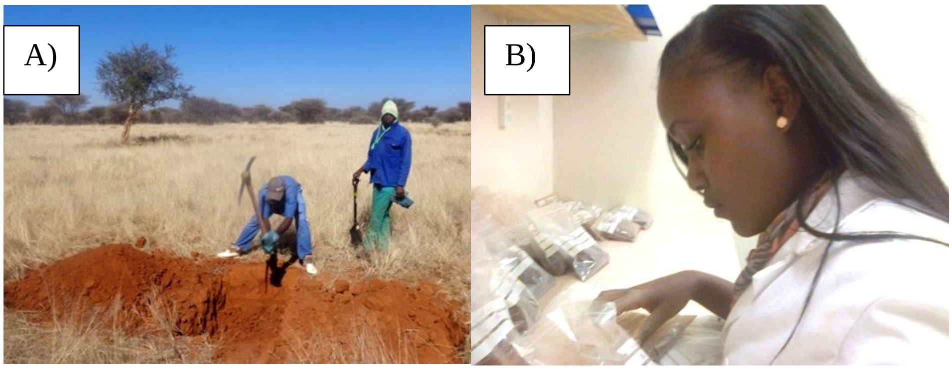

Figure 5: A) field assistants digging a soil pit and B) sorting soil samples

3.3.2. Laboratory analysis

Soil samples underwent chemical analysis at the Ministry of Agriculture, Water and Forestry's analytical laboratory following standard procedures (Appendix 1). In order to prepare them, samples were air dried at temperatures not exceeding 35°C. A 2 mm sieve was used to sieve samples and retain parts called the fine earth fraction for analysis. Fractions measuring more than 2 mm were referred to as *stones* and *gravel*, and were not used in the analysis. Samples were analysed for pH levels, exchangeable calcium (Ca), magnesium (Mg), and sodium (Na), and total nitrogen (N), phosphorus (P), potassium (K), soil organic matter (SOM), available micronutrients (zinc (Zn), manganese (Mn), copper (Cu), iron (Fe)), selected exchangeable ions, cation exchange capacity (CEC), electrical conductivity (EC), sodium adsorption ratio (SAR), and soil texture (sand, silt, clay) (Appendix 1).

3.4. Above-ground woody carbon stock and biomass determination

The mass of living organisms in a biome is called biomass (Condit, 2008). For the purpose of this study, the terms *above-ground biomass* and *below-ground biomass* refer to standing woody vegetation, and its roots, respectively. The woody biomass and carbon stock were determined using a modification of Hairiah et al. (2001)'s method and allometric equations by Brown (1997).

3.4.1. Vegetation survey

The distinction between trees and shrubs was made on the basis of a description by Curtis and Mannheimer (2005). For the purpose of this study, a tree was considered a woody plant if it had one or several thick stems usually branching well from the ground. As well, it could only be classified a tree if it had relatively large stems with basal diameter of more than 5 cm and a height potential of more than 3 m (Liu et al., 2013; González-Roglich et al., 2014). Shrubs on the other hand were classified as any woody plant with relatively many small stems arising from and branching near the ground, having a basal diameter less than 5 cm (Liu et al., 2013).

Woody vegetation in the 30 plots were measured using a non-destructive method (Hairiah et al., 2001b; Kuyah et al., 2012). For each of the woody plants, the height and diameter (stem and canopy) were measured. The stem diameter was measured at the base of the main trunk, defined as the tallest or at the ground level of the group of stems if the plant was multi-stemmed (González-Roglich et al., 2014). Stem circumference measurements were repeated to ensure that the tape was positioned properly and correctly read. The measurement was recorded and divided by

3.1416 (π rounded) to determine the diameter. Multi-stemmed woody vegetation stems were measured separately to calculate total circumference.

Tree canopy diameter was measured by the length of the longest spread from edge to edge across the crown (Figure 6). The tree's edges that were perpendicular to the first canopy diameter measurement were also measured and recorded. The height of first leaves of potential leaf bearing stems was also measured and recorded. The height of each woody plant was determined using a graduated range pole. Plants whose height was less than 30 cm was measured with a ruler. However, the height of plants taller than the graduated range pole was estimated to the nearest half metre (Kaholongo and Mapaure, 2014).

Figure 6: Measuring the bush canopy in the field

3.4.2. Above- and below-ground biomass and woody carbon stock estimation

Various studies and manuals have compiled biomass data sets and allometric equations (Equation 1, 2, 3,4 and 5) to predict above-ground carbon in forest trees to estimate carbon stocks for national greenhouse gas estimates (Beets et al., 2012; Eamus et al., 2000; Henry et al., 2011; Condit, 2008; Kuyah, et al., 2012;Takimoto et al., 2008). It is therefore important that accurate allometric equations be available to estimate carbon stocks from national plot inventory data (Beets et al., 2012).

According to Amadhila (2012) and Brown (1997), two methods can be used to estimate tree biomass and carbon stock. Such methods are direct and indirect methods. The direct method, which is also known as the destructive method, involves harvesting trees to determine biomass. On the other hand, the indirect method involves establishing the biomass through the use of formulas or allometric equations to independent variables such as the diameter and height measured in inventory plots (Amadhila, 2012; Beets et al., 2012; Condit, 2008; Kuyah et al., 2012). In this study the indirect, or non-destructive method of establishing biomass and carbon stock, was used. The indirect biomass estimation and carbon stock method used for this study was adapted from Brown (1997), where a general equation applicable to tropical dry ecological zones was used, as species-specific allometric equations were not available for all tree species in the savanna biome (Takimoto et al., 2008). Also, the BECVOL (Biomass Estimation from Canopy Volume) software to estimate tree biomass for savanna biome, invented by Smit (1994), is outdated and is less utilised due to its shortcomings.

3.5. Data Analysis

Above- and below-ground biomass and woody carbon stock estimation were estimated as follows:

Estimated merchantable volume (VOB m^3/ha) were converted to oven-dry mass, using Africa's mean wood density (WD) of 0.58 t/m^3 (Brown 1997). The merchantable growing stock was then expanded to account for non-merchantable components of the tree with the Biomass Expansion Factor (BEF). BEF is defined as the ratio of above-ground oven-dry biomass density to the oven-dry biomass density of the inventoried volume (Brown, 1997).

The following equations were adapted from Brown (1997) and used to estimate above-ground biomass (AGB), below-ground biomass (BGB), and carbon stock for this study (Figure 10):

Equation 1: Above-ground biomass (AGB) = VOB x WD x BEF

Equation 2: Above-ground carbon stock = AGB x 0.47

Equation 3: Below-ground biomass (BGB) = AGB x 0.24

Equation 4: Below-ground carbon stock = BGB x 0.47

Equation 5: Total carbon stock = Above-ground carbon stock + Below-ground carbon stock

The notation WD in Equation 1 represents the mean wood density for Africa, which is 0.58 t/m^3, whereas BEF is the biomass expansion factor which is calculated using the formula BEF = EXP {3.213-0.506 x In (BV)}. On the other hand, BV in the

BEF formula is the biomass of inventoried volume calculated as the product of wood density and inventoried volume (BV = VOB (m^3/ha) x WD (0.58 t/m^3). This formula was used to calculate BEF because BV was < 190 t/ha (Brown 1997; Amadhila, 2012).

3.5.1 Above-ground biomass (AGB) and carbon stock

To estimate AGB, Equation 1 was used where VOB is the estimated merchantable volume per hectare (m^3/ha). This was calculated using Smit's (1996) formula for trees and shrubs that had a stem diameter of the foliage at the height of first leaves or potential leaf bearing stems (C) bigger than zero, but smaller than the maximum canopy diameter (D). This is because most of the trees and shrubs encountered met this requirement. Thus:

$$\text{VOB} = (\tfrac{1}{3}) \times (\tfrac{22}{7}) \times \text{G} \times ((\tfrac{D}{2})^2 + (\tfrac{D}{2}) \times (\tfrac{E}{2}) + ((\tfrac{E}{2})^2)$$

Whereas;

G = height of tree base (difference between height of maximum canopy diameter (B) and height of first leaves or potential leaf bearing stems (C))

D = maximum canopy diameter

E = base diameter of the foliage at height C

The above-ground carbon stock which is the function of the above-ground biomass was estimated using Equation 2, where 0.47 is the estimated total percentage of carbon in live wood as per FAO (Amadhila, 2012).

3.5.2 Below-ground biomass (BGB) and carbon stock

According to Henry et al. (2011) and Button, Liao, Filley & Archer (2009), below-ground biomass is a function of above-ground biomass. Since the indirect method of estimating biomass was used, the biomass obtained using Equation 1 was that of above-ground only. Thus, to obtain the total carbon stock, below-ground biomass and carbon stock needed to be established. This was achieved using Equation 3 and 4 where 0.24 is the constant coefficient ratio of below-ground biomass to above-ground biomass and 0.47 is the estimated total percentage of carbon in live wood used for the country's FAO reporting format (Amadhila, 2012).

3.5.3 Total woody carbon stock

Equation 5 was used to calculate total standing woody vegetation carbon stock, and was determined using the values from Equations 2 and 4 (Figure 7).

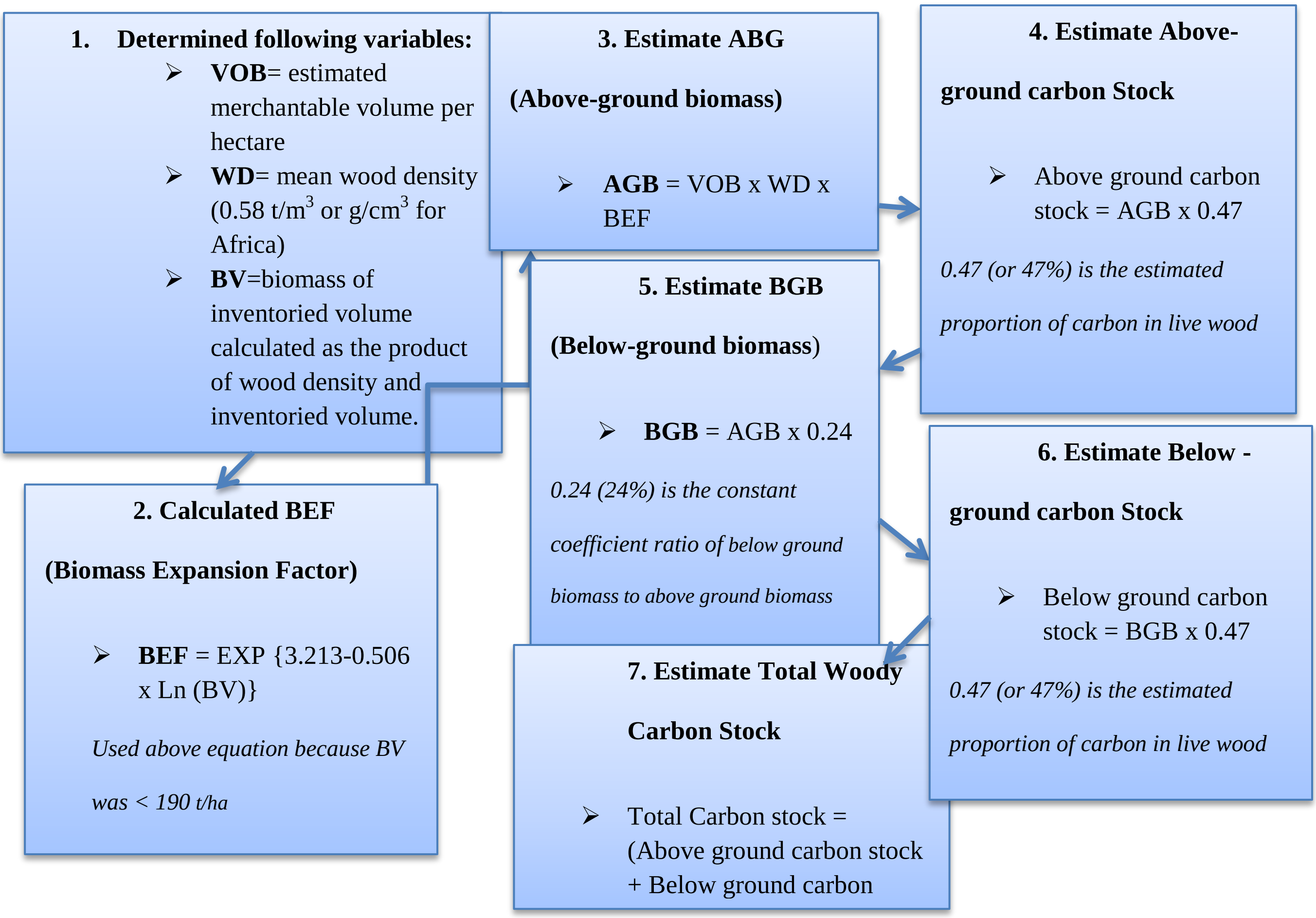

Figure 7: Steps followed to estimate above and below ground biomass, as well as woody carbon stock for the vegetation sampled

3.5.4 Statistical Analysis

Above-ground and below-ground biomass, carbon stock data, as well as the different soil properties were tested for normality using the Shapiro Wilk's test, whereas the Levene test was used to test for homogeneity of variance. The Shapiro Wilk test was used because the sample sizes were less than 2,000 (Dytham, 1999). For normally-distributed data, a one-way analysis of variance (ANOVA) was used to test whether there was a significant difference in the mean values of measured variables among the high bush encroached, medium, and low bush encroached sites (Zar, 1999). From the measured variables, only pH, Cu, Fe, Cl^- and Mn were normally-distributed. Where conditions for parametric tests were not met, the nonparametric Kruskal-Wallis test for numerical data was used to test for the significant difference (Figure 8). The Kruskal-Wallis test was perfomed on the total woody carbon stock and biomass, above and below-ground carbon stock, SOM, total N, P, K, Ca, Na, Mg, Na, ions (Ca^{2+}, Mg^{2+}, and K^+), SAR and EC.

When the null hypothesis of no difference among group means and medians was rejected, the Tukey-Kramer HSD post hoc test–or Dunnett T3–of all pairwise of means, as well as the multiple comparison test for medians, was used to assess which treatment means were different among the three encroached sites (Figure 8). All statistical tests were performed using the predictive analysis software SPSS (Field, 2009).

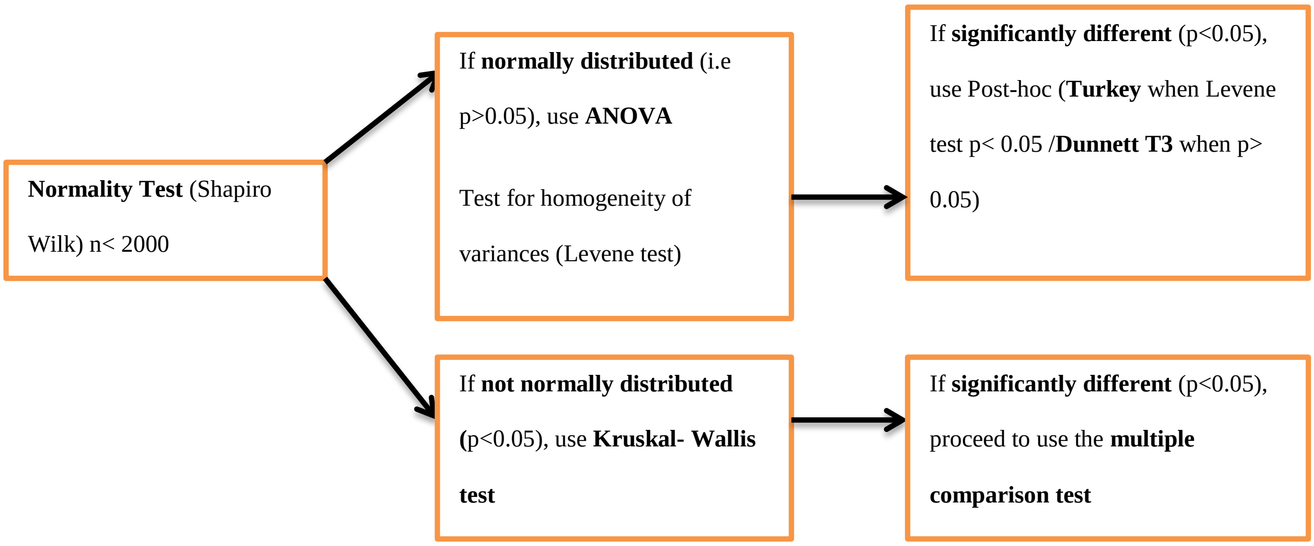

Figure 8: Steps followed in analysing the data

CHAPTER 4: RESULTS

4.1. Vegetation parameters

4.1.1. Total woody vegetation carbon stock

The Kruskal-Wallis test for the total woody vegetation carbon stock in the high, medium, and low encroached sites indicated that there was no significant difference across the sites ($H = 2.449$, $df = 2$, $P = 0.294 > 0.05$). The high encroached site sequestered 34.23 t/ha while the medium encroached site had 10.16 t/ha. The low encroached site sequestered 0.53 t/ha.

4.1.2. Above- and below-ground woody carbon stock

The median of above-ground carbon stock was similar across three sites (Kruskal-Wallis: $H = 0.663$, $df = 2$, $P = 0.718 > 0.05$) while, below-ground carbon stock was significantly different (Kruskal-Wallis: $H = 56.202$, $df = 2$, $P\ 0.000 < 0.05$). The all pairwise test further revealed the below-ground carbon stock to be significantly higher in the medium encroached site than in the high encroached sites ($P < 0.05$). Figure 9; show the below-ground carbon stock across the three sites.

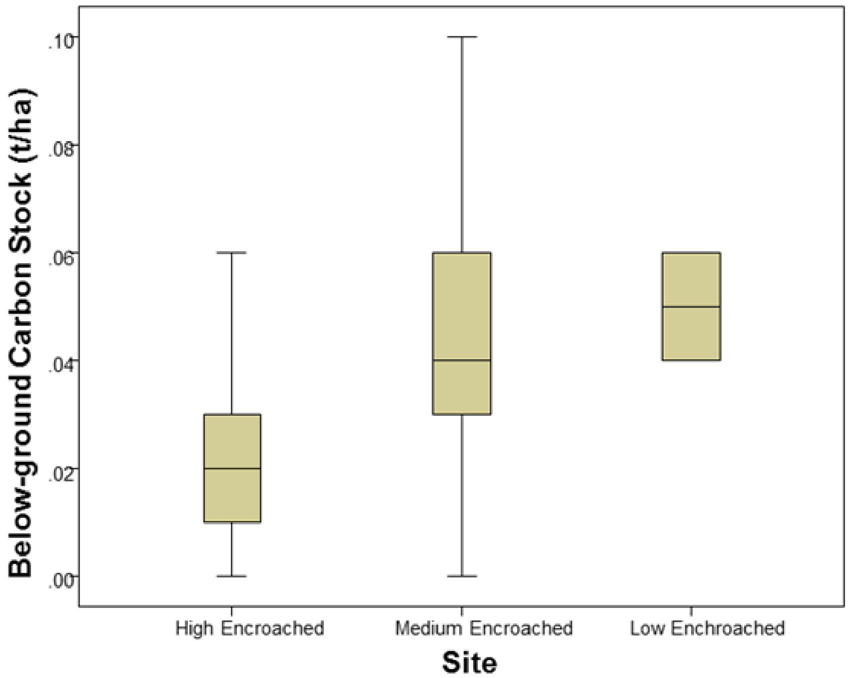

Figure 9: Comparison of the below-ground carbon stock across the three sampled sites at Erichsfelde farm in the Otjozondjupa region

4.2. Soil properties

4.2.1. Soil texture and soil composition

The medium and low encroached sites had the same type of soils (sandy loam and sandy clay loam), while the high encroached site had sandy and loamy sand soils (Table 1). The soil analysis also revealed that the three sites soil textures were dominated by sandy soil particles, then clay soil particles, and finally silt soil particles (Figures 11 A), B) and C)).

Table 1: The comparison of soil texture in the bush encroached sites at Erichsfelde farm

Site	**Soil Texture**						
	Clay loam	**Sand**	**Loamy sand**	**Sandy loam**	**Loam**	**Silt loam**	**Sandy clay loam**
High encroached	Absent	Present	Present	Absent	Absent	Absent	Absent
Medium encroached	Absent	Absent	Absent	Present	Absent	Absent	Present
Low encroached	Absent	Absent	Absent	Present	Absent	Absent	Present

4.2.2. Soil pH

The medium encroached site had a mean pH of 7.44 while the high encroached site had a mean pH of 7.53 finally, the low encroached site had a mean pH of 7.48 (high encroached site). There was no statistically significant difference in mean pH of soils amongst the three sites (ANOVA: $F = 0.669$, $df = 2$, $P = 0.517 > 0.05$).

4.2.3. Soil total nitrogen (N) content

The N of soil samples ranged from 0.04% (medium encroached site) to 0.03% (high and low encroached site). There was no statistically significant difference in the median N among sites (Kruskal-Wallis: $H = 3.375$, $df = 2$, $P = 0.185 > 0.05$).

4.2.4. Soil organic matter (SOM) content

The SOM content of the soil ranged from 0.2 - 1.5 % m/m in the high encroached site, 0.3 – 1.6 % m/m in the medium encroached site and 0.4 – 1.0 % m/m in the low encroached site. The Kruskal-Wallis test indicated that SOM in soils sampled from the high, medium and low encroached sites was similar ($H = 2.678$, $df = 2$, $P = 0.262 > 0.05$).

4.2.5. Soil phosphorus (P)

There was a significant difference in the median soil P among the three sites (Kruskal-Wallis: H = 9.317, df = 2, P = 0.009 < 0.05) where P content ranged between 0 – 11 ppm in the high encroached site, 0 – 3 ppm in the medium encroached site and 0 – 2 ppm in the low encroached site (Figure 10). The Multiple Comparison Test showed that the high encroached site had greater soil P than the low (P = 0.009) and medium (P = 0.047) encroached site.

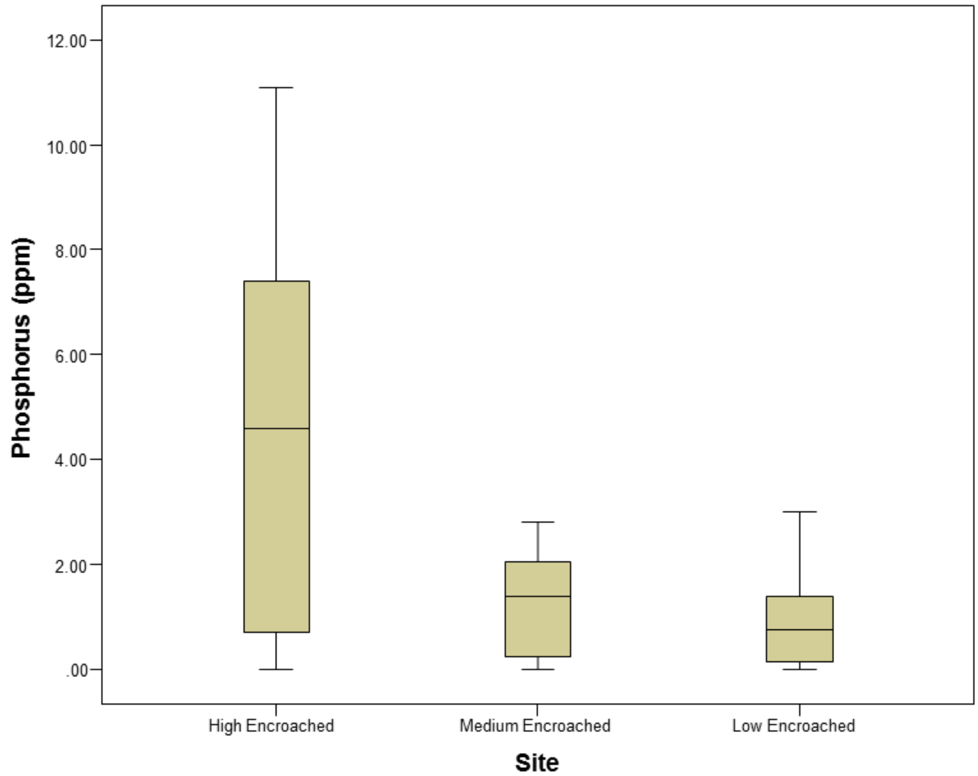

Figure 10: The comparison of soil P in the high, medium and low encroached sites at Erichsfelde farm in the Otjozondjupa region

4.2.6. Soil potassium (K)

The K content in the study area ranged from about 128 – 191 ppm in the high encroached, 216 – 338 ppm, in medium encroached site and 317 – 388 ppm in the low encroached site (Figure 11). The non-parametric Kruskal-Wallis test indicated that there was a significant difference in the median K content in soils sampled from the three sites ($H = 21.584$, $df = 2$, $P < 0.05$). The Multiple Comparison Test (all pairwise) showed this difference to be between the high encroached soils that was significantly lower than the medium ($P = 0.024 < 0.05$) and low encroached soils ($P < 0.05$).

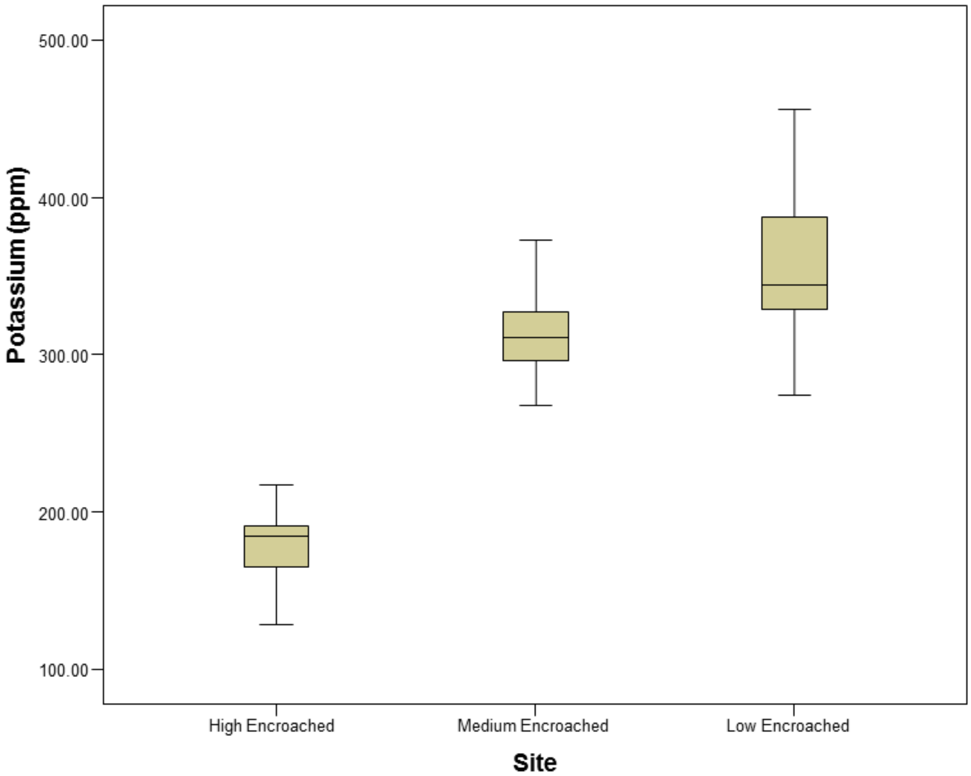

Figure 11: Assessment of the soil K in the soils sampled in the high, medium and low encroached sites at Erichsfelde farm in the Otjozondjupa region

4.2.7. Soil calcium (Ca)

Median Ca content was significantlly different across the three sites (Kruskal-Wallis: H = 14.338, df = 2, P = 0.001< 0.05). The multiple comparison post hoc test showed that the high encroached site had lower Ca content than the medium (P = 0.001 < 0.05) and the low encroached (P = 0.006 < 0.05) sites (Figure 12).

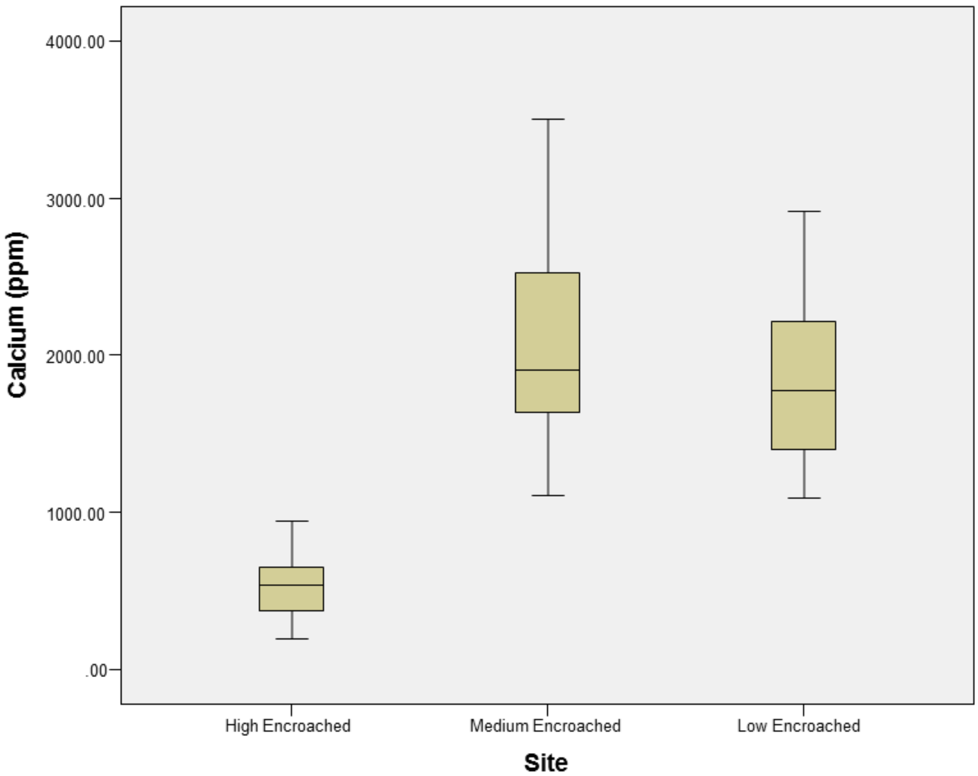

Figure 12: Ca content of the sampled soil in the low, medium and high bush encroached sites at Erichsfelde farm

4.2.8. Soil magnesium (Mg)

The non-parametric Kruskal-Wallis test result indicated that there was a significant difference in the median content of Mg in soils sampled from the three sites studied (H = 29.085, df = 2, $P < 0.05$). This is shown by the Mg content ranges from 85 -154 ppm in the high encroached site, 241 – 609 ppm in the medium encroached site and 312 – 615 ppm in the low encroached site (Figure 13). Furthermore, multiple comparison analysis revealed that the Mg content in the high encroached site was lower than the medium ($P < 0.05$) and low ($P < 0.05$) encroached sites.

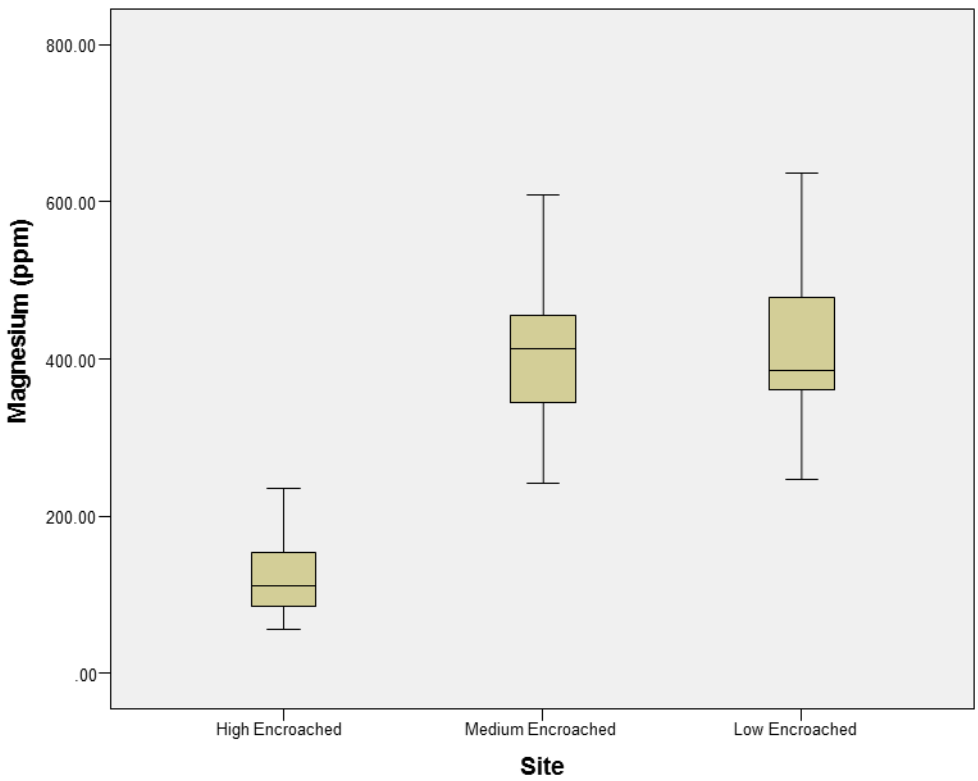

Figure 13: Soil Mg content of the low, medium and highly bush encroached sites at Erichsfelde farm

4.2.9. Soil sodium (Na)

There was a significant difference in the median content of Na in soils sampled from the different sites (Kruskal-Wallis: $H = 27.567$, $df = 2$, $P < 0.05$). In particular, the Na content in the medium encroached site was significantly lower than the high ($P = 0.004$) and low ($P < 0.05$) encroached sites. With the Na content ranging from 10 – 164 ppm (high encroached), 0 – 84 ppm (medium encroached) and 83 – 162 ppm (low encroached) (Figure 14).

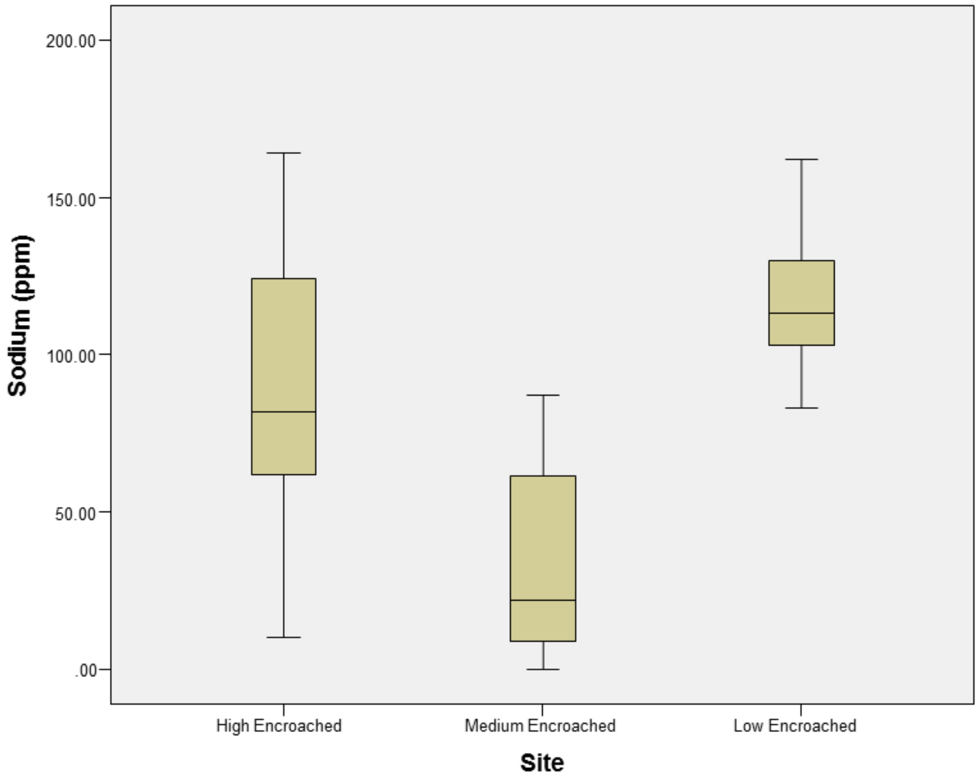

Figure 14: The comparison of Na content in the soil sampled in the high, medium and low bush encroached sites at Erichsfelde in the Otjozondjupa region

4.2.10. Soil iron (Fe)

The one-way ANOVA showed that there was a significant difference in the mean soil Fe content across the three sites ($F = 13.554$, $df = 2$, $P < 0.05$). The Dunnett T3 post hoc test further revealed that the soil Fe content in the low encroached site was significantly higher than in the high ($P < 0.05$) and the medium encroached sites ($P = 0.001$) (Figure 15).

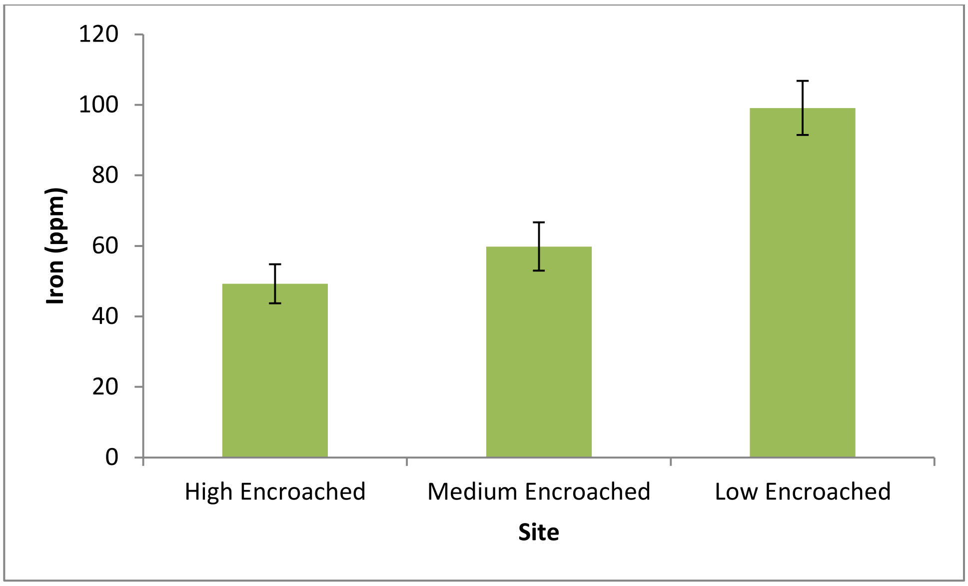

Figure 15: Fe content of the soil sampled in the three sites. Error bars denote standard error 95% confidence intervals

4.2.11. Soil copper (Cu)

The one-way ANOVA showed that the mean Cu content was significantly different across the three sites ($F = 10.086$, $df = 2$, $P < 0.05$). Furthermore, the Dunnett T3 test indicates the high encroached site to be significantly lower in Fe content than both the medium ($P = 0.017 < 0.05$) and low ($P = 0.001 < 0.05$) encroached sites (Figure 16).

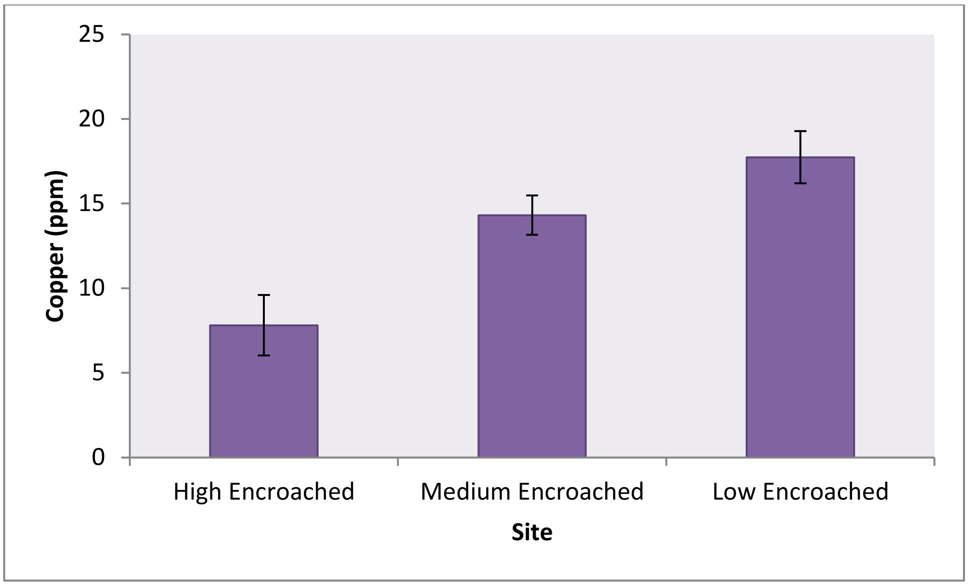

Figure 16: Assessment of Cu content in high, medium and low encroached site soils at Erichsfelde farm. Error bars denote standard error 95% confidence intervals

4.2.12. Soil manganese (Mn)

The Mn content ranged between 48 – 351 ppm for the high bush encroached site, 116 – 232 ppm for the medium encroached and 127 – 277 ppm for the low bush encroached site. The one-way ANOVA depicted no significant difference in the mean content of Mn in soils sampled from the three different sites ($F = 0.201$, $df = 2$, $P = 0.818 > 0.05$).

4.2.13. Electrical conductivity (EC)

There was no significant difference in the mean EC of the soils in the high, medium and low encroached sites at Erichsfelde farm (Kruskal-Wallis: $H = 1.715$, $df = 2$, $P = 0.424 > 0.05$).

4.2.14. Soil calcium ion (Ca^{2+})

The Kruskal-Wallis test showed no significant difference in the median Ca^{2+} ion content across the three sites sampled at Erichsfelde farm ($H = 0.618$, $df = 2$, $P = 0.734 > 0.05$).

4.2.15. Magnesium ion (Mg^{2+})

There was a similarity in the median Mg^{2+} ion content across the high, medium and low bush encroached sites (Kruskal-Wallis: $H = 1.992$, $df = 2$, $P = 0.369 > 0.05$).

4.2.16. Sodium ion (Na^{+})

The median Na^{+} content showed similarities in the soils sampled in the high, medium and low bush encroached sites at Erichsfelde (Kruskal-Wallis: $H = 2.454$, $df = 2$, $P = 0.293 > 0.05$).

4.2.17. Potassium ion (K^+)

The Kruskal-Wallis test indicated a significant difference in the median content of K^+ in soils sampled from the high, medium and low bush encroached sites ($H = 6.021$, $df = 2$, $P = 0.049 < 0.05$). The multiple comparison test revealed that the difference in the K^+ content occurred between the medium and high encroached sites ($P = 0.044 < 0.05$) with K^+ content of the high encroached site being significantly greater than that of the medium encroached site (Figure 17).

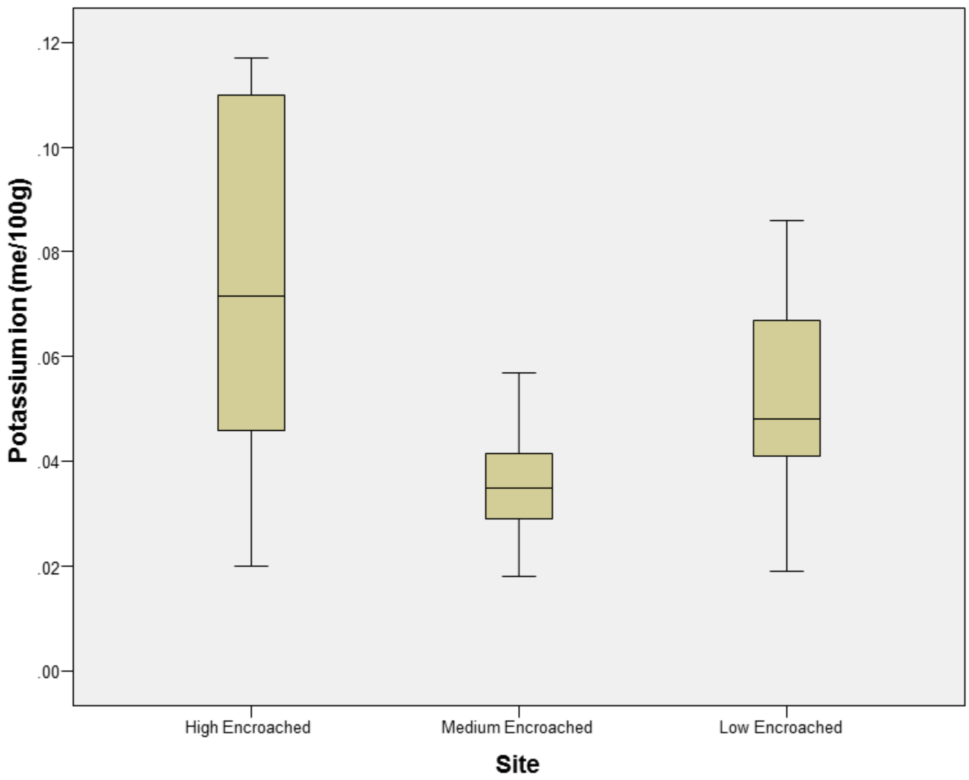

Figure 17: Comparison of the K^+ ion content in the soil sampled form the high, medium and low bush encroached sites at Erichsfelde farm

4.2.18. Soil cloride ion (Cl^-)

The Kruskal-Wallis test ($H = 1.348$, $df = 2$, $P = 0.269 > 0.05$) revealed a similarity in the median Cl^- content across the high, medium and low encroached sites at Erichsfelde farm.

4.2.19. Sodium Adsorption Ratio (SAR)

The Kruskal-Wallis test indicated that there was a significant difference in the content of SAR in soils sampled from the high, medium and low encroached sites ($H = 14.394$, $df = 2$, $P = 0.008 < 0.05$). The multiple comparison test further revealed that the low encroached site was significantly higher than the medium and high encroached sites ($P < 0.05$) (Figure 18).

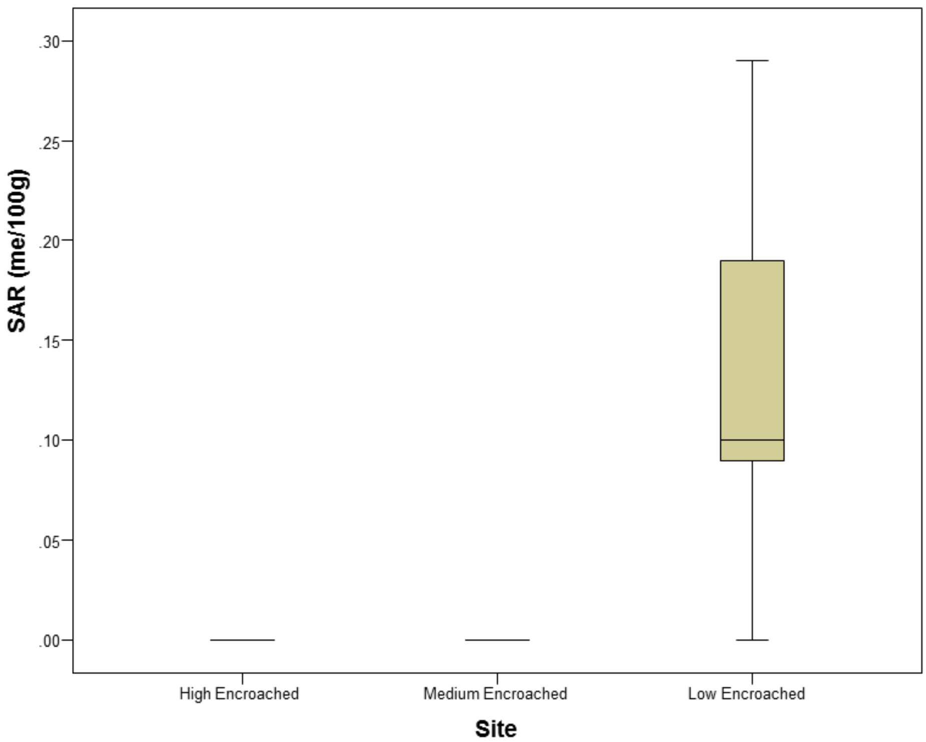

Figure 18: The SAR content of the sampled soils in the high, medium and low encroached sites at Erichsfelde in the Otjozondjupa region

CHAPTER 5: DISCUSSION

5.1. Carbon stock

The results of this study have shown no significant difference in the mean total woody and above-ground carbon stock across the three sites. In a related study, Shackleton & Scholes (2011) mentioned that intensive land use such as overgrazing can lead to a mosaic of land use types with varying levels of woody biomass posing a great potential for carbon sequestration. Although, the mosaic and distinct encroached sites were observable at Erichsfelde farm as a result of bush encroachment and clearing of land for grazing, there was no significant difference among the three sites with regards to total woody and above-ground carbon sequestration. According to San Jose et al. (1998), savanna vegetation types have the potential to influence gobal carbon budget. This is because with an increase in woody vegetation, there tend to be a greater carbon stocks accumulation in the ecosystem. Even though this was not the case with Erichsfelde which showed no significant difference across the three sites, results of neighbouring South Africa's Coetsee et al. (2013)'s study, San Jose et al. (1998) and González-Roglich et al. (2014) showed a significant increase in the mean total carbon stock with increasing woody layer of protected savannas. This suggest that the increase in mean total carbon stock in these studies was mainly driven by woody plant biomass even though mean total carbon stock have been found to depend on precipitation, soil properties, vegetation type and management (González-Roglich et al., 2014). The high encroached site at Erichsfelde characterised by it's high number of bushes (Appendix 2) was expected to store more carbon due to the presence of more biomass yet, no significant difference was found across the three sites. This could be because bushes had smaller woody materials and could not store as much carbon compared to the few large trees found in the low and medium encroached sites. By contrast, trees in the low and medium encroached sites had large woody materials but were fewer in number. Hence approximately the same amount of carbon was sequestered among the three sites.

In de Klerk (2004)'s review on bush encroachment, he mentioned that Namibia's aridity and the slow growth rate of shrubs do not allow the country to be an effective carbon sink for carbon sequestration. This is in agreement with Shackleton & Scholes' (2011) and Takimoto et al. (2008) studies were they mentioned that arid and semi-arid sites such as Erichsfelde have very low ecological production potential with carbon stock pools of approximately 9 t/ha. This amount was close to the mean total carbon stock found in the medium encroached site (10.16 t/ha). However, Erichsfelde's high encroached site sequestered higher amounts of mean total carbon stock then whats anticipated (34.23 t/ha).

Although there was no significant difference across the three sites, the world is facing different environmental challenges such as land degradation and climate change, and inventory of carbon stock is a basic step in the promotion of carbon sequestration and the mitigation of land degradation and climate change (Shackleton & Scholes, 2011). This carbon stock inventory data, can be used as carbon credits to provide farmers in developing countries with an incentive for sustainable land management while promoting carbon sequestration (Lal, 2009; von Oertzen, 2009; Takimoto et al., 2008). Currently bonds are sold at US$ 1.00 per Mg of carbon conserved (Oelbermann et al., 2004). Carbon sequestration is not only an economic benefit to farmers but will also be an environmental benefit to the global community at large as well managed savanna ecosystems can remove CO_2 from the atmosphere to compensate for CO_2 emissions thus, controlling global CO_2 concentration (Takimoto ey al., 200). To date, Namibia is not yet involve in carbon trading but countries such as Costa Rica, Canada and United States have existing markets for carbon credits forestry projects (Oelbermann et al., 2004).

5.2. Above- and below-ground woody carbon stock

Total standing above-ground carbon stock of woody vegetation elements is often one of the largest carbon pools, due to its large biomass (Hairah et al., 2001; San Jose et al., 1998). Studies in tropical and temperate regions found a higher above-ground carbon stock then below-ground carbon stock as a consequence of the replacement of grasses by woody species (San Jose et al., 1998; González-Roglich, 2014). However, for this study there was no significant difference in the above-ground carbon stock between the high, medium and low encroached sites. According to González-Roglich (2014), African woody plant encroachment generates a low gain in ecosystem or total carbon stock, suggesting that carbon accumulation in these ecosystems is very slow. The low carbon accumulation might explain why there was a similarity in the above-ground carbon stock among the three bush encroached sites. The low bush encroached site at Erichsfelde was de-bushed and replanted with buffalo grass in 2006, and is used for dairy cows, calving, and weaning each year (R. Scheidt[1], Personal communication, 31 July, 2014). This may explain why the low encroached

[1] R.Scheidt is Erichsfelde farm manager.

site had only two large *Vachellia erioloba* trees (Appendix 2 & 4) and was dominated by grasses. Likewise, this explains the similarity in the above-ground carbon stock of the three sites, as carbon stock is dependent on biomass. Carbon stock can be estimated from biomass calculations (Brown, 1997; Hairiah et al., 2001b). Few big trees in the low encroached site and many shrubs and trees in the high and medium encroached sites all sequester approximately the same amount of carbon due to them having the same amount of above-ground biomass. In their study, Kuyah et al. (2012) mentioned that published equations overestimate landscape biomass due to errors in either smaller trees with dbh < 10 cm which dominate landscapes (example with the high bush encroached site) or the few larger trees wih dbh >40 cm which constitute a small percentage of all the trees but hold most of the biomass (example with the low bush encroached site). This means that the small differences in the equations for small trees could actually add up to a large amount of carbon when looking at a landscape. This might explain why no significant difference was found in above-ground carbon stock across the three sites. Likewise, similarity in the above-ground carbon stock could be attributed to the fact that arid savanna vegetation invest more in the below-ground biomass than the above-ground biomass as a protection mechanism against fire and herbivory (Grace et. al, 2006). The above-ground carbon stock results of Erichsfelde correspond to Takimoto et al. (2008)'s results where the authors found a similarity in the fenced, fodder bank and degraded land above-ground carbon stock. This similarity could be because vegetation in the savanna biome translocates about 30-50% of its carbon stock below-ground (Belay & Kebede, 2010). This translocation of biomass underground also explains why the below-ground carbon stock was found to be higher in the medium encroached site at Erichsfelde (Figure 9). Erichsfelde's medium encroached site was dominated by grasses, shrubs and trees compared to the high encroached site that was dominated by shrubs and little to no grass. The medium encroached site was dominated by trees, shrubs and grasses as it allow them to co-exist and partition resources thus, minimising competition between the different life forms. The high encroached site had little grass because the shrubs out-compete the grass when the grass layer is over utilised, living bare spaces for shrubs to grow (de Klerk, 2004; NAU, 2010).

Mills & Cowling's (2010) study in the Eastern Cape of South Africa, found a low below-ground carbon stock (2.7 ± 0.3 t/ha) in the degraded area that was changed into cultivation land. San Jose et al. (1998) also found similar results in their study were the burned area had a low below-ground carbon stock (0.15 t/ha) than the forest area. Both this results correspond to Erichsfelde's low encroached site (1.01 t/ha) that was cleared for grazing and the highly bush encroached site (5.41 t/ha). Correspondingly, Takimoto et al. (2008) had similar results for their fodder bank and the high density live fenced sites. The low below-ground carbon stock in the fodder bank and low bush encroached site was because this sites are fairly young and not likely to store as much carbon as the well-established medium enchroached site (Takimoto et al., 2008). Moreover, the fodder bank and low bush encroached sites had fewer trees, shrubs and more grass as they are used for grazing. Grasses do not have large biomass and stores less carbon stock. The high density live fenced sites in Tikomoto et al. (2008)'s study, which is similar to the high bush enchroached site at Erichsfelde's woody biomass accumulation per tree was comparatively less, resulting in a low below-ground carbon stock. Mills & Cowling (2010) accounted for the low below-ground carbon stock in the degraded area to the changes in vegetation structure which causes reduced carbon input from roots and leaf litter, greater microbial activity due to increased soil temperature, and the loss of topsoil through erosion. The same reasons can be attributed for the low below-ground carbon stock in the low encroached site. This is because the vegetation structure of low encroached site is different from the medium encroached site as it is dominated by grasses while the medium encroached site is dominated by shrubs, trees and grasses. The low encroached site also has more open spaces because there are fewer trees and shrubs and this reduces carbon input from tree roots and leaf litter. This can allow more solar radiation to penetrate into the system thereby increasing microbial activity and as a result, reducing the carbon stock. Furthermore, in most ecosystems, roots comprise up to 30% of total tree biomass and play a role in increasing the below-ground carbon stock (Oelbermann et al., 2004). The low bush encroached site grasses and their small fibrous, shallow roots might have resulted in this site sequestering low amounts of below-ground carbon. Grasses fibrous and shallow roots store less below-ground biomass than trees and shrub's lateral roots, and will therefore sequester low below-ground carbon stock compared to shrubs and trees of the medium bush encroached sites. Additionally, the increased accumulation of carbon stock in the medium bush encroached site as oppose to the low bush encroached site could be because of the fact that the low encroached site had fewer species determining the carbon partitioning in the vegetation compared to the medium encroached site that is invaded by a large number of different species storing carbon (San Jose et al., 1998). Also, the expansion of shrubs and trees as found in the medium encroached site, led to huge increase in carbons stock as a result of increase biomass below-ground and decreases in turnover and decomposition (Coetsee et al., 2013).

The distribution and spatial patterns of woody biomass in savanna ecosystems is driven by climate, topography, soils, competition, herbivory, and fire (Colgan et al., 2012; Scholes & Archer, 1997). Even though savanna trees store most of their biomass underground to prevent it from herbivory and fire, the high enchroached site with its high shrub density could not store much carbon underground because it is difficult for animals to penetrate the bushes and roam freely in this site. Consequently, the high encroached site has not been used for livestock grazing due to the dense bushes that hinder movement (R. Scheidt, personal communication, 31 July, 2014). Therefore, herbivory is not much of a challenge in the high bush encroached site and the shrubs do not invest greatly in the below-ground biomass accounting for the low below-ground carbon stock in this site. In addition to that, the high bush encroached site is densely populated by shrubs and inter-species competition is expected to be higher, resulting in a low biomass.

5.3. Soil fertility indicators and selected nutrients

Soil results revealed a similarity in the EC, SOM, pH, total N, and Mn across all three sites. Likewise, all soil ions (Cl^-, Ca^{2+}, Mg^{2+}, and Na^+) except K^+ ion (Figure 19) were found to be similar among the low, medium, and high bush encroached sites. With levels of major soil parameters such as SOM, total N and EC being similar among the three sites, positive contributions of bushes to major nutrients is not indicated in this study. The similarity in the levels of these soil parameters can be attributed to the fact that all three sites are situated within a semi-arid savanna and that savannas are thought to be nutrient limited (Vourlitis et al., 2015). All three sites had similar particle size distribution of the topsoil with dominant sandy soils (Figure 11 & Table 1), suggesting that topsoils of these sites were derived from the same parent material (Mills & Cowling, 2010). This might explain the similarity in the low ion concentration among the three sites. The low ion concentration among the three sites is a reflection of the unavailability of cation nutrients for plant growth because sandy soils in this study area are expected to have a low cation exchange capacity (Peacock & Christensen, n.d).

On the other hand, parameters such as P, K, Ca, Mg, Na, Fe, Cu, and SAR differed across the three sites (Figures 12, 13, 14, 15, 16, 17, 18 and 20). The high bush encroached site was found to significantly differ from the other two sites for most of the soil parameters that had a difference. The soil samples in the high bush encroached site were found to have greater amounts of P and K^+ nutrients. K^+ ion was the only cation that was elevated in the high bush encroached site while all other measured cations and anions were similar across the three sites. As the only readily available cation, its contribution to plant growth will be overlooked because plants need more than just one cation. On the other hand, the same samples of the high bush encroached site had lower amounts of K, Ca, Mg and Cu content which may not be readily available for plant uptake due to it's low concentration in the soil. With the high bush encroached site having lower concentration of major nutrients, this is an indication that high bush encroached areas are nutrient deficient and offer no positive contribution regarding soil nutrients. According to Vourlitis et al. (2015) and Rolo, López-Díaz & Moreno (2012), vegetation enhance surface P and K^+ availability but not Ca^{2+} and Mg^{2+}, this might explain why greater amounts of P and K^+ concentrations were found in the high bush encroached site that had a high density of woody vegetation. This also explain why there was no significant difference in the Ca^{2+} and Mg^{2+} concentrations which are not necessarily enriched by vegetation. Additionally, the high P and K^+ nutrient content in the high bush encroached soils could also be due to the decay of the lateral roots of the bushes in this site. Many woody encroaching species such as *D. cinerea* which was encountered in the high bush encroached site has the ability to increase soil P conditions as litter falls to the ground and decompose, explaining the greater amounts of P in the high bush encroached site (Blaser, Shanungu, Edwards & Olde Venterink 2014). In his study, Ward (2009) supported having high P levels in the high bush encroached site (Figure 12) when he explained that the distribution and concentration of soil N, K, and P is strongly associated with the presence of shrubs in arid and semiarid environments, due to the accumulation of plants' SOM. Moreover, shrubs from the high bush encroached site may provide a high amount of cover for nesting, bedding, temperature regulation, burrowing, and protection of small mammals from predators (Hoffman et al., 2010; Smit, 2004). This subsequently gives rise to the high bush encroached site having numerous animal life forms that contribute to the high P and K^+ concentration nutrient in the soil.

The low concentration of K, Ca, Mg and Cu, in the high bush encroached site is probably due to the differences in the original soil types between the low, medium and high bush encroached sites. Additionally, savanna shrubs have small leaves and less above-ground biomass that could return nutrients to the soil. Rolo et al. (2012) also explained that shallow rooted shrubs have the ability to deplete the N and Mg availability in the soil. This could explain the lower amounts of Mg in the shrub-dominated high bush encroached site. Also, the degraded highly bush encroached site might have went through soil erosion and leaching which took away the surface layer of the soil, causing the remaining layer to form a hard surface pan that lacks nutrients. Soil nutrients in the low and medium bush encroached site were found to be similar except for Fe that was found to be higher in the low bush encroached site and Na that was lower in the medium bush encroached site (Figures 17 and 16). High soil Fe is usually associated with acidic soils. However, the soils at Ericshfelde had a mean pH of 6.8. Thus, soil pH might not be responsible for the higher Fe concentration in the low bush encroached site. The lower Na content in the medium encroached site might be because of micro-climatic reasons as mentioned by Brady & Weil (1999)'s study that indicated that prolonged periods of high evaporation and low precipitation in semi-arid ecosystems promote the accumulation of salt elements such as Ca and Na. In addition, grasses do not return a large amount of nutrients to the soil when decomposed which may explain the low nutrient content of the grass-dominated low bush encroached site.

Woody plant encroachment in savannas has the potential to alter abiotic factors such as mineral nutrients (Belay & Kebede, 2010; Rolo et al., 2012). However, major nutrients such as SOM and N were similar among three sites at Erichsfelde. According to Belay & Kebede (2010), in their study they found a significant difference in the SOC which is a product of SOM between a cleared area and a woody encroached area. However, Belay & Kebede (2010) that studies by McCarron et al., (2003) and Smit &

Johnson (2003) found no significant difference in the SOM. The latter studies' results correspond to Erichsfelde's results. Higher standing biomass, increased litter deposition and reduced soil erosion are other factors that can increase N in soils (Mureithi et al., 2014). The three bush encroached sites at Erichsfelde had similar above-ground biomass and carbon stock. Also, the vegetation in these sites had small leaves that do not contribute a significant amount to leaf litter, which could result in a low nutrient return to the soil explain the similarity in the SOM and N content among three site. Other factors such as low precipitation, a high evaporation rate, and a low decomposition rate also contribute to low levels of nutrients. These very factors are also coupled with low primary productivity and low biomass production, and may explain the low biomass, SOM, and nutrients in this study area. Other studies results differed from Erichsfelde's results were such studies found a significant difference in the SOM and N between the forest and the cleared degraded area (de Wit et al., 2005; Coetsee et al., 2013). In these studies, the forest area had a significantly higher SOM than the degraded area. de Wit et al. (2005) explained that SOM is related to woody debris and if they are easily degradeable and slowly accumulated in the soil, less change in the SOM content is expected. It was found that there was no significant difference in the above-ground carbon stock of Erichsfelde's because of it's low above-ground biomass, this means that the vegetation in all the three sites contribute less residue and litter to the soil to be degraded and turned into SOM. The low bush encroached site at Erichsfelde was dominated by non-woody material that decomposes quickly and a high SOM and nutrient content was expected in this site. However, the non-woody material make up less biomass and if compared to the woody materials of the medium and high bush encroached sites that has more biomass but degrade at a slow rate, the amount of SOM that will be return to the soil will be the same hence, the similarities in the SOM and soil nutrients among the three sites.

Furthermore, some authors had found tree age to have an influence on the tree canopies because as trees age, eventually the canopies closes causing a more evenly distributed organic material input from litterfall (Oelbermann et al., 2004; González-Roglich et al., 2014). Erichsfelde’s tree and shrub ages were not established therefore tree age can not be accounted for the similarity in the SOM content among the three sites. In addition, the similarities in the major nutrients may be due to several factors such as species dependent changes in microclimate, litter production, topography, soil types, and soil biota. Most encroaching woody species fix N symbiotically while others aquire N from the soil (Blaser et al., 2014). Studies on N has showed a correlation between N and C indicating that N from the soil orginates from SOM (Blaser et al., 2014). Studies that found a significant difference in SOM and N reported that higher levels of these nutrients were found under the canopy of trees and shrubs (Belay & Kebede, 2010; Blaser et al., 2014; Buyer et al., 2016). For this study, soil pits were dug far from trees and this might have had a negative influence on the level of SOM and N content found in the three sites, as leaf litter and N accumulation under canopies might not have had a direct influence on the SOM and N as it was found in studies were soil samples were collected under tree canopies. The soils in Erichsfelde had more sandy content then the soils in studies done in other arid and semi-arid regions were Takimoto et al. (2008) found the soils in the degraded lands of Mali to have more silt and clay content. Soils rich in silt and clay tend to be rich in SOM compared to sandy soils (Takimoto et al., 2008; González-Roglich et al., 2014).

Erichsfelde’s soil in all three sites was dominated by sandy soils and similar low amounts of SOM was found among the three sites. Oelbermann et al. (2004) and Kuyah et al. (2012) mentioned that a continual addition of organic material from tree prunings can increase the SOM content. However, no pruning is done in the bush encroached sites of Erichsfelde thus; no addition organic material is added to the soil. This might explain the similarities in the SOM among the three sites. The results of this study clearly indicated that the bushes’ advantages are outweighed by their disadvantages as they do not positively contribute to soil nutrients, therefore causing invader bushes to be a nuisance to farmers and to the environment at large.

CHAPTER 6: CONCLUSION

The study concluded that total woody and above-ground carbon stock was similar across the three bush encroached sites at Erichsfelde farm. However, the study found the medium encroached site to sequester significantly higher below-ground carbon stock. The higher below-ground stock could be attributed to the fact that savanna vegetation store most of its biomass underground, and vegetation in the medium encroached site had larger biomass resulting in a higher below-ground carbon stock. Although to date, Namibia is not yet involve in carbon trading, this study suggests that Erichsfelde's medium encroached site can potentially contribute to the missing carbon sinks.

The highly dense bush encroached savanna could not store greater amounts of below-ground carbon stock this could be because dense shrubs in the high encroached site do not invest in the below-ground biomass and carbon stock because of low herbivory as livestock will find it difficult to roam in there as oppose to the medium encroached site that feature higher below-ground carbon storage to protect the vegetation from herbivory and fire. The fact that the medium encroached site that striked a balance between trees, shrubs and grasses sequestered more carbon stock than the high encroached site, shows that dense shrubs in the high encroached site need to be cleared and a balance between trees, shrubs and grasses strived to increase carbon sequestration in this ecosystem.

Furthermore, this study also concluded that there was no significant difference in the SOM, EC, pH, total N, Mn and ions (Cl^-, Ca^{2+}, Mg^{2+}, and Na^+) concentration across the three sites. The high bush encroached site was found to have greater amounts of soil P and K^+ and lower amounts of K, Ca, Mg, Cu and SAR. The difference in these major nutrient contents indicated a soil nutrient deficiency due to the fact that savanna bushes have less biomass that returns low nutrients to the soil. Thus it can be concluded that, despite the ecological importance of bushes, they do not positively contribute to soil nutrients and carbon sequestration, therefore interventions to clear and manage bushes in order to strike a balance between trees, shrubs and grasses need to be put in place by both farmers and decision-makers to ensure optimum carbon sequestration.

CHAPTER 7: RECOMMENDATIONS

1. The study recommends that a more comprehensive long-term and spatial explicit study involving larger and more replicates should be carried out.
2. Country and species-specific allometric equations to calculate vegetation biomass and carbon stock in a savanna biome need to be calibrated for accurate accounting of carbon stock to yield better results.
3. User-friendly softwares that can be used by managers and farmers to estimate carbon stock at the local and regional levels need to be developed.
4. The study recommends that dense shrubs be cleared. However, different species-specific methods of controlling and managing these bushes need to be employed to avoid introducing a new problem at the expense of eradicating another.
5. There is a need for the country to implement a carbon trading system with economic incentives through sustainable land management and protection of savannas to encourage farmers to sustainably manage their rangelands.

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APPENDICES

Appendix 1: Steps followed in the analysis of the soil samples at the Ministry of Agriculture, Water and Forestry's analytical laboratory.

pH

Soil pH was measured for all soil samples. The pH (KCl) of the soil was measured in a 1:2.5 soil to 1M potassium chloride ratio suspension on a mass-to-volume basis. The pH (water) was measured in a 1:2.5 soil-to-water ratio suspension on a mass-to-volume basis.

Available phosphorus

The Olsen Method was used to determine avaliabe phosphorus. This method was used because it is suitable for alkaline to neutral soils (Shoenau & Karamanos, 1993). A 5 g soil sample was measured, to which 0.3 g of charcoal was added. This charcoal had previously been washed with hot water, and was added to the sample to act as a decolouring agent (Shoenau & Karamanos, 1993). This mixture was then placed on a shaker for 30 minutes, filtered with a Whatman No: 42 filter paper, and placed in a freezer for 24 hr to allow settling of suspended particles. Five millilitres of the solution was then placed in a test tube to which 5 drops of 4 M sulphuric acid was added. Fifty millimeteres of distilled water was added to the solution and 10 ml was extracted from that solution. The extracted solution was added to a phosphate reagent and left to react for 15 minutes. Afterwards, the solution was placed in a cuvette and pipetted in a UVmini 1240 to measure the amount of light at a wavelength of 825 nm. Phosphate was measured spectrophotometrically using the Phosphomolybdate Blue Method (International Soil Reference and Information Centre, 2002).

Organic carbon (organic matter content)

The Walkley-Black Method (sulphuric acid-potassium dichromate oxidation) was used to determine the percent organic matter content in the soil samples (Nelson & Sommers, 1982). A factor of 1.74 was used to account for incomplete oxidation. Soil organic matter content was calculated as organic carbon x 1.74. The soil organic matter was estimated by measuring the weight loss when dried samples were heated in a muffle furnace at 360°C for 4 hr.

A subsample of 2.5 g of soil per sample was prepared, to which 10 ml of potassium chromate was added. Ten millilitres of concentrated sulphuric acid was added to this solution under a fume hood, and left to react for 30 minutes. This solution was cooled down by adding 200 ml of distilled water, and let to stand for 10 to 15 min. After cooling, 10 ml of concentrated orthoplosponic acid and a 0.5 ml barium diphenylamine sulphamate indicator were added to the solution in an Erlenmeyer flask, and titrated into the burette titration flask. At the beginning of the titration process, the solution was blue. The indicator solution was then added to the solution until a green colour appeared. This titration process was performed in order to determine the presence or absence of organic carbon in the sample.

Total nitrogen

The Kjeldahl acid digestion method was used to convert organic nitrogen into ammonium in order to determine total nitrogen in soil samples (Shaenau & Karamanos, 1993; Soil Science Society of South Africa, 1990). Ammonia was determined by alkaline steam distillation into boric acid and titration back to the original pH. This procedure determines the total nitrogen, specifically for ammonium nitrogen, nitrate-nitrogen, nitrite-nitrogen, and organic nitrogen content of the soil. Five grams of the soil sample was placed into a digestion tube, after which 20 ml of salicylic sulphuric acid was added and placed in a freezer overnight. Some 0.5 g of sodium was added to the solution, which was placed in a digestion chamber and heated to 200°C for 30 min. After heating, the sample was left to cool and two tablets were then added to the solution. The solution was once again heated to 400°C for 120 min and allowed to cool before placing on a distillation unit. Distilled water was added and left for 5 min in order to neutralise the acid. The solution was poured into 20 ml of boric acid to which 0.1 g of indicator solution (0.2 g of Methyl red plus 100 ml of ethanol plus 0.1 g of methylene blue) was added. The solution was subsequently titrated against 0.1 M of hydrochloric acid until the solution turned colour from green to purple. The entire digestion process took three to four hours, and the total nitrogen content was determined on an optical emission spectrometer (ICP-OES).

Extractable cations /macronutrients (available K, Mg, Ca) and Cation Exchange Capacity (CEC)

Fifty millilitres of 1 M ammonium acetate of pH 7 was added to 5 g of the soil sample for extraction. The sample mixture was shaken for one hour on a Model 1346 shaker, filtered via gravity, and placed in a freezer overnight. On the following day, 20 ml of the extract was placed in a vial to which 1 ml of 65% nitric acid was added. A dilute solution (1:10) of the extract was prepared from 1 M ammonium acetate blank (1,000 ml of ammonium extraction solution plus 50 ml of 65% nitric acid), where the content of macronutrients was determined on an optical emission spectrometer (ICP-OES) (Soil Science Society of South Africa, 1990; Shoenau & Karamanos, 1993).

Extraction with 1 M ammonium acetate at pH 7 (if pH (H_2O) <6.8 and EC<0.4 mS/cm) was performed for CEC. Extraction with a 50:50 1 M ammonium acetate and ethanol at pH 7 (if pH (H_2O)>6.8 and EC>0.4 mS/cm) were used. Calcium, magnesium, potassium, and sodium were measured by atomic absorption spectroscopy.

Soil texture and particle size analysis

Determination of silt and clay was performed using the pipette method. Soil texture was determined in terms of particle size analysis, where percentage of sand, silt, and clay were recorded. A 20 g soil sample was placed in a 250 ml volumetric flask, to which 20 ml of dispersing agent was added. The dispersing agent consisted of 40 g of sodium hexametaphosphate (sodium carbonate) in 100 ml of distilled water and 10 g of sodium carbonate mixed in a volumetric flask to yield a 1 L solution. The dispersing solution was shaken manually for 30 min and left to stand overnight. Eighty millilitres of distilled water was added to the extract and then shaken for 30 seconds. Afterwards, 5 ml of it was extracted into an aluminium dish and placed in an oven at 100°C for 3 hr, after which the silt content was weighed. These steps were repeated after five hours in order to extract the clay content.

For sand content, the whole extract was poured into, and washed through, a 5 µm sieve using distilled water, then placed into an aluminium dish. This extract was placed in an oven at 100°C for 3 hr after which the sand content was weighed. The sand fraction was determined by sieving in order to retain >53 micron fractions and textural class using the USDA soil classification system.

Electrical conductivity

Measurements were taken in the supernatant of the 1:2.5 soil: water suspension prior to measurement of pH. Units of measurement are mS/cm (1 mS=1,000 uS). For the salinity analysis, saturated soil: water paste was prepared and the extract recovered by vacuum filtration. For each soil sample, a saturated paste was prepared to determine electrical conductivity. A saturated paste was prepared by mixing the soil sample with 150 ml of water in a polyethylene plastic bottle and left to stand overnight. The next morning, the paste solution was poured into a funnel for filtering. The funnel was placed on a vacuum extractor utilising a reusable extraction tube (SAMPLETEK) connected to a syringe. The extract was placed in the funnel for one hour, which yielded approximately 60 ml of the sample. The pH and electrical conductivity (EC) were determined for the extracted solution using an electric pH meter (Multi-lab, model 540). Afterwards, 20 ml of the extract was placed in a vial with 1 ml of 65% nitric acid, and the concentration of cations was determined on the ICP-OES (Sonon, 2015). High values of more than 0.75 dS/m indicate a possible salinity hazard and were then repeated on the extract of saturated soil paste. The Sodium Adsorption Ration (SAR) is a diagnostic criterion for assessing salinity, and is equal to the concentration of sodium divided by the square root of one half the combined calcium and magnesium in the extract Sonon (2015). All concentrations were measured in milligrams per litre (mg/l).

Available micronutrients

Extraction with 0.5 M ammonium acetate and 0.5M acetic acid 0.02M EDTA at pH 4.65 at a 1:5 extraction ratio was carried out. Iron, manganese, copper, and zinc were measured by atomic absorption spectroscopy. The available calcium, potassium, and magnesium were also measured in the extract. Fifty millilitres of 1 M of acetate at pH 4.6 was added to 5 g of the soil sample. The solution-mixture was shaken for one hour and subsequently filtered via gravity. The filtered extract was placed in the freezer overnight, and 20 ml of the extract was placed in a vial the next day, to which 1 ml of distilled water was added. The diluted solution was analysed and its micronutrient content was determined using the ICP-OES according to the International Soil Reference and Information Centre (2002).

Carbonate

Carbonate estimation was conducted using treatment of dry soil with 10% hydrochloric acid and observation of effervescence.

Available sulphur

A 1:2 weight-to-volume extraction of soil with 0.01 M calcium chloride was performed to test for sulphur. Sulphate-S was estimated by measuring turbidity at 600 nm following treatment with acidified barium chloride. For an estimation of sulphate content, a soil to water extract from pH/EC measurement was made with respect to calcium by adding 1 M of calcium chloride. The filtered extract reacted with acid barium chloride and turbidity was visually compared with a standard solution of sulphate-S.

Appendix 2: Trees/shrubs species encountered in the high, medium and low bush encroached sites at at Erichsfelde farm during the survey.

High Encroached Site	**Medium Encroached Site**	**Low Encroached Site**
Species Name	**Species Name**	**Species Name**
	S. mellifera (Vahl) Seigler &Ebinger	*V. erioloba* (E.Mey.) P.J.H.Hurter
B.foetida Schinz	*V.erubescens* Welw. ex Oliv.	*B.albitrunca* (Burch.) Gilg & Ben.
V. erubescens Welw. ex Oliv.	*C.alexandri* D.Don	
V. luederitzii (Engl.) Kyal. & Boatwr.	*Z.mucronata*	
B.albitrunca (Burch.) Gilg & Ben.	*S. mellifera* (Vahl) Seigler &Ebinger	
V. luederitzii (Engl.) Kyal. & Boatwr.	*V.karoo* (Hayne) Banfi & Galasso	
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger	
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger	
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger	
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger	
S. mellifera (Vahl) Seigler &Ebinger	*S. mellifera* (Vahl) Seigler &Ebinger	
V. luederitzii (Engl.) Kyal. & Boatwr.	*S. mellifera* (Vahl) Seigler &Ebinger	

V. luederitzii (Engl.) Kyal. & Boatwr.	*S. mellifera* (Vahl) Seigler &Ebinger
A.anthelmintica Brong.	*S. mellifera* (Vahl) Seigler &Ebinger
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger
V. luederitzii (Engl.) Kyal. & Boatwr.	*S. mellifera* (Vahl) Seigler &Ebinger
B.foetida Schinz	*S. mellifera* (Vahl) Seigler &Ebinger
A.anthelmintica Brong.	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don
A.anthelmintica Brong.	*C.alexandri* D.Don
B.foetida Schinz	*Z.mucronata* Willd.
A.anthelmintica Brong.	*V.karoo* (Hayne) Banfi & Galasso
V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
A.anthelmintica Brong.	*C.alexandri* D.Don
A.anthelmintica Brong.	*C.alexandri* D.Don
A.anthelmintica Brong.	*C.alexandri* D.Don
V. luederitzii (Engl.) Kyal. & Boatwr.	*S. hereroensis* (Engl.) Kyal. & Boatwr.
V.karoo (Hayne) Banfi & Galasso	*C. alexandri* D.Don
V.tortilis (Forssk.)	*S. mellifera* (Vahl) Seigler

Galasso & Banfi	&Ebinger
S. mellifera (Vahl) Seigler &Ebinger	*X.americana*
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger
A.anthelmintica Brong.	*S. mellifera* (Vahl) Seigler &Ebinger
A.anthelmintica Brong.	*S. mellifera* (Vahl) Seigler &Ebinger
S. mellifera (Vahl) Seigler &Ebinger	*C.alexandri* D.Don
B.foetida Schinz	*G. flava* DC.
Unknown 1	*S. mellifera* (Vahl) Seigler &Ebinger
V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
Unknown 2	*C.alexandri* D.Don
V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don
S. mellifera (Vahl) Seigler &Ebinger	*C.alexandri* D.Don
G. bicolor Juss.	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don

V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
V.karoo (Hayne) Banfi & Galasso	*C.alexandri* D.Don
S. cinerea (Schinz) Kyal. & Boatwr.	*C.alexandri* D.Don
S. cinerea (Schinz) Kyal. & Boatwr.	*C.alexandri* D.Don
S. cinerea (Schinz) Kyal. & Boatwr.	*S. mellifera* (Vahl) Seigler &Ebinger
S. mellifera (Vahl) Seigler &Ebinger	*S. mellifera* (Vahl) Seigler &Ebinger
V.tortilis (Forssk.) Galasso & Banfi	*S. mellifera* (Vahl) Seigler &Ebinger
Unknown 2	*G. flava* DC.
Unknown 2	*G. flava* DC.
Unknown 2	*Z.mucronata*
Unknown 2	*C.alexandri* D.Don
Unknown 2	*C.alexandri* D.Don
V.tortilis (Forssk.) Galasso & Banfi	*C.alexandri* D.Don
S. mellifera (Vahl) Seigler &Ebinger	*C.alexandri* D.Don
B.albitrunca (Burch.) Gilg & Ben.	*V.hebeclada* (DC.) Kyal. & Boatwr.
B.albitrunca (Burch.) Gilg & Ben.	*V.hebeclada* (DC.) Kyal. & Boatwr.
D.cinerea Wight et Arn.	*V.hebeclada* (DC.) Kyal. & Boatwr.
V.karoo (Hayne) Banfi & Galasso	*S. mellifera* (Vahl) Seigler &Ebinger
V.tortilis (Forssk.)	*S. mellifera* (Vahl) Seigler

Galasso & Banfi	&Ebinger
V. luederitzii (Engl.) Kyal. & Boatwr.	*S. mellifera* (Vahl) Seigler &Ebinger
S. mellifera (Vahl) Seigler &Ebinger	*G. flava* DC.
S. mellifera (Vahl) Seigler &Ebinger	*G. flava* DC.
Unknown 2	*G. flava* DC.
Unknown 2	*G. flava* DC.
V.karoo (Hayne) Banfi & Galasso	*S. mellifera* (Vahl) Seigler &Ebinger
V.karoo (Hayne) Banfi & Galasso	*B.albitrunca* (Burch.) Gilg & Ben.
Unknown1	*S. mellifera* (Vahl) Seigler &Ebinger
Unknown 2	*S. mellifera* (Vahl) Seigler &Ebinger
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V.tortilis (Forssk.) Galasso & Banfi
V.erubescens Welw. Ex Oliv.
V. erioloba (E.Mey.) P.J.H.Hurter
Unknown 2

Appendix 3. Outcome of the Shapiro-Wilk and Levene test results for soil properties and woody vegetation carbon stock

Variable	Shapiro-Wilk P-value	Decision rule (distribution)	Levene test of Homogeneity of variance	Decision rule
Total woody carbon stock	< 0.05	Not normally	N/A	N/A
Aboveground carbon stock	< 0.05	Not normally	N/A	N/A
Belowground carbon stock	< 0.05	Not normally	N/A	N/A
Copper (Cu)	0.155	Normally	0.274	Non-homogenous
Iron (Fe)	0.340	Normally	0.089	Non-homogenous
Chloride ion (Cl^-)	0.0.056	Normally	0.091	Non-homogenous
Manganese (Mn)	0.061	Normally	0.006	Homogenous
pH	0.062	Normally	0.030	Homogenous
Soil Organic Matter (SOM)	< 0.05	Not normally	N/A	N/A
Total nitrogen (N)	0.003	Not normally	N/A	N/A
Phosphorus (P)	< 0.05	Not normally	N/A	N/A
Potassium (K)	0.045	Not normally	N/A	N/A
Calcium (Ca)	< 0.05	Not normally	N/A	N/A
Sodium (Na)	0.021	Not normally	N/A	N/A
Magnesium (Mg)	0.046	Not normally	N/A	N/A
Calcium ion (Ca^{2+})	<0.05	Not normally	N/A	N/A
Magnesium ion (Mg^{2+})	< 0.05	Not normally	N/A	N/A
Sodium ion (Na^+)	< 0.05	Not normally	N/A	N/A
Potassium ion (K^+)	< 0.05	Not normally	N/A	N/A
Sodium adsorption ratio (SAR)	< 0.05	Not normally	N/A	N/A
Electrical Conductivity	< 0.05	Not normally	N/A	N/A

Appendix 4. A map of the sampling points in the A) low and medium B) high encroached sites at Erichsfelde.