GRASSLAND DEGRADATION AND REHABILITATION OF SOIL ORGANIC CARBON AND NITROGEN STOCKS

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PREFACE

I, *Phesheya Dlamini*, declare that

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ABSTRACT

Land degradation is widely considered to adversely affect soil fertility, soil quality, constrain productivity, subsequently leading to a decline in soil organic carbon (SOC) and nutrients in soils, yet little is known about the stocks, environmental controls, destabilization mechanisms and carbon sequestration potential of degraded grassland soils. The aim of this dissertation was to evaluate (1) the impact of land degradation on SOC and nitrogen (N) stocks, distribution and SOC quality, to elucidate the environmental controls, in a communal rangeland with varying intensities of degradation, (2) to examine the rehabilitation potential of the same rangeland (3) to assess the spatial variability and replenishment potential of SOC and N stocks in a typically degraded grassland catchment.

A meta-analysis was conducted to provide a quantitative review of the impact of land degradation on SOC stocks in grassland soils, worldwide. Subsequently, the impact of degradation on SOC and N stocks and organic matter quality was investigated in a communal rangeland in the KwaZulu-Natal province, South Africa with varying intensities of degradation. Thereafter, different rehabilitation techniques were applied in the same communal rangeland to replenish SOC and N stocks. Advantage was also taken of 23 ha degraded grassland catchment to assess the spatial variability, carbon replenishment potential of SOC and N and to elucidate the main environmental controls.

Degradation resulted in a significant depletion of SOC stocks in grassland soils, both in the meta-analysis and field experiment. The meta-analysis indicated that the depletion of SOC stocks as a result of degradation was more pronounced in sandy acidic soils under dry climate than clayey soils under wet climate. The field experiment showed that degradation significantly depleted SOC stocks by 89% and N stocks by 76% in sandy acidic soils at the study site. The reduction of the stocks due to degradation was accompanied by an increase in soil bulk density, a decrease in soil aggregate stability and concomitant decrease of macro and micronutrients (e.g, Ca by 67%; Mn, 77%; Cu, 66% and Zn, 82%). SOC and N stocks decreased sigmodially with a linear decrease in grass aerial cover. After two years, the "Savory and fertilization techniques increased SOC stocks by 6.5% and 3.9%, respectively. At catchment level, degradation led to high spatial variability of SOC and N stocks controlled primarily by soil surface characteristics, including grass cover, soil surface crusting and secondarily by topography. The carbon replenishment potential of

degraded grassland catchment was estimated to be 4.6 t C ha^{-1}, with clay-rich Acrisols having a greater capacity to replenish SOC stocks than sandy Luvisols and Gleysols.

In conclusion, the results of this dissertation indicate that degradation results in high depletion of SOC and N stocks. However, rehabilitation has the potential for carbon sequestration and can lead to more sustainable grassland ecosystems.

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CHAPTER 1

1. INTRODUCTION

Soil organic matter (SOM) for which soil organic carbon (SOC) constitutes a major proportion is at the heart of key functions in the terrestrial ecosystem. SOM serves as a source of energy for soil microorganisms. By breaking-down fresh SOM, soil organisms input essential nutrients to soils that enhance plant productivity (Wardle et al., 2004). Moreover, SOM serves as a filter and buffer for chemical and biological contaminants in the soil, thus controlling the quality of surface and ground waters (Palm et al., 2007). Lastly, SOM constitute a major reservoir of carbon (~1500 Gt of SOC), three times as much as in the atmosphere and in the terrestrial vegetation (Houghton, 2007). SOC plays an important role in the global carbon cycle and associated climate change. In this context, any loss of SOC will have dramatic consequences on ecosystem functioning.

Grassland soils, which occupy about 40% of the world's land surface and store approximately 10% of the global SOC (Suttie et al., 2005) are threatened by intensive degradation (Lal, 2004). It is estimated that up to 30% of grasslands have already been affected by degradation (Suttie et al., 2005). There is evidence that degradation results in significant losses of stocks, especially in the topsoil (Daily, 1995; Lal, 2004), but this still remains unexplored. For instance, in a Chinese alpine grassland Wu and Tiessen (2002) reported that degradation decreased SOC stocks by 33%, while a more recent study by Dong et al. (2012) showed a decrease of 90%.

Yet, there is little information on the rate, factors and mechanisms controlling the depletion of SOC stocks by degradation in grassland soils, with a host of key research questions which remain to be answered:

- What is the impact of grassland degradation on changes in SOC and soil organic nitrogen (SON) stocks and the quality of OM in degraded grassland soils?
- How best can the degraded grasslands be rehabilitated to replenish associated soil stocks and improve ecosystem functioning?

This thesis aimed to investigate these research questions in the South African context, while giving a broader perspective to improve our understanding of the effects of land degradation on SOC and nutrient dynamics, to elucidate the environmental controls and to inform on grassland rehabilitation and soil stock replenishment.

For many smallholder farmers in South Africa, grasslands make a significant contribution to food security by providing part of the feed requirements of livestock used for meat and milk production (O' Mara, 2012). However, large parts of the communal rangelands, especially in the foothills of the Drakensberg Mountains are being degraded (Figure 1.1) due to an increase in the anthropogenic pressure on the land and to poor land management (Suttie et al., 2005). Figure 1.1 shows a degraded communal grassland in Potshini, Drakensberg where there is no rotation of grazing areas. In this study, the term 'degradation' refers to the depletion of SOC and nutrient stocks due to soil disturbances including grazing, livestock trampling, soil erosion and land mismanagement, ultimately impairing the soil fertility, productivity and consequently reducing the capacity of grasslands to carry out their key ecosystem functions (Daily, 1995; UNEP, 2007). The loss of grass cover on these steep Drakensberg Mountain slopes has resulted in the reduction of soil water infiltration, increased runoff, soil and SOC erosion (Everson et al., 2007; Dlamini et al., 2011; Mchunu and Chaplot, 2012). As such, grassland degradation is jeopardizing both the environment and the economic development of the rural livelihoods because smallholder farmers not only lose land that could be used for crop production, but also for livestock production (Everson et al., 2007).

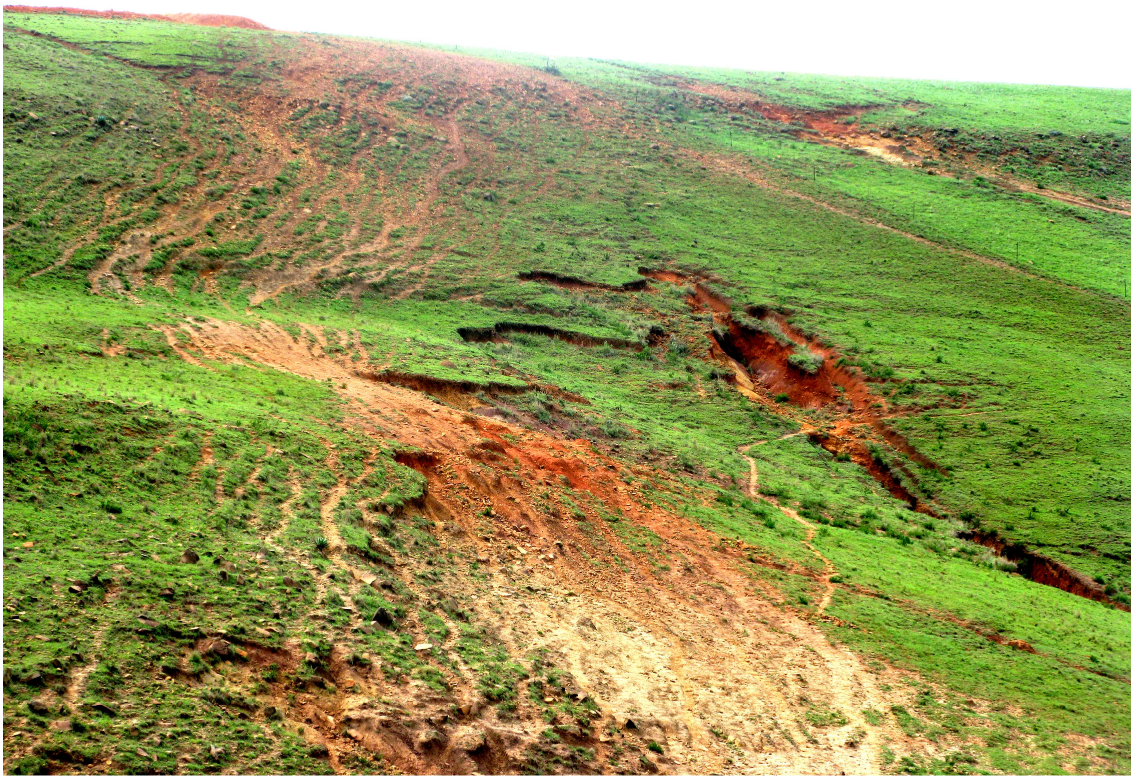


Figure.1.1 A picture of a communal rangeland in the Potshini catchment in the KwaZulu-Natal Province, South Africa, showing different land degradation intensities.

The following four chapters in this thesis will address these knowledge gaps and research questions.

Chapter 2 provides a quantitative review to elucidate the impact of grassland degradation on changes in SOC stocks and the main environmental controls, worldwide. This was based on a comprehensive meta-analysis conducted using 29 studies, with 630 soil profiles from 131 temperate, sub-tropical and semi-arid sites, to compare SOC stocks in the topsoil of non-degraded and degraded grassland soils. This chapter which aimed at assessing the impact of degradation on SOC stocks and its main factors of control has been submitted for publication in Agriculture Ecosystem and Environment.

In chapter 3, the issue of land degradation impact on SOC depletion is investigated in South Africa in the Potshini communal rangeland in the foothills of the Drakensberg Mountains in the KwaZulu-Natal province. The main objective was to evaluate the consequences of grassland degradation, i.e. the decrease in grass aerial cover, on SOC and SON stocks. The depletion in stocks was investigated for grass aerial covers from 100%

(Cov100, corresponding to a non-degraded grassland) to 50-75% (Cov75), 25-50% (Cov50) and 0-5% (Cov5, corresponding to a heavily degraded grassland). This chapter has been accepted for publication in Geoderma.

Chapter 4 investigates the effect of grassland rehabilitation techniques on the same communal rangeland and the replenishment of both SOC and SON stocks. A technique (“Savory”), which involves short-duration (5 days $year^{-1}$), high intensity grazing (1200 cows ha^{-1}) and followed by livestock exclusion for 362 days was compared to five common grassland rehabilitation strategies: (1) traditional communal free grazing; (2) livestock exclosure; (3) livestock exclosure + topsoil tillage; (4) livestock exclosure + NPK fertilization (2:3:3, 22 at 2 t ha^{-1}); (6) annual grassland burning. This chapter is in preparation to be submitted for publication in Geoderma.

Finally in chapter 5, the spatial variation of both C and N stocks due degradation and its replenishment potential were assessed in an entirely grazed catchment. This chapter has been submitted for publication in Geoderma.

During the course of this work, additional information was collected at the study site in which accrued results have been published in the following peer reviewed scientific publications (Appendix):

Dlamini, P. Chivenge, P. Manson, A. Chaplot, V. 2014. Land degradation impact on soil organic carbon and nitrogen stocks of sub-tropical humid grasslands in South Africa. *Geoderma*. 235-236:372-381.

Dlamini, P and Chaplot, V. 2012. On the interpolation of volumetric water content in research catchments. *Physics and Chemistry of the Earth*. 50-52:165-174.

Dlamini, P. Orchard, C. Jewitt, G. Lorentz, S. Titshall, L. Chaplot, V. 2011. Controlling factors of sheet erosion under degraded grasslands in the sloping lands of KwaZulu-Natal, South Africa. *Agricultural Water Management*. 98:1711-1718.

CHAPTER 2

2. ASSESSMENT OF THE IMPACT OF GRASSLAND DEGRADATION ON SOIL ORGANIC CARBON STOCKS AND CONTROLLING ENVIRONMENTAL FACTORS: A META-ANALYSIS

Abstract

Grasslands occupy about 40% of the world's land surface and store approximately 10% of the global soil organic carbon (SOC) stock. This SOC pool, in which a larger proportion (ca 60-70%) is held in the topsoil (0-0.3 m), is strongly influenced by grassland management. Despite this, it is not yet fully understood how grassland soils respond to degradation, particularly for the different environmental conditions found globally. The objective of this review was to elucidate the impact of grassland degradation on changes in SOC stocks and the main environmental controls, worldwide. A comprehensive meta-analysis was conducted using 29 studies with 630 soil profiles from 131 temperate, sub-tropical and semi-arid sites, to compare SOC stocks in the topsoil of non-degraded and degraded grassland soils. Grassland degradation significantly reduced SOC stocks by 16% in dry climates (<600 mm) compared to 8% in wet climates (>1000 mm). The depletion of SOC stock induced by degradation was more pronounced in sandy (<20% clay) soils with a high SOC depletion of 10% compared to 1% in clayey (≥32% clay) soils. Furthermore, grassland degradation significantly reduced SOC by 14% in acidic (pH ≤ 5) soils, while SOC changes were negligible for higher pH. Degradation caused SOC losses of up to 14% in C_3 grass species compared to 4% in C_4 grasses. Given that 30% of grasslands worldwide are degraded, the amount of SOC likely to be lost would be 4.05 Gt C with a 95% confidence between 1.8 and 6.3 Gt C (i.e. from 1.2 to 4.2% of the whole grassland soil stock). There is, therefore, considerable potential that changes in SOC stocks induced by grassland degradation will have a significant impact on SOC dynamics in such soils with positive feedbacks to the atmospheric C pool. These results have implications for grassland management and underscore the need to establish effective rehabilitation strategies especially in heavily degraded grasslands where SOC stocks are highly vulnerable to SOC depletion.

Keywords: SOC stocks; spatial variation; controlling factors; grasslands; grassland degradation

2.1 Introduction

Soil is the third largest reservoir of carbon (C) next to the lithosphere and the oceans. Globally, soil contains about twice the amount of C in the atmosphere and more than three times in above-ground biomass (Batjes, 1996; Batjes and Sombroek, 1997; Jobbagy and Jackson, 2000). Historically, terrestrial C pools, have been largely depleted by anthropogenic activities such as deforestation, tillage and overgrazing (Lal, 2004). It has been widely argued that a shift in land use or land management in agroecosystems could potentially sequester as much as 30 to 40% organic C back into the soil (Lal, 2004). A meta-analysis of 74 studies by Guo and Gifford (2002) reported that conversion of croplands to grasslands could result to SOC gains of 19%, while a global analysis of 115 studies by Conant et al. (2001) estimated much lower SOC gains varying from 3 to 5%. Under degraded croplands in the Highveld region of South Africa characterized by a temperate climate, with 6 to 8 months dry spells Preger et al. (2010) indicated that consideration of the initial level of degradation was important, as they observed 30% SOC stock gains (i.e. 300 kg C ha^{-1} yr^{-1}) when less degraded croplands were converted to grasslands to as much as 70% (i.e. 500 kg C ha^{-1} yr^{-1}) for heavily degraded croplands. The variable response of SOC stocks to shifts in land use may be related to environmental factors including precipitation as shown by Guo and Gifford (2002), who reported greater SOC gains in areas receiving low mean annual precipitation (<500 mm) than in areas receiving high mean annual precipitation (>500 mm).

The cessation of tillage practices in agroecosystems has been shown to increase SOC stocks. Six et al. (2002) found that the no-tillage in croplands increased SOC stocks by an average of about 325±113 kg C ha^{-1} yr^{-1} irrespective of the climatic conditions (tropical *vs*. temperate). A review of 140 comparative studies examining the impact of tillage on SOC stocks by Baker et al. (2007) similarly found significant enrichment of SOC in no-tilled soils, but only in the 0-0.3m topsoil layer. Interestingly, there were no significant differences in SOC stocks between tilled and no-tilled soils when the entire soil profile (top 1m of the soil) was considered. Indeed, it has been argued that because of such discrepancies in the results, the actual SOC sequestration potential of soil remains largely uncertain (Powlson et al., 2011).

Grasslands cover about 40% of the world's land surface (Suttie et al., 2005) and store approximately 10% of the global SOC stock of 1,500 Gt (Lal 2004; Batjes, 1996). Grasslands are an essential component of the biogeochemical cycle and provide key ecosystem goods and services (Suttie et al., 2005; FAO, 2010). Grasslands including both pastures and rangelands support biodiversity and are used extensively for the production of forage to sustain the world's livestock (Asner et al., 2004; Bradford and Thurow, 2006). Not only is the SOC pool in grassland soil critical for climate change, but it yields important feedbacks to soil fertility, plant productivity, soil aggregate stability, water holding capacity and overland flow regulation (Sombroek et al., 1993; Lal, 2004).

It is estimated that up to 30% of grasslands worldwide have been affected by grassland degradation (Suttie et al., 2005). So far, the relative effect of degradation on SOC stocks has been difficult to predict because of the paucity of data (FAO, 2010), particularly in grassland soils. Because a greater proportion (ca 60-70%) of the SOC stock is held in the top 0.3 m of the soil (Gill et al., 1999), it has been postulated that grassland soils could be highly sensitive to, and strongly vulnerable to grassland degradation than previously thought.

Interestingly, depletion of SOC stocks in grassland soils has been shown by many studies to be largely affected by grazing. However, grazing effects on SOC stocks have been found to be highly variable with some studies showing a decrease in SOC with grazing, for example, Martinsen et al (2011) found that SOC stocks declined by 84% after 7 years of grazing in Norway, with 0.76 kg C m^{-2} in ungrazed compared to 0.64 kg C m^{-2} in heavily grazed grasslands. Steffens et al. (2008) found that 30 years of grazing in semi-arid Mongolian grasslands resulted in a 55% decrease in SOC stocks, with 0.64 kg C m^{-2} in grazed compared to 1.17 kg C m^{-2} in ungrazed grasslands. Franzluebbers and Stuedemann (2009) observed that 44% of SOC stocks were lost after 12 years of grazing, with 0.051 kg C m^{-2} in heavily grazed compared with 0.117 kg C m^{-2} under ungrazed grasslands. In contrast, some studies have shown that grazing results in an increase in SOC stocks (Smoliak et al., 1972; Bauer et al., 1987; Frank et al., 1995 and Derner et al., 1997), while others have reported no difference in SOC stocks after grazing (e.g. Johnston et al., 1974; Domaar et al., 1977). These contradictory findings demonstrate that the underlying processes affecting the response of SOC stocks to grassland degradation are not well understood. Thus, there is a need to study grassland degradation over a wide range of environmental and management conditions.

SOC losses due to degradation are dependent on soil texture. A number of studies have shown that fine-textured soils relatively have greater SOC stocks than coarse textured soils (Hassink, 1992; Hassink, 1997; Bird et al., 2000 and Brye and Kuric, 2003). However, the effect of grassland degradation in soils differing in texture is largely unknown. In a previous study, Parton et al. (1987) using 560 soil profiles showed that SOC stocks and soil texture were correlated, with SOC stocks greater in fine textured soils than sandy textured soils. Across two degraded grassland soils with contrasting texture in the USA, Potter et al. (2001) found that grassland degradation reduced SOC stocks by 41% in coarse-textured than fine-textured soils. Clayey soils have a greater stabilizing influence on SOC than sandy soils, probably due to a large surface area, which form stable organo-mineral complexes that protect C from microbial decomposition (Feller and Beare, 1997; Six et al., 2000). Soil texture may interact and be confounded with other environmental factors such as climate, which may profoundly affect SOC depletion in grassland soils (Feller and Beare, 1997). Climate can impose constraints on the processes that control SOC stabilization, which may result in different changes of SOC under different environmental conditions (Virto et al., 2012). In a review of 12 studies totalling 22 data points, Conant and Paustian (2002) identified mean annual precipitation (MAP) as the main factor controlling C sequestration in degraded grasslands. Since the publication of this review, studies covering a wider range of environmental conditions have become available, thus allowing a robust evaluation of the impact of not only climate, but also, altitude and soil properties, time and grass type (C_3 *vs* C_4) and their likely interactions on SOC dynamics.

Because the response of soils to grassland degradation is expected to vary from site to site, the main objective of this review was to assess the level of SOC stock depletion in grassland soils worldwide and to identify the main environmental factors of control. This study considered analytical data from 621 soil profiles gathered from 55 studies in tropical, temperate and semi-arid grasslands globally.

2.2 Materials and Methods

2.2.1 Literature search and database construction

An exhaustive literature search was conducted using online search engines (Google Scholar, ISI Web of Knowledge) and electronic bibliographic databases (Science Direct, Springerlink). The key words that were used to search the literature were SOC, C sequestration, soil C storage, C depletion, grazing and grasslands. The studies had to meet specific criteria to be included in the data set and include those that, i.e.: (1) reported the concentration of SOC expressed as percent of the total soil mass (%) or g C kg^{-1} and the soil bulk density (ρ_b); (2) reported SOC stock, which is the quantity of carbon per unit area expressed in kg C m^{-2} or kg C ha^{-1} (Equation 1); (3) determined SOC by dry combustion using the C and N elemental analyzer or the Walkley and Black oxidation method.

$$SOC_S = x_1 \times x_2 \times x_3 (1 - \frac{x_4}{100}) \qquad (1)$$

where SOC_s is the C stock (kg C m^{-2}); x_1 is the C concentration in the <2 mm soil material (g C kg^{-1} soil); x_2 is the soil bulk density (kg m^{-3}); x_3 is the thickness of the soil layer (m); x_4 is the proportion (%) of fragments of > 2mm.

A quantitative database was then established in excel based on the published literature (Table 2.1). The following environmental site characteristics were, when available, extracted from the research papers: altitude above sea level: Z; mean annual precipitation: MAP; mean annual temperature: MAT; latitude: LAT; CLAY, SILT, SAND: clay, silt, sand content in the topsoil; bulk density: ρ_b and sampling depth. When not reported, the information was gathered from global assessments, such as the WORLDCLIM database with a spatial resolution of 30 arc seconds (approximately 1 km) for MAP and MAT (Hijmans et al., 2005), and the global digital elevation model of the U.S Geological Survey with a spatial resolution of 30 arcs seconds for terrain morphology (http://eros.usgs.gov/#Find_Data/Products_and_Data_Available/GTOPO30). The global land cover database (Figure 2.2) of the International Geosphere Biosphere Program at 1 km × 1 km spatial resolution was used for the classification of grassland areas globally (IGBP, 1998).

Environmental conditions were summarized by the number of categorical variables as described in Table 2.6: Degradation intensity was classified into three classes: lightly degraded (exhibit a small decline in grassland productivity and retaining full potential for recovery), moderately degraded exhibiting a reduction in grassland productivity and amenable to restoration), heavily degraded (exhibiting a substantial reduction in grassland productivity) following Daily et al. (1995). The degradation classes are roughly comparable to those defined in the soil survey of each site. Degradation intensity refers to the level of disturbance as specified by the authors of the source literature, which were classified into three categorical degradation classes: heavy moderate and light. Soil texture, which was classified into three categorical textural classes: sand (<20% clay), loam (20-32% clay) and clay (>32% clay) based on the textural triangle (Shirazi and Boersma, 1984). Soil pH which was classified into three categorical pH classes: strong acidic (≤5), weak acidic to weak alkaline (5-7) and strong alkaline (≥7). MAP, which was divided into three precipitation classes: dry (<600 mm), intermediate (600-1000 mm) and wet (>1000 mm) following FAO guidelines for agro-climatic zoning (Fischer et al., 2001). Grass type, which refers to the type of grasses dominant in the community, was classified into three categories: C_3, C_4 and mixed C_3-C_4 depending on the authors' classification (McSherry and Ritchie, 2013). Studies were classified into short-term (<10 years) and long-term (>10 years) according to the duration of the study as reported in the literature.

Given that various studies reported SOC for different soil surface layers (e.g. from 0.05m, Dlamini et al., 2011, to 0.30 m, Maia et al., 2009), SOC_S data were transformed into SOC density (SOC_D) (Equation 2) following Sombroek et al. (1993) and Batjes (1996) to facilitate comparison among the results. The analysis was further restricted to the 0.30 m topsoil layer since that depth increment contains the highest (70%) stock of SOC in grassland soils (Gill et al., 1999), and potentially has the strongest response to grassland degradation. SOC_D in kg C m^{-3} allowed for comparison of SOC stocks in horizons and soil layers with different depth increments (Equation 2).

The density of SOC in the topsoil was calculated as

$$SOC_D = SOC_S \times \frac{1}{x_3} \tag{2}$$

where SOC_D is the SOC density (kg C m^{-3}), SOC_S is the SOC stock (kg C m^{-2}) and x_3 is the thickness of the soil layer (m).

The database on SOC stocks in grasslands contained 628 data points from 45 study sites across the world originating from semi-arid, temperate and tropical grassland environments. The geographical distribution of the data was as follows. A majority of the studies came from North America (38%), Europe (22%), and only a few from Africa (16%), South America (7%) and New Zealand (4%). The geographic distribution of the study sites included in the review is depicted in Figure 2.1. The database consisted of SOC_S, SOC_D, MAP, MAT, Z, LAT, LONG, ρ_b, SAND, SILT and CLAY. The grassland sites exhibited a wide range of environmental conditions (Table 2.2). Mean annual precipitation ranged from 240 to 2000 mm, with an average of 960 mm, while MAT ranged from -1.6 to 22.7^{o}C, with an average of 11.5^{o}C. Altitude ranged between 10 and 2721 m, with an average of 847 m. Soil texture exhibited substantial variations with CLAY ranging from 3 to 85%, while SILT and SAND ranged from 3 to 73% and from 2 to 81%, respectively.

2.2.2 Determination of grassland degradation impact on SOC stock

In order to investigate the potential change in SOC_D induced by grassland degradation, a subset of studies that examined SOC_D in both degraded and non-degraded grasslands were extracted from the original database of 628 SOC data points. In this study, the term 'grassland degradation' refers to the depletion of SOC stocks due to soil disturbances including grazing, livestock trampling, soil erosion and land mismanagement, ultimately impairing the soil fertility, productivity and consequently reducing the capacity of grasslands to carry out their key ecosystem functions (Daily, 1995; UNEP, 2007). It was necessarily assumed that all "non-degraded grasslands" referred to in the source literature are largely grasslands that had experienced minimal or no soil disturbance, although it is recognized that non-degraded grasslands are likely to be genuinely pristine. Degraded grasslands include grasslands disturbed by grazing, soil erosion and land mismanagement.
The effect of grassland degradation on SOC stocks was compared among the studies by using the change in the SOC stock as a result of grassland degradation relative to initial value of the SOC stock. This variable (SOC_{DC}) was calculated as follows:

$$SOC_{DC} = \frac{SOC_{D-ND} - SOC_{D-D}}{SOC_{D-ND}} \times 100 \qquad (3)$$

where SOC_{DC} is the change in SOC_D, $SOC_{D\text{-}ND}$ is SOC_D in the non-degraded grassland grasslands and $SOC_{D\text{-}D}$ is SOC_D in degraded grasslands. An assumption was made that the level of degradation considered in these studies was in accordance with what was commonly found in the site. Since this variable (SOC_D) can now be compared between different sites, a meta-analysis was performed, which included other environmental factors (mean annual precipitation, mean annual temperature, soil texture, grass type, soil pH and grazing intensity). Twenty-nine studies met these criteria, representing 218 comparative sites reporting SOC stocks in both degraded to non-degraded grassland soils.

2.2.3 Statistical analysis

Statistical analysis was performed on the 628 data set. First, the basic statistics was computed and this included minimum, maximum, mean, median, variance, standard deviation, skewness, 25^{th} quartile and 75^{th} percentiles, kurtosis, standard error (SE) and coefficient of variation (CV). Second, a correlation matrix was applied to the data set to identify the univariate relations between the SOC stocks in grassland soils and the environmental factors. Third, a principal component analysis (PCA) was applied to the data to identify the multiple relationships between SOC stocks and the controlling environmental factors using STATISTICA 7.0 (StatSoft, Inc., Tulsa, OK). A PCA is a statistical tool for data analysis, a dimensionality reduction technique that identifies structure in large sets of correlated multivariate data (Webster, 2001). Beyond that, multiple regression analysis was applied to the data to model and spatially display the influence of grassland degradation on SOC_D and SOC_{DC}.

2.2.4 Meta-analysis

The data was analyzed using MetaWin 2.1 software (Rosenberg et al., 2000). The meta-analysis was used to determine the mean effect of grassland degradation on SOC stocks in grassland soils. The natural log (InR) of the response ratio was used as an effect size for the meta-analysis. The natural log linearizes the metric by treating deviations in the numerator and denominator the same and also provides more normal sampling in small samples (Hedges et al., 1999). The response ratio was calculated as the ratio of SOC between degraded and non-degraded grasslands using the following equation:

$$r = Xe / Xc \quad (4)$$

Where Xe is the mean for the treatment and Xc is the mean of the control group (Rosenberg et al., 2000). A resampling based on 4999 bootstrap samples was used to generate the mean effect size of each categorical variable and 95% confidence intervals. The bootstrapping technique was used to generate confidence intervals on the mean effect size of the whole data set and for each categorical variable. The number of iterations used for bootstrapping was 4999 (Rosenberg et al., 2000). Grassland degradation effect on a response variable was considered significant if the 95% confidence interval did not overlap zero. The means of categories were considered significantly different if their 95 confidence intervals did not overlap 0 (Hedges et al., 1999). Meta-analysis was performed using a non-parametric weighting function and confidence intervals (CIs) were generated using bootstrapping. Effect sizes were weighted by replication. For ease of interpretation the response ratio was transformed to percentage. These percentages represent the mean percentage change for a given site that has been degraded.

2.3 Results and Discussion

2.3.1 Global distribution of SOC stocks in grassland soils

SOC content (SOC_C) ranged between 0.2 and 293 g C kg^{-1} with a coefficient of variation (CV) of 115%. The average SOC_C in the topsoil of both degraded and non-degraded grasslands worldwide computed from 625 observations was 34.9 g C kg^{-1}standard error (±) 1.6 g C kg^{-1} (Table 2.3). SOC stocks (SOC_S) ranged between 0.1 and 38.8 kg C m^{-2}, with an average of 5.0±0.2 kg C m^{-2}. SOC density (SOC_D) ranged from a minimum of 0.7 kg C m^{-3}, reported by Baisden and Amundson (2002) for sandy loamy grassland soils under a Mediterranean climate (hot, dry summers and cool, wet winters) in USA to a maximum of 194.0 kg C m^{-3} reported by Schipper et al. (2007) under a temperate climate, with a latitudinal gradient of 36-46°S in New Zealand grassland soils. The average SOC_D was 32.2± 1.3 kg C m^{-3} with a CV of 88%, suggesting that SOC stocks are spatially highly variable in grassland soils worldwide.

SOC_S were found to be highest in temperate regions (Figure 2.3). This trend of greater SOC stocks in temperate climates can be explained by the lower average temperatures in this region, which slows the rate of decomposition, hence accumulating SOC as pointed out by Davidson and Janssens (2006). SOC stocks were also found to be high in the lowland areas of the humid tropics. Greater SOC stocks in humid tropics could be explained by generally faster turnover of soil organic matter and enhanced decomposition due to the higher moisture regimes (Trumbore, 1993). Post et al. (1982) observed that the higher annual precipitation in the humid tropics favours high biomass production which in turn increases C inputs into the soil. The lowest SOC stocks were observed in arid to semi-arid grassland soils. This trend of lower SOC stocks in semi-arid to arid grassland environments may be explained by the low rainfall amounts which decrease biomass production and organic matter decomposition, thus reducing C inputs into the soil (Amundson et al., 1989; De Deyn et al., 2008).

2.3.2 Relationship between SOC stocks and selected environmental factors

MAP and MAT

The climatic variables, MAP and MAT explained much of the variability of SOC_D in grassland soils worldwide. Correlation matrix (Table 2.4) showed that SOC_D was correlated positively with MAP (r = 0.20; P < 0.05), LONG (r = 0.39; P < 0.05) and correlated negatively with LAT (r = -0.20; P < 0.05) and Z (r = -0.20; P < 0.05). Further insights on the relationship between SOC_D the selected environmental factors was explored using a PCA (Figure 2.4). The first two axes of the PCA generated using data from non-degraded grasslands only (Figure 2.4A), explained 72% of the total data variation within the data set. The first PCA axis (Axis 1), which accounted for 41% of the variance was positively correlated with MAP and MAT and negatively correlated with LAT. Axis1 was thus, interpreted as an axis of "*tropicality*" which is defined as "the quality characteristic of the tropics", with MAP and MAT increasing as LAT decreases. The second PCA axis (Axis 2) which accounted for 31% of the variance was correlated with Z, and was interpreted as an axis of elevation. While SOD_D seemed to be somewhat slightly correlated to altitude, indicating that there was a tendency for SOD_D to increase as *tropicality* increased and altitude decreased. This suggests that SOD_D was greater in the lowland areas of the humid tropics. The other scatter diagram shown in figure 2.4B which accounted for Axis 2 and Axis 3 revealed that SOC_D was opposed to MAT, suggesting that there was a tendency for SOC_D to increase as MAT decreases.

Such an impact of climatic factors (MAP and MAT) on the spatial variation of SOC_D in grassland soils is consistent with classical studies based on soil-forming factors (Jenny, 1941). Based on a comprehensive analysis of 2700 soil profiles, Post et al. (1982) found that SOC_D increased with increasing MAP and decreasing MAT. Likewise, Jobbágy and Jackson (2000) found that SOC_D increased with precipitation and decreased with temperature. Therefore, climate exerts a strong control on the amount of SOC stocks in grassland soils.

Altitude.

The distribution of the SOC_D in grassland soils was also found to be related to Z (F = 33, p < 0.01). The complementary multivariate analysis which explained 72% of the variability of SOC stocks in grassland soils showed that there is tendency for SOD_D to increase as the altitude decreased (Figure 2.4). This increasing pattern of SOC stocks at low altitude grassland regions is corroborated by results of Garcia-Pausas et al. (2007) in the Pyrenees mountain grasslands of Spain, who reported higher SOC stocks at lower altitudes compared to lower SOC stocks at higher altitudes. They suggested that the lower SOC stock at higher altitudes was a result of the low MAT conditions ranging between -0.7^{o}C and 5^{o}C, which limited net primary productivity. Along an altitude gradient varying between 1665 and 2525 m.a.s.l across a Swiss alpine grassland, Hitz et al. (2001) found that C inputs and root turnover times decreases, with increasing altitude.

The direct interactions between cold temperatures, water logging conditions and substrate quality, the combination of which favours soil organic matter accumulation (Hobbie et al., 2000; Grosse et al., 2011; Baumann et al., 2009). Firstly, the low temperatures in the high latitude cold regions slows the rate of decomposition, thus accumulates C into the soil. Secondly, the poorly drained soils in altitude regions restrict the decomposition of organic matter due to the lack of oxygen for soil organisms. Thirdly, grassland regions contain a substantial fraction of substrate quality that decomposes slowly, making it not to be readily incorporated into the soil (Hobbie et al., 2000).

Soil texture.

SOC_S in grassland soils were found to be significantly positively correlated with soil texture, specifically CLAY (r = 20, P < 0.05). This observation accords with results obtained from other studies that have shown that fine textured soils have a greater capacity to stabilize SOC than coarse textured soils. Hassink (1992) in the topsoil layer of various grassland sites in the Nertherlands found that clayey soils had on average higher SOC (101 g kg^{-1}) compared to 60.2 g kg^{-1} in sandy soils. For similar neighbouring Dutch grassland soils, Hassink (1997) soils found higher SOC (37 g kg^{-1}) in fine-textured soils (<10 g kg^{-1}) than in coarse textured soil. Garcia-Pausas et al. (2007) found higher SOC (93.9 g kg^{-1}) in silty loamy soils compared to 60 g kg^{-1} in sandy soils in the Pyrenees mountain grasslands of Spain. Bird et al. (2000) found on average a twofold increase in SOC in clayey soils compared to sandy soils in a tropical grassland soil in Zimbabwe (19.7 *vs* 7.26 g kg^{-1}). Brye and Kucharik (2003) across two grassland soil topochronosequences in the USA found in average higher SOC (26.4 g kg^{-1}) in fine textured soils compared to 8.9 g kg^{-1} in coarse textured soils. Greater SOC stocks in fine textured soils compared to coarse textured soils has been corroborated by (Parton et al., 1987), and is attributed to the interaction of SOC polymers with clay surfaces and the effective stabilization of SOC by clay and silt content, consequently protecting organic matter from decomposers (Six et al., 2002). In addition, SOC is generally retained much more efficiently in clayey soils because of their higher nutrient and water holding capacities (Skjemstad et al., 1996).

2.3.3 Degradation impact on SOC stocks

Overall, the mean effect size of changes in SOC stocks induced by grassland degradation for all 131 direct comparisons from 29 studies was -9%, with a 95% confidence interval of -14% to -4% (Figure 2.5). The meta-analysis revealed that changes in SOC stocks were affected by the intensity of degradation, with SOC stocks being significantly reduced when the grassland was heavily degraded. Grassland degradation significantly reduced SOC stocks by 13% in heavily degraded soils and by 7% in lightly degraded ones (Figure 2.5). Such a result suggests that minimizing the intensity of degradation can decrease the depletion of SOC stock by 6%. The initial level of degradation has been shown to affect the C sequestration potential of grassland soils. In a temperate soil, characterized by different intensities of degradation, Preger et al. (2010) found SOC stock gains of 30% after the conversion of lightly degraded croplands to grasslands and up to 70% when heavily degraded croplands were converted to grasslands.

2.3.4 Impact of controlling factors on SOC stocks depletion

Grassland degradation reduced SOC stocks in grassland soils, although the effect size varied with degradation intensity, soil texture, soil pH, climatic factors (MAP and MAT), duration and grass type.

MAP, Changes in SOC stocks induced by grassland degradation were found to be significantly correlated with MAP (Figure 2.8), with areas receiving ≤600 mm MAP showing a greater decline in SOC stocks (-16%) than areas receiving 600-1000 mm MAP (-1%). These results are also corroborated by Ruiz-Sinoga and Diaz (2010) who investigated the soil degradation level at eight sites (469 topsoil samples) along a Mediterranean precipitation gradient varying from 240 to 1100 mm yr^{-1} in southern Spain. They found that grassland degradation significantly reduced SOC stocks by 18% in dry climates compared to wet climates. The higher SOC stocks in areas receiving high MAP (1100 mm yr^{-1}) was shown to be related to higher carbon inputs as a result of increased aboveground plant biomass which enhanced ecosystem stability by mitigating soil degradation. Below a threshold MAP value of 550 mm yr^{-1}, they found that plant biomass was no longer associated with higher soil moisture content, but was dependant on the chemical and physical properties of the soil. The results of this meta-analysis are also in agreement to an early review of 22 studies examining the SOC sequestration potential of degraded grasslands worldwide by Conant and Paustian (2002), who observed that mean annual precipitation was the main factor controlling soil carbon sequestration in degraded grasslands. Similar to the results obtained in this meta-analysis, Conant and Paustian (2002) found that grassland degradation led to a greater depletion of SOC stocks under dry climates (<333 mm). In contrast, they reported a greater SOC sequestration potential of up to 93% for grassland sites in wet climates (≤1800 mm yr^{-1}). Greater SOC stocks under wet climates can be attributable to the high productivity of grasslands in wet environments, which allocate a high proportion of C below ground (Guo and Gifford 2002; Jobbágy and Jackson, 2000). On the other hand, grassland soils in dry climates do not receive adequate C inputs owing to the low precipitation to replenish the SOC lost through grassland degradation.

Soil texture

Meta-analysis revealed that grassland degradation had a significant negative effect on coarser textured than clayey textured grassland soils (Figure 2.6). On average, grassland degradation resulted to a 12% decline in SOC stocks in loamy soils (20-32% clay), 10% in sandy soils (<20% clay) and there was a negligible effect (1%) in clayey soils (≥32% clay). A comparable response of loamy and sandy soils was observed, however, the difference was not significant. The correlation of SOC and soil texture corroborates most previous studies on grassland soils, which have indicated that grassland degradation depletes SOC more in coarse textured soils. For example, Potter et al. (2001) who examined the impact of grassland degradation on SOC stocks across two degraded grassland soils with strongly contrasting soil textures in the USA found that degradation significantly reduced SOC stocks by 41%, with 56.7 t ha^{-1} in coarse textured compared to 95.7 t ha^{-1} in fine-textured soils.

Not only can intensification of degradation lead to significant depletion of SOC stocks, but it can induce shifts in the distribution of soil texture. A recent study examining the effects of grassland degradation on soil quality in the Qinghai-Tibetan Plateau in China by Dong et al. (2012) found that grassland degradation led to a shift in soil texture from loamy soils (40% sand, 40% silt and 20% clay) towards sandy loamy soils (60% sand, 30% silt and 10% clay) along a degradation gradient from non-degraded to heavily degraded grasslands. One possible mechanism explaining such a shift in soil texture could be large amounts of the nutrient-rich surface material being removed under heavily degraded soils, thus exposing the subsoil. They postulated that this shift towards more sandy textured soils induced reductions in the C storage capacity of the soil.

The limited effect of grassland degradation on the depletion of SOC stocks in clayey soils can be explained by several reasons. Clay particles associate with organic compounds, thereby contribute to the formation of stable organo-mineral complexes (Six et al., 2002). These stable complexes are an important mechanism that leads to the stabilization of SOC through physical protection against decomposition (von Lutzow et al., 2006). Clay-sized particles also have greater reactive surface areas which provide greater capacity to chemically stabilise SOC and form building blocks for aggregates, thereby increasing physical protection of SOC by occlusion in aggregates, especially micro-aggregates (Feller and Beare, 1997; Six et al., 2000). Furthermore, soils with high clay content have been shown to have better water holding capacity and infiltration rates, which might stimulate biomass production and consequently increase C inputs into the soil (Burke et al., 1989; Schimel et al., 1994). The greater depletion of SOC stocks in coarse textured soils may be related to that the initial SOC levels are low in these soil conditions such that grassland degradation causes greater proportional losses of SOC.

Soil pH

Across all comparisons, changes in SOC stocks were greater in strongly acidic soils (that is soils with a pH<5). On average, grassland degradation reduced SOC stocks by 14% in acidic soils (Figure 2.7). Several studies have shown that soil pH influences soil carbon dynamics via decomposition of soil organic matter (Motavalli et al., 1995; Andersson and Ingvar Nilsson, 2001; Aciego Pietri and Brookes, 2008) as well through the hydrolysis and protonation processes. The protonation process regulates solubilization and complexation, which affect the stability of soil carbon via sorption and desorption of SOC on mineral surfaces (van Bergen et al., 1997). The level of SOC stocks at low pH is due to a number of factors, including microbial activity and associated rates of organic matter decomposition (Andersson and Ingvar Nilsson, 2001; Aciego Pietri and Brookes, 2008). Motavalli et al. (1995) suggested that under acidic soils the decomposition rates of freshly added organic material is reduced, which might explain the lower SOC stocks. According to Janssens et al. (2010) soil acidification is considered to be a stabilization mechanism because it reduces decomposition of plant litter and soil organic matter. Given that soil pH is crucial to enzyme functioning, soil acidification could have detrimental effect on microbial activity and thus decomposition of SOM (Janssens et al., 2010).

Another possible explanation could be that at low soil pH, base cations such as Ca, K and Mg are weakly bound to the soil (Berthrong et al., 2009). Jobbágy and Jackson, (2003) found that a greater decline in pH was associated with losses of exchangeable base cations, particularly Ca. Under heavily degraded grasslands in China, Wu and Tiessen (2002) found that grassland degradation significantly reduced the cation exchange capacity (CEC) by 18% and a decline in CEC could trigger irreversible SOC and nutrient losses (Jobbágy and Jackson, 2003). The loss of SOC lowers nutrient availability and CEC, this could then lower biomass production (lower vegetative growth), which, overtime may lower organic inputs thereby lowering SOC.

Interestingly, there was no significant effect of changes in SOC induced by grassland degradation stocks for soils with a pH ranging between 5 and 7. The limited impact of grassland degradation on SOC stocks in soils with a pH ranging between 5 and 7 is in agreement with Dong et al. (2012) who found that grassland degradation did not alter the

SOC stock of soils with pH varying from 6.5 to 7.0 in a Chinese grassland soil. However, Mchunu and Chaplot (2012) found that grassland degradation reduced SOC stocks by 63% in acidic soils (pH = 3.84) of a South African degraded grassland.

Study duration

The effect of grassland degradation on SOC stocks was found to be greater in the short term (<10 years) than in the long term (>10 years) studies. In the short term studies, grassland degradation induced a 12% reduction in SOC stocks compared to 5% in long-term studies (Figure 2.9). Such a difference in short and long term changes in SOC stocks could reflect a situation where the system equilibrium has not been attained in the short term while in the long term the equilibrium has been reached.

Grass type

The depletion of SOC stocks were found to be related to the different photosynthetic pathways of C_3 grasses (adapted to cool-season conditions) and C_4 grasses (adapted to warm-season conditions). Grassland degradation significantly reduced SOC by 14% in C_3 grasses and by 5% in C_4 grasses (Figure 2.10). This is consistent with a recent review by McSherry and Ritchie (2013) which found that grazing significantly reduced SOC by 18% in C_3 grasses.

The main factor responsible for the different response of C_3 and C_4 grasses to SOC losses is grazing intensity (McSherry and Ritchie, 2013). Grazing strongly affects SOC loss in grassland through defoliation (Bargett and Wardle, 2003). The influence of grazing on the C cycle has been shown to be greater where herbivory has induced changes in the functional composition of plant communities (Chapin et al., 1997). Grazing may modify the functional group composition by altering the relative abundance of C_3 and C_4 grass species. Grass species with these distinct photosynthetic pathways differ markedly with their functional attributes, including C, nutrient and water use characteristics. Consequently, the relative proportion of C_3 and C_4 grasses has the potential to influence the amount and dynamics of SOC stocks because of the variability in the quality and quantity of their C inputs, and SOC losses (Derner et al., 2006; De Deyn et al., 2008).

Grass species characteristics regulate SOC storage by controlling C assimilation, transfer and storage in belowground biomass and its release through soil respiration (De Deyn et al., 2008). The depletion of SOC in degraded grasslands is largely influenced by the difference between inputs via plant litter and losses through decomposition, and therefore is expected to differ significantly between grass species. Using isotope signature of ^{13}C, Frank et al. (1995) investigated the effect of changes in grass species from C_3 grasses to C_4 grasses in moderately and heavily degraded grassland. An increase in total SOC of approximately 20% was observed for C_4 grass species in moderately degraded grasslands, while C_4 grasses in heavily degraded grassland were associated with an increase of 24% in total SOC. They suggested that the dense shallow root system of C_4 grasses increases C inputs belowground, and thus maintains high SOC levels in the soil.

Derner et al. (2006) found that the above-ground biomass of C_4 grasses was 44% lower in degraded than in non-degraded grasslands, while above-ground biomass of C_3 grasses was 76% lower in degraded than in non-degraded grasslands. A higher aboveground biomass should therefore generate higher SOC inputs originating from the plant litter. Grass species that have a higher aboveground biomass have a greater tendency to sequester SOC in the soil, which may contribute to better maintenance of ecosystem services such as plant productivity, soil fertility and increased soil aggregation (Conant et al., 2001). The preservation of SOC stocks in grassland soil is crucial because increased SOC stocks will limit soil degradation and ensure long-term sustainability of grasslands (Conant et al., 2001).

In loamy soils of the Qinghai-Tibetan Plateau in China, Dong et al (2012) found that grassland degradation resulted in an increase in C_3 grasses (*L. Virgaurea*) and a decrease in C_4 grasses (*Kobresia capillifolia*). This change in grass species composition reduced C accumulation in the soil. C_3 and C_4 grasses have different C allocation strategies, therefore, an increase in C_3 grasses and a decrease in C_4 grasses will result to lower SOC stocks because C_4 grasses have a higher root-to-shoot ratio and greater transfer of photosynthate belowground (Frank et al., 1995; Reeder et al., 2004).

In grasslands, the intensity of degradation causes loss of grass species diversity which ultimately affects C storage in the soil, and the mechanism of this loss has been explained by Klumpp et al. (2009), who proposed that depending on the intensity of soil disturbance grasslands ecosystems tend towards two contrasting systems that differ in soil C storage. On the one hand, degraded grasslands are usually dominated by fast-growing plant species that produce high quality litter (low C/N ratio and lignin content), which is quickly decomposed by bacteria. As a result, C storage in these productive systems is relatively low (C-releasing ecosystem). On the other hand, less degraded grasslands are dominated by slow-growing plants species and fungi, thus exhibit larger C storage (C-storing ecosystem) and lower above-ground net primary productivity.

2.4 Conclusion

In this study of 630 soil profiles from 131 temperate, sub-tropical and semi-arid sites our main objective was to quantify the impact of grassland degradation on SOC stocks and to identify the main environmental factors of control, worldwide. Two main conclusions can be drawn. The first one is that the worldwide average grassland SOC stock depletion was 9% with values ranging between 13% for heavily degraded to 7% in lightly degraded soils. The second conclusion is that grassland degradation had a more pronounced impact on the reduction of SOC stocks under dry climates, on sandy acidic soils, compared to wet climates and clayey soils.

Given that 30% of grasslands worldwide have been affected by grassland degradation, the amount of SOC likely to be lost is estimated as 4.05 Gt C with a 95% confidence between 1.8 and 6.3 Gt C (i.e. from 1.2 to 4.2% of the whole grassland soil stock). This implies that a similar amount of atmospheric C could be potentially sequestrated in soils through grassland rehabilitation. These results on the impact of grassland degradation on SOC stocks have implications for grassland management and they are expected to inform the international community on the regions where particular attention should be expended for the development of adapted protection measures and efficient land rehabilitation strategies to mitigate grassland degradation. Global carbon models could also benefit from this newly acquired knowledge.

Table 2.1 Compilation of references included in the database for analysis of the factors controlling SOC stocks in grasslands.

Author (s)	Country	Sample size	MAP	MAT	Z	LONG	LAT	SOC_C			SOC_S			SOC_D			CLAY
		n	mm	mm	m.a.s.l	-----Degree-----		---------g C kg^{-1}---------			-------kg C m^{-2}-------			-------kg C m^{-3}--------			%
								Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	
Abril and Bucher (1999)	Argentina	2	550	22.7	217	-62.7	-23.3	33.8	22.5	45.0	6.8	5.5	8.2	34.2	27.5	41.0	17.6
Baisden and Amundson (2002)	USA	26	300	16.0	1591	-117.9	37.9	13.7	0.7	60.5	1.6	0.3	5.1	14.2	0.7	52.0	17.6
Bauer et al. (1987)	USA	2	538	3.4	670	-101.0	46.0	2.2	2.0	2.3	1.2	1.1	1.3	2.6	2.4	2.8	26.5
Bird et al. (2000)	Zimbabwe	41	630	17.7	1400	28.3	-20.2	12.9	3.3	45.4	1.1	0.3	2.0	16.4	4.9	40.0	20.0
Chuluun et al. (1999)	China	4	307	2.6	1165	112.8	44.3	61.1	7.8	82.6	3.9	0.5	5.3	65.4	8.3	88.3	10.3
Conant et al. (2003)	USA	8	1075	13.5	84	-77.8	37.6	6.5	4.8	8.8	4.4	3.2	6.0	8.8	6.4	11.9	10.3
Covaleda et al. (2011)	Mexico	1	844	16.8	1674	-100.8	19.6	48.6	48.6	48.6	2.4	2.4	2.4	24.3	24.3	24.3	34.8
Cui et al. (2005)	China	8	350	0.2	1255	116.7	43.5	12.4	7.6	16.7	3.9	3.5	4.6	14.3	8.8	19.4	21.0
Don et al. (2007)	Germany	10	600	8.0	267	10.4	51.0	11.3	0.9	44.2	1.6	0.5	2.9	15.7	1.2	57.4	26.8
Dong et al. (2012)	China	20	570	-0.6	4200	100.2	34.5	54.0	14.2	164.0	5.6	2.1	15.2	52.4	13.3	137.8	20.0
Frank et al. (1995)	USA	12	404	4.4	573	-99.2	46.8	22.0	11.4	36.1	4.7	3.3	6.1	29.7	15.4	48.7	10.0
Franzluebbers and Stuedemann (2009)	USA	12	1250	16.5	153	-82.6	33.4	13.2	4.3	24.1	3.3	1.8	4.9	18.2	6.1	32.8	10.0
Fynn et al. (2003)	South Africa	59	790	17.6	2280	29.4	-29.6	34.1	27.1	58.0	2.3	0.8	5.9	39.5	31.4	67.3	33.0
Ganjegunte et al. (2005)	USA	3	384	15.0	1930	-104.0	41.1	22.2	19.8	26.0	1.2	1.1	1.4	23.7	21.6	27.6	35.0
Garcia-Pausas et al. (2007)	Spain	26	1595	2.6	1461	-0.6	42.8	85.0	38.0	165.0	13.9	5.9	23.4	52.1	26.0	97.4	31.9
Gill (2007)	USA	20	932	1.3	1600	-110.5	39.3	33.4	16.4	53.3	5.4	3.3	10.5	35.7	21.7	69.8	24.7
Gill et al. (1999)	USA	5	321	8.2	628	-103.2	49.8	5.5	1.7	15.9	0.7	0.4	1.4	5.5	2.0	14.0	19.4
Hafner et al. (2012)	China	6	582	1.7	3440	99.8	35.5	28.0	10.1	62.1	3.1	2.1	4.2	27.2	11.3	52.2	25.0
Hassink (1997)	Netherlands	14	750	8.0	67	5.6	51.1	37.2	15.0	60.7	3.7	1.5	6.1	37.2	15.0	60.7	8.0
Hiltbrunner et al. (2012)	Switzerland	9	1250	6.0	1600	7.3	46.6	24.3	6.4	51.4	2.3	1.4	3.2	20.7	7.2	38.0	52.9
Ingram et al. (2008)	USA	9	425	15.0	1930	-103.1	41.2	14.5	7.7	26.1	1.6	1.1	2.2	17.8	11.1	27.6	10.0
Kaye et al (2002)	USA	10	382	10.1	1186	-103.1	40.7	11.9	7.9	20.6	3.2	2.1	5.5	16.0	10.6	27.7	28.7
Leifeld and Kögel-Knabner (2005)	Germany	2	833	7.5	462	11.3	48.5	31.7	24.6	38.8	8.5	8.1	8.9	149.8	29.7	270.0	18.0
Maia et al. (2009)	Brazil	63	1950	15.1	171	-49.7	-9.1	13.0	4.1	30.0	3.9	1.4	7.5	14.9	4.7	34.2	18.0
Manley et al. (1995)	USA	2	384	13.0	1930	-103.1	41.2	14.6	14.0	15.1	5.7	5.7	5.7	18.9	18.9	19.0	10.0
Manson et al. (2009)	South Africa	4	1380	10.0	1841	29.3	-29.0	94.8	77.0	114.0	6.8	3.2	9.4	57.7	47.0	65.0	61.3
Martinsen et al. (2011)	Norway	6	1000	-1.5	1211	7.9	60.8	247.8	209.0	293.0	0.6	0.5	0.8	12.3	10.2	15.0	3.0
Masiello et al. (2004)	USA	16	1000	12.0	580	-123.7	41.3	48.8	3.4	82.1	12.5	2.3	31.8	53.1	4.7	82.1	17.6
Mchunu and Chaplot (2012)	South Africa	2	684	13.0	1300	29.6	-26.7	8.2	4.1	12.3	0.2	0.1	0.3	10.3	5.5	15.0	16.6
Medina-Roldán et al. (2012)	England	2	1840	2.8	400	2.4	54.2	193.2	163.6	222.9	6.1	5.9	6.2	30.3	29.5	31.2	10.0
Mestdagh et al. (2006)	Belgium	6	780	9.8	13	3.7	50.9	29.9	13.8	54.2	5.2	2.7	8.5	28.6	17.5	46.8	12.3
Mills and Fey (2004)	South Africa	2	1050	15.1	1718	29.5	-28.3	34.5	22.0	47.0	4.1	2.7	5.4	40.5	27.0	54.0	19.0
Mills et al. (2005)	South Africa	2	1050	15.1	1718	29.5	-28.3	34.5	22.0	47.0	4.1	2.7	5.4	40.5	27.0	54.0	19.0
Muñoz García and Faz Cano (2012)	Bolivia	16	505	4.5	167	-67.5	-13.3	52.7	30.0	91.7	1.8	0.5	3.8	32.6	5.0	76.0	16.0
Naeth et al. (1991)	Canada	7	355	4.0	745	-112.0	51.0	40.1	31.3	49.2	5.1	4.0	6.3	51.3	40.0	63.0	15.8
Neff et al. (2005)	USA	3	207	11.7	1500	-109.9	38.3	2.3	1.1	3.6	0.3	0.2	0.5	3.2	1.5	5.0	4.8
Percival et al. (2000)	New Zealand	22	1191	12.5	1365	172.5	-42.4	41.5	17.0	83.0	7.9	3.2	15.8	39.4	16.2	78.9	24.4
Piñeiro et al. (2009)	Uraguay	6	1100	17.3	110	-56.9	-32.0	23.6	15.4	30.3	9.2	6.0	11.8	30.7	20.0	39.3	25.5
Potter et al. (2001)	USA	40	842	17.0	2438	-97.2	34.2	11.2	4.7	27.1	1.8	0.3	4.6	16.3	7.4	33.3	23.3
Preger et al. (2010)	South Africa	9	641	15.5	1456	27.2	-26.4	14.5	6.0	29.7	1.6	0.9	2.2	18.3	9.0	41.6	19.3
Raiesi and Asadi (2006)	Iran	3	860	6.7	2500	51.0	31.8	21.1	19.4	23.6	7.0	6.4	8.0	23.4	21.3	26.7	50.0
Reeder and Schuman (2002)	USA	7	343	15.0	1930	-104.0	41.1	12.7	9.6	17.3	5.0	3.7	6.7	16.5	12.4	22.5	10.0
Schimel et al. (1985)	USA	8	310	8.5	1238	-103.2	40.8	5.9	0.8	20.4	1.2	0.5	2.1	8.0	1.2	26.4	23.2
Schipper et al. (2007)	New Zealand	31	1266	12.6	10	175.3	-36.2	121.2	62.6	204.2	23.0	11.9	38.8	115.1	59.5	194.0	24.2
Shi et al. (2012)	China	6	353	-0.4	1038	106.4	40.5	36.1	12.6	68.4	3.6	1.4	7.2	34.2	16.3	57.5	25.0
Sinoga et al. (2011)	Spain	10	598	16.1	902	-2.2	37.5	13.3	3.4	32.4	1.6	0.5	3.5	16.1	3.8	35.0	19.7
Smoliak et al. (1972)	Canada	4	550	1.3	926	-109.5	49.1	12.0	11.0	13.8	1.5	1.4	1.8	15.2	14.0	17.7	15.8

Steffens et al. (2008)	China	4	343	0.7	1270	116.7	43.6	24.1	17.0	31.0	1.0	0.9	1.2	25.9	21.5	28.8	15.0
Teague et al. (2011)	USA	6	820	18.1	315	-32.7	98.1	41.1	24.5	56.2	8.4	6.0	10.8	39.8	26.0	51.4	30.0
von Lützow et al. (2002)	Germany	3	803	7.4	462	11.3	48.5	31.7	28.0	38.0	9.4	8.3	11.3	47.2	41.7	56.6	14.0
Wiesmeier et al. (2012)	China	4	350	0.7	1260	116.7	43.6	17.0	13.7	21.3	1.9	1.7	2.0	18.8	17.5	20.0	20.4
Wood and Blackburn (1984)	USA	10	624	17.0	316	-98.6	34.0	33.1	23.0	45.0	1.5	1.2	1.8	51.1	41.4	61.2	30.0
Wu and Tiessen (2002)	China	3	416	-0.3	2940	102.8	37.2	67.7	37.0	85.0	7.4	4.1	9.5	49.3	27.0	63.2	27.3
Yong-Zhong et al. (2005)	China	3	366	6.5	360	120.7	43.0	2.4	2.1	2.8	0.5	0.5	0.6	3.3	2.9	3.7	2.5
Zimmermann et al. (2007)	Switzerland	2	1337	4.5	1656	8.5	46.2	35.4	27.3	43.4	10.5	8.1	12.8	52.3	40.4	64.2	26.0

Table 2.2 Statistical summary of the site environmental characteristics: mean annual precipitation (MAP); mean annual temperature (MAT); latitude (Lat); longitude (Long); altitude above sea level (Z); soil bulk density (*ρ_b*) and clay content (CLAY) from the global data set.

	MAP	**MAT**	**Z**	**LONG**	**LAT**	**CLAY**	**ρ_b**
	mm	^{o}C	m.a.s.l	----degree----		%	g cm^{-3}
Minimum	207	-1.6	10	-124	-42	2.0	0.14
Maximum	2000	22.7	4200	175	98	70.4	1.90
Mean	898	11.4	1248	-11	15	21.9	1.16
Median	790	14.3	1365	-1	34	18.9	1.16
Variance	248290	39	984789	7783	1160	127	0
Standard deviation	498.3	6.3	992.4	88.2	34.1	11.3	0.3
Skewness	1	-0.7	1	1	0	1.2	-0.52
Quartile1	550	6.0	171	-97	-20	15.0	1.00
Quartile3	1149	17.0	1841	29	41	30.0	1.34
Kurtosis	0	-1.0	1	-1	-1	2.3	1.58
CV	56	55	80	-805	234	51	23
SE	19.9	0.3	39.6	3.5	1.4	0.4	0.0

Table 2.3 Statistical summary of soil organic carbon content (SOC_C); soil organic carbon stocks (SOC_S) and soil organic carbon density: SOC_D from the global data set.

	SOC_C	**SOC_S**	**SOC_D**
	g C kg^{-1}	kg C m^{-2}	kg C m^{-3}
Minimum	0.2	0.1	0.7
Maximum	293.0	38.8	194.0
Mean	34.9	5.0	32.2
Median	22.9	3.1	25.2
Variance	1602	35	807
Standard deviation	40.0	5.9	28.4.
Skewness	2.9	2.9	2.5
Quartile1	11.0	1.6	12.8
Quartile3	42.6	5.5	41.7
Kurtosis	10.8	10.3	9.1
CV	115	120	88
SE	1.6	0.2	1.1

Table 2.4 Correlation matrix of soil organic carbon content (SOC_C); soil organic carbon stocks (SOC_S); soil organic carbon density (SOC_D) and selected environmental factors: altitude (Z); mean annual precipitation (MAP); mean annual temperature (MAT); longitude (LONG); latitude (LAT), clay content (CLAY) and soil bulk density (ρ_b).

	Z	**MAP**	**MAT**	**LONG**	**LAT**	**CLAY**	**ρ_b**
SOC_C	0.10*	0.16*	-0.34*	0.47*	-0.17*	-0.16*	-0.73*
SOC_S	-0.30*	0.27*	0.16*	-0.05	0.08*	0.11*	0.06
SOC_D	-0.10*	0.20*	0.01	0.39*	-0.23*	0.11	-0.23*

*Significant correlation at $P < 0.05$

Table 2.5 SOC and environmental characteristics of A horizons for both degraded and non-degraded grasslands. SOC_{D-ND} is the SOC density for non-degraded grassland A horizons; SOC_{D-D} is the SOC density for degraded grassland A horizons.

Reference	Country	Location	Sample size	MAP	MAT	Z	Clay	SOC_{D-ND}	SOC_{D-D}
			n	mm	°C	m	%	-------kg C m^{-3}-------	
Abril and Bucher (1999)	Argentina	Salta	12	550	22.7	217	17.6	41.0	27.5
Bauer et al. (1987)	USA	North Dakota	6	538	3.4	670	26.5	2.8	2.4
Chuluun et al. (1999)	China	Mongolia	14	307	2.6	1165	10.3	88.3	8.3
Cui et al. (2005)	China	Inner Mongolia	29	350	0.2	1255	21.0	17.6	11.0
Dong et al. (2012)	China	Qinghai-Tibetan	21	570	-0.6	4200	20.0	137.8	13.3
Frank et al. (1995)	USA	Mandan, N.D	2	404	4.4	573	10.0	48.7	15.4
Franzluebbers and Stuedemann (2009)	USA	Georgia	10	1250	16.5	153	10.0	32.0	6.1
Ganjegunte et al. (2005)	USA	Cheyenne	28	384	15.0	1930	35.0	21.6	21.8
Gill (2007)	USA	Utah	11	932	1.3	1600	24.5	50.3	21.7
Hafner et al. (2012)	China	Qinghai-Tibetan	25	582	1.7	3440	25.0	41.0	11.3
Hiltbrunner et al. (2012)	Switzerland	Fribourg	16	1250	6.0	1600	52.9	32.0	7.2
Ingram et al. (2008)	USA	Cheyenne	18	425	15.0	1930	10.0	21.6	11.1
Manley et al. (1995)	USA	Cheyenne	3	384	13.0	1930	10.0	18.9	19.0
Martinsen et al. (2011)	Norway	Burskerud County	7	1000	-1.5	1211	3.0	13.8	10.2
Mchunu and Chaplot (2012)	South Africa	Bergville	15	684	13.0	1300	16.6	15.0	5.5
Medina-Roldán et al. (2012)	England	Yorkshire Dales	20	1840	2.8	400	10.0	29.5	31.2
Naeth et al. (1991)	Canada	Alberta	13	355	4.0	745	15.8	55.0	40.0
Neff et al. (2005)	USA	Utah	19	207	11.7	1500	4.8	5.0	1.5
Piñeiro et al. (2009)	Uraguay	Rio de la Plata	23	1100	17.3	110	25.5	36.3	23.0
Potter et al. (2001)	USA	Oklahoma	26	842	17.0	2438	23.3	33.3	7.9
Raiesi and Asadi (2006)	Iran	Shahrekord	27	860	6.7	2500	50.0	26.7	22.2
Reeder and Schuman (2002)	USA	Cheyenne	5	343	15.0	1930	10.0	19.4	12.5
Smoliak et al. (1972)	Canada	Alberta	1	550	1.3	926	15.8	14.0	14.4
Steffens et al. (2008)	China	Xilinhot	8	343	0.7	1270	15.0	28.8	21.5
Teague et al. (2011)	Texas	USA	22	820	18.1	315	30.0	50.6	26.0
Wiesmeier et al. (2012)	China	Inner Mongolia	9	350	0.7	1260	20.4	20.0	18.1
Wood and Blackburn (1984)	USA	Texas	4	624	17.0	316	30.0	55.9	46.8
Wu and Tiessen (2002)	China	Tianzhu	17	416	-0.3	2940	27.3	57.8	27.0
Yong-Zhong et al. (2005)	China	Naiman County	24	366	6.5	360	2.5	3.7	2.9

Table 2.6 List of the categorical variables describing the environmental conditions.

Category	Class	Definition
Degradation intensity	Heavy	
	Moderate	
	Light	
Soil texture	Low	<20% clay
	Medium	20-32% clay
	High	>32% clay
Soil pH	Strong acidic	,<5
	Weak acidic to weak alkaline	5-7
	Strong alkaline	>7
Mean annual precipitation (MAP)	Low	<600 mm
	Medium	600-1000 mm
	High	>1000 mm
Grass type	C_3	
	C_4	
	Mixed C_3-C_4	
Duration of study	Short-term	<10 years
	Long-term	>10 years

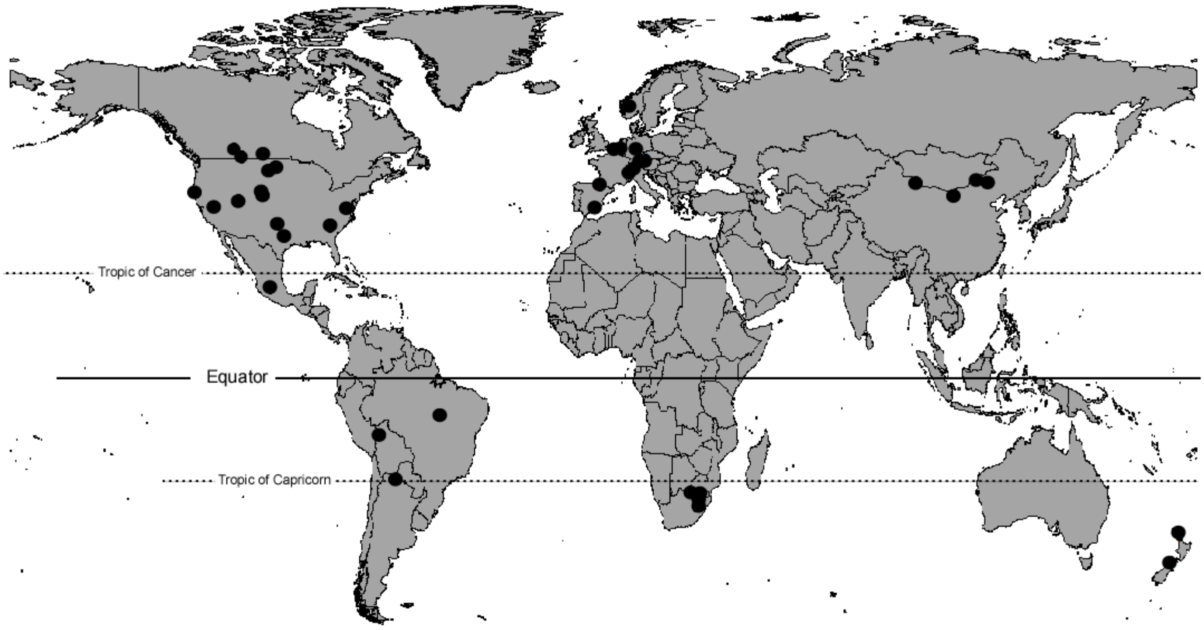


Figure 2.1 Location of the study sites included in the literature review.

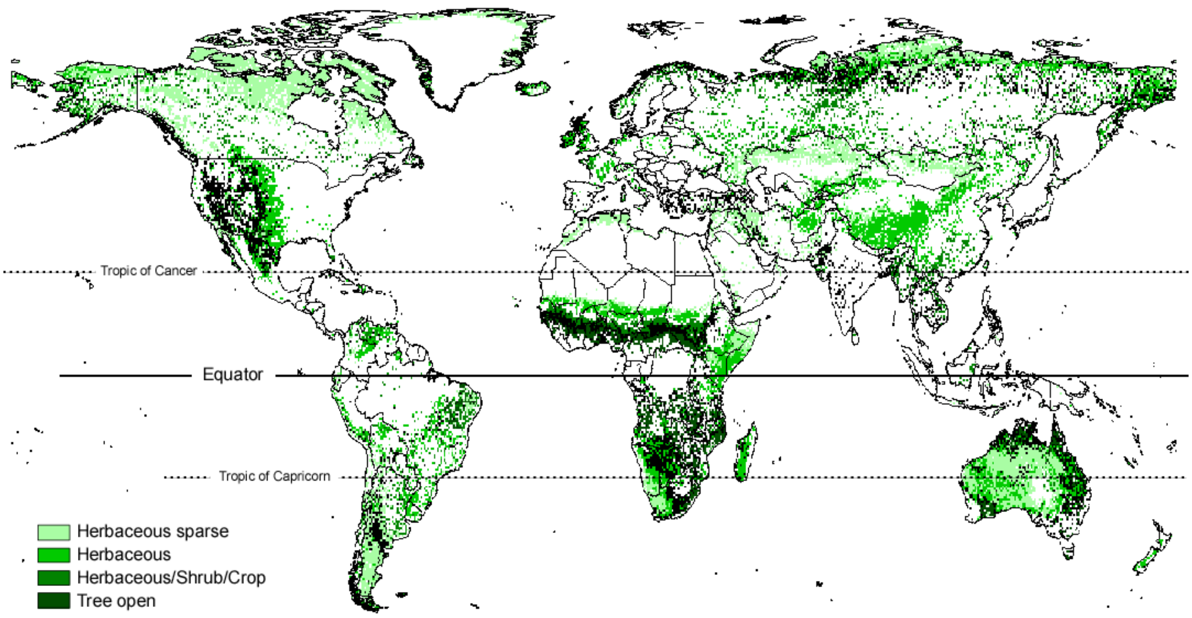


Figure 2.2 The global extent of grasslands. The global land cover database of the International Geosphere Biosphere Program at 1 km × 1 km spatial resolution was used for the classification of grassland areas globally (IGBP, 1998).

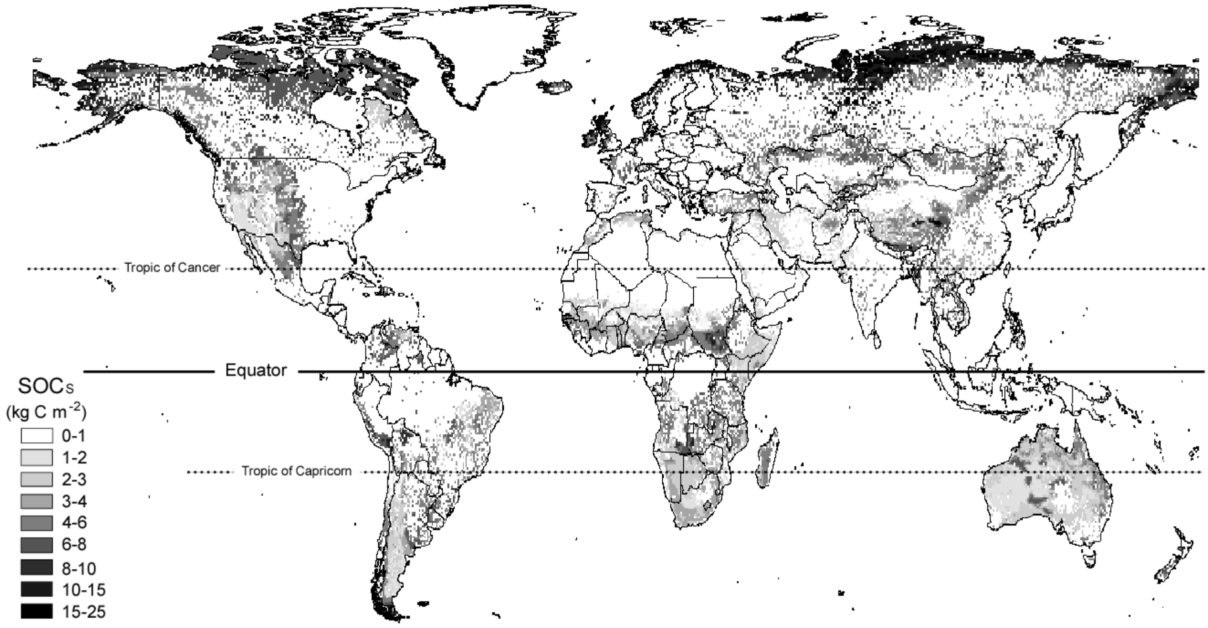


Figure 2.3 Map showing the spatial variation of SOC stocks throughout grassland soils worldwide.

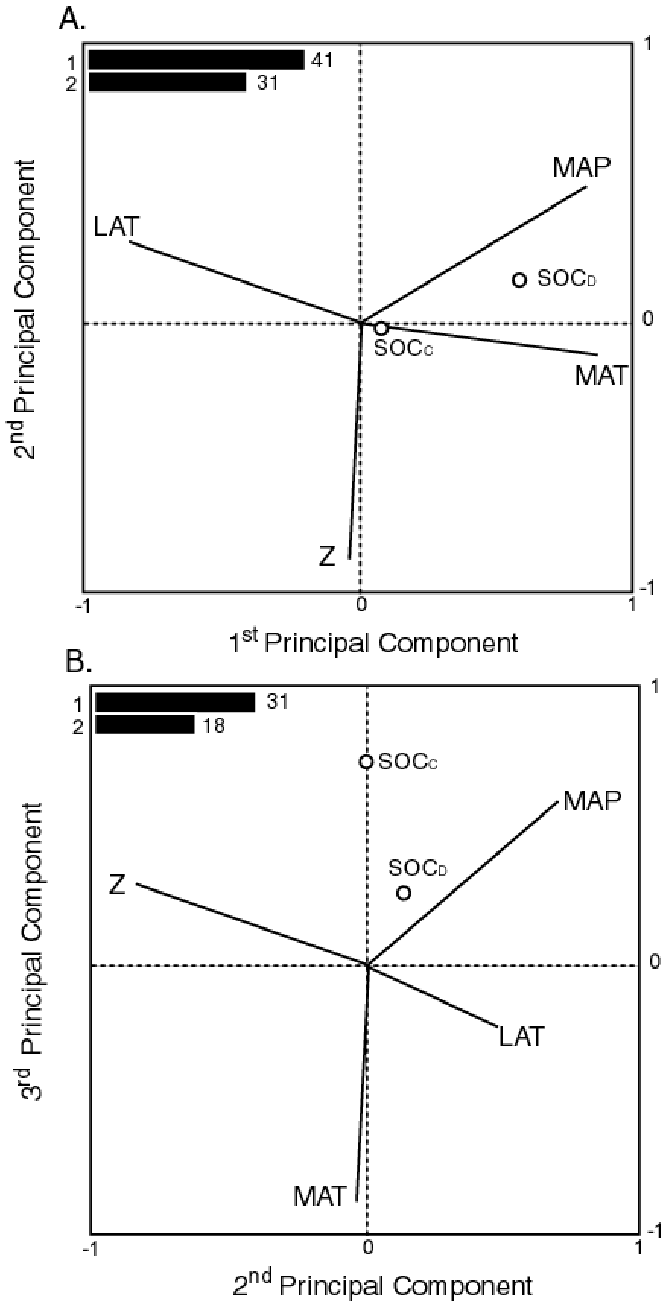


Figure 2.4 Principal component analysis (PCA) scatter diagrams for soil organic carbon content (SOC_C) and density (SOC_D) on the one hand and selected environmental factors on the other hand. A: scatter diagram with the two first PCA axes; B: scatter diagram with Axes 2 and 3.

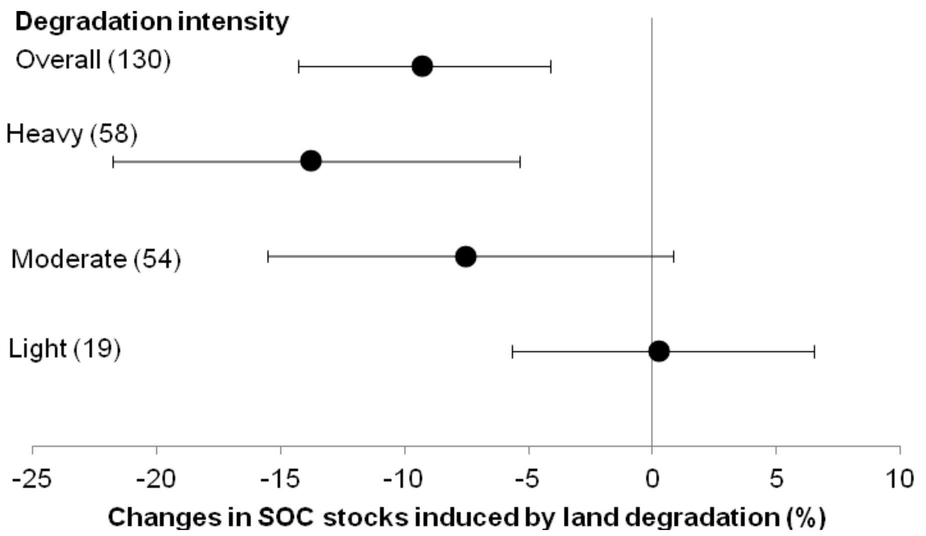


Figure 2.5 Influence of degradation intensity on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

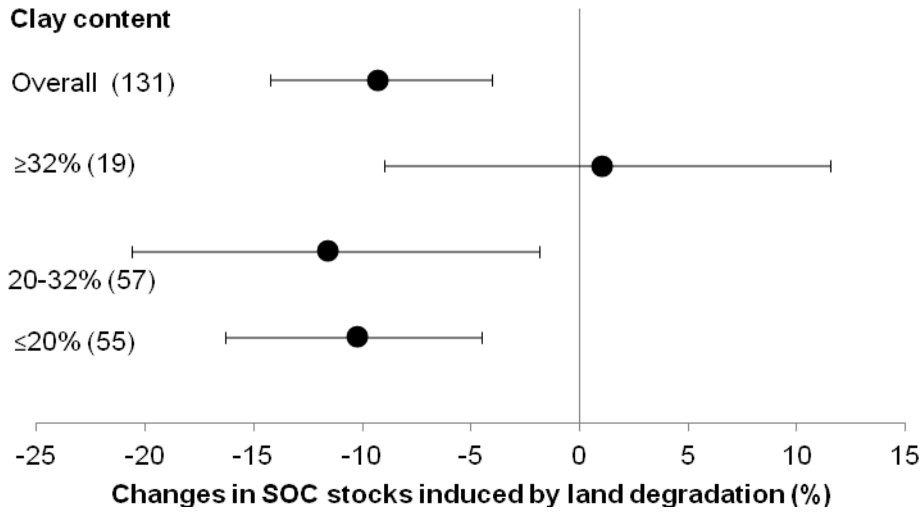


Figure 2.6 Influence of soil texture on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

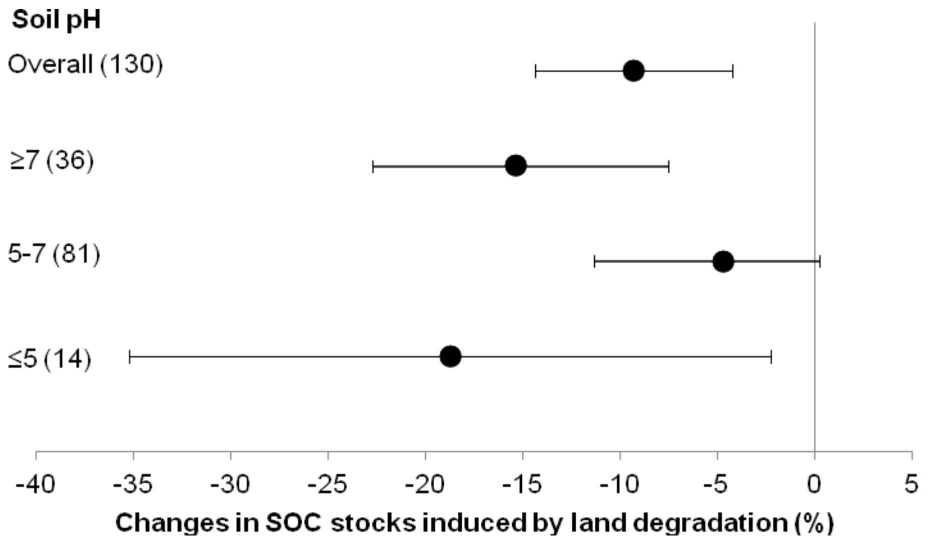


Figure 2.7 Influence of soil pH on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

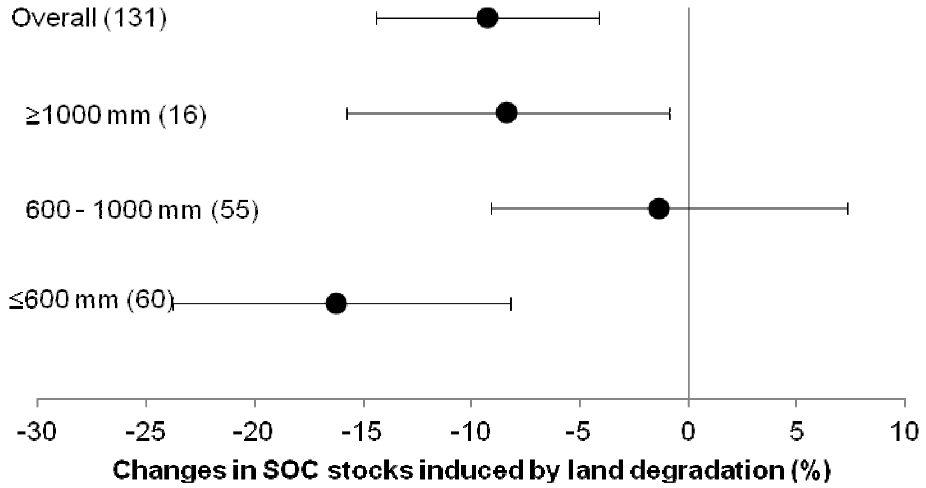


Figure 2.8 Influence of climatic factors: mean annual precipitation (MAP) on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

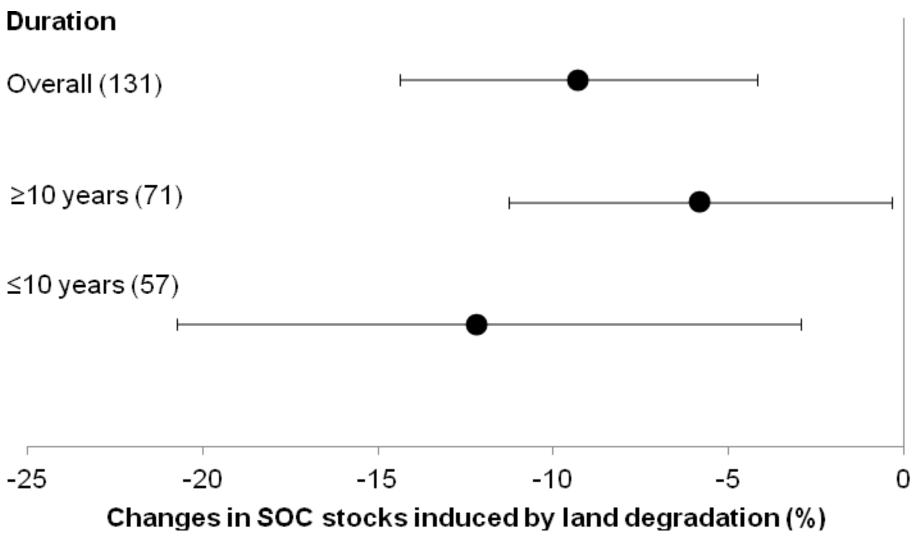


Figure 2.9 Influence of duration on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

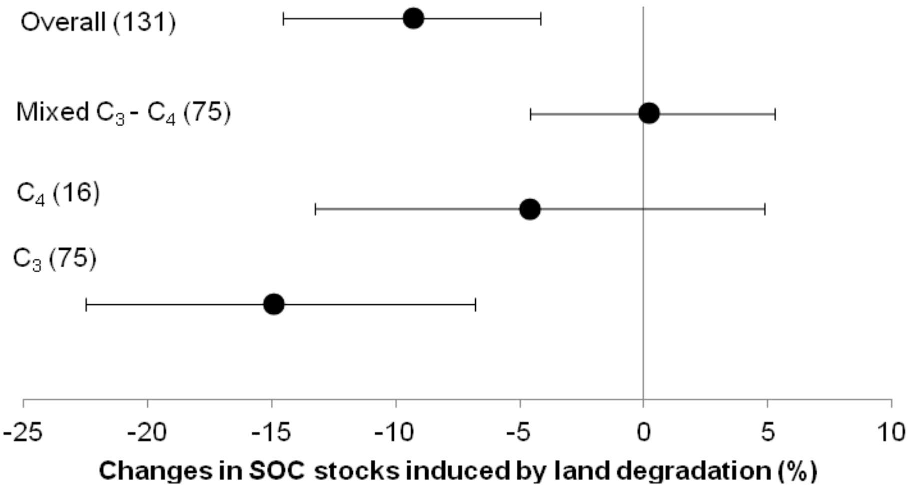


Figure 2.10 Influence of grass type on changes in SOC stocks. Values are mean effect sizes with 95% confidence intervals (CI). A significant response is when the CI does not overlap 1. The number of observations in each class is shown in parenthesis.

CHAPTER 3

3. GRASSLAND DEGRADATION IMPACTS ON SOIL ORGANIC CARBON AND NITROGEN STOCKS IN THE POTSHINI CATCHMENT, SOUTH AFRICA

Abstract

Land degradation is recognized as a main environmental problem that adversely depletes soil organic carbon (SOC) and nitrogen (SON) stocks, which in turn directly affects soils, their fertility, productivity and overall quality. While it is expanding worldwide at rapid pace, quantitative information on the impact of land degradation on the depletion of SOC and SON stocks remains largely unavailable, limiting the ability to predict the impacts of land management on the C losses to the atmosphere and the associated global warming. The main objective of this study was to evaluate the consequences of a decrease in grass aerial cover on SOC and SON stocks. A degraded grassland showing an aerial cover gradient from 100% (Cov100, corresponding to a non-degraded grassland) to 50-75% (Cov75), 25-50% (Cov50) and 0-5% (Cov5, corresponding to a heavily degraded grassland), was selected in South Africa. Soil samples were collected in the 0.05 m soil layer at 48 locations along the aerial cover gradient and were subsequently separated into the clay + silt (2-20 µm) and sand (20-2000 µm) fractions, prior to total C and N analysis (n=288). The decline in grass aerial cover from 100% to 0-5% had a significant ($P<0.05$) impact on SOC and SON stocks, with losses by as much as 1.25 kg m^{-2} for SOC and 0.074 kg m^{-2} for SON, which corresponded to depletion rates of 89 and 76%, respectively. Furthermore, both the C:N ratio and the proportion of SOC and SON in the silt + clay fraction declined with grass aerial cover, which was indicative of a preferential loss of not easily decomposable organic matter. The staggering decline in SOC and SON stocks raises concerns about the ability of these acidic sandy loam soils to sustain their main ecosystem functions. The associated decrease in chemical elements (e.g., Ca by a maximum of 67%; Mn, 77%; Cu, 66%; and Zn, 82%) was finally used to discuss the mechanisms at stake in grassland degradation and the associated stock depletion of SOC and SON stocks, a prerequisite to grassland rehabilitation and stock replenishment.

Keywords*: carbon cycle, pasture; rangeland; climate change; ecosystems*

3.1 Introduction

Grasslands occupy about 40% of world’s land surface and store approximately 10% of the global soil carbon (C) stock of 1500 Gt (Suttie et al., 2005). Consequently, grasslands are considered to have greater potential to sequester SOC, depending on management strategies (Franzluebbers and Doraiswamy, 2007), making them an important component of the global C cycle. Additionally, grasslands provide key ecosystem goods and services by supporting biodiversity, and serving as rangelands for the production of forage to sustain the world’s livestock (Suttie et al., 2005; FAO, 2010; Asner et al., 2004; Bradford and Thurow, 2006). However, land degradation severely impacts on the productivity of grasslands (UNEP, 2007).

Grassland degradation, defined here as the reduction in the capacity of grasslands to carry out their key ecosystem functions, is commonly attributed to disturbances including overgrazing livestock trampling, and soil erosion (Daily, 1995; UNEP, 2007). For instance a recent study by Kotzé et al. (2013) investigated the impacts of rangeland management on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa. They found that continuously grazed communal farms were generally depleted of nutrient stocks, and nutrient depletion generally increased with increasing grazing intensity. Grassland management practices substantially influence the amount, distribution and turnover rate of soil organic matter and nutrients in soils (Blair et al., 1995). Moreover, because the larger proportion (ca 60-70%) of SOC and nutrient stocks in grassland soils is concentrated in the top 0.3 m (Gill et al., 1999), any external disturbance is likely to cause dramatic soil fertility and SOC depletion, which in turn will constrain grassland productivity, including biodiversity loss and forage production (Ruiz-Sinoga and Romero Diaz, 2010; Dong et al., 2012).

Yet, contradictory results have been reported on the impact of grassland degradation on SOC stocks with some studies showing a decrease in SOC with overgrazing (Martinsen et al., 2011; Steffens et al., 2008), some no change (Johnston et al., 1971; Domaar et al., 1977) whereas some show an increase (Smoliak et al. 1972; Derner et al., 1997).

For instance, SOC stocks declined by 15% after seven years of grazing in Norway, with 0.76 kg C m^{-2} in ungrazed compared to 0.64 kg C m^{-2} in heavily grazed grasslands (Martinsen et al., 2011). Steffens et al. (2008) found that 30 years of overgrazing in a semi-arid Chinese grassland resulted in 50% decrease in SOC stocks, with 0.64 kg C m^{-2} in grazed compared to 1.17 kg C m^{-2} in ungrazed grasslands. A similar depletion rate was found in the USA, where Franzluebbers and Stuedemann (2009) observed that heavy grazing reduced SOC stocks to 0.051 kg C m^{-2} after 12 years of grazing, compared with 0.117 kg C m^{-2} on ungrazed grasslands. Wu and Tiessen (2002) reported that land degradation reduced SOC and N by 33% and 28%, respectively in a degraded Chinese alpine grassland. Finally, Dong et al. (2012) found an extreme SOC depletion rate of 90% in a degraded Chinese grassland.

In contrast, grazing increased SOC stocks under several environments (Smoliak et al. 1972; Bauer et al., 1987; Frank et al., 1995 and Derner et al., 1997), by rates ranging from 14% to 91%. However, in the latter, moderate grazing is reported to be beneficial to grassland soils rather than contributing to their degradation.

While the studies focusing on grassland degradation have reported associated losses in SOC, little is known on the impact of different degradation intensities on SOC stocks, with the underlying research question being at what threshold of grassland degradation do SOC stocks dramatically decrease?

To further improve the understanding of grassland degradation impact on SOC losses from soils, more work needs to be done on the mechanisms controlling organic matter destabilization. As such, the changes in organic matter quality as a consequence of grassland degradation could be early indicators of SOC stock depletion in both natural and agricultural ecosystems, as suggested by Christensen et al. (2001). Furthermore, a better understanding of the rates of SOC and N depletion and the associated destabilization mechanisms are expected to enhance efforts to circumvent land degradation and accelerate the recovery of degraded soils (Schmidt et al., 2011), whilst maintaining a viable forage production for livestock and supporting biodiversity (Lal, 2004).

For many smallholder farmers in Africa, grasslands make a significant contribution to food security by providing part of the feed requirements of livestock used for meat and milk production (O' Mara, 2012). However, many of the grasslands are in poor condition and showing signs of degradation due an increase in anthropogenic pressures on marginal lands, overgrazing and the associated problems of soil erosion (Suttie et al., 2005). As a consequence, this is jeopardizing both the environment and the economical development of rural livelihoods.

In this study of a communal rangeland in the uplands of the Drakensburg region, KwaZulu-Natal Province, South Africa managed by smallholder farmers, our main objective was to evaluate the consequences of a decrease in grass aerial cover on SOC and N depletion rate and the associated organic matter quality. Grass aerial cover was used as an indicator of grassland degradation.

3.2 Materials and methods

3.2.1 Site description

The study area is located in the Potshini catchment, 10 km north of the Bergville district in the KwaZulu-Natal Province of South Africa (Long: 29^{o} 21'; Lat: -28^{o} 48'). This area has a sub-tropical climate, characterized by cold dry winters and warm rainy summers (October to March), with a mean annual precipitation of 684 mm, a mean annual potential evaporation of 1600 mm and a mean annual temperature of 13^{o}C (Schulze, 1997). The altitude ranges from 1080 to 1455 m.a.s.l and the average slope gradient is 8%. The underlying geology is sandstone and mudstone, and the soils are classified as Acrisols (WRB, 2006). The vegetation in this area is dominated by Moist Highveld Sourveld (Camp and Hardy, 1999). The dominant vegetation species of Moist Highveld Sourveld include *Hyparrhenia hirta* and *Sporobolus africanus*.

3.2.2 Experimental design and sampling strategy

A degraded grassland site with a surface area of 1500 m^2 (30m × 50 m) and homogeneous soils was selected in the uplands of the Drakensburg region of South Africa (Figure 3.1). This site was selected because it exhibited a grassland degradation gradient varying from highly degraded areas with bare soils in the north to areas fully covered by grass in the south. Such areas are a common feature of many communal rangelands in this part of South Africa. For the purposes of this study, only grass aerial cover was used as an indicator of grassland degradation. Grass aerial cover is defined here as the area of the ground covered by the vertical projection of the aerial portion of the plants (USDA, 1996). Aerial cover was measured by placing a 1 × 1 m plot frame at fixed intervals along each corresponding aerial cover category, while aerial cover of the plants in the plot was recorded as an estimate of the % total area (Daubenmire, 1959). For soil sampling, four categories of grass aerial cover were identified and evaluated in the site, i.e. 75-100% (Cov100), 50-75% (Cov75), 25-50% (Cov50), and 0-5% (Cov5). At each cover category, three sampling points were randomly selected. For each selected sampling position, four replicate soil samples were collected in the 0-0.05m soil layer 1-m apart in a radial basis sampling strategy to yield twelve samples per category. The sampling resulted in a total of

48 soil samples. Furthermore, for each category, additional soil samples for bulk density were sampled using a 0.075 m diameter metallic cylindrical core (height, 0.05 m) following similar sampling strategy. The surface layer was intensively sampled because the effects of land degradation on SOC and nutrient stocks have been shown to be more pronounced in this soil layer (Snyman and du Preez, 2005; Dong et al., 2012). For the analysis of SOC and N stocks, with depth, additional soil samples were collected at depth increments of 0-0.05 m, 0.05-0.15 m, and then every 0.15 m down to 1.2 m using a hand shovel from the face of a 1 m × 1 m × 1.2 m soil pit. Triplicate soil bulk density samples were also collected in the different depth increments of soil profiles using 220.89 cm^{-3} metal cylindrical cores (height 0.05 m, diameter 0.75 m). Soil samples for bulk density were taken to the laboratory, immediately oven-dried at 105°C to determine the oven dry weight using the gravimetric method (Blake and Hartge, 1986). Once in the laboratory, the field moist samples were passed through an 8-mm sieve by gently breaking apart the soil along lines weakness for soil aggregate stability tests. The remaining soil samples were air-dried and ground to pass through a 2 mm sieve for further soil analysis. The penetration resistance (PR) of the soil, a proxy for soil compaction was measured in the field using a cone penetrometer (Herrick and Jones, 2002). The PR was evaluated by randomly selecting fifteen positions in each grass aerial cover category for penetration readings of the soil surface. The PR measurements were taken before the soil surface was disturbed for soil sample collection from the 0.05 m soil layer.

3.2.3 Soil physical and chemical analysis

The particle size distribution was determined by the sieve and pipette method (Gee and Bauder, 1986). The soil pH was measured in a 1:2.5 (10 g) to 1 M KCl (25 mL) suspension using a Calimatic pHM766 pH meter. Exchangeable Ca, Mg and exchangeable acidity were determined by extraction in 1M KCL while P, K, Zn, Mn and Cu were determined by extraction in Ambic 2 - extract containing 0.25M NH_4HCO_3, with detection by atomic absorption spectrometry (Manson and Roberts, 2000). The concentration of P and K were determined by inductively-coupled plasma optical emission spectrometry (ICP-OES). Effective cation exchange capacity (ECEC) was calculated as the sum of extractable cations and some rapid measure of exchangeable acidity, while the percentage acid saturation was calculated as the exchangeable acidity × 100/ (Ca + Mg + K + extractable acidity). Total C and N were measured in the bulk soil using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI).

3.2.4 Particle size fractionation

Physical fractionation was applied to the soil samples as a proxy of soil organic matter mineralization potential (Feller and Beare, 1997). Soil samples were separated into two size fractions; sand (20-2000 µm) and clay+silt fraction (2-20 µm) by combining wet sieving and sedimentation, following Schmidt et al. (1999). Twenty grams of soil were dispersed in 25 ml of distilled water using ultrasound (LABSONIC B1510), with a power output of 600 W and an output-energy of 22 J ml^{-1}, which was below the energy threshold that may disrupt coarse sand sized soil organic matter (Amelung and Zech, 1999). The dispersed suspension was then wet sieved to obtain the sand fraction (> 53-2000 µm), while the remaining material containing the silt and clay fractions (<53 µm) was separated by wet sieving following the standard pipette method (Gee and Bauder, 1986). Thereafter, the fractions were oven-dried at 40^{o}C, ground to <0.5 mm and analyzed for total C and N in triplicate using a LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI).

Particular attention was undertaken to ensure the potential losses during the particle-size fractionation procedure did not induce bias. To determine the recovery efficiency, the mass and organic C content of whole soil was compared with the sums of mass and C content in the two fractions to (%). The recovery efficiency obtained was on average 105%.

3.2.5 Aggregate stability

Aggregates 3- 8 mm in size were obtained by breaking soil aggregates by hand along lines of weakness after air-drying them in the laboratory at room temperature. The stability of soil aggregates was measured on the 3-8 mm aggregates following the ISO standard method (ISO/DIS 10930:2012) described by Le Bissonnais (1996). The fast wetting, slow wetting and stirring of pre-wetted aggregate tests were applied for each soil sample analysed. They correspond to a specific disaggregation mechanism, *viz*, slaking, differential clay swelling and mechanical breakdown, respectively. For the fast wetting test, aggregates (10 g) were immersed in 50 ml distillated water for 10 minutes. For the slow wetting test, aggregates were placed on top of foam humidified with water for 1 hour. For the stirring test, aggregates were first immersed in ethanol, then in water and gently shaken up and down 10 times. The weight of aggregates collected on each sieve (sizes: 2 mm, 1 mm, 0.5 mm, 0.2 mm, 0.1 mm and 0.05 mm) was subsequently measured and expressed as the percentage of the initial sample dry mass to compute the mean weight diameter (MWD), which was calculated as follows:

$$\text{MWD} = \frac{\sum (x_i \times w_i)}{100} \tag{1}$$

where x, is the mean inter-sieve size and w_i the percentage of fragments retained by the sieve *i*. The greater the MDW the more resistant to disaggregation the aggregates are.

3.2.6 Calculation of SOC and N stocks

The SOC and N stocks were calculated using the following equation by (Batjes, 1996):

$$C_s = x_1 x_2 x_3 (1 - \frac{x_4}{100}) \times b \tag{2}$$

where C_s is the C stock (kg C m^{-2}); x_1 is the C concentration in the <2 mm soil material (g C kg^{-1} soil); x_2 is the soil bulk density (kg m^{-3}); x_3 is the thickness of the soil layer (m); x_4 is the proportion (%) of fragments of > 2mm.

3.2.7 Calculation of changes in SOC and N stocks induced by land degradation

The effect of land degradation on SOC stocks was determined by using the change in the SOC stock along the degradation gradient relative to initial value of the SOC stock under Cov100. We assumed that 100% aerial cover represented a non- degraded grassland. The change in SOC stock was calculated as follows:

$$\boldsymbol{SOC_{SC}} = \frac{\boldsymbol{SOC_{S-ND}} - \boldsymbol{SOC_{S-D}}}{\boldsymbol{SOC_{S-ND}}} \times 100 \tag{3}$$

where SOC_{SC} is the change in SOC stocks (SOC_S), $SOC_{S\text{-}ND}$ is SOC_S in the non-degraded soils and $SOC_{S\text{-}D}$ is SOC_S in degraded soils. The same equation was used for N stocks. The change in SOC stocks induced by degradation is based on the premise that the loss of SOC stocks from non-degraded soil is of a less consequence than the loss of the same amount of stocks from a soil already depleted of SOC. Thus, the more a soil is depleted of SOC stocks the more difficult it is to rehabilitate (Blair et al., 1995).

3.2.8 Statistical analysis

A correlation matrix was generated to determine the univariate relationship between changes in SOC and N stocks and selected soil properties. The data was analyzed using the software packages Sigma Plot 8.0 (Systat Software Inc, Richmond, California, USA) and STATISTICA 7.0. (StatSoft, Inc, Tulsa, OK). Mean separations were done using the SAS statistical package (SAS Institute, 2003) and were considered to be significant at $P < 0.05$.

3.3 Results

3.3.1 General soil characteristics of the degraded grassland site

The soils (Acrisols) are characterized by a dark brown (7.5YR 4/4) 0-0.3 m thick A horizon, with a weak sub-angular blocky structure. This horizon is underlain by a reddish (5YR 4/6) B-horizon 0.3-0.7 m. Underlying this horizon is the C horizon 0.7-1.2 m characterized by sandy saprolite showing signs of wetness. Sand content ranged between 49% in Cov5 and 73% in Cov75, silt content ranged between 13% in Cov75 and 17% in Cov5 and clay content ranged between 14% in Cov75 and 34% in Cov5 in the 0.05 m soil layer (Table 3.1). The soil pH is acidic with values in the 0-0.05 m layer as low as 3.74 in Cov5 to 3.94 in Cov75. Effective cation exchange capacity (ECEC) ranged between 2.36 $cmol_c$ kg^{-1} in Cov50 and 5.35 $cmol_c$ kg^{-1} in Cov5, while acid saturation ranged between 26% in Cov100 and 77% in Cov5 (Table 3.1).

3.3.2 Impact of aerial cover on SOC and N stocks

The SOC content and stocks in the 0.05 m soil layer decreased with decreasing grass aerial cover in the following order: Cov100 (19.87 g kg^{-1}; 1.39 kg m^{-2}) > Cov75 (11.19 g kg^{-1}; 0.79 kg m^{-2}) > Cov50 (5.17 g kg^{-1}; 0.40 kg m^{-2}) > Cov5 (1.73 g kg^{-1}; 0.14 kg m^{-2}), with differences significant at $P<0.05$ level (Figure 3.2). Similarly, N content and stocks decreased with decreasing grass aerial cover as follows: Cov100 (1.53 g kg^{-1}; 0.106 kg m^{-2}) > Cov75 (0.95g kg^{-1}; 0.068 kg m^{-2}) > Cov50 (0.48 g kg^{-1}; 0.037 kg m^{-2}) > Cov5A (0.39 g kg^{-1}; 0.032 kg m^{-2}). SOC and N stocks varied greatly along the degradation gradient decreasing sigmodially in non-degraded grassland soils (Cov100) to heavily degraded grassland soils (Cov5) (Figure 3.2). Thus, C:N ratios decreased with decreasing grass cover, from an average of 13.0 at Cov100 to 11.8 at Cov75 to 10.6 at Cov50 and 4.4 at Cov5.

On average, land degradation resulted to a decrease in SOC stocks of 79% for Cov5, 42% for Cov50 and a negligible effect for Cov75 (Figure 3.3). Similarly, degradation led to a decrease in N stocks of 48% for Cov5 and 39% for Cov50. This result suggests that the critical grass aerial cover threshold for which degradation greatly affects SOC and N stocks is 50%. While, there were significant differences in SOC and N stocks in the 0.05 m soil layer, no significant differences were found with depth.

3.3.3 SOC and N distribution in the particle-size fractions

Results of land degradation impact on SOC_C changes in the different soil fractions are presented in Figure 3.4A. Soil organic C in the sand fraction decreased from around 6.09 g C kg^{-1} fraction in non-degraded soils (Cov100) to 0.37 g C kg^{-1} fraction in degraded soils (Cov5), which corresponded to a 94% decrease, significant at $P<0.05$. Concentrations of SOC in the silt + clay size fraction decreased from 61.4 g C kg^{-1} fraction in non-degraded soils to 2.82 g C kg^{-1} fraction in degraded soils, which corresponded to a 95% decrease, significant at $P<0.05$.

As a proportion of total soil carbon, SOC in the sand fraction decreased from 19% in non-degraded soils to 11% in heavily degraded soils (Figure 3.4B). In contrast, SOC in the silt + clay fractions varied little from 84% in non-degraded soils to 78% in heavily degraded soils. Nitrogen in sand fraction decreased from around 0.43 g N kg^{-1} fraction in non-degraded soils to 0.26 g N kg^{-1} fraction degraded soils, which correspond to a 40% decrease, significant at $P<0.05$ (Figure 3.5A). Concentrations of N in the silt + clay sized fraction decreased from 5.31 g N kg^{-1} fraction in non-degraded soils to 0.91 g N kg^{-1} fraction in degraded soils, which corresponded to a 83% decrease, significant at $P<0.05$. As a proportion of total N, the N in the sand fraction varied relatively little from 17% in non-degraded soils to 22% in heavily degraded soils (Figure 3.5B). Nitrogen in the silt + clay sized fraction varied from 84% in non-degraded soils to 78% in heavily degraded soils.

3.3.4 Impact of grass aerial cover decrease on soil physical and chemical properties

Changes in soil physical properties were observed along the grassland degradation gradient (Table 3.1). The sand distribution was similar, with 71%, 73%, and 72% for Cov100, Cov75 and Cov50, respectively, while, the sand content for Cov5 was 49%. The silt distribution was almost similar along the degradation gradient. The clay content was higher (34%) in Cov5 and similar for Cov100 (15%), Cov75 (14%) and Cov50 (15%). The aggregate stability decreased with decreasing grass aerial cover from 1.36 mm in Cov100 to 0.71 mm in Cov5. This pattern was consistent for penetrometer resistance (PR), with 16.8 kg cm^{-2} in Cov100, 18.63 kg cm^{-2} in Cov50 and 19.47 kg cm^{-2} in Cov5.

Potassium varied along the degradation gradient, with concentrations ranging between 143-167 mg kg^{-1}in Cov100 and Cov75, then declined to about 62 mg kg^{-1}in Cov5. Calcium decreased from around 226 mg kg^{-1}in non-degraded soils to 75 mg kg^{-1}in degraded grassland soils (Table 1). Magnesium decreased from about 104 mg kg^{-1} in Cov100 to 31 mg kg^{-1} in Cov50, then increased to 80 mg kg^{-1} in Cov5. Phosphorus decreased from 5.25 mg kg^{-1}in non-degraded soils to 2.17 in degraded soils. Zn and Mn, respectively significantly decreased with decreasing grass aerial cover in the following order: Cov100 (1.67 mg kg^{-1}; 14.3 mg kg^{-1}) > Cov75 (0.77 mg kg^{-1}; 10.6 mg kg^{-1}) > Cov50 (0.22 mg kg^{-1}; 4.08 mg kg^{-1}) > (0.29 mg kg^{-1}; 3.25 mg kg^{-1}). In contrast, acid saturation increased with decreasing grass aerial cover from around 26% in non degraded soils to 78% in degraded soils. ECEC also varied relatively little along the degradation gradient from 3.18 $cmol_c$ kg^{-1} in non-degraded soils to 5.35 $cmol_c$ kg^{-1}in degraded soils.

3.3.5 Other environmental factors controlling SOC and SON stocks

A correlation matrix (Table 3.2) revealed that changes in SOC and N stocks induced by degradation were significantly positively correlated to mean weight diameter (MWD), a measure of soil aggregate stability (r^2 = 0.67; P < 0.05) and negatively correlated with clay (r^2 = -0.54; P < 0.05), soil bulk density (ρ_b) (r^2 = -0.40; P < 0.05), penetration resistance (PR), a proxy of soil compaction (r^2 = -0.29; P < 0.05) acid saturation (Acid sat) (r^2 = -0.74; P < 0.05), ECEC (r^2 = -0.36; P < 0.05), exchangeable acidity (Exch acid) (r^2 = -0.66; P < 0.05).

3.4 Discussion

3.4.1 Impact of land degradation on the depletion of SOC and SON stocks

In this study, the decrease in grass aerial cover resulted in a greater depletion in SOC stocks of 89% in the surface layer. This depletion of SOC stocks is relatively high compared to what has been reported in other studies. For example, Snyman and du Preez (2005) found that land degradation decreased SOC and N stocks by 22% and 13%, respectively in fine sandy loamy grassland soils under a semi-arid climate in Bloemfontein, South Africa. Wu and Tiessen et al. (2002) found that land degradation reduced SOC stocks by 33% and N stocks by 28% in a Chinese alpine grassland. Interestingly, the higher SOC depletion rate of 89% found at our study site is of similar magnitude to those reported by Dong et al. (2012), who found that land degradation significantly reduced SOC stocks by 90% in loamy soils under a continental climate in China. Similarly, Wen et al. (2012) found that land degradation led to a 89% decline in SOC stocks for sandy grassland soils in China.

The greater depletion of SOC and N stocks in the heavily degraded grassland soils may be due to a number of reasons. First, the soils at our site are characterized by coarse texture (up to 73% sand). The greater SOC loss in such soils is due to the lack of the physical protection of organic matter (Feller and Beare, 1997). Secondly, the soils are acidic (pH <3.9). Previous studies have shown that in acidic soils, base cations such as Ca, K and Mg are weakly bound to the soil (Berthrong et al., 2009), causing weak interactions with organic matter in the soil. In this study, it was found in the heavily degraded grassland soils, Ca was reduced by 67%, K by 56% and Mg 23%. The loss of SOC lowers nutrient availability and cation exchange capacity, and this then lowers biomass production, which overtime, may lower organic inputs thereby lowering SOC. The third reason involves the removal of the nutrient rich A horizon by water erosion. Indeed, as shown by Dlamini et al. (2011) in the same study site, water erosion potentiated soil losses at rates up to 13 t ha^{-1} yr^{-1}.

3.4.2 Impact of grass aerial cover decrease on soil organic matter quality

The decrease in grass aerial cover was accompanied by a decrease in the C:N ratio and in the proportion of the organic matter present in the sand fraction of the soil. Such a shift in organic matter quality towards less proportion of fresh and easily decomposable compounds, can be explained by the decrease in biomass production and residue inputs to the soil. The fact that sorption is the main process that preserves organic matter to mineral surfaces (Christensen, 2001, Kaiser and Guggenberger, 2003) in the silt + clay fraction of the soil, suggests that the SOC remaining in the soil at the greatest degradation intensity is likely to be preserved from biological decomposition.

3.4.3 Mechanisms of SOC and SON stocks depletion

Several mechanisms are likely to explain the depletion of SOC and SON stocks as a consequence of the decrease in grass aerial cover. From the available literature, the reduction in grass aerial cover and associated decline in biomass production is likely to have a direct effect on soil stocks through the decline of organic C inputs to soils. Not only can the reduction in grass aerial cover directly impact soil stocks, but also indirectly. The loss of grass aerial cover can indeed influence the dynamic nature of the plant-soil interactions through, for instance, the modification of the water cycle and the fluxes of other elements. Hiltbrunner et al. (2012) in Swiss sub-alpine grassland observed an increase in soil bulk density by as much as 20%, with associated changes in soil functions such as biomass production. The associated changes in soil porosity tend to decrease soil infiltration by water, thus potentiating, SOC and SON losses by water erosion. Podwojewski et al. (2011) and Mchunu and Chaplot (2012) by using rainfall simulation at the same site found that the decrease of grass aerial cover from 100 to 5% decreased soil infiltration from 21.6 to 6 mm hr^{-1}, with an associated increase in SOC losses by 213% from Cov100 to Cov5, lost via erosion processes in particulate forms (Mchunu and Chaplot, 2012).

The disruption of soil aggregates through the process of either water erosion or trampling by the livestock constitutes another likely mechanism of SOC and SON stock depletion. Soil aggregates undeniably serve as physical protection for organic matter through a range of interactions from inclusion to sorption (Tisdall and Oades 1982; Jastrow, 1996; Torn et al., 1997; Baldock and Skjemstad, 2000; Masiello, 2004; Mikutta et al., 2006). The disaggregation process results in SOC losses from soils either through organic matter decomposition and associated CO_2 emissions to the atmosphere as more SOC becomes unprotected, or through preferential SOC erosion, owing to the light nature of soil organic matter. Similarly, the release of the hitherto encapsulated organic material can follow the mechanical breakdown of aggregates during trampling by cattle.

According to the conceptual model of Kinnell (2001), water erosion can breakdown soil aggregates through (1) the physical action of raindrop impact (splash); (2) the combined action of splash and shallow runoff on the soil surface, which increases the disaggregation efficiency of raindrops; (3) the action of the overland flow. While previous studies performed under clayey soil conditions, for instance, by Brunet et al. (2004, 2006) pointed to the absence of preferential SOC erosion. Studies under sandy soil conditions by Boegling et al. (2005) and by Mchunu and Chaplot (2012) at the same site demonstrated preferential SOC erosion, with an SOC enrichment of the exported soil material relative to the bulk soil, increasing by a factor greater than 5.

Another potential mechanism of SOC and SON stock depletion consecutive to the decrease in grass aerial cover lies into the alteration of the local micro-climate. Under similar grassland conditions in South Africa, Mills and Fey (2004) showed that grassland degradation is accompanied by an increase in soil temperature, which in turn increases the rate of organic matter mineralization.

While the depletion of SOC and SON stocks can be consequential to the decrease in grass aerial cover, whose origin might either be overgrazing or other land mismanagement, there remains the possibility for internal soil processes to induce the depletion of stocks. Not only can disaggregation be physical, but it can also be chemical and biological. Tisdall and Oades (1982) postulated that primary particles (<20 µm) can be agglomerated to form micro-aggregates (20-250 µm) by persistent binding agents such as oxides, polyvalent metal cation complexes and aluminosilicates; these micro-aggregates being in turn, agglomerated into macro-aggregates (>250 µm) by biological binding agents such as hyphae and roots. The biological agents tend to rapidly decompose, and when not replaced as the root density decreases, aggregates tend to breakdown, thus releasing the encapsulated SOC and SON. Leaching of oxides, metal cations and aluminosilicates could constitute a natural disaggregation mechanism in acidic sandy soils (Rienks et al., 2000), leading to SOC and SON losses with consequences and feedbacks to soil fertility and grassland degradation.

The present study by pointing to a sharp decrease in Zn concentration at the initial stages of grassland degradation, highlights the importance of micronutrients in the overall grassland ecosystem functioning. Because Zn is an essential micronutrient required by plants, any depletion below a certain threshold leads to a considerable decrease in plant productivity (Alloway, 2009), potentially explaining the decrease in grass aerial cover. What could then explain the sharp initial decrease in Zn in the study soils? The decrease in nutrients could be caused by high grazing intensity and its accompanied depletion of plant cover and litter input as well as trampling of the soil (Zhou et al., 2010; Kotzé et al., 2013). The nutrient losses could also be the result of erosion and low nutrient input of plants (Snyman and du Preez, 2005; Tefera et al., 2010). In the case Zn ions, which exist in the soil primarily as stable complexes with proteins and nucleic acids (Alloway, 2009). Soil acidification, a natural process in the region (Rienks et al. 2000) has been suggested to be the main mechanism promoting breakdown of Zn complexes, leading to Zn leaching (Cakmak et al., 1997). Therefore, the depletion of SOC and nutrient stocks could be the result of the initial and internal leaching of micronutrients, resulting in the decrease of both the protective grass cover and the soil aggregate stability, which in turn potentiates soil erosion by water and its known consequences on SOC and SON stock depletion as previously shown by Mchunu and Chaplot (2012) at the same site.

3.5 Conclusion

In this study of a degraded grassland in South Africa, our main objective was to quantify the impact of grassland degradation on the depletion of SOC and SON stocks and to evaluate some of the associated mechanisms for remediation purposes.

The present study revealed a sigmodial decrease in topsoil SOC and SON stocks with the linear decrease in grass aerial cover, with depletion rates up to 89 and 76%, respectively The stock depletion was accompanied by an increase in soil bulk density, a decrease in soil aggregate stability, a preferential enrichment in stabilized organic matter and a decrease in chemical elements such as Ca, Mg, K, Mn, Cu, and Zn essential for soil aggregate stability and plant growth.

While little evidence exists on the mechanisms responsible for the decrease in SOC and SON stocks, the sharp decline in micronutrients such as Zn, at the initial stages of grassland degradation, suggest that leaching of the essential aggregate binding and plant growth elements is the main cause of grassland degradation as well as the associated losses of SOC and SON. Further research is needed to elucidate such mechanistic linkages in order to establish effective grassland rehabilitation strategies that could replenish the depleted SOC and nutrient stocks.

Table 3.1 Characterization of soils with the different grass covers in the 0.05 m soil layer of a degraded grassland in Potshini, South Africa. Data are means (±SE; n = 48).

Soil property	**Grass aerial cover (%)**			
	100	**50-75**	**25-50**	**0-5**
Sand, %	71±1.43b	73±2.81b	72±1.10b	49±0.87a
Silt, %	14±0.93ab	13±1.71a	14±0.58ab	17±0.38b
Clay, %	15±0.74a	14±1.15a	15±0.75a	34±0.95b
SWC, %	14.72±0.62a	12.31±4.10a	9.38±0.99a	10.32±0.29a
P, mg kg^{-1}	5.25±0.22b	5.08±0.29b	1.92±0.15a	2.17±0.32a
K, mg kg^{-1}	143.17±6.97c	167.33±8.26d	88.17±11.95b	62.42±2.14a
Ca, mg kg^{-1}	225.83±10.61c	170.17±9.76b	82.67±8.71a	75.25±5.27a
Mg, mg kg^{-1}	103.67±5.02c	67.08±4.79b	30.83±4.32a	79.83±7.45b
Exch acidity, $cmol_c kg^{-1}$	0.83±0.06a	0.65±0.06a	1.47±0.11b	4.16±0.12c
ECEC, $cmol_c kg^{-1}$	3.18±0.11b	2.48±0.13a	2.36±0.08a	5.35±0.11c
Acid sat, %	25.83±1.51a	26.33±2.35a	62.25±4.43b	77.58±0.96c
pH (KCl)	3.81±0.02b	3.94±0.02d	3.88±0.02c	3.74±0.00a
Zn, mg kg^{-1}	1.67±0.20c	0.77±0.09b	0.22±0.06a	0.29±0.06a
Mn, mg kg^{-1}	14.33±1.12c	10.58±1.00b	4.08±0.43a	3.25±0.68a
Cu, mg kg^{-1}	1.34±0.10b	0.58±0.03a	0.98±0.23b	0.45±0.07a
MWD, mm	1.36±0.06b	1.35±0.10b	0.89±0.06a	0.71±0.08a
PR, kg cm^{-2}	16.77±0.81b	11.30±0.72b	18.63±1.22b	19.47±1.33a

Values indicate mean ± standard error. Sand content (Sand), silt content (silt), clay content (clay), soil water content (SWC), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), exchangeable acidity (Exch acidity), effective cation exchange capacity (ECEC), acid saturation (Acid sat), soil pH in KCl (pH KCl), zinc (Zn), manganese (Mn), copper (Cu), mean weight diameter (MWD) and penetrative resistance (PR). Statistical analyses were performed for comparisons between the different aerial grass covers. Within each grass cover class, values followed by a different letter are significantly different at $P < 0.05$.

Table 3.2 Correlation matrix between changes in SOC and N stock induced by land degradation and selected soil properties.

		N change	Cov	Clay	P	K	Ca	Mg	Ex acidity	Tot cat	Acid sat	pH	Zn	Mn	Cu	BD	SOC	SON	MWD	PR
SOC change	1.00																			
SON change	0.97	1.00																		
Cov	0.67	0.60	1.00																	
Clay	-0.54	-0.43	-0.88	1.00																
P	0.77	0.77	0.68	-0.43	1.00															
K	0.55	0.56	0.78	-0.61	0.73	1.00														
Ca	0.87	0.88	0.66	-0.47	0.81	0.73	1.00													
Mg	0.47	0.54	0.00	0.17	0.55	0.30	0.61	1.00												
Ex acidity	-0.66	-0.57	-0.96	0.96	-0.60	-0.76	-0.63	0.00	1.00											
Tot cat	-0.36	-0.24	-0.84	0.94	-0.28	-0.51	-0.24	0.41	0.90	1.00										
Acid sat	-0.74	-0.71	-0.85	0.73	-0.81	-0.91	-0.86	-0.40	0.85	0.57	1.00									
pH	0.05	-0.03	0.65	-0.72	0.22	0.57	0.11	-0.34	-0.69	-0.77	-0.54	1.00								
Zn	0.81	0.80	0.45	-0.34	0.67	0.49	0.84	0.62	-0.46	-0.11	-0.65	-0.03	1.00							
Mn	0.88	0.88	0.64	-0.44	0.80	0.65	0.92	0.52	-0.59	-0.25	-0.76	0.04	0.81	1.00						
Cu	0.48	0.46	0.27	-0.33	0.24	0.15	0.38	0.12	-0.35	-0.25	-0.26	-0.01	0.50	0.37	1.00					
BD	-0.40	-0.33	-0.48	0.39	-0.49	-0.52	-0.48	-0.31	0.49	0.32	0.53	-0.20	-0.42	-0.45	-0.13	1.00				
SOC	0.99	0.95	0.68	-0.53	0.80	0.59	0.89	0.50	-0.66	-0.35	-0.76	0.05	0.83	0.89	0.48	-0.50	1.00			
N	0.96	0.98	0.63	-0.45	0.80	0.60	0.91	0.57	-0.60	-0.25	-0.75	-0.01	0.83	0.90	0.47	-0.48	0.97	1.00		
MWD	0.67	0.64	0.68	-0.49	0.71	0.55	0.68	0.15	-0.60	-0.42	-0.67	0.17	0.58	0.71	0.22	-0.33	0.69	0.66	1.00	
PR	-0.29	-0.30	-0.55	0.34	-0.51	-0.57	-0.40	-0.13	0.45	0.32	0.53	-0.36	-0.11	-0.35	0.25	0.35	-0.30	-0.31	-0.30	1.00

*Significant correlation at $P < 0.05$

SOC change, changes in soil organic carbon; SON change, changes in soil organic nitrogen; Cov, grass cover; Clay, clay content; P, phosphorus; K, potassium; Mg, magnesium; Ex acidity, exchangeable acidity; ECEC, effective cation exchange capacity; Acid sat, acid saturation; pH; Zn, zinc; Mn, manganese; Cu, copper; BD, soil bulk density, SOC_C, soil organic carbon content, N, nitrogen content, MWD, mean weight diameter; PR, penetration resistance.

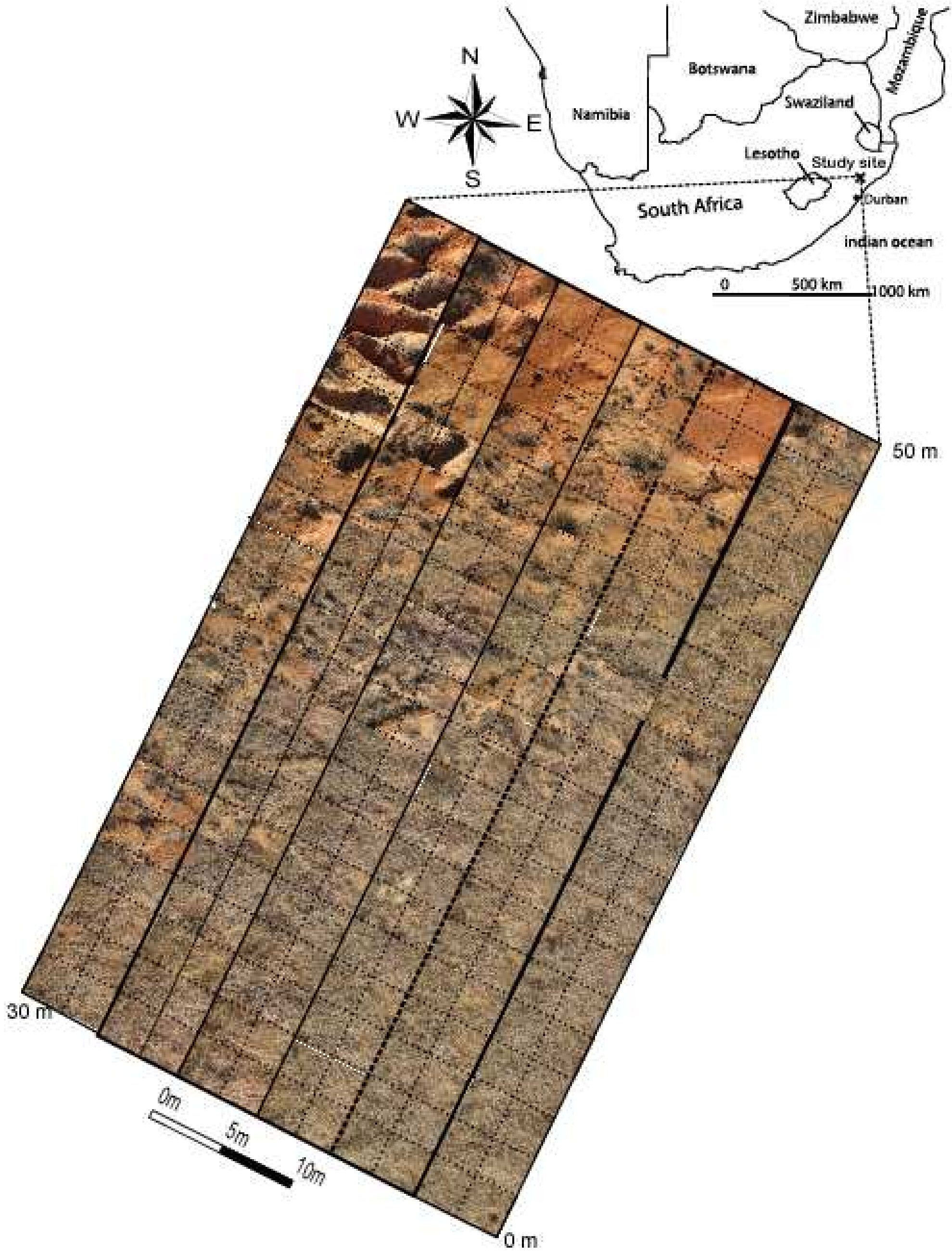


Figure 3.1 Mosaic of pictures showing the different land degradation intensities or grass aerial cover from 0-5% in the north to 100% in the south along a degraded grassland site Potshini, South Africa.

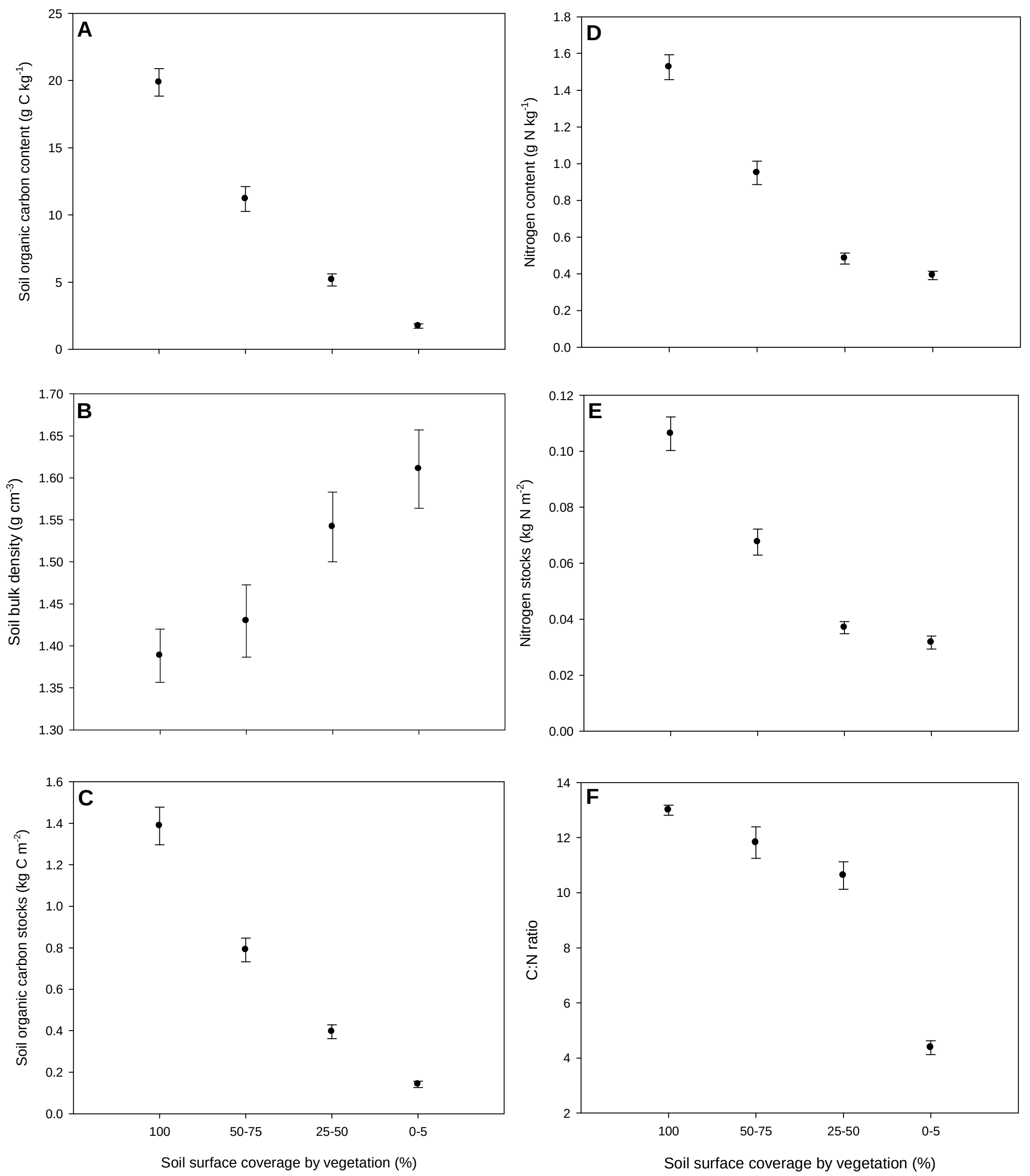


Figure 3.2 Mean ± standard error of (A) soil organic carbon content (SOC_C); (B) soil bulk density (BD); (C) soil organic carbon stocks (SOC_S); (D) nitrogen (N) content; (E) nitrogen stocks (N_S) and (F) carbon to nitrogen ratio. Values are the mean ± standard error of four replicate soil samples along the four categories of grass aerial cover (n=48).

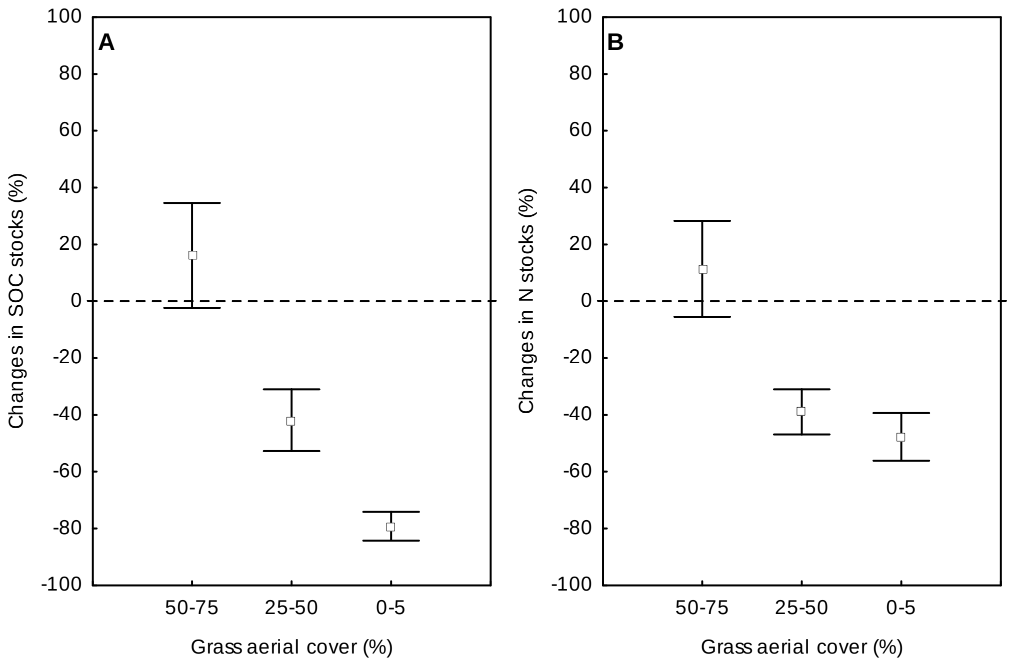


Figure 3.3 Influence of land degradation on changes in (A) soil organic stocks (SOC_S) and (B) N stocks (N_S) in the upper 0.05 m soil layer along the degradation gradient. Values for SOC and N are the mean ± standard error of four replicate soil samples from 50-75, 25-50 and 0-5 grass aerial covers, relative to 100% grass aeriel cover values.

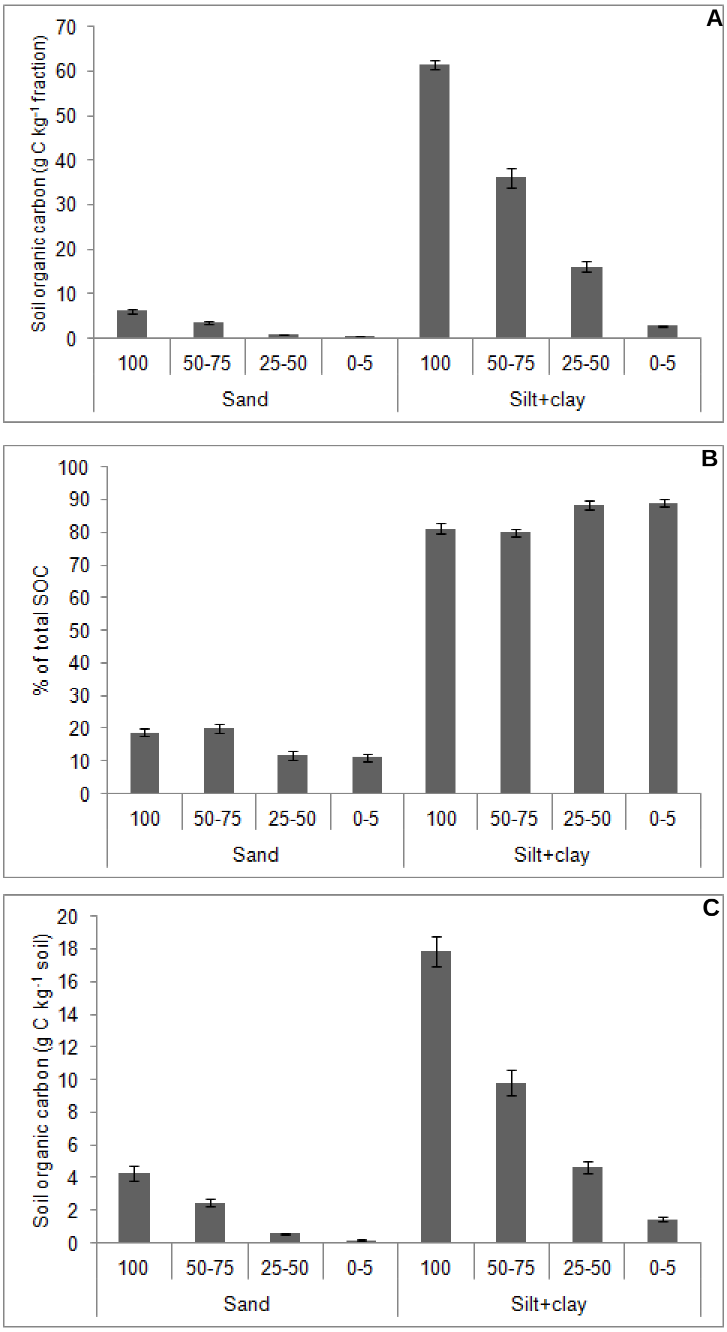


Figure 3.4 Relative distribution of SOC stored within the sand and silt + clay particle size fractions for the different grass aerial covers, showing values expressed as (A) concentrations (g C kg^{-1}) in particle size fractions, (B) proportions (%) of the total C content and (C) concentrations of SOC (g C kg^{-1}) in bulk soil. Bars represent mean ± standard error of four replicate soil samples.

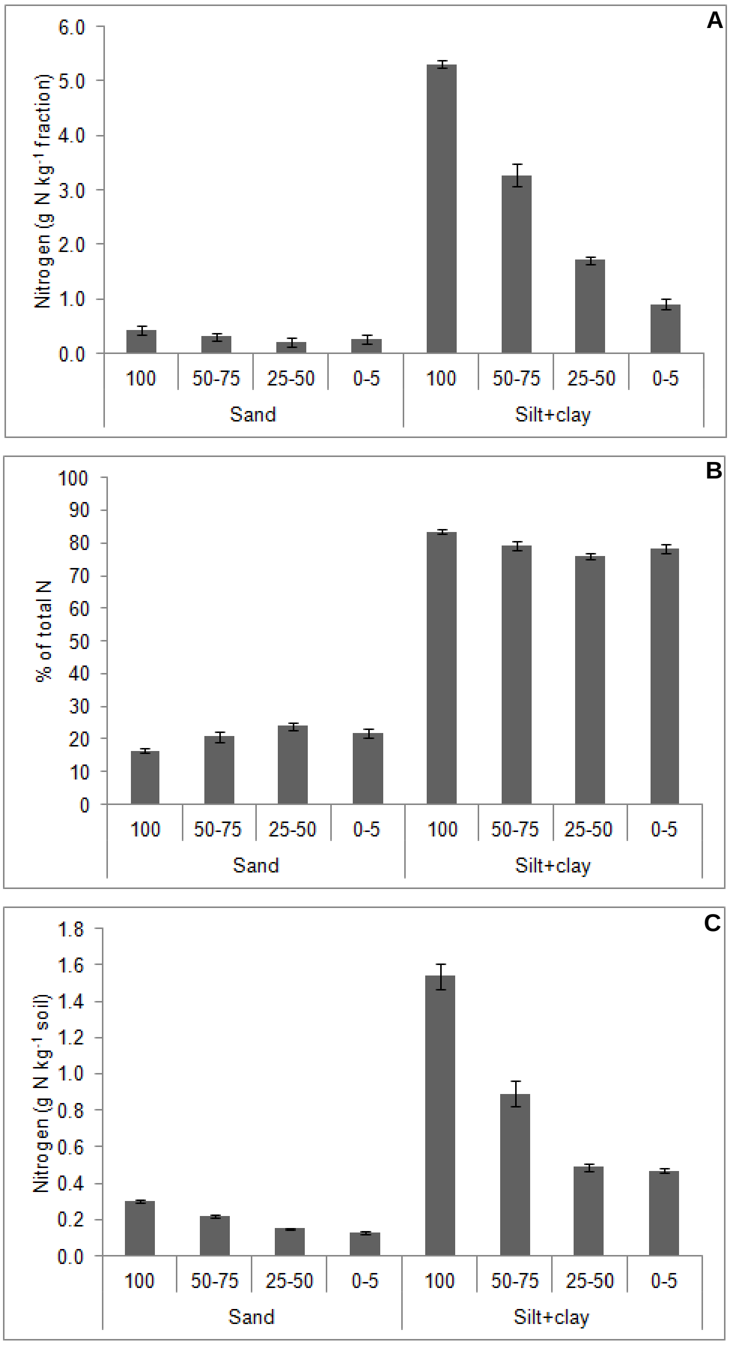


Figure 3.5 Relative distribution of N stored within the sand, silt + clay particle size fractions, for the difference grass aerial, showing values expressed as (A) concentrations (g N kg^{-1}) in particle size fractions, (B) proportions (%) of the total N content and (C) concentrations of N content (g N kg^{-1}) in bulk soil. Bars represent mean ± standard error of four replicate soil samples.

CHAPTER 4

4. DOES GRASSLAND REHABILITATION LEAD TO C AND N SOIL STOCK REPLENISHMENT?

Abstract

Grassland degradation results in significant losses of soil organic carbon (SOC) and nutrient stocks, thus reducing the capacity of grassland soils to provide key ecosystem services such as forage production and C sequestration. Grassland rehabilitation has thus emerged as an important strategy for replenishing the lost stocks of SOC and nutrients. The main objective of this study was to determine the effect of grassland rehabilitation on a communal rangeland in South Africa. A technique (“Savory”), which involves short-duration (5 days), high intensity cattle (1200 cows ha^{-1}) grazing and followed by livestock exclusion for 362 days was compared to five common grassland rehabilitation strategies: (1) traditional communal free grazing; (2) livestock exclosure; (3) livestock exclosure + topsoil tillage; (4) livestock exclosure + NPK fertilization (2:3:3, 22 at 2 t ha^{-1}); (5) annual grassland burning were applied from 2011. Changes in SOC and soil nitrogen content (SON) were assessed in 2013 along with selected grassland characteristics. A total of 540 soil samples were collected in the 0.05 m soil layer for the six treatments between 2011 and 2013. After two years, SOC and SON stocks in the 0.05 m surface layer were significantly increased under “Savory” and fertilization treatment, by an average of 6.5% (0.091 kg C m^{-2}) and 3.9% (0.055 kg C m^{-2}), respectively for SOC, which corresponded to significant differences at $P<0.05$. Livestock exclosure, tillage and burning reduced SOC stocks by 1.4%, 3.5% and 6.9%, respectively. A maximum SOC repletion rate of 14.8% was observed for Savory for moderately degraded soils as the land was re-vegetated, but no gains were observed for the heavily and non-degraded grasslands. The time controlled, short-duration and high intensity grazing of cattle in Southern Africa seem to be a viable strategy for grassland rehabilitation compared to burning, tillage and livestock exclosure. While only a few years might be required to replenish SOC stocks under moderately degraded grasslands, more is to be done to accelerate grassland rehabilitation and thus SOC replenishment on the heavily degraded grassland soils.

Keywords: *grassland, degradation, rehabilitation, grazing*

4.1 Introduction

Grassland soils store approximately10% of the global soil carbon (C) stock of 1500 Gt, but there is evidence that over the last few decades that large amounts of C have been lost from grassland soils through degradation and poor grassland management practices (Daily, 1995; Lal, 2004; Conant et al., 2001). For instance, Wu and Tiessen (2002) reported that grassland degradation reduced SOC stocks by 33% in Chinese alpine grassland, while a more recent study by Dong et al. (2012) showed that grassland degradation decreased SOC stocks by 90% in a Chinese grassland. The depletion of SOC stocks in grassland soils is exacerbated by poor grassland management practices (Conant et al., 2001). As a result, one of the major implications of such SOC losses is the reduced capacity of grassland soils to provide key ecosystem services such as forage production and C sequestration (FAO, 2010).

Grassland rehabilitation - defined here as the process of assisting the recovery of an ecosystem that has been degraded (SER, 2004), has been posited as a strategy that could potentially sequester more C in soil, thus improve the functioning of grasslands and mitigate grassland degradation (FAO, 2010). One of the prerequisites to rehabilitate degraded lands is to increase soil C stocks (Lal, 2004). It has been suggested that if rehabilitation measures are appropriately applied to degraded soils, SOC and nutrient stocks can be maintained or enhanced (Follet, 2001). Indeed, effective rehabilitation can be achieved through the adoption of appropriate management practices, which are considered as part of a strategy to reduce C loss from grassland soils by increasing residue inputs and reducing decomposition rates (Paustian et al., 2000, FAO, 2010).

Rehabilitation of grasslands has been shown to enhance plant species diversity (Smith et al., 2008). Some studies have showed that grassland rehabilitation improves soil aggregation (An et al., 2013) and increases SOC stocks (Dong et al., 2012), especially in the uppermost surface layer of the soil. Therefore, grassland rehabilitation needs to be promoted to stabilize erodible soil, improve soil quality and sequester atmospheric carbon (Lal et al., 1999; Post and Know, 2000).

The SOC sequestration potential of the world's grasslands is approximately estimated to be 0.01–0.3 Gt C year^{-1} (Lal, 2004), and is hypothesized to be achieved through rehabilitation practices such as fertilization, grazing exclusion by fencing, tillage and burning have been shown to be important for grassland rehabilitation (Bruce et al., 1999; Conant et al., 2001; Potthoff et al., 2005; Smith et al., 2008;). These rehabilitation strategies have been widely implemented to improve ecosystem services in grassland (Feng et al., 2013). Nitrogen fertilization has been used as a management strategy to enhance primary production, to improve the forage quality and ground cover, thereby promoting the rehabilitation of degraded grasslands (Conant et al., 2001; Bai et al., 2010). In addition, nitrogen fertilization has been shown to inhibit soil microbial respiration (Ramirez et al., 2010). Nitrogen additions have also been reported to significantly increase the decomposition of light soil carbon fractions, with decadal turnover times, while it has been shown to further stabilize soil carbon compounds in heavier mineral associated fractions, with multidecadal to century lifetimes (Neff et al., 2002). As a result, application of nitrogen fertilizer is often recommended to increase SOC, particularly on degraded lands that have experienced significant losses of SOC. The application of appropriate grazing regimes in degraded grasslands is seen as an ecologically viable solution for their rehabilitation (Papanastasis, 2009). Grazing plays a critical role as a disturbance mechanism that opens up swards and enables seeds to germinate in the gaps created (Bullock et al., 1995; Kotenen, 1996; Bekker et al., 1997). Over the past years, excluding livestock through the establishment of fences has become a common grassland management strategy for the rehabilitation of native vegetation (Spooner et al., 2002). Steffens et al. (2008) found that 25 years of grazing exclusion in a semi-arid grassland in Inner Mongolia dominated by *Leymus chinensis* increased SOC stocks by 82%, while fencing of a degraded alpine grassland in China decreased SOC by 25% (Li et al., 2013).

Fire is another common management practice in grasslands that is used regularly by livestock farmers and wildlife managers, especially in African savanna to regularly control bush encroachment and to remove dead and dying vegetation that has low forage quality and is unpalatable to animals (Tainton, 1999). The burning of grasslands to remove dead and dying vegetation often results in grass growth earlier in the growing season, which increases dry-matter production (Ojima et al., 1994).

While many studies have considered strategies to rehabilitate grasslands (Smith et al., 2000; 2002; 2003; 2008), only a few have reported the consequences of rehabilitation on

SOC and SON stocks (De Deyn et al., 2011). Information on effective stock replenishment through rehabilitation for different environments is indeed crucial to refine the worldwide stock replenishment potential of grassland soils. Moreover, research efforts are needed to identify the underlying mechanisms controlling grass recovery as well as C and N replenishment in soils, especially those that will increase C input to the soil and reduce C losses (De Deyn et al., 2008).

Smallholder farmers in South Africa and in many drylands have no access to fertilization, fencing and mechanization. The objective of this study was to investigate how a shift in the cattle management by smallholder farmers can lead to grassland rehabilitation and C sequestration in soils. In this study, the “Savory” holistic management technique (Savory and Parsons, 1980; Savory, 1983), which involves short-duration, high intensity cattle grazing and followed by livestock exclusion was implemented. Several common grassland rehabilitation strategies were also applied for comparison and to further improve understanding of the mechanisms involved in grassland rehabilitation and stock replenishment.

4.2 Materials and methods

4.2.1 Site selection and description

The grassland site is located in the Potshini catchment, which is 10 km north of the Bergville district in the KwaZulu-Natal Province of South Africa (Long: 29^{o} 21'; Lat: -28^{o} 48'). This area has a temperate climate, with cold dry winters and warm rainy summers, a mean annual precipitation of 684 mm, most of which falls in the summer months (October and March), a mean annual potential evaporation of 1600 mm and a mean annual temperature of $13^{o}C$ (Schulze, 1997). The altitude ranges from 1080 to 1455 m.a.s.l and the average slope gradient is 8%. The vegetation in this area is classified as a Moist Highland Sourveld (Camp, 1999). The dominant vegetation of Moist Highland Sourveld species include *Hyparrhenia hirta* and *Sporobolus africanus*.

Soils on this site are acidic (pH 3.78-3.86 in KCL) Acrisols characterized by a high sand content (60%) derived on sandstone and mudstone, (WRB, 2006). Kaolinite is the dominant clay mineral in these Acrisols. The soils are characterized by a dark brown (7.5YR 4/4) 0-0.3 m thick A horizon, with a weak sub-angular blocky structure. This horizon is underlain by a reddish (5YR 4/6) B-horizon 0.3-0.7 m. Underlying this horizon is the C horizon 0.7-1.2 m characterized by sandy saprolite showing signs of wetness. Effective cation exchange capacity (ECEC) ranged between 1.86 and 5.86 $cmol_c\ kg^{-1}$, while acid saturation ranged between 11 and 83% (Table 1).

4.2.2 Experimental design of rehabilitation treatments

The experimental site was established in July 2011 within a communal degraded grazing rangeland in the KwaZulu-Natal Province, South Africa. The site consists of a 50 m × 30 m plot, which exhibits clear shifts in aerial grass cover from non-degraded grasslands ,with grass aerial cover (Cov) of 100%) to heavily degraded grasslands where Cov was as low as 0%. The plot was further sub-divided into six 5 m × 50 m plots, showing the different intensities of degradation. The rehabilitation treatments were (1) traditional communal free grazing; (2) "Savory" technique, which involves short-duration (5 days), high intensity cattle (1200 cows ha^{-1}) grazing and followed by livestock exclusion for 362 days (2) livestock exclosure; (3) livestock exclosure + topsoil tillage; (4) livestock exclosure + NPK fertilization (2:3:3, 22 at 2 t ha^{-1}); (5) annual grassland burning were applied from 2011.

In brief, the treatments were as follows: In the control treatment, the livestock are allowed to continuously graze as is common practice in the community. The Savory treatment is based on the idea of the Savory holistic management method (Savory and Parsons, 1980; Savory, 1983). This holistic method entails managing livestock on the land, in such a way that they can be used to reverse the degradation of grasslands with or without fencing. Such an intervention is based on the hypothesis that livestock grazes and tramples on the grass and in so doing enhances litter cover and stimulates their growth as sunlight reaches the low-growing parts. The cows then cycle the dead plants back to the soil surface through deposition of dung and urine. In so doing they enhance the organic matter and porosity of the soils while keeping water in the system (Savory and Parsons 1980; Savory, 1983; Fynn, 2008). In the Savory treatment, 38 Nguni cattle from the local community were left overnight for 5 days, in the 2011-2012 and 2012-2013 seasons. The fenced treatment includes fencing to exclude livestock. In the tilled treatment, the soil was tilled by hand-hoeing to a depth of 0.1 m. The fertilized treatment included livestock exclusion and fertilization with nitrogen (N), phosphorus (P) and potassium (K) combined +NPK fertilizer (2:3:3, 22 at 2 t ha^{-1}). The fertilizer was decided based on what is commonly used in the community. The burned treatment included annual grassland burning in the 2011-2012 and 2012-2013 seasons. Burning is an important management practice that is commonly used in African savanna by both livestock farmers and wildlife managers to regularly control bush encroachment and to remove dead, dying vegetation that has low forage quality and is unpalatable to animals (Tainton, 1999). The monitoring of the all the rehabilitation techniques started in August 2011 through to October 2013.

4.2.3 Soil sampling

Soil samples were collected in July 2011 to establish the baseline conditions in selected soil properties before the implementation of the different rehabilitation treatments. According to Sanderman and Baldock (2010) without the baseline at the inception of the experiment, it is not possible, for instance, to determine whether or not the current measured SOC between treatments has resulted in a net sequestration of CO_2. It is from the baseline that the impact of the applied rehabilitation treatments on changes in SOC and SON stocks was calculated.

For soil sampling, four categories of grass aerial cover or intensities of degradation were considered, i.e. 75-100% (Cov100), 50-75% (Cov75), 25-50% (Cov50), and 0-5% (Cov5). This factorial design of treatments allowed for testing of changes in SOC and SON in soils of varying degradation intensities. At each category, three sampling points were randomly selected. For each selected sampling position, three replicate soil samples were collected in the 0-0.05 m soil layer 1-m apart in a radial basis sampling strategy to yield nine samples per category. The sampling resulted in a total of 45 soil samples per rehabilitation treatment. Soil samples were also collected at depths of 0.1, 0.25, 0.45, 0.6, 0.9 and 1.2 m layer using a 220.89 m^{-3} metallic cylindrical cores (height 0.05 m, diameter 0.75) (Table 1). The site was re-sampled in the August of 2013, two years after the onset of the rehabilitation strategies to detect differences between treatments and to monitor changes in SOC and SON overtime. Soil samples were returned immediately to the laboratory where roots and stones were removed by hand.

4.2.4 Assessment of plant basal cover

To address potential changes in both plant aboveground biomass and basal cover composition across rehabilitation treatments, which may help explain changes in SOC, a vegetation survey was carried out on a monthly visit. The plant basal cover was estimated using metal wire square sub-divided into 0.5 m × 0.5 m quadrats, which were placed in exactly the same position over each plot and the proportion of the surface covered by vegetation was recorded in every other quadrat over time following the four categories of grassland aerial cover. Moreover, pictures were taken every 2.5 m × 2.5 m to produce a vegetation map of the baseline conditions before and after the treatments were applied.

4.2.5 Above-ground biomass

Above-ground biomass was harvested once a year in August 2012 and 2013 to coincide with the grass cutting regimes in the community. Above-ground biomass was collected from each treatment using systematically positioned 0.5 m × 0.5 m quadrats by clipping all shoot material above the soil surface to the crown, oven-dried at 60^{o}C and weighed. The dry mass of all living plants per quadrat was averaged over 15 replicates to estimate above-ground biomass. In the field, the harvested above-ground biomass was removed from the six treatments by rake and hand, which resulted in almost no standing biomass. For each grass aerial cover class and treatment, above-ground biomass was monitored over time.

4.2.6 Leaf area index

Leaf area index was measured using an AccuPAR L-P-80 ceptometer and photosynthetic active radiation sensor. Leaf area index allows for estimation of biomass production without destroying the plant. Each treatment was divided into five quadrants and in each quadrant five replicate measurements were taken.

4.2.7 Basal cover

Basal cover was determined using the method of Hardy and Tainton (1993), with 100 sample points per site located by lowering a sharp spike every 1-m in a radial basis, and the distance to the nearest tuft and diameter of that tuft were measured. The percentage basal cover was then calculated using the empirically derived equation of Hardy and Tainton (1993). This gives the basal cover of grasses. Forbs contribute very little to basal cover in these grasslands, hence were not included in the assessment.

4.2.8 Species diversity index

The number and frequency of grass species at each treatment was determined using the Simpson’s diversity index (Simpson, 1949). The number of species present per ecological category (decreaser, increaser I, increaser II and increaser III species) and the abundance of each species were determined.

4.2.9 Aggregate stability

The aggregate stability of the soil samples was measured on the 3-5 mm aggregates following the method described in Le Bissonnais (1996). Briefly, three procedures that distinguish various mechanisms of breakdown were used: slaking due to fast wetting (treatment 1), micro-cracking due to slow wetting (treatment 2) and mechanical breakdown by stirring of pre-wetted aggregates (treatment 3). The stability of each breakdown mechanism was expressed using the resulting fragment distribution of the six classes ranging from <0.1 mm to 2-5 mm. The mean weight diameter (MWD), which is the sum of the mass fraction of soil remaining on each sieve after sieving multiplied by the mean aperture of the adjacent meshes was calculated from the fragment size distribution.

4.2.10 Soil physical and chemical analysis

The soils were air-dried and sieved to pass through a 2 mm mesh. Particle size distribution was determined by the pipette method (Gee and Bauder, 1986). Total C and N were determined on air dried soil by complete combustion using a LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). The soil bulk density was determined by the gravimetric method (Blake and Hartge, 1986). The soil pH was measured in KCl and H_2O using a Calimatic pHM766 pH meter, whereby a solution ratio of 1:2.5 was used (10 g soil: 25 mL solution). Exchangeable Ca, Mg and extractable acidity were determined by extraction in 1M KCl while P, K, Zn, Mn and Cu were determined by extraction in Ambic 2 - extract containing 0.25M NH4HCO3, with detection by atomic absorption spectrometry and inductively-coupled plasma optical emission spectrometry (Manson and Roberts, 2000). Effective cation exchange capacity (ECEC) was calculated as the sum of extractable cations and some rapid measure of exchangeable acidity, while the percentage acid saturation was calculated as the exchangeable acidity × 100/ (Ca + Mg + K + 'extractable acidity').

4.2.11 Calculation of in soil organic carbon and nitrogen stocks

The SOC and N stocks were calculated using the following equation by (Batjes, 1996):

$$C_s = x_1 x_2 x_3 (1 - \frac{x_4}{100}) \times b \qquad (2)$$

where C_s is the C stock (kg C m^{-2}); x_1 is the C concentration in the <2 mm soil material (g C kg-1); x_2 is the soil bulk density (kg m^{-3}); x_3 is the thickness of the soil layer (m); x_4 is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

4.2.12 Statistical analysis

Above-ground biomass, SOC, N, SOC_S, N_S, soil bulk density and MWD data were subjected to an analysis of variance using the PROC MIXED procedure of SAS software (SAS Institute, 2003). Above-ground biomass, SOC, N, SOC_S, N_S, soil bulk density and MWD were considered as fixed effects, whereas treatment, vegetation cover and season were considered as random effects. Differences between means were tested with the DIFF option of LSMEANS statement with a significance level of $P < 0.05$.

4.3 Results

4.3.1 Effect of treatments on leaf area index, species diversity and basal cover

After two years of rehabilitation, the "Savory" treatment significantly increased LAI by 274% from an average of 0.38 ± 0.11 in the control treatment to 1.42 ± 0.50 in the Savory treatment (Figure 4.1A). The fertilized treatment significantly increased LAI by 321%, with an average of 1.60 ± 0.63 compared to the 0.38 ± 0.11 in the control treatment. Both the fenced and tilled treatments significantly increased LAI relative to the control by 134% and 89%, respectively.

Plant species diversity, measured using the Simpson index was lower in the "Savory" treatment (0.25) and fertilized treatment (0.22) compared to the control (0.25). The plant species diversity was greater in the tilled (0.41) and burned treatments (0.34) compared to the control, while there was no significant difference between the control and the fenced treatment (Figure 4.1B).

The percentage of basal cover in the "Savory" treatment (12.6%) and fertilized treatment (12.7%) was higher compared to the control treatment (10.6%). The basal cover in the burned treatment (3.8%) was significantly lower compared to the control. There were no significant differences between the fenced (10.8%) and tilled treatments (10%) relative to the control (Figure 4.1C).

4.3.2 Effect of rehabilitation treatments on above-ground biomass

Repeated measure of ANOVA of above-ground biomass, using rehabilitation treatment, grass cover and season, and all interactions as fixed-factors, showed that the effects of grass cover, treatment and season were highly significant and that all interactions were also highly significant (Table 4.2).

After one year of rehabilitation, in Cov5, "Savory" treatment significantly increased ($P < 0.05$) above-ground biomass production from 7.3 kg m^{-2} in the control to 18.4 kg m^{-2}, 19.5 kg m^{-2} in the control to 26.1 kg m^{-2} in Cov50, 22.8 kg m^{-2} in the control to 50.5 kg m^{-2} in Cov75 and. 22.5 kg m^{-2} in the control to 40.1 kg m^{-2} in Cov100 (Fig 4.2A). Fertilization significantly increased above-ground biomass from 7.3 kg m^{-2} in the control to 64.6 kg m^{-2} in Cov5, 19.5 kg m^{-2} in the control to 128.8 kg m^{-2} in Cov50, 22.8 kg m^{-2} in the control to 92.7 kg m^{-2} in Cov75 and 22.5 kg m^{-2} in the control to 143.2 kg m^{-2} in Cov100. Burning significantly decreased above-ground biomass from 7.3 kg m^{-2} in the control to 5.2 kg m^{-2} in Cov5, 22.8 kg m^{-2} in the control to 4.6 kg m^{-2} in Cov50 and 22.5 kg m^{-2} in the control to 18.5 kg m^{-2} in Cov75. No significant differences were observed between the tilled, fenced and the control.

After two years rehabilitation, "Savory" treatment significantly increased (P<0.05) above-ground biomass production from 7.2 kg m^{-2} in the control to 27.3 kg m^{-2} in Cov5, 23.3 kg m^{-2} in the control to 122.9 kg m^{-2} in Cov50, 13.0 kg m^{-2} in the control to 107.9 kg m^{-2} in Cov75 and 34.7 kg m^{-2} in the control to 132.8 kg m^{-2} in Cov100 (Fig 4.2A). Fertilization significantly increased above-ground biomass production from 7.2 kg m^{-2} in the control to 120.6 kg m^{-2} in Cov5, 23.3 kg m^{-2} in the control to 129.0 kg m^{-2} in Cov50, 13.0 kg m^{-2} in the control to 88.8 kg m^{-2} in Cov75 and 34.7 kg m^{-2} in the control to 135.5 kg m^{-2} in Cov100. Burning decreased above-ground biomass from 7.2 kg m^{-2} in the control to 2.5 kg m^{-2} in Cov5, 1.8 kg m^{-2} in the control to 26.1 kg m^{-2} in Cov50, 13.0 kg m^{-2} in the control to 7.8 kg m^{-2} in Cov75 and 34.7 kg m^{-2} in the control to 82.0 kg m^{-2}. No significant differences were observed between the tilled, fenced and the control.

4.3.3 Effect of rehabilitation strategies on changes in SOC and SON stocks

After two years, the "Savory" treatment increased SOC stocks by 6.5% relative to the baseline SOC stocks, while +NPK fertilization increased SOC stocks by 3.9%. In contrast, livestock exclosure, tillage and burning decreased SOC stocks by 1.4, 3.5, and 6.9% respectively (Figure 4.3).

After two years, no significant differences were observed for SON stocks between the "Savory" treatment and the control. Fencing, tillage, fertilization and burning slightly decreased SON stocks by 1.5, 3.8, 1.9 and 1.3%, respectively (Figure 4.4).

4.3.4 Effect of rehabilitation on soil aggregate stability

Within each rehabilitation treatment, the mean weight diameter, a measure of soil aggregate stability among the four grass aerial covers was different in the following order: 100 > 50-75 > 25-50 > 0-5%. After two years of rehabilitation, "Savory" significantly increased ($P < 0.05$) and fenced treatments resulted in a slight increase in soil aggregate stability compared to the control (Figure 4.5), the effect not significant at $P < 0.05$.

4.4 Discussion

The carbon sequestration potential of the grasslands in our study site, with rehabilitation by the "Savory" technique was 0.046 kg C m^{-2} y^{-1} and 0.028 kg C m^{-2} y^{-1} through fertilization. This SOC accumulation rate within the short-term rehabilitation was similar to the 0.049 kg C m^{-2} y^{-1} reported by Steinbeiss et al. (2008) in Germany, 0.058 kg C m^{-2} y^{-1} reported by Fornara et al. (2013) in UK grasslands and 0.06 kg C m^{-2} y^{-1} estimated by Janssens et al. (2005) for European grasslands but significantly lower than that of 0.317 kg C m^{-2} y^{-1} reported by De Deyn et al. (2011) in UK grasslands.

4.4.1 Effect of "Savory" on changes in SOC stocks

In this study of a degraded communal rangeland, grassland rehabilitation through short-duration, high intensity cattle grazing significantly increased SOC stocks in the 0.05 m soil surface layer. The increase in SOC stocks under "Savory" may be attributed to the deposition of manure and urine by cattle, which improved the soil fertility and thus increased biomass production. The deposition of dung by cattle on the soil surface provides a continuous supply of carbon and nutrients that may sustain micro organisms (Bardgett et al., 1998). The retention of nutrients provides an important ecosystem service. The efficient cycling of nutrients sustains plant growth. As a consequence, there was an increase in biomass production, which results in greater C inputs to the soil through plant litter and root production (Smoliak et al., 1972). High intensity cattle grazing for several days also opened up the sward canopy, which allowed sunlight to penetrate to low-growing grasses and forbs (Savory and Parsons, 1980; Savory, 1983; Menke, 1992, Fynn, 2008). In addition, high intensity grazing also removed standing stems, leaving a thick litter layer to build-up uniformly on the soil surface. Livestock 'hoof action' or trampling puts dead material in contact with decomposer bacteria and invertebrates in the soil which speeds nutrient recycling and litter turnover (Menke, 1992). So long as the high-intensity grazing is infrequent, perennial grass plant carbohydrate metabolism is not severely disrupted and plants are maintained in the community (Menke, 1992).

The "Savory" technique also resulted in increased above-ground biomass and basal cover. This was probably due to the increased C inputs, breaking of the soil surface by hoof action from the livestock, which allowed the vegetation to spread out and light to penetrate into the soil. In the longer term, it is expected that the "Savory" technique will have far reaching consequences for ecosystem services in the grassland. The continuous increase in above-ground biomass, basal cover and litter build up is likely to increase the water holding capacity of the soil. Increased plant cover combined with the litter layer in the bare patches is also likely to increase water infiltration, decrease surface erosion and thus reduce soil erosion.

After two years of rehabilitation, the "Savory" technique slightly increased the soil aggregate stability compared to the control. In the long-term, improved soil aggregation can promote the physical protection of C and N in the soil. An increase in SOC stocks will lead to an improvement in soil quality, including an increase in the water holding capacity, soil aggregate stability and nutrient retention.

4.4.2 Effect of fertilization on changes in SOC stocks

After two years of rehabilitation, fertilization increased SOC stocks in these acidic sandy grassland soils. A potential mechanism underlying the accrual of SOC could be accumulation of partially undecomposed plant-derived detritus over time that was often observed in topsoil layers of the fertilized treatment (Fornara et al., 2013). This observation is supported by previous studies which have shown that fertilizer additions, especially of N reduce microbial mineralization of soil organic pools, effectively slowing down decomposition (Janssens et al., 2010; Liu and Greaver, 2010) and thus promoting organic C accumulation in the soil. The increase in SOC stocks following fertilization could also be attributed to increases in plant productivity, which increased C inputs into the soil. The addition of P and K through the combined +NPK fertilizer could have differently affected the rates of plant litter decomposition (Kaspari et al., 2008), thus potentially influencing C accumulation.

In this study, fertilization significantly increased above-ground biomass after two years of grassland rehabilitation. The increase in biomass production is expected to increase microbial activity and consequently CO_2 concentration. Fertilization on the other hand reduced plant species diversity. Previous studies have also shown that fertilizer addition reduces species richness (Bai et al., 2010). The mechanisms responsible for reduction of grassland diversity following fertilization could involve acidification or the accumulation of plant litter (Hautier et al., 2009), however, more work needs to be done to understand the underlying mechanisms.

4.4.3 Effect of tillage on changes in SOC stocks

Tillage decreased SOC stocks in this short-term rehabilitation experiment, which is consistent with the results obtained by Potthoff et al. (2005) in restored perennial grassland in California, USA. A possible explanation for the reduction of SOC stocks due to tillage is the mechanical disturbance caused by tillage which increases the rate of decomposition of SOC by destroying the physical structure of the soil and exposing soil organic matter to microbial decomposition (Six et al., 1999). Tillage also increases aeration by breaking down soil aggregates, which leads to rapid mineralization of C previously encapsulated within the aggregates and enhancement of CO_2 fluxes to the atmosphere (Elliot, 1986).

In this degraded communal rangeland, two years of grassland rehabilitation through tillage increased plant species diversity. The increase in plant species diversity may be attributed to tillage disturbance. A previous study by Fynn et al. (2004) suggested that disturbance is necessary to achieve maximum grass species richness by removing litter and increasing the availability of light (Fynn et al., 2004), while, complete protection from disturbance results in high litter levels on the soil surface, which reduce seedling germination and emergence (Fynn et al. 2003).

4.4.4 Effects of burning

In this study, burning resulted in a decrease in SOC and SON stocks. This is consistent with previous long-term studies that have similarly shown that fire reduces total soil carbon, nitrogen and rates of nitrogen mineralization (Ojima et al., 1994; Fynn et al., 2003; O’ Connor et al., 2004). This is ascribed to reduced organic matter inputs, but most importantly to an increase in the rate of soil organic matter mineralization. Increased mineralization is likely to take place due to greater soil temperature, an increased number of wetting and drying cycles, an increase in soil pH and a possible reduction in microbeinhibiting root exudates (Mills and Fey, 2004). Burning also decreased plant basal cover and above-ground biomass. Fire dries out the soil surface by removing surface litter, which greatly increases evaporation from the soil surface (Snyman, 2002), thus affecting biomass productivity.

4.5 Conclusion

In this study of a communal rangeland in South Africa with varying intensities of degradation, the main objective was to determine the effect of grassland rehabilitation on a communal rangeland in South Africa. It can be concluded that the "Savory" and fertilized treatments increased SOC stocks by an average of 6.5% (0.091 kg C m^{-2}) and 3.9% (0.055 kg C m^{-2}) after two years of rehabilitation. In addition, both the "Savory" and fertilization had strong effects on above-ground community properties, increasing basal cover and biomass productivity. Therefore, these rehabilitation techniques have the potential for C sequestration and can lead to more sustainable communal rangelands. Future research should focus on improving understanding into the mechanisms involved into the enhancement of C sequestration.

Table 4.1 General soil characteristics of the experimental site before rehabilitation.

Depth	Sand	Silt	Clay	BD	SOC	N	SOCs	Ns	C/N	pH(KCl)	P	K	Ca	Mg	Exch. Acidity	Total cations	Acid sat.	Zn	Mn	Cu
m	----------%----------			g cm^{-3}	----g kg^{-1}----		----kg m^{-2}----				--------mg kg^{-1}--------				------------$cmol_c kg^{-1}$------------		%	-----mg kg^{-1}-----		
0-0.05	66	15	20	1.49	9.49	0.1	0.71	0.06	11	3.84	4	115	138	70	1.78	3.3	48	0.7	8	0.8
0.1-0.25	67	16	17	1.45	4.4	0.04	0.95	0.09	11	3.86	1	49	38	18	1.54	2	77	0	1	0.4
0.25-0.45	66	17	17	1.40	3	0.04	0.84	0.11	8	3.84	1	39	26	17	1.49	1.85	80	0	1	0.6
0.45-0.6	67	17	16	1.38	1.7	0.02	0.35	0.04	9	3.81	1	39	27	23	1.66	2.08	80	0.1	1	0.5
0.6-0.9	64	21	14	1.52	1.8	0.03	0.82	0.14	6	3.85	1	25	31	22	1.29	1.68	76	0.1	1	0.4
0.9-1.2	58	24	18	1.59	2.5	0.04	1.19	0.19	6	3.78	1	30	46	28	1.86	2.4	78	0.1	1	0.4

Table 4.2 Repeated measure analysis of variance for above-ground biomass using treatment, grass cover and season, and all interactions as fixed-effects after two years of experimental treatments.

Effect	Df	*F* value	*P*
Treatment	5	150.32	< 0.0001
Grass cover	3	45.80	< 0.0001
Season	1	69.51	< 0.0001
Treatment × grass cover	15	4.03	< 0.0001
Treatment × season	5	14.20	< 0.0001
Grass cover × season	3	3.75	0.0135
Treatment × grass cover × season	15	4.63	< 0.0001

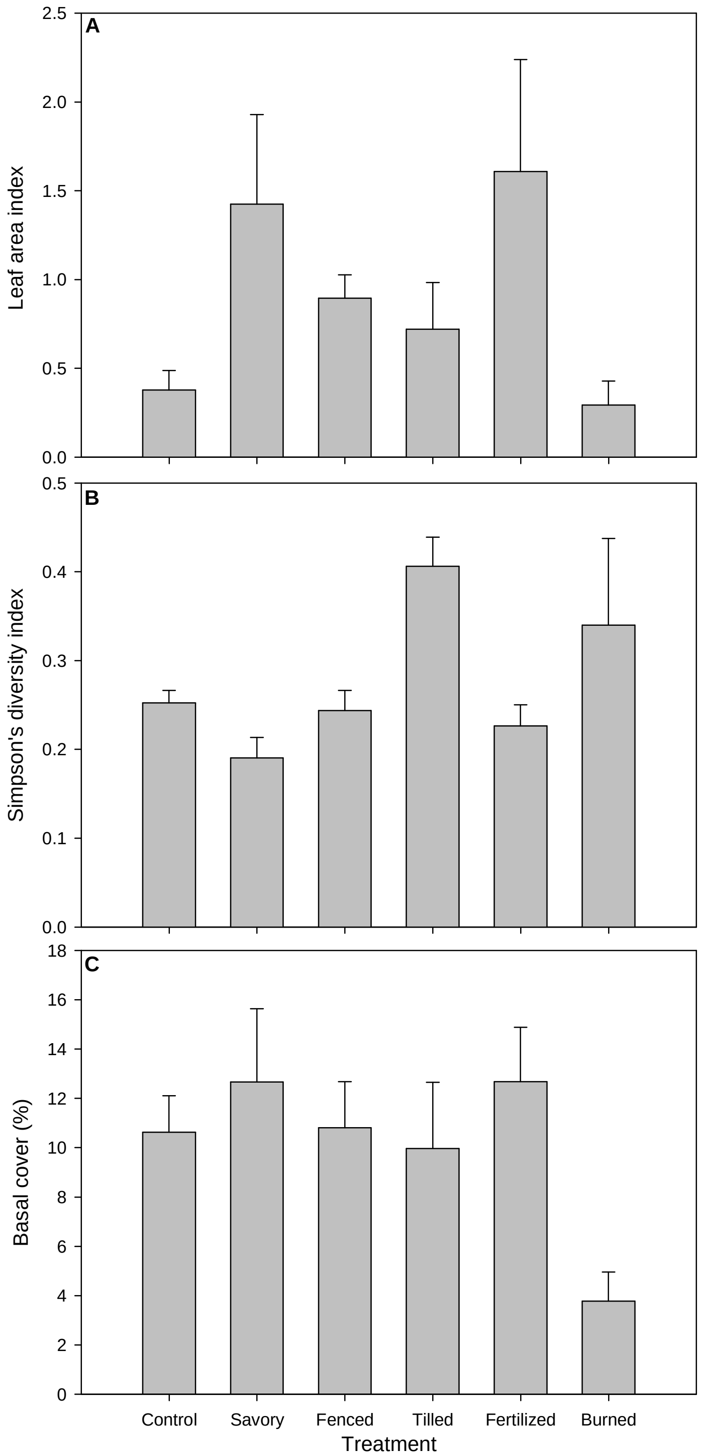


Figure 4.1 Effects of rehabilitation on (A) leaf area index, (B) plant diversity calculated using the Simpson's index and (C) basal cover after two years.

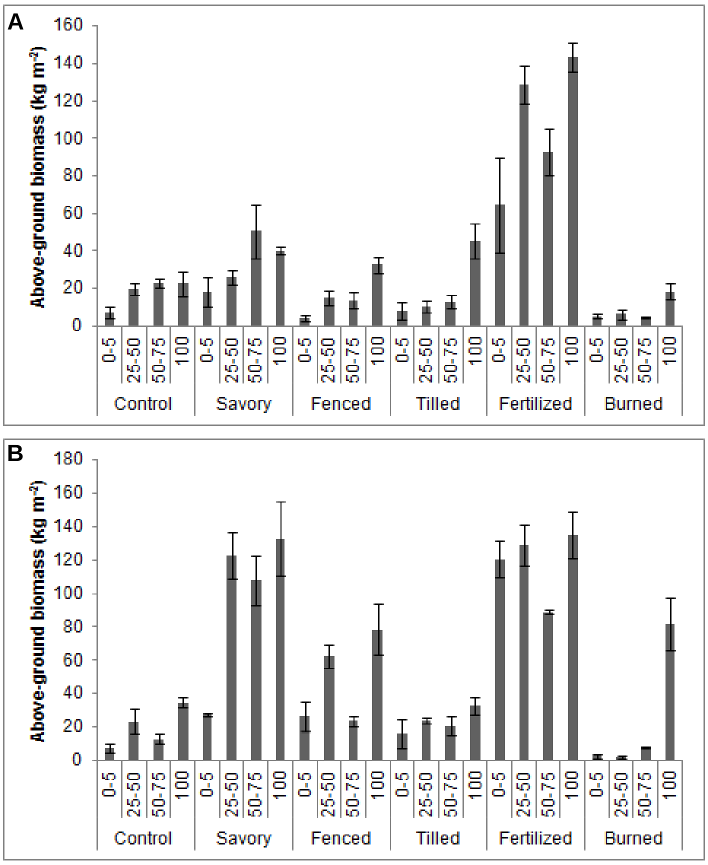


Figure 4.2 Effect of rehabilitation treatments on above-ground biomass for (A) the 2011-2012 and (B) 2012-2013 growing seasons.

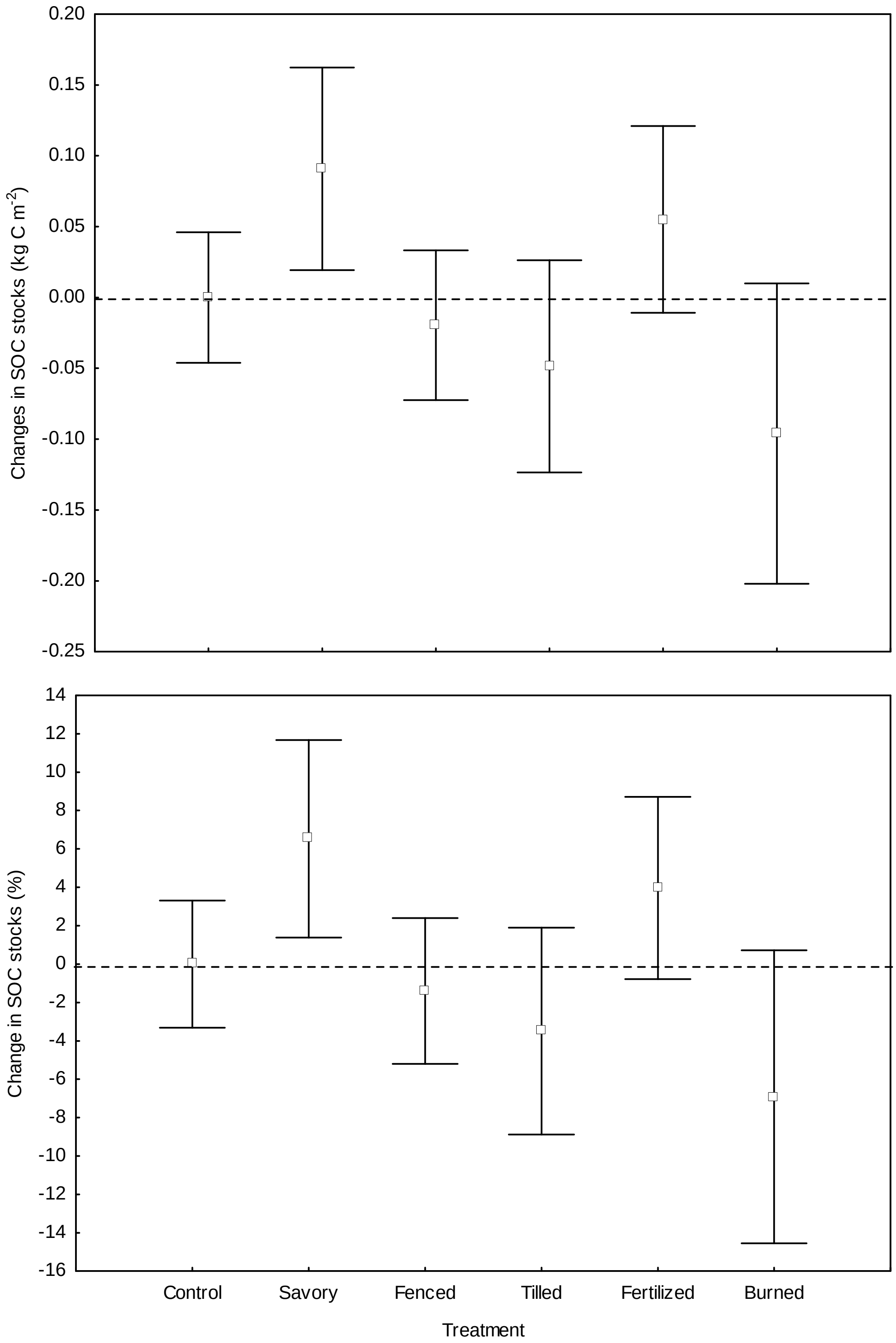


Figure 4.3 Effects of treatments on changes in SOC stocks in the 0.05 m soil layer after rehabilitation. The error bars are the standard errors of the mean.

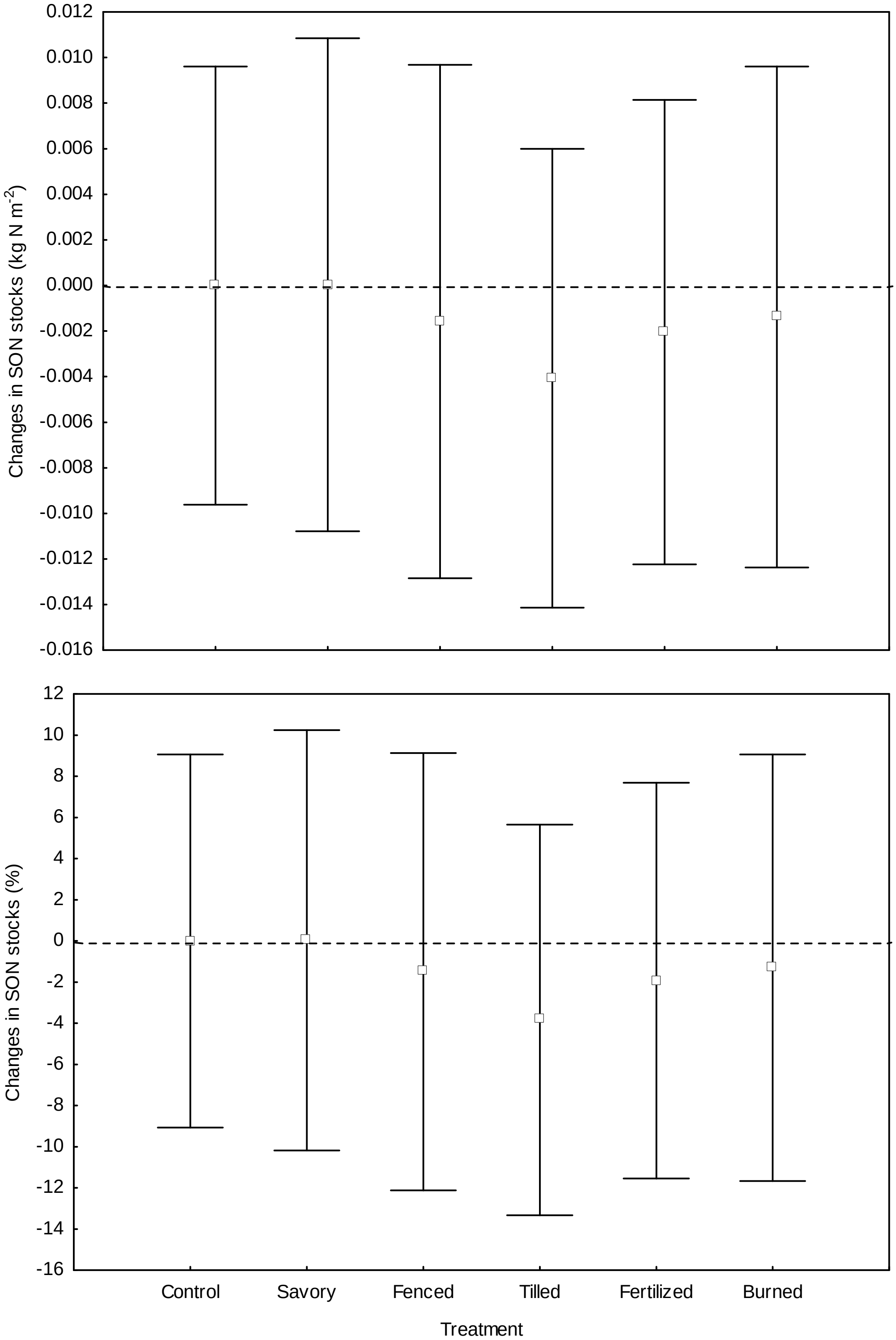


Figure 4.4 Effects of treatments on changes in SON stocks in the 0.05 m soil layer after rehabilitation. The error bars are the standard errors of the mean.

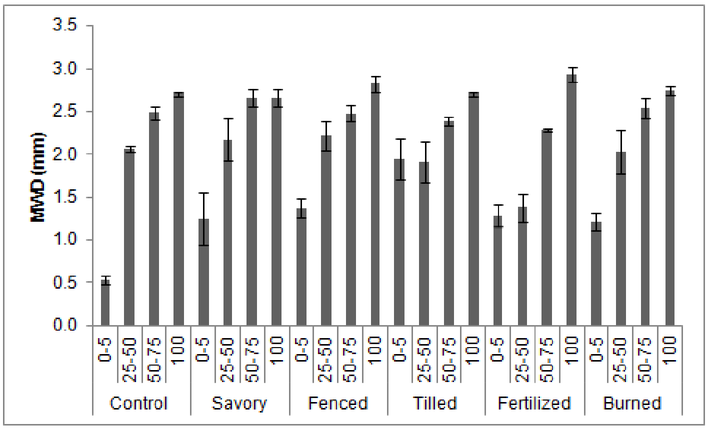


Figure 4.5 Effect of rehabilitation treatments on soil aggregate stability. The error bars are standard errors of the mean.

CHAPTER 5

5. SPATIAL VARIATION AND REPLENISHMENT POTENTIAL OF C AND N STOCKS IN A DEGRADED GRASSLAND CATCHMENT

Abstract

Grassland soils, which hold a major proportion of terrestrial carbon (C) stocks, have a potential to sequester atmospheric C and thus mitigate climate change. Despite this, it is not yet fully understood how the C pool in grassland soils is affected by degradation. Thus, the objectives of this study were: (1) to assess the spatial variation of C and N stocks, to investigate their relationship with soil type, parent material and selected terrain attributes (mean slope gradient, stream power index, compound topographic index, slope length factor and hillslope curvatures) of a typical 23 ha degraded grassland catchment in the Drakensberg foothills, South Africa and (2) to estimate the replenishment potential of heavily degraded compared with non-degraded grasslands. The topsoil (0-0.05m) of a 23 ha degraded grassland was sampled regularly at the nodes of a 20×20 m grid, resulting in 716 data points. Additional, soil samples were collected at different depths along two catenas from the 0-0.1 m surface layer to deeper soil layers, from 0-0.3, 0-0.6 and 1 m. Terrain attributes were extracted from a 5 m digital elevation model (DEM). C stocks in the 0.05 soil layer ranged between 0.28 and 2.1 kg C m^{-2}, while N stocks ranged between 0.02 and 0.19 kg N m^{-2}. Despite the weak univariate correlation between C and N stocks and selected environmental factors, a complementary principal component analysis (PCA), which explained 70% of the data variability, indicated that C and N stocks were higher at concave hillslopes, specifically in the bottomland areas of the catchment. No significant correlation was found between C stocks, soil types and parent material, but there was a tendency for C and N stocks to decrease with decreasing grass aerial cover (Cov) and increasing soil crusting (Crust). Furthermore, the average carbon replenishment potential of the degraded soils was estimated to be 4.6 t C ha^{-1}. The clay-rich red Acrisols were found to have a greater capacity to replenish C stocks compared to the sandy Luvisols and Gleysols. Both static (terrain attributes) and dynamic parameters (soil surface characteristics) might have strong repercussions on C and N stocks if environmental conditions are to become more favorable to degradation in the future. These

results could be used to digitally map the spatial variability of C and N stocks for similar degraded grasslands environments in order to develop effective rehabilitation measures.
Key words: soil organic carbon, nitrogen, grassland degradation; digital soil mapping; South Africa.

5.1 Introduction

The carbon (C) stored in grassland ecosystems account for a major proportion of the global terrestrial C stocks (Suttie et al., 2005). It is estimated that grassland soils contain approximately 10% of the global soil C stock of 1500 Gt (Lal, 2004). Historically, this C pool has largely been affected by environmental disturbances such as land degradation (Daily, 1995). Grassland soils continue to experience a dramatic decline in C and nutrient stocks due to increasing soil degradation exacerbated by poor management and anthropogenic pressures. The frequent disturbance of C and nutrient stocks by degradation is likely to affect their spatial distribution in the landscape.

Naturally, the spatial distribution of soil properties in the landscape is not uniform (Atteia and Dubois, 1994). Soil properties are highly variable, ranging from small to large scales (Cerri et al., 2004). The spatial variability of soil properties, including C and nutrient stocks is dependent on variety of factors including land use and land management (Stenger et al., 2002; Wang et al., 2010), soil type (Cerri et al., 1999), soil erosion (Chaplot et al., 2009) and pedogenic processes (Trangmar et al., 1985), which significantly result in marked spatial variation. Furthermore, soil C and nutrient stocks are highly heterogeneous in the landscape as a result of the local-scale variability in the surrounding soil environment induced by topography, stoniness, parent material (Garcia-Pausas et al., 2007). Due to the high spatial variability of C and nutrient stocks it is mandatory to consider various environmental factors to predict C accurately at catchment scale (Doetterl et al., 2013). The assessment of soil C and nutrient stocks at catchment scale is also important to provide complementary information for calibration, verification and application of soil landscape models (Thompson and Kolka, 2005; Garcia-Pausas et al., 2007).

A few studies have quantitatively investigated the link between the spatial distribution of soil properties and their surrounding environmental conditions. Chaplot et al. (2010) using 3471 soil profiles in Laos (230 566km^2) showed that C stocks were significantly affected by land use, with greater C stocks in forests and lower C stocks under shifting cultivation. Similarly, Wang et al. (2010) in a 2.02 km^2 catchment in China also found that the spatial variation of C was influenced by land use, with higher C stocks under mature forested land and grassland compared to immature forest land, orchard land and terraced cropland. Percival et al. (2000) in New Zealand grassland soils across a range of climatic conditions and soils types found C stocks were greater in allophanic soils (12.8 kg C m^{-2}) compared to semi-arid soils (3.4 kg C m^{-2}). Percival et al. (2000) found C stocks were weakly correlated to both temperature and rainfall across all soils. In addition, with regard to soil properties, they found that C stocks were weakly correlated with soil clay content. Chaplot et al. (2009) in a steep hillslope in Laos found that C stocks were significantly influenced by soil erosion, with greater C stocks at depositional sites (16.2 kg C m^{-2}) compared to eroded sites (10.9 kg C m^{-2}).

Degradation of soil is widespread in South Africa, as it is in many other dryland regions. Many studies have reported that the depletion of C in most South African soils is caused by the removal of vegetation cover through frequent burning, intensive grazing, ploughing and soil erosion (Mills and Fey., 2003; Le Roux et al., 2007 and Scholes et al., 2007). Despite the predominance of degraded grassland soils in South Africa, little quantitative information exists on the spatial variation of C and nutrient stocks affected by degradation, especially at catchment scale. In degraded grassland landscapes, C may be transported from erosional to depositional landscapes resulting in a heterogeneous pattern of the distributed C stocks (Doetterl et al., 2013). The few studies in South Africa that have attempted to quantify C stocks in degraded grassland soils have been conducted at soil profile level (Snyman and du Preez, 2005). At this scale, the effect of environmental factors such as topography is usually not considered (Doetterl et al., 2012). Yet, at the landscape level, both erosion and deposition occur and most of the C and nutrient stocks may only be translocated and redistributed within the landscape and not lost (Beuselinck et al., 2000, Steegen and Govers, 2001). Therefore, not considering the landscape scale in degradation studies severely biases all attempts to budget net soil C and nutrient losses from the soil (van Oost et al., 2000; Lal et al., 2000). Moreover, in sloping degraded landscapes, erosion and deposition processes may lead to spatially dynamic patterns of C stocks. Such environments with high soil redistribution rates induced by degradation, can exhibit a large variability in C stocks due to the burial of C rich topsoil material at foothill positions and C depletion at eroding slope positions due to the removal of C rich soil layers (Doetterl et al., 2012).

Thus, detailed and precise maps of the spatial distribution of C stocks are a prerequisite for the assessment of the soil quality, adaptation of management practices and to explain how environmental variables are linked to degradation processes affecting the distribution of C and nutrient stocks (Doetterl et al., 2013). The main objectives of this study were: (1) to assess the spatial variation of C and N stocks, to investigate their relationship with soil type, parent material and selected terrain attributes (mean slope gradient, stream power index, compound topographic index, slope length factor and hillslope curvatures) and (2) to estimate the replenishment potential of heavily degraded compared with non-degraded grasslands of a typical 23 ha communal rangeland in KwaZulu-Natal province of South Africa replete with different intensities of degradation.

5.2 Materials and methods

5.2.1 Study site description

A 23 ha grassland catchment exhibiting different grassland degradation intensities (Figure 5.2) was selected within the communal rangeland of the Potshini Catchment which is situated 10 km south of the Bergville district in the KwaZulu-Natal Province, South Africa (Long: 29.38°; Lat:28.81°). The mean annual precipitation over the past 30 years is 684mm per annum, with a potential evaporation of 1600 mm per annum and a mean annual temperature of 13^{o}C (Schulze, 1997). This landscape is characterized by a relatively gentle relief with a mean slope gradient of about 15.7 % and an altitude ranging from about 1381 to 1492 m.a.s.l. The underlying bedrock is predominantly sandstone with dolerite dolerite intrusions (Figure 5.3). Soils vary along the catchment hillslope from Gleyic Luvisols and Gleysols in the footslope, to deep red Acrisols and Luvisols midslope, and to Yellow Acrisols or Luvisols, upslope (WRB, 2006). The vegetation reflects the intensity of the grassland degradation to which most of the area has been subjected to livestock grazing.

5.2.2 Sampling strategy for C and N stocks

The soil samples for C and N content analysis were regularly selected at the nodes of a 20×20 m grid in July 2009 from the 23 ha grassland catchment. The soil samples were taken to a sampling depth of 0.05 m from the soil surface. This gave 716 observations, the positions of which are shown in Figure 5.1. Additional, but limited soil samples were collected at different depths from two catenas in the catchment. Soil samples were collected from 10 soil profiles at three geomorpholocally distinct landform positions of the catenas: footslope, midslope and upslope. From each profile, soil samples were collected from the 0-0.1 m surface layer to deeper soil layers, from 0-0.3, 0-0.6 and 1 m. In the field, the soils of two representative catenas across the study site were described pedologically (WRB, 2006). The colour of dry soil samples was determined with the Munsell colour chart. Triplicate soil bulk density samples were also collected in the different soil horizons of the soil profiles using 220.89 cm^{-3} metal cylindrical cores (height 0.05 m, diameter 0.75 m).

5.2.3 Soil chemical and physical characteristics

The soil bulk density samples were immediately oven-dried in the laboratory and gravimetric water content was determined by following Blake and Hartge, (1986). Prior to soil analyses, the soil samples were air-dried and ground to pass through a (<2 mm) sieve. The particle size distribution was determined by the pipette method (Gee and Bauder, 1986). The soil pH was measured in both distilled water and KCl using a Calimatic pHM766 meter, whereby a solution ratio of 1:2.5 (10 g soil:25 mL solution). The soils were analyzed for total C and N dry combustion using a LECO CNS-2000 Dumas analyzer (Lerco Corp, St. Joseph, MI). The P and K content were measured by extraction with strontium chloride. The concentrations of P and K were determined by inductively-coupled plasma optical emission spectrometry (ICP-OES).

5.2.4 Determination of soil organic carbon and nutrient stocks

To estimate C stocks, for the 0.05 m topsoil layer and the deeper soil layers, the C concentration is multiplied with the soil bulk density (ρ_b), and the respective thickness of the soil layer according to Batjes (1996);

$$C_s = x_1 x_2 x_3 (1 - \frac{x_4}{100}) \qquad (2)$$

where C_s is the C stock (kg C m^{-2}) of the 0-0.05 m soil layer; x_1 is the C concentration in the ≤2 mm soil material (g C kg^{-1}); x_2 is the soil bulk density (Mg m^{-3}); x_3 is the thickness of the soil layer (m); x_4 is the proportion of fragments of >2mm in percent.

5.2.5 Determination of environmental factors

Terrain attributes

To quantify the effect of the terrain attributes, a high resolution DEM with a 5-m mesh was developed for the study area in which topographical data was obtained. This DEM was generated from 50,000 data points gathered within the catchment with differential global positioning system (DGPS), with precision in Z of +/- 0.05 m, horizontal accuracy of 0.1m and vertical accuracy of

0.2, and interpolated using the inverse distance weighting (IDW) interpolation method in ArcView3.2 (ESRI, 2004). The following terrain attributes were derived from the DEM: mean slope gradient (S); aspect hillslope curvature (Curv); plan curvature ($Curv_{pl}$); profile curvature ($Curv_{pr}$); stream power index (SPI); compound topographic index (CTI) and slope length factor (SLF). The terrain attributes were estimated using a variety of spatial analyst tools such as the terrain analysis tool to determine the CTI, SPI and SLF and the DEMAT tool to determine S, Aspect and curvatures in ArcView 3.2 (ESRI, 2004). A map for each terrain attribute was generated for the catchment. According to Moore et al. (1991; 1993), terrain attributes can be classified into primary and secondary attributes. Primary attributes are directly calculated from a DEM and include variables such as elevation, slope, plan and profile curvature, slope length factor and specific catchment area. Profile curvature is a measure of the rate of change of the potential gradient and it is thus important for water flow and sediment transport processes. Plan curvature is a measure of the convergence or divergence and hence the concentration of water in the landscape (Moore et al., 1991). Slope length factor accounts for the effects of topography on erosion (Wischmeier and Smith, 1978). Secondary attributes entail combinations of the primary attributes and can be used to characterize the spatial variability of specific processes occurring in the landscape (Moore et al., 1993). Secondary attributes include the CTI or wetness index and SPI. The compound topographic index is a useful topographic variable that is a guide to water and sediment movement in landscapes (McKenzie and Ryan, 1999). It is used to characterize the spatial distribution of surface saturation zones and soil water content in landscapes (Moore et al., 1993). Stream power index is a measure of the erosive power of overland flow (Moore et al., 1991), which is likely to be a good indicator of land degradation due to water erosion.

5.2.6 Soil surface characteristics

In this study, the influence of soil surface characteristics on soil stocks was investigated by qualitatively assessing the proportion of grass aerial cover (Cov) and soil surface crusting (Crust), two indicators of grassland degradation in the region (Dlamini et al., 2011), using a 1×1m quadrats, following the method by Auzet et al. (2004). Detailed information on Cov and Crust were gathered in the study site in June 2009, one week before the soil sampling for C and N stock estimation. A total of 200 field observations were randomly selected throughout the catchment, with locations captured using the global positioning system (GPS) with a 0.2 m lateral accuracy. From the nine main types of crusts defined by Casenave and Valentin (1992), only three crust types were found at the study site: structural (rough surface made of coalescing partially slaked aggregates), erosion (smooth surface made of a single seal of fine cemented particles) and sedimentary (laminated with layers of different texture). The proportion of crusts reported here corresponded to the sum of the three crust types. GIS layers for Crust and Cov were then generated from data points by interpolating using the ordinary kriging function of ArcMap (ESRI, 2004), a function adapted for soil properties and lower sampling densities (McBratney and Webster, 1983).

5.2.7 Determining the spatial structure of data

Variograms were computed to determine the spatial scale and strength of C and N pattern across the 23ha catchment. A variogram is a geostatistical characteristic used to describe the spatial structure or spatial dependence of a data set (Atkinson and Tate, 2000; Kravchenko, 2003). The variogram is an integral geostatistical tool because it reveals the nature or pattern of topographic variation of a soil property across a landscape, by quantifying the scale and intensity of the spatial variation. Furthermore, it provides the essential spatial information for optimal estimation and interpolation, and can also be used for optimizing sampling schemes (Burgess and Webster, 1980; Oliver, 1987; Oliver et al., 1989). Using the variogram, the spatial structure can be described by fitting observation data to a model. The shape of the variogram gives an indication of the spatial structure of the soil property (Warrick and Nielsen, 1980).

The first step in assessing the spatial distribution for C and nutrient stocks was to determine whether there was a significant spatial structure (Goovaerts, 1999). In order to detect the random function that caused the spatial structure of the observations, an appropriate model was fitted to the experimental variograms using routines from a standard computer program, GS+7.0 geostatistical software (Robertson, 2007). The data was fitted to permissible variogram models commonly used in practice such as the spherical, exponential, Gaussian which show the spatial correlation of the data, and the linear variogram model which generally indicate drift in the data (Gassner and Schnug, 2006). An isotropic distribution was selected and the parameters of the directional variogram were determined. The best fitted model from the geostatistical analysis of the data was used. Because soil properties do not vary isotropically in the landscape, to detect anisotropy, variograms were analysed for four directions of 0^o, 45^o, 90^o and 135^o. Oliver et al. (1989) recommends that in order to detect directional differences or anisotropy the variogram should be estimated in at least three directions.

The variogram parameters for characterizing the spatial structure are the nugget variance, sill, nugget/sill (N/S) ratio and spatial correlation range. It is important to differentiate between these parameters. The nugget variance represents the random variable of the data. Firstly, from the measurement error, and secondly from the spatial variability at distances smaller than the shortest sampling interval. The sill is the plateau which every bounded model will reach. The N/S ratio is defined as the proportion of short-range variability that cannot be described by a geostatistical model based on the isotropic variogram (Kravchenko, 2003). The ratio of nugget semi-variance to sill semi-variance (N/S) was calculated from the sill and nugget values to determine the spatial dependence within the data. If $N/S < 25\%$, the variable is considered strongly spatially dependent; if $25\% < N/S < 75\%$, the variable is considered moderately spatially dependent and if $N/S > 75\%$, the variable is considered weakly spatially dependent (Kravchenko, 2003). The range is the distance of spatial autocorrelation at which the model reaches the sill or plateau (Gassner and Schnug, 2006). It is an indication of the distance over which soil property data points are spatially depended on each other (Kravchenko, 2003; Gassner and Schnug, 2006). Generally, a large spatial correlation range and small N/S ratios indicate that great accuracy can be achieved in mapping the variable of interest (Isaaks and Srivastava, 1989). In this study, prediction accuracy of various interpolation techniques was tested and the strongest interpolation techniques were used to map the spatial distribution of C and N across the 23 ha grassland catchment. Maps showing the spatial distribution of C and N were generated using ArcView GIS 3.2 (ESRI, 2004).

5.2.8 Replenishment potential of heavily degraded compared with non-degraded soils

Following the results of the spatial variation of C and N stocks estimation, we aggregated the soil C and N stocks derived from heavily degraded and non-degraded grassland soils to estimate the replenishment potential. This was done by taking the maximum C and N stock values at non-degraded and the lowest values at heavily degraded grassland soils (n = 612 data points). This information was further used to estimate the replenishment potential of C and N stocks in the catchment.

5.2.9 Statistical analysis

A correlation matrix was generated to identify univariate relationship between C and N stocks and the selected environmental factors. A multivariate analysis was applied to the data to find relationships between C and N stocks, soil characteristics such as clay content, as well as environmental characteristics such as the mean slope gradient, compound topographic index, and stream power index. A principal component analysis was used as it has been previously shown to be well adapted to large sets of variables and to identify the structure or dependence in data sets (Webster, 2001). The PCA converted the connectivities into the so-called factors or principal components, which together explained the total variance of the data (Jambu, 1991). In this multivariate statistical tool, the first and second factors often explain most of the variance and therefore most of the information contained in the data. The ADE4 software (Chessel et al., 2004) was used for this study.

5.3 Results

5.3.1 Description of soil variations along the catena's

The soils have been mapped in detail in the 23 ha grassland catchment using catenas and extensive soil surveys (Figure 5.4; Tables 5.1 and 5.2). Luvisols are the dominant soil type, found mainly in the backslope and footslope position of the catchment. The profiles on these Luvisols exhibited dark brown (7.5YR 4/5) A horizon, with fine angular blocky structure underlain by reddish brown (2.5YR 3/3) subsoil horizon, with medium blocky structure due to the intrusion of dolerite in this area and a high clay content (>30%). The footslopes are characterized by deep (>1.5m) Gleysols and Gleyic Luvisols characterized by dark brown (7.5 YR 3/2) A horizon, dark grey (7.5 YR 4/1) B horizon, with a high sand content (>40%) and redoximorphic features evident in the entire soil profile (Table 5.1and 5.2). There is a transition for a few meters (~10 m) to Acrisols at footslope, which are the second dominant soil type. They are characterized by dark brown (7.5YR 4/6) topsoil horizon, with better drainage conditions. Yellow Acrisols and Luvisols are found in the upslope position of the hillslope. These soils upslope of the site are shallow, characterized by dark brown (7.5 YR 4/4) A horizon, with medium angular blocky structure, a yellowish red (5YR 3/3) B horizon, and have a high clay content of 30%. From the two described catenas it was observed that soil depth increases down the hillslope.

5.3.2 Descriptive statistics of C and N in the 0-05 m soil layer

Soil carbon stocks in the 0.05 m soil layer across the 23 ha grassland catchment ranged between 0.28 and 2.1 kg C m^{-2}, with an average of 1.2 kg C m^{-2} and a coefficient of variation (CV) of 27.7%. Nitrogen stocks ranged between 0.02 and 0.19 kg N m^{-2}, with an average of 0.08 kg N m^{-2} and a CV of 27.8%. Both C and N stocks were found to be positively skewed, with positive skewness coefficients of 0.03 and 0.35, respectively (Table 5.3).

5.3.3 Variation of C and N with soil depth

Soil organic carbon content decreased with depth at all sampled soil profiles, from an average of 19.4 g C kg^{-1} at 0.10 m to 14.8 g C kg^{-1} at 0.3 m and 9.1 g C kg^{-1} at 0.6 m (Table 5.1 and 5.2; Figure 5.4). Similarly, nitrogen content decreased with depth from 1.36 g N kg^{-1} at 0.10 m to 1.03 g N kg^{-1} at 0.3 m and 0.72 g C kg^{-1} at 0.6 m. This pattern was consistent for the thirteen soil profiles along the two catena's. The soil bulk density varied significantly with depth from the topsoil layer to the deeper soil layers, rendering the C and N stocks to also vary with depth. C stocks in the 0.10 m surface layer ranged between 0.77 kg C m^{-2} and 3.52 kg C m^{-2}, with an average of 2.19 kg C m^{-2}. In the 0.3 m soil layer, the C stocks ranged between 1.31 kg C m^{-2} and 4.97 kg C m^{-2}, with an average of 3.05 kg C m^{-2}. In the 0.6 m soil layer, the C stocks ranged between 1.01 kg C m^{-2} and 5.27 kg C m^{-2}, with an average of 2.36 kg C m^{-2}. N stocks in the 0.10 m surface layer ranged between 0.08 kg N m^{-2} and 0.24 kg N m^{-2}, with an average of 0.16 kg N m^{-2}. In the 0.3 m soil layer, N stocks ranged between 0.1 kg N m^{-2} and 0.33 kg N m^{-2}, with an average of 0.21 kg N m^{-2}. In the 0.6 m soil layer, N stocks ranged between 0.07 kg N m^{-2} and 0.32 kg N m^{-2}, with an average of 0.19 kg N m^{-2}.

5.3.4 Interpretation of spatial structure and variability

To determine the spatial pattern of C and N across the 23 ha catchment, directional variograms of C and N were computed using a generation set of 646 data points (Figure 5.5). The isotropic variogram for C displays an increase in semi-variance from 0.5 to 0.7, with an increasing distance from 25 m to about 230 m. It plateaus after 230 m, which marks the limit of the spatial dependence. There was a marked anisotropy which was greatest between the 0^o and 135^o direction. The N/S ratio was 10.3, which indicates a moderate spatial dependent structure. The isotropic variogram for N displays an increase in semi-variance from 0.0.0024 to 0.0028, with an increasing distance from 25 m to about 100 m, which tails of gradually to 300 m marking the limit of the spatial dependence. There was a marked anisotropy which was greatest between the 0^o and 135^o direction. The N/S ratio was 10.3, which indicates a moderate spatial dependent structure. This implies that C and N stocks under degraded conditions in the catchment have a significant spatial autocorrelation or dependency (the tendency or likelihood that sampled points at neighbouring locations in space are much more similar to one another than those further apart) up to a distance of 150 m, and beyond this distance C and N do not depict a significant spatial autocorrelation.

As is evident in Figure 5.6, the spatial variability of C stocks in the 23 ha grassland catchment is high ranging between 0.35 kg C m^{-2} and 3.04 kg C m^{-2}, with an average of 1.20 kg C m^{-2}. Similarly, N stocks varied greatly between 0.03 kg N m^{-2} and 0.21 kg N m^{-2}, with an average of 0.08 kg N m^{-2}. This can be attributed to the landscape being heterogeneous as result of the varying intensities of degradation due to the management practices such as cattle pathways evident in Figure 5.2.

Beyond analyzing the spatial variability of C and N stocks in the grassland catchment, an effort was made to estimate the replenishment potential of the soils. Figure 5.6 show the C and N replenishment potential, calculated by aggregating the maximum C and N values for non-degraded soils and subtracting the minimum values gathered from the heavily degraded soils. The C replenishment potential of the degraded soils in the catchment ranged between 0.17 t C ha^{-1} (1%) and 12.8 kg t C ha^{-1} (78.3%), with an average of 4.6 kg t C ha^{-1} (28.6%). Higher C replenishment potential was estimated for the red Acrisols, while a lower potential was estimated for Luvisols and Gleysols (Table 5.5). The N replenishment potential of the degraded soils ranged between 0.0014 t C ha^{-1} (0.8%) and 0.9 t C ha^{-1} (78.6%), with an average of 0.3 kg t C ha^{-1}(31.5%). No significant difference in the N replenishment potential was observed for the different soil types.

5.3.5 Correlation between soil stocks and selected environmental factors

The spatial distribution of C and N stocks was related to parent material, soil type, soil surface characteristics and selected environmental factors seeking possible explanations of the variation. Generally, the r coefficients are relatively low ranging between 0 and 0.14 (Table 5.4). However, owing to the large number of observation data points used in this study (n=716), C stocks were significantly correlated with SPI (r=0.10), aspect (r=0.09), and SLF (r=0.09), while N stocks significantly correlated with S (r=0.15), SPI (0.13) and SLF (r=0.14).

Principal component analysis (PCA) was applied to the data to further examine the relationship between soil stocks and environmental factors (Figure 5.7). Two PCAs were generated from auxiliary variables of Cov, Crust, S, SPI, CTI, Curv, and SLF. The first PCA (PCA1) included all the 716 data points. Its two first axes explained 60% of the data variation (Figure 5.7A). The first axis which accounts for 41% of the total data variation showed a trend associated with SLF, Crust and Cov. Crust was negatively correlated to axis1 (x= -0.89), while Cov was positively correlated (x=0.89). Axis 1 can be interpreted as an axis of land degradation. Topographic variables, SPI and SFL were negatively correlated with axis 2 (y= -0.76 and -0.69, respectively), which accounts for 19% of data variance, while slope curvature was positively correlated (r=0.70). Axis 2 discriminates between hillslope positions; the lower hillslope positions of high SPI and SLF and low Curv which is opposed to the higher hillslope positions. As shown by Figure 5.7A, higher C and nutrient stocks were found at lower hillslope positions. In summary, it can be concluded from this multivariate analysis that higher C and N stocks in the catchment are found in bottomland areas under waterlogged conditions which are characterised by high soil surface coverage.

To exclude the effect of waterlogging conditions and to focus on other processes and factors that might influence the spatial variation of C and N stocks across the catchment, another PCA was constructed which excluded soils with redoximorphic features (Figure 5.7B). The first axis of PCA (PCA 2) accounting for 47% of the data variance showed a trend associated with slope gradient and soil surface coverage by vegetation. The second axis of PCA 2 which accounts for 20% of data variance showed a trend associated with hillslope curvatures. The weakly convex and strongly concave areas are characterized by thicker soils, which is evident in the soils found in the footslope of the catchment. Soil stocks of C and N, correlated with axis 1. There was, however, a slight decrease of C and N stocks as Cov decreased and Crust increased. Finally, both the soil type and the bedrock type (Figure 5.8) had no significant (P<0.05) impact on both C and N stocks. Overall, what is evident from the multivariate analysis is that there is no single environmental factor which seems to override or transcend the other as the main controlling factor, thus the factors cannot be considered in isolation as they are linked with one another. No one soil forming factor acts in isolation although locally one may exert a stronger influence (Jenny, 1941).

5.4 Discussion

5.4.1 Characteristics of C stocks

The average C stock of 1.2 kg C m^{-2} found in our study site was of similar magnitude as the C stocks of 1.3 kg C m^{-2} reported by Mills and Fey (2004) in neighbouring grassland area of South Africa, but an order of magnitude lower than the C stock values reported by Leifeld et al. (7.5 kg C m^{-2}) in Swiss alpine grasslands, by Rodriguez-Murillo (2001) in grasslands of Spain (7.3 kg C m^{-2}), Townsend et al. (1995) in the tropical grasslands of Hawaii (14 kg C m^{-2}), by Jobbàgy and Jackson (2000) in temperate grasslands (11.7 kg C m^{-2}), by Tate et al. (2000) in pastures of New Zealand (20.0 kg C m^{-2}), and the average C stock of 10.6 kg C m^{-2} reported by Batjes (1996) for global soils. The lower C stocks in this study may be attributable to the acidic ($3<pH<5$), sandy (>60%) soil conditions, steep slope conditions (15.7%) and soil losses of up to 13 t ha^{-1} y^{-1} induced by soil erosion (Dlamini et al., 2011) and 2.30 t ha t ha^{-1} y^{-1} induced by gulley erosion (Chaplot, 2013).

5.4.2 Spatial variation of C and N

Using the best interpolation technique, maps showing the distribution of C and N stocks across the 23 ha grassland catchment were produced (Figure 6). It is evident from Fig 6 that the spatial dependence in the variogram arises because there are patches in the catchment where C and N stocks are high and other areas where the stocks are low. Seemingly, the spatial variability of both C and N stocks across the catchment is not purely random. Indeed, the C stocks are spatially highly variable across the catchment. High C stocks are found in the lower positions of the catchment (valley bottom) where waterlogged conditions are strongly expressed, and also restricted to crest locations at the midslope of the catchment. Lower C stocks occurred in the steep slope positions in the east-west direction of the catchment.

The higher C and N stocks at lower hillslope positions might be explained by a combination of two mechanisms: (1). Preferential detachment and downslope transport of the more labile, and lighter, SOC fractions which are typically enriched in SOC relative to the bulk soil (Gregorich et

al., 1998) and soil erosion, which redistributes the uppermost nutrient-rich material and accumulates it down the hillslope.
(2) hydromorphic conditions of waterlogged soils on the footslope restricts the decomposition of organic matter due to the lack of oxygen for soil micro-organisms, thus accumulating nutrient rich organic material.

5.4.3 Controlling environmental factors

From the correlation matrix, it is evident that some of the topographic variables are significantly correlated with C and N stocks. The low correlation coefficients, especially between terrain attributes and soil stocks were unexpected. Thompson et al. (1997) in Minesota found soil C stocks to be significantly correlated to slope gradient (r=-0.50) and distance to local depression (r=-0.56). Under a temperate climate and gentle slope conditions, Chaplot et al. (2001) found r coefficients between topographic factors and soil C stocks of r=0.08 for slope curvature and r=0.89 for elevation above the bottomland. Slope gradient was also found to be the main controlling factor of C stocks in the silty alluvial deposits of South West France (Arrouays et al., 1999).

The significant correlations between soil stocks and SPI, which is a measure of the erosive power of overland flow (Moore et al., 1991) and SLF which accounts for the effects of topography on erosion (Wischmeier and Smith, 1978) confirms the relevance of mechanism 1. Similar studies carried out in other parts of the world (i.e. Thompson et al., 1997; Chaplot et al., 2009) found that C and N stocks increase downslope. The second mechanism is confirmed by apparent lack of relationship between soil stocks and terrain attributes when the multivariate analysis was performed by excluding waterlogged soils. Based on this and field observations, it seems that both sets of processes are operative across the catchment, but to differing degrees leading to the high spatial variability of SOC stocks induced by degradation.

Following the PCA analysis, higher C and N stocks are found at concave areas of the catchment which correspond to the bottomland, while lower stocks were encountered at convex areas. Lower C and N stocks were found at higher curvatures which corresponded to midslope convex areas. In general, stronger convex areas have thin soils (Heimsath et al., 1997) because of greater soil erodibility, which has been confirmed at the study site using in-situ measurements (Dlamini et al., 2011). Overall, 40% of the variability of C and N stocks remained unexplained.

In our study, there was no relationship between C stocks and soil clay content. This is consistent with the results presented by Percival et al. (2000) for temperate grassland soils in New Zealand. Clay content does not appear to be the only factor responsible for the stabilization of C in soils. It is suggested that the stability of C in the soil could rather be linked to soil mineralogy than clay content only. Some studies have elucidated positive relationship between soil C and clay content, especially when other factors such as climate, vegetation and hydrology are similar (Davidson, 1995). This has been attributed to greater C stabilization effect of clay content as a result of the presence of a large specific surface area capable of forming stable organo-mineral complexes (Feller and Beare, 1997; Six et al., 2000).

In the study site, C stocks were found to slightly vary with soil type, dolerite derived soils having greater stocks than sandstone derived ones. This finding even though was less apparent is consistent with what has been reported elsewhere in literature. For example, Mills and Fey (2004) in a South African grassland found that stocks of soil C in dolerite-derived soils characterized by reddish colours were greater (16.4 kg C m^{-2}) than in sandstone-derived soils (9.7 kg C m^{-2}) characterized by greyish colours in the top 0.5 m layer of soil. In two grassland sites with two different geologies in Germany, Don et al. (2007) found that higher C stocks of 2.9 kg C m^{-2} in the 0-0.05 m soil layer of dolerite derived soils (clay rich site) compared to 1.6 kg C m^{-2} in sandstone derived soils (sandy site). The higher C stocks in the clayey site were attributed to C stabilization by clay (Six et al., 2002). While our observations were consistent with what has been found in other studies, the differences between soil stocks from the two geologies were not significant. The soils in the study site overlie rocks of varied geological substrate ranging from sandstone to dolerite intrusions. However, there is no clear limit between the two geologies, with the dolerite intrusions evident in some parts of the catchment. The unexplained variability of high stocks in other areas of the catchment besides the bottomlands and waterlogged conditions may probably be explained by the intrusion of the dolerite in those areas. In South Africa, dolerite is one of the many common basic rocks, which contain a high percentage of Ca- and Mg-rich primary minerals (Bühmann et al., 2004). Soil mineralogy is important in determining the quantity of C stored in the soil (Torn et al., 1997). Dolerite derived soils tend to have greater sesquioxide content than sedimentary rocks (Percival et al., 2000), which plays a stabilizing role, hence the greater C stocks under dolerite derived soils.

Finally, considering the spatial extent of the various soil types along the hillslope of the 23 ha catchment, the obtained data show that the heavily degraded soils have a high replenishment potential. The C replenishment potential for the heavily degraded grassland soils was 4.6 t C ha^{-1}. The main environmental controls that can contribute to the replenishment of C and N stocks include increasing vegetation cover to reduce soil surface crusting, soil and C losses.

5.5 Conclusion

In this study of a 23 ha degraded grassland catchment in South Africa the main objective was to assess the spatial variability of C and N stocks and to find the link with several terrain attributes and soil properties including soil surface crusting and vegetation cover, two indicators of land degradation in the region. Three main conclusions can be drawn from this study:

(1) C and N stocks were highly variable at the catchment level (CV of 27.7% for C and 27.8% for N stocks);

(2) The spatial distribution of C and N stocks were associated primarily to terrain morphology and secondary to land degradation. Soil stocks exhibited low univariate correlations with the terrain attributes, but a higher correlation was found with a bean of attributes, with higher stocks being found in the concave bottomlands while lower stocks corresponded to higher landscape convexities;

(3) Surprisingly, there was no significant relationship found between C, N stocks and soil clay content, soils type and geological bedrock;

(4) The study demonstrated that carbon replenishment potential of the degraded soils was 4.6 t C ha^{-1}. The clay-rich red Acrisols were found to have a greater capacity to replenish C stocks compared to the sandy Luvisols and Gleysols.

More needs to be done to understand the physical and biogeochemical processes controlling soil C and nutrients stocks. Overall, the spatial patterns of C and N stocks at the landscape level was shown to be controlled by factors which affect the movement of water and redistribution of soil material movements in sloping landscapes. This suggests the strong impact of soil redistribution by water erosion and waterlogging mechanisms on the spatial distribution of soil nutrients, two mechanisms overriding the expression of the other classical soil-forming factors of the Jenny model.

The impact of grassland degradation on C and nutrients stocks requires as well further investigation. Owing to that 81% of the C stocks in the first meter of the soil are found in the 0-0.3 m layer, any increase in soil degradation puts the stock inventories at risk. Finally, more is to be done to integrate this knowledge into digital soil mapping: (1) on the integration of both static (such as topography) and dynamic (e.g., vegetation cover; soil crusting) environmental factors; (2) for large scale prediction. Despite the high possibilities of extrapolating this knowledge to the whole Drakensberg foothills region (similar terrain morphology, land use and soils), the contribution of other environmental factors like climate needs to be explored. Overall, such an understanding of the spatial variability of C and nutrient stocks is expected to drive policies in order to develop the best conservation strategies, so as to effectively manage the soil and receiving water resources while fostering ecological functioning of landscapes.

Table 5.1 Some chemical and physical characteristics of soil horizons for catena 1 at the site.

Profile	Horizons	Depth	pH		BD	Sand	Silt	Clay	C	N	C/N	Cs	Ns	Matrix colour
		cm	H_2O	KCl	g cm^{-3}	---------%---------			-----g kg^{-1}------			--kg m^{-2}--		
1A							**Footslope**							
	1	0-10	4.5	5.5	1.23	50	22	28	14.8	1.14	13	1.82	0.14	5YR 3/3
	2	10-25	4.3	5.6	1.27	48	22	30	12.0	0.91	13	2.29	0.17	5YR 4/3
	3	25-40	4.3	5.7	1.30	48	20	32	8.9	0.78	11	1.74	0.15	5YR 4/4
	4	40-55	4.3	5.7		47	18	35	4.9	0.43	11			2.5YR 3/4
	5	55-90	4.1	5.6		46	17	37	2.6	0.33	8			2.5YR3/5
	6	90-170	4.0	5.7		41	18	40		0.11				5YR 6/8
	7	170-220	4.0	5.7		41	15	44		0.07				
	8	220-285	4.3	6.1		40	14	47						
1B							**Midslope**							
	10	0-14	4.5	5.7	1.19	53	20	27	12.1	0.95	13	2.02	0.16	5YR 3/3
	11	14-30	4.2	5.5	1.27	49	17	34	10.0	0.64	16	2.03	0.13	5YR 3/4
	12	30-46	4.2	5.5	1.21	47	16	37	10.4	0.75	14	2.01	0.15	5YR 3/4
	13	46-60	4.1	5.4	1.23	44	16	40		0.31				2.5YR 4/6
	14	60-90	4.0	5.4	1.23	40	16	43	1.6	0.49	3	0.59	0.18	2.5YR4/6
	15	90-120	4.0	5.4		40	18	42	0.2	0.14	1			2.5YR4/8
1C							**Midslope**							
	16	0 -10	4.9	5.9	0.96	31	29	40	28.4	2.09	14	2.73	0.20	5YR 3/3
	17	10-20	5.0	6.1	1.10	33	24	43	22.6	1.52	15	4.97	0.33	2.5YR 3/3
	18	20-45	4.7	5.9		34	26	40	17.3	1.16	15			2.5YR 3/4
	19	45-170	4.8	6.0		29	17	54	0.8	0.35	2			2.5YR 4/6
	20	170+	4.3	6.5		65	20	15						10YR 6/8
1D							**Upslope**							
	21	0 -10	4.1	5.4	0.96	18	29	54	36.7	2.25	16	3.52	0.22	5YR 3/4
	22	10-25	4.2	5.8	0.94	19	24	56						2.5YR 3/6
	23	25-50	4.2	5.1	0.88	19	21	60	9.7	0.51	19	2.13	0.11	2.5YR 4/6
	24	50-140	4.1	5.2	0.89	13	22	64	1.0	0.22	5	0.80	0.18	2.5YR 4/6
1E							**Upslope**							
	25	0-10	4.0	4.9	1.15	50	19	30	17.7	1.16	15	2.04	0.13	7.5YR 4/4
	26	10-25	4.1	4.9	1.18	46	19	35	12.6	0.78	16	2.23	0.14	7.5YR 4/5
	27	25-40	4.2	5.0	1.09	49	16	35	7.9	0.54	15	1.29	0.09	7.5YR 4/6
	28	40-60	4.1	5.0		46	15	39	3.7	0.79	5			5YR 5/7
	29	60-85	4.0	5.0		41	20	39	1.6	0.8	2			2.5YR 4/8

Table 5.2 Some chemical and physical characteristics of soil horizons for catena 2 at the site.

Profile	Horizons	Depth	pH		BD	Sand	Silt	Clay	C	N	C:N	Cs	Ns	Matrix colour
		cm	H_2O	KCl	g cm^{-3}	----------%----------			--g kg^{-1}--			--kg m^{-2}--		
2A							**Footslope**							
	1	0 -10	5.7	4.0	1.30	52	21	27	9.8	0.7	14	1.27	0.09	5YR 4/1
	2	10-24	5.9	4.4	1.33	55	24	22	22.8	1.4	16	4.25	0.26	7.5YR 3/3
	3	24-60	6.1	4.7	1.54	51	27	21	9.5	0.5	19	5.27	0.28	7.5YR 3/3
2B							**Midslope**							
	4	0-10	5.8	4.5	1.18	45	22	33	15.4	1.0	15	1.82	0.12	7.5YR 3/2
	5	10-30	5.9	4.3	1.21	52	25	23	12.5	0.8	16	4.54	0.29	5YR 5/6
	6	30-40	5.7	4.3	1.21	43	21	37	9.2	0.6	15	1.11	0.07	5YR 5/6
	7	40-80	5.8	4.3	1.11	40	21	39	5.7	0.5	11	2.53	0.22	7.5YR 4/6
2C							**Midslope**							
	8	0-10	5.6	4.3	1.14	29	18	54	18.2	1.3	14	2.07	0.15	7.5YR 3/3
	9	10-30	4.8	4.3	1.08	28	20	52	9.6	0.8	12	3.11	0.26	7.5YR 3/2
	10	30-70	5.1	4.1	1.08	26	23	51	5.2	0.5	10	2.25	0.22	2.5YR 3/4
	11	70-140	5.6	4.1	1.42	15	43	42	2.7	0.5	5	2.68	0.50	2.5YR 4/6
2D							**Midslope**							
	12	0-10	5.5	4.5	1.33	41	28	31	20.6	1.4	15	2.74	0.19	7.5YR 3/2
	13	10-22	6.3	4.6	1.23	38	24	38	8.9	0.7	13	1.31	0.10	7.5YR 4/6
	14	22-54	6.2	4.5	1.31	36	26	39	8.4	0.7	12	3.52	0.29	2.5YR 3/4
	15	54-80	5.9	4.7	1.30	34	29	37	5.6	0.5	11	1.89	0.17	2.5YR 4/6
2E							**Upslope**							
	16	0-10	5.2	4.0	1.17	15	35	49	6.6	0.7	9	0.77	0.08	2.5YR 4/6
	17	10-32	4.6	4.0	1.09	26	25	50	16.2	1.2	14	3.88	0.29	2.5YR 4/6
	18	32-54	4.7	4.0	0.94	31	28	42	4.9	0.6	8	1.01	0.12	2.5YR 4/8

Table 5.3 Summary statistics of the soil carbon content (C), nitrogen content (N), carbon stocks (C_S) and nitrogen stocks (N_S) in the 23 ha catchment of higher sampling density (646 points).

	C	N	C_S	N_S
	---g kg^{-1}---		---kg m^{-2}---	
Minimum	6.6	0.6	0.28	0.02
Maximum	72.4	5.2	2.1	0.19
Mean	29	2	1.2	0.08
Median	28.5	1.9	1.2	0.08
Variance	56	0.3	0.11	0.00053
Standard deviation	7.5	0.5	0.33	0.02
Skewness	0.5	0.8	0.03	0.35
Kurtosis	2.7	3.3	0.26	0.99
CV	25.8	25.4	27.7	27.8
SE	0.3	0	0.01	0.001

Table 5.4 Correlation matrix of soil carbon stocks (C_S); nitrogen stocks (N_S); and SL, estimated yearly soil losses (SL) and selected environmental factors: grass aerial cover (Cov); proportion of soil surface covered by crusts (Crust); mean slope gradient (S); slope length factor (SLF); Aspect (Asp); Stream Power Index (SPI); Compound Topographic Index (CTI); tangential curvature (Curvt), plan curvatures (Curvpl); and profile curvature (Curvpr), for 646 observation data points.

	SL	Cov	Crust	S	SLF	Asp	SPI	CTI	$Curv_t$	$Curv_{pl}$	$Curv_{pr}$
C_S	-0.12*	0.12*	-0.12*	0.06	0.09*	0.09*	0.10*	0.00	-0.05	-0.05	0.03
N_S	-0.24*	0.24*	-0.24*	-0.15*	0.14*	-0.02	0.13	-0.09	-0.05	-0.05	0.04

* Significant correlation at $P < 0.05$.

Table 5.5 Changes in carbon stocks (C_S) and nitrogen stocks (N_S) in the 0.05 m soil layer of non-degraded versus heavily degraded soils for the different soil types at the 23 ha grassland catchment.

Soil type	Non-degraded	Heavily degraded	Non-degraded	Heavily degraded
	C_S		**N_S**	
	---------kg C m^{-2}---------		----------kg N m^{-2}---------	
Red Yellow Livisols	1.64	0.85	0.12	0.06
Red Luvisols	1.50	0.69	0.11	0.05
Yellow Luvisols	1.53	0.67	0.11	0.05
Gleyic Luvisols	1.64	0.80	0.12	0.05
Yellow Acrisols	1.64	0.81	0.11	0.05
Red Acrisols	1.82	1.16	0.12	0.07
Gleysols	1.52	1.11	0.10	0.07

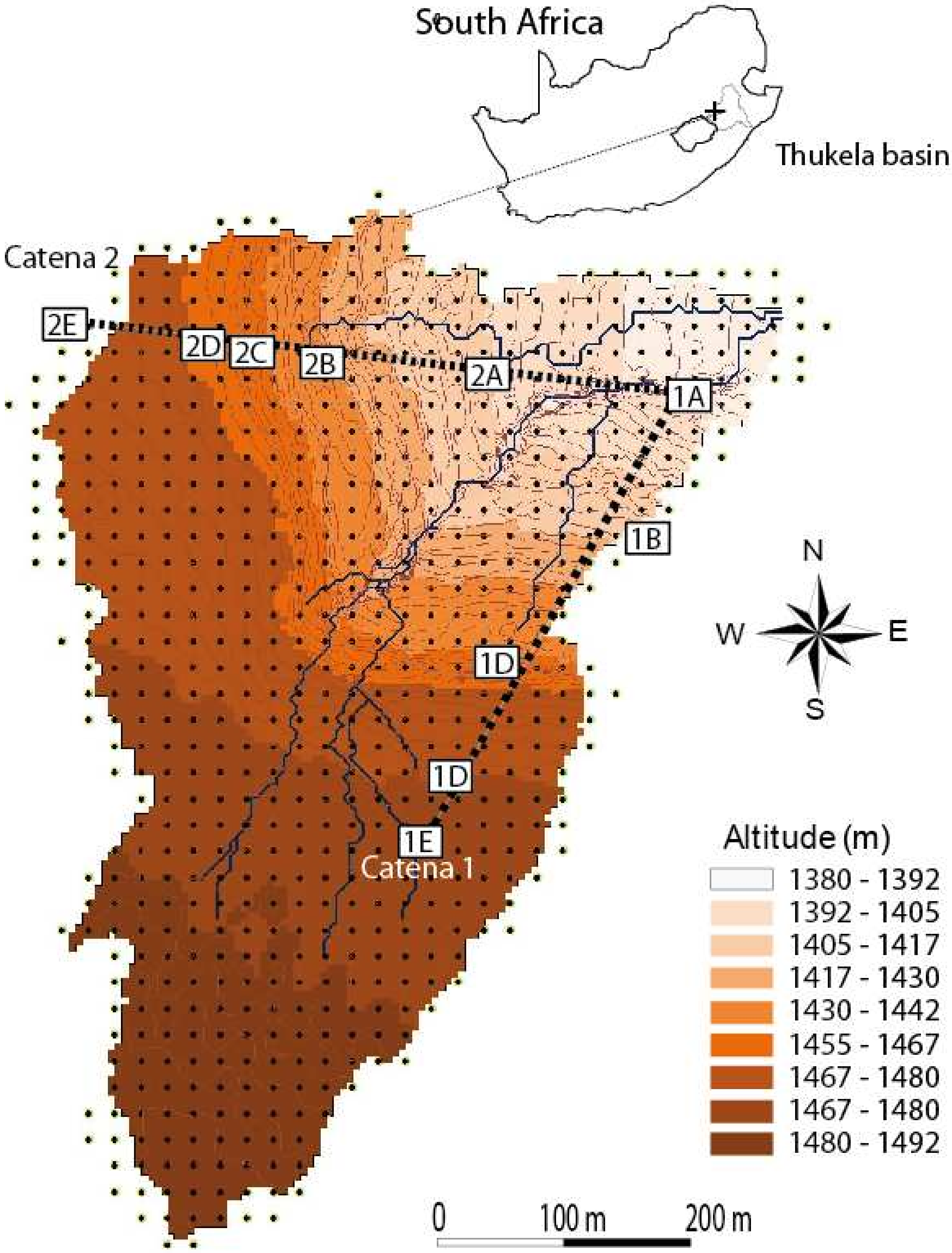


Figure 5.1 Location of the 23 ha grassland catchment, DEM with a 5m mesh size, contour lines at 2m intervals, position of the two catenas and sampling points of soil profiles.

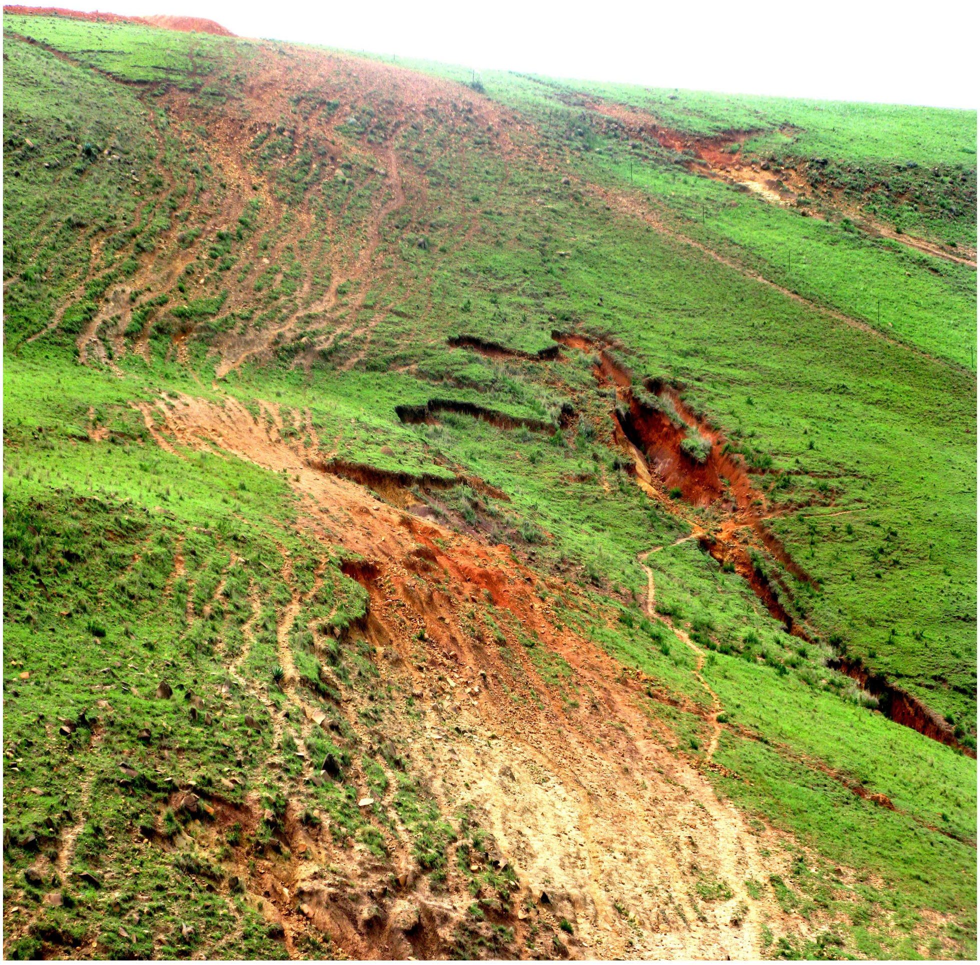


Figure 5.2 Land degradation features in the 23 ha grassland catchment.

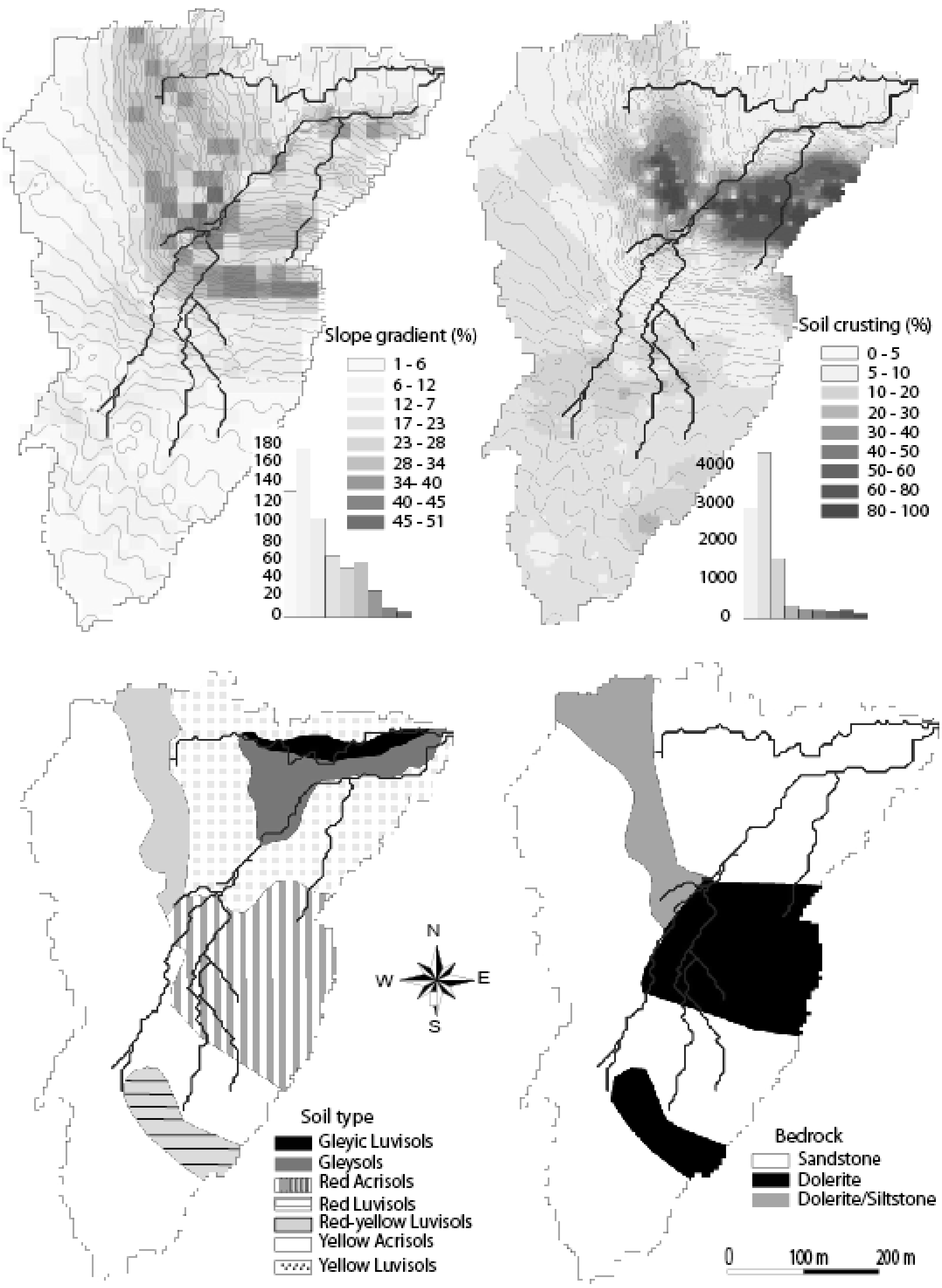


Figure 5.3 Spatial distribution of slope gradient, the proportion of the soil surface with crusts, soil type and geological bedrock. Kriged maps interpolated using georeferenced field observations.

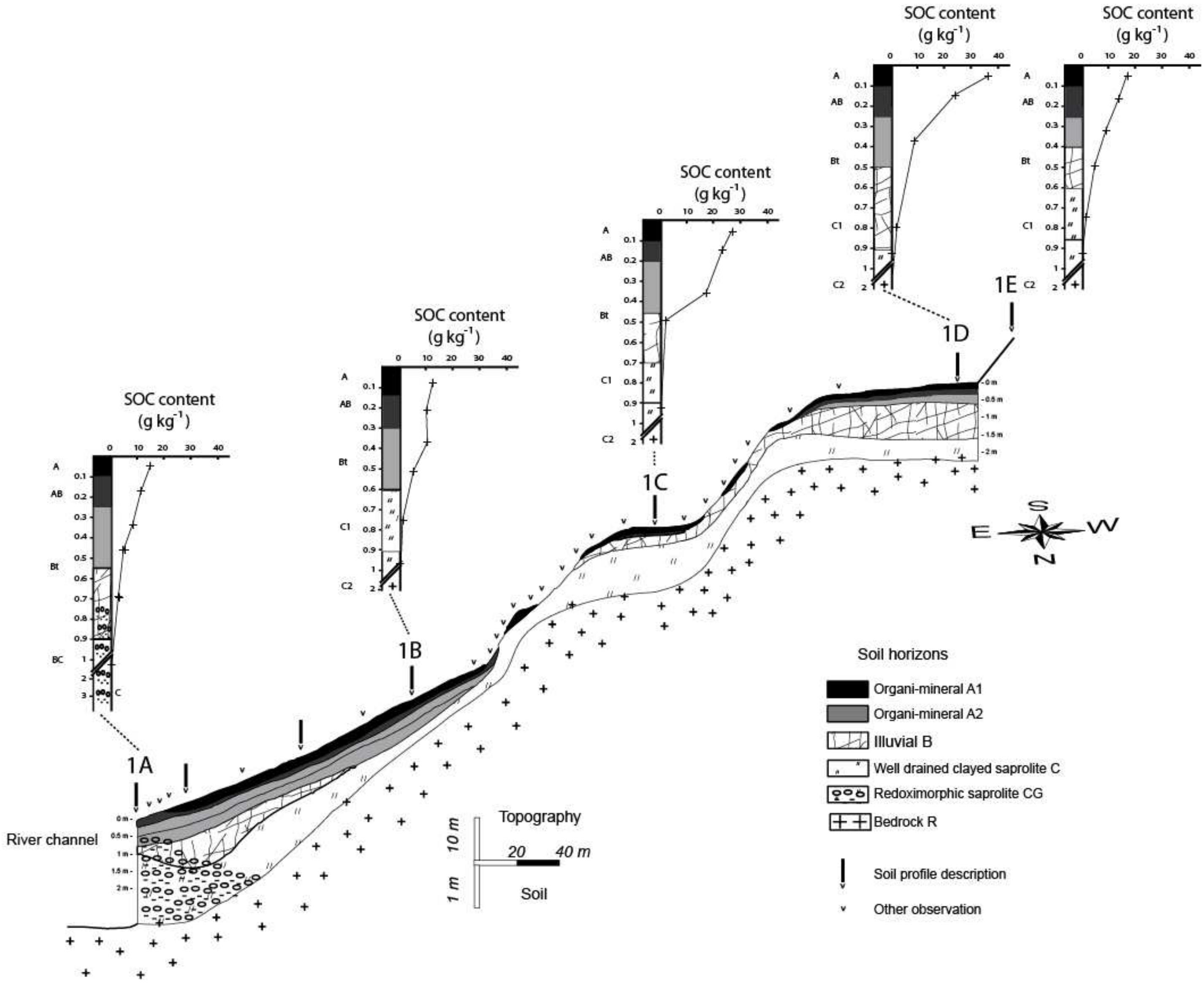


Figure 5.4 Typical catena showing distribution of C content with depth.

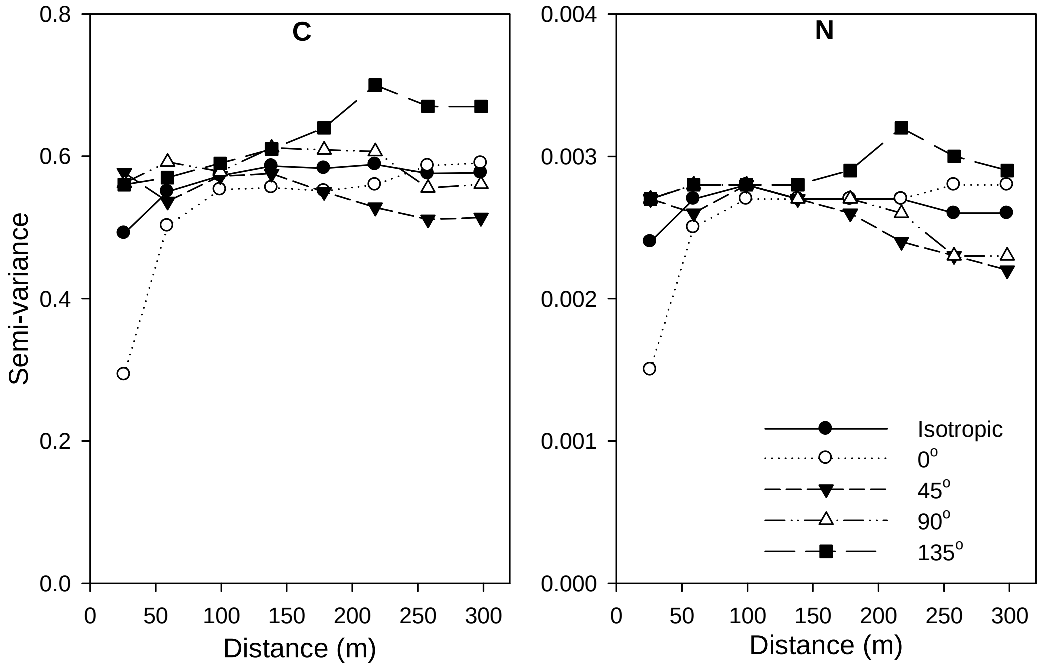


Figure 5.5 Directional variograms of C and N computed using 716 data points.

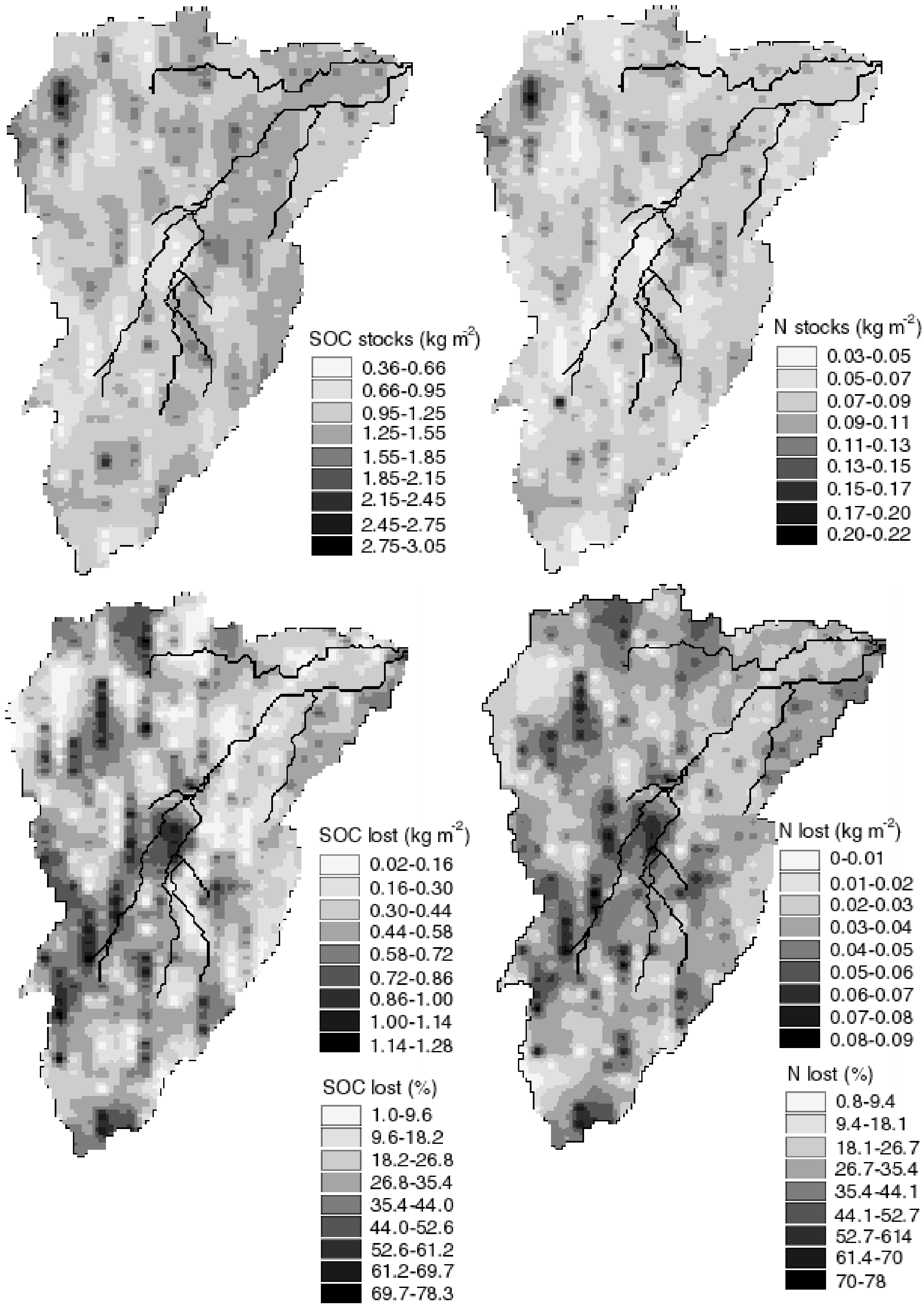


Figure 5.6 Digital maps at a resolution of 5m by 5m for C and N stocks and changes in C and N stocks in the 0-0.05 m soil layer obtained using the best interpolation technique.

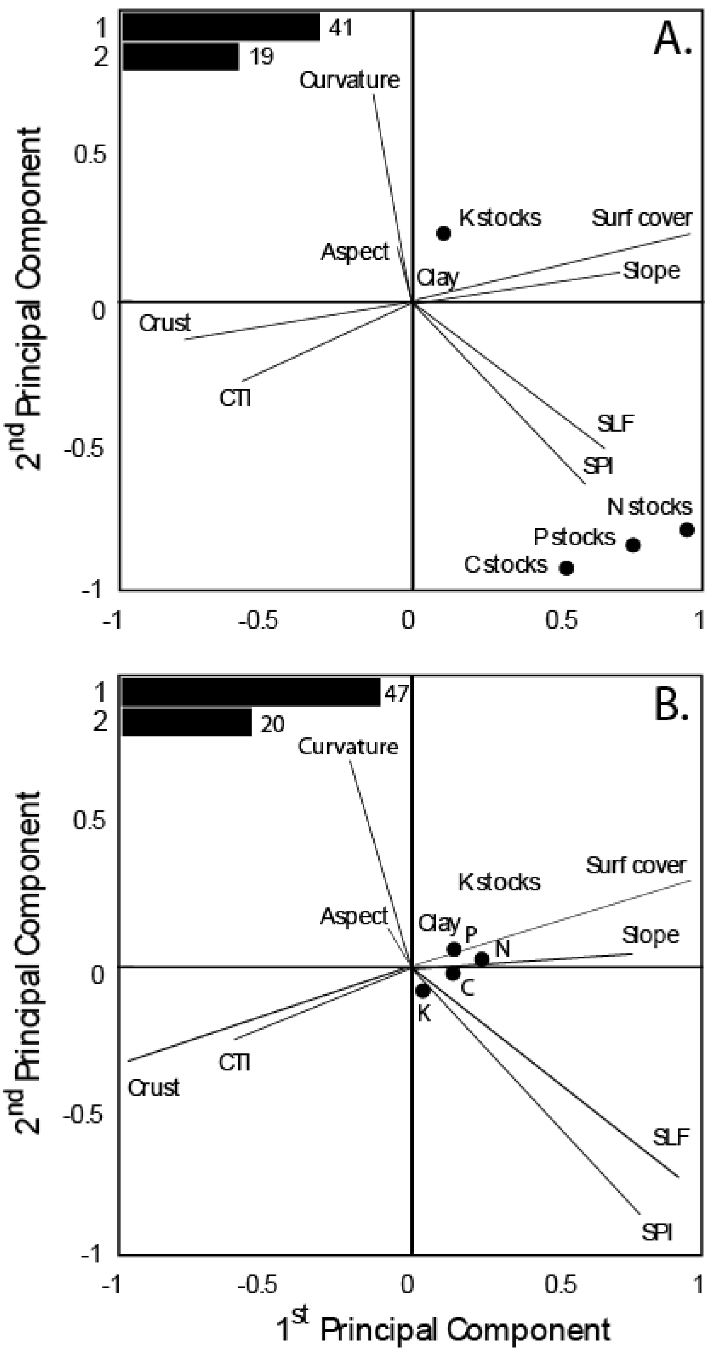
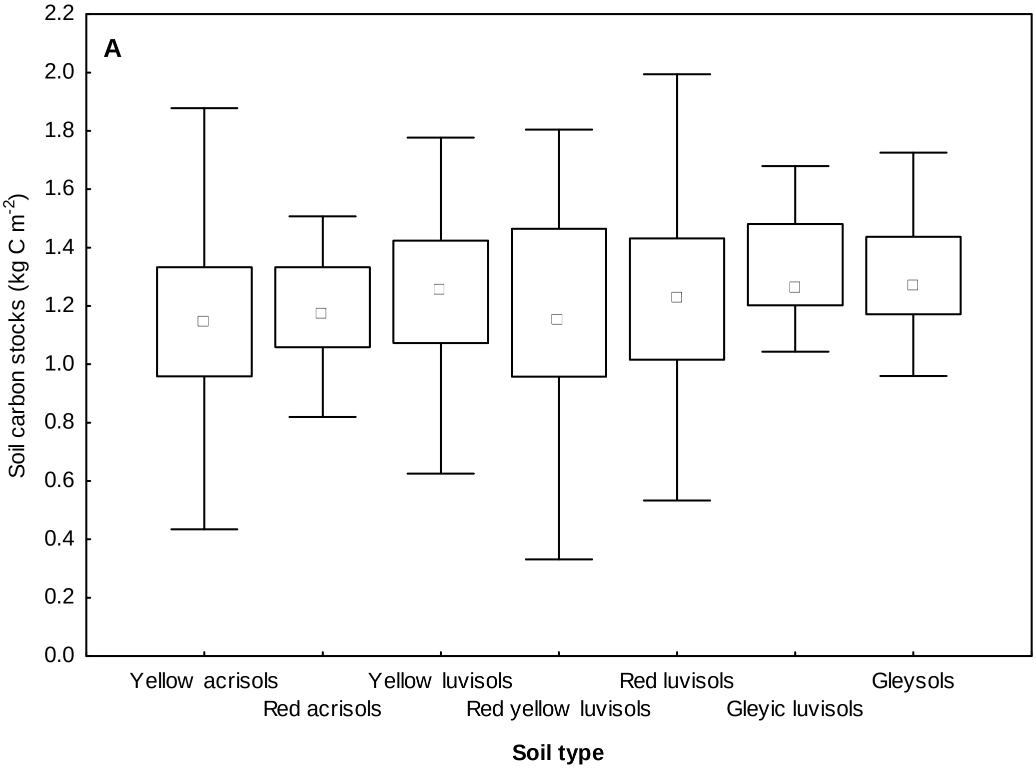


Figure 5.7 Principal component analysis (PCA) of selected environmental factors and C and nutrient stocks for the 0-0.5 m depth (n = 716).



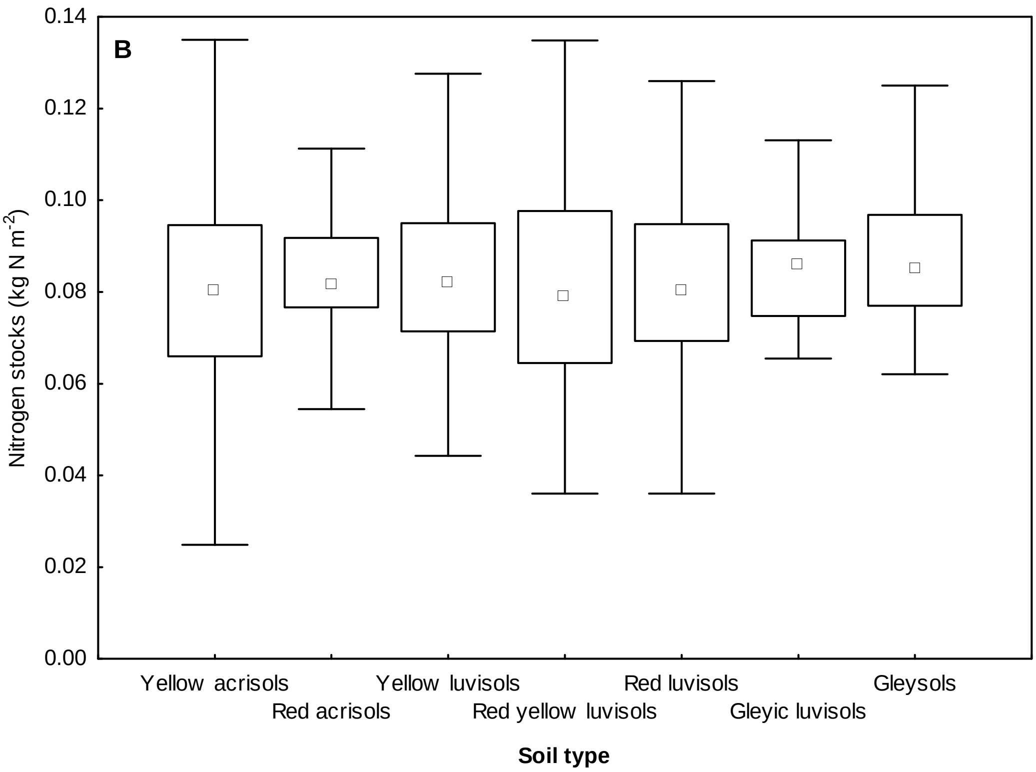


Figure 5.8 Box plots and whisker plots of (A) SOC stocks and (B) N stocks (n=716) as function of soil type. The two vertical lines which form the top and bottom ends of each box represent 25th and 75th percentiles, respectively, of the distribution in each class. The small box across the whisker box represents the median (50th percentile).

CHAPTER 6

6. CONCLUSIONS

This dissertation aimed to better understand the impact of grassland degradation and rehabilitation on soil organic carbon and nutrient stocks. The second chapter, based on a literature review, investigated the impact of grassland degradation on changes in SOC stocks and to elucidate the main environmental controls, worldwide. The third chapter evaluated the consequences of degradation on SOC and SON stocks on a communal rangeland. The fourth chapter investigated the effect of grassland rehabilitation on the same communal rangeland. Finally, the fifth chapter assessed the spatial variation of both SOC stock degradation and its replenishment potential on an entirely grazed catchment.

Collectively, the results obtained in this thesis showed that grassland degradation results in significant losses of SOC and nutrient stocks. More specifically, the results of chapter 2 showed that the worldwide average grassland SOC stock depletion was 9%, with values ranging between 13% for heavily degraded to 7% in lightly degraded soils, with an absolute minimum value of 1% and an absolute maximum value of 90%. Grassland degradation had a more pronounced impact on the depletion of SOC stocks under dry climates and sandy acidic soils compared to wet climates and clayey soils.

The maximum SOC depletion rate of up to 89% found in our site in the Drakensberg foothills, with sandy acidic soils falls in the upper range of the results reported in the existing literature. This is consistent with the fact that higher SOC losses occur under sandy acidic soils. The measurements showed a sigmodial decrease in topsoil SOC and SON stocks with the linear decrease in grass aerial cover. The stock depletion was accompanied by an increase in soil bulk density, a decrease in soil aggregate stability, a preferential enrichment in stabilized organic matter and a decrease in chemical elements such as Ca, Mg, K, Mn, Cu, and Zn essential for soil aggregate stability and plant growth. This staggering decline in SOC stocks raises concerns about the ability of these acidic sandy grassland soils to sustain their main ecosystem functions, such as soil fertility for biomass and food production.

While some authors have reported that grassland degradation is irreversible, even though grazing exclosure (fencing), chapter 4 showed that effective rehabilitation of both grassland and soil stocks is possible through changes in cattle management and fertilization. After two years of rehabilitation topsoil SOC stocks increased by an average of 6.5% (i.e., 0.091 kg C m^{-2}) under “Savory” and 3.9% (0.055 kg C m^{-2}) under fertilization. These techniques should be considered in the future to rehabilitate grasslands in order to replenish SOC stocks.

Finally, the results of chapter 5 informed on the both the spatial variation and replenishment potential of a typically degraded grassland catchment in the foothills of the Drakensberg region, showing typical association between degraded and non-degraded grasslands and variations in soils and topography. A high spatial variability of C and N stocks in the topsoil layer was demonstrated, with values ranging between 0.28 and 2.1 kg C m^{-2} for C stocks, while N stocks varied between 0.02 and 0.19 kg N m^{-2}. The C and N stocks under degraded conditions in the catchment had significant spatial autocorrelation up to a distance of 150 m, and beyond this distance C and N do not depict a significant spatial autocorrelation. The spatial variability of C and N stocks was primarily related to soil surface characteristics, including grass cover and secondarily to topographic attributes. Beyond analysing the spatial distribution of stocks in the catchment, an attempt was made to estimate the carbon replenishment potential of the degraded soils. The C replenishment potential of the degraded soils in the catchment ranged between 0.17 t C ha^{-1} (1%) and 12.8 kg t C ha^{-1} (78.3%), with an average of 4.6 kg t C ha^{-1} (28.6%). The clay-rich red Acrisols in our site have a greater capacity to replenish C stocks than the sandy Luvisols and Gleysols.

Limitations and Perspectives

The work presented in this dissertation is important because there is a paucity of quantitative information on the actual impact of environmental disturbances (in this case of degradation) on SOC and nutrient stocks in grassland soils (UNEP, 2007), limiting our ability to predict how they will respond to future environmental changes. This work has also contributed to improved understanding of the impact of degradation on SOC and nutrient stocks in grassland soils and the interacting environmental controls.

Inevitably, each research work has its limits. In the course of this work, the limitations of establishing baseline conditions, SOC sampling and analysis, difficulty in tracing the SOC inputs and outputs (isotopic signature), time, cost and labour became evident.

Furthermore, the two year rehabilitation experiment was short. Short-term experiments may in some cases have a limited value for understanding interactions over longer timescales, because of potential biogeochemical and/or plant compositional shifts that occur in ecosystems over time that might alter long-term responses to any given environmental factor (Reich and Hobbie, 2013). But the promising results from the short-term rehabilitation experiment in our site offers an opportunity to monitor changes overtime, and this will improve understanding of the plant-soil interactions and the carbon cycle.

Unfortunately due to time and financial constraints, a few sites were investigated, which limits their global representatives. At the heart of the issue lies the question of how much the rehabilitation potential will be under different environmental conditions? Will the rehabilitation of “Savory” technique be successful under different conditions? Will degradation lead to a loss of the stable carbon under different environmental conditions? How much C can be sequestered through rehabilitation in the long-term?

Soils deliver several ecosystem services including carbon sequestration and nutrient cycling, which are of central importance to climate mitigation and food production. The role of soils in the terrestrial global carbon cycle has now become the front line of global environmental change research (Schmidt et al., 2011). There are still many unknowns, however, about how soils will respond to future changes in climate, vegetation and environmental disturbances such as degradation. We need to predict how soils will respond to environmental changes so that we can better understand their role in the terrestrial system and ensure they continue to provide key ecosystem functions.

Previous studies have presented the SOC stabilization mechanisms and the associated SOC dynamics. SOC has been shown to be physically stabilized through microaggregation, or by intimate interaction with silt and clay particles and can be biochemically stabilized through the formation of recalcitrance SOM compounds (Sollins et al., 1996; Six et al., 2002). However, the mechanisms of SOC destabilization and the associated SOC dynamics have received less attention (Sollins et al., 2007; Smernik and Skjemstad, 2009), especially in grassland soils where a greater proportion of SOC stocks is held in the uppermost soil surface layer. The findings of this thesis add further insight into the flip side of the coin - the destabilization mechanisms of SOC, especially in grassland soils.

The information gathered from the field study on the spatial variability of SOC and nutrient stocks and quality needs to be integrated into carbon dynamic models (i.e. the Century, Roth C) to be transmittable to similar landscapes. Future research should focus at regional scale (i.e. Drakensberg region) by identifying environmental factors which control SOC stocks and examine how these factors can be used in a carbon dynamic model to predict SOC stocks. This will require the availability of spatial datasets of explanatory variables at high resolution. The improved process understanding presented in this thesis might form the basis of the model.

Finally, in the context of sustainable development, it is essential to advance collaboration with social scientists and farmer support groups to identify practical ways of promoting adoption of the effective sustainable management strategies for degradation prevention (i.e. “Savory” and fertilization) to farmers and policy makers. Payment for ecosystem services created through C sequestration in the soil is one important strategy.

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8. APPENDIX

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Land degradation impact on soil organic carbon and nitrogen stocks of sub-tropical humid grasslands in South Africa

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ABSTRACT

Land degradation is recognized as a main environmental problem that adversely depletes soil organic carbon (SOC) and nitrogen (SON) stocks, which in turn directly affects soils, their fertility, productivity and overall quality. While it is expanding worldwide at rapid pace, quantitative information on the impact of land degradation on the depletion of SOC and SON stocks remains largely unavailable, limiting the ability to predict the impacts of land management on the C losses to the atmosphere and associated global warming. The main objective of this study was to evaluate the consequences of a decrease in grass aerial cover on SOC and SON stocks. A degraded grassland showing an aerial cover gradient from 100% (Cov100, corresponding to a non-degraded grassland) to 50–75% (Cov75), 25–50% (Cov50) and 0–5% (Cov5, corresponding to a heavily degraded grassland), was selected in South Africa. Soil samples were collected in the 0.05 m soil layer at 48 locations along the aerial cover gradient and were subsequently separated into the clay + silt (2–20 μm) and sand (20–2000 μm) fractions, prior to total C and N analysis (n = 288). The decline in grass aerial cover from 100% to 0–5% had a significant ($P < 0.05$) impact on SOC and SON stocks, with losses by as much as 1.25 kg m^{-2} for SOC and 0.074 kg m^{-2} for SON, which corresponded to depletion rates of 89 and 76%, respectively. Furthermore, both the C:N ratio and the proportion of SOC and SON in the silt + clay fraction declined with grass aerial cover, which was indicative of a preferential loss of easily decomposable organic matter. The staggering decline in SOC and SON stocks raises concerns about the ability of these acidic sandy loam soils to sustain their main ecosystem functions. The associated decrease in chemical elements (e.g., Ca by a maximum of 67%; Mn, 77%; Cu, 66%; and Zn, 82%) was finally used to discuss the mechanisms at stake in land degradation and the associated stock depletion of SOC and SON stocks, a prerequisite to land rehabilitation and stock replenishment.

1. Introduction

Grasslands occupy about 40% of world's land surface and store approximately 10% of the global soil carbon (C) stock of 1500 Gt (Suttie et al., 2005). Consequently, grasslands are considered to have greater potential to sequester SOC, depending on management strategies (Franzluebbers and Doraiswamy, 2007), making them an important component of the global C cycle. Additionally, grasslands provide key ecosystem goods and services by supporting biodiversity, and serving as rangelands for the production of forage to sustain the world's livestock (Asner et al., 2004; Bradford and Thurow, 2006; FAO, 2010; Suttie et al., 2005). However, land degradation severely impacts on the productivity of grasslands (UNEP, 2007).

Land degradation, defined here as the reduction in the capacity of grasslands to carry out their key ecosystem functions, is commonly attributed to disturbances including overgrazing, livestock trampling and soil erosion (Daily, 1995; UNEP, 2007). For instance, a recent study by Kotzé et al. (2013) investigated the impacts of rangeland management on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa. They found that communal farms with continuous grazing were generally depleted of nutrient stocks, and nutrient depletion generally increased with increasing grazing intensity. Grassland management practices substantially influence the amount, distribution and turnover rate of soil organic matter and nutrients in soils (Blair et al., 1995). Moreover, because the larger proportion (ca 60–70%) of SOC and nutrient stocks in grassland soils is concentrated in the top 0.3 m (Gill et al., 1999), any external disturbance is likely to cause dramatic soil fertility and SOC depletion, which in turn will constrain grassland productivity, including biodiversity loss and forage production (Dong et al., 2012; Ruiz-Sinoga and Romero Diaz, 2010).

Yet, contradictory results have been reported on the impact of land degradation on SOC stocks with some studies showing a decrease in SOC with overgrazing (Martinsen et al., 2011; Steffens et al., 2008),

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some no change (Dormaar et al., 1977; Johnston et al., 1971) whereas some show an increase (Derner et al., 1997; Smoliak et al., 1972).

For instance, SOC stocks declined by 15% after seven years of grazing in Norway, with 0.76 kg C m^{-2} in ungrazed compared to 0.64 kg C m^{-2} in heavily grazed grasslands (Martinsen et al., 2011). Steffens et al. (2008) found that 30 years of overgrazing in a semi-arid Chinese grassland resulted in 50% decrease in SOC stocks, with 0.64 kg C m^{-2} in grazed compared to 1.17 kg C m^{-2} in ungrazed grasslands. A similar depletion rate was found in the USA, where Franzluebbers and Stuedemann (2009) observed that heavy grazing reduced SOC stocks to 0.051 kg C m^{-2} after 12 years of grazing, compared with 0.117 kg C m^{-2} on ungrazed grasslands. Wu and Tiessen (2002) reported that land degradation reduced SOC and N by 33% and 28%, respectively in a degraded Chinese alpine grassland. Finally, Dong et al. (2012) found an extreme SOC depletion rate of 90% in a degraded Chinese grassland.

In contrast, grazing increased SOC stocks under several environments (Bauer et al., 1987; Derner et al., 1997; Frank et al., 1995; Smoliak et al., 1972), by rates ranging from 14% to 91%. However, in the latter, moderate grazing is reported to be beneficial to grassland soils rather than contributing to their degradation.

While the studies focusing on land degradation have reported associated losses in SOC, little is known on the impact of different degradation intensities on SOC stocks, with the underlying research question being at what threshold of land degradation do SOC stocks dramatically decrease?

To further improve the understanding of land degradation impact on SOC losses from soils, more work needs to be done on the mechanisms controlling organic matter destabilization. As such, the changes in organic matter quality as a consequence of land degradation could be early indicators of SOC stock depletion in both natural and agricultural ecosystems, as suggested by Christensen (2001). Furthermore, a better understanding of the rates of SOC and SON depletion and the associated destabilization mechanisms is expected to enhance efforts to circumvent land degradation and accelerate the recovery of degraded soils (Schmidt et al., 2011), while maintaining a viable forage production for livestock and supporting biodiversity (Lal, 2004).

For many smallholder farmers in Africa, grasslands make a significant contribution to food security by providing part of the feed requirements of livestock used for meat and milk production (O'Mara, 2012). However, many of the grasslands are in poor condition and showing signs of degradation due to an increase in anthropogenic pressures on marginal lands, overgrazing and the associated problems of soil erosion (Suttie et al., 2005). As a consequence, this is jeopardizing both the environment and the economical development of rural livelihoods.

In this study of a communal rangeland in the uplands of the Drakensburg region, KwaZulu-Natal Province, South Africa managed by smallholder farmers, our main objective was to evaluate the consequences of a decrease in grass aerial cover on SOC and N depletion rates and the associated organic matter quality. Grass aerial cover was used as an indicator of land degradation.

2. Materials and methods

2.1. Site description

The study area is located in the Potshini catchment, 10 km north of the Bergville district in the KwaZulu-Natal Province of South Africa (Long: 29° 21′; Lat: −28° 48′). This area has a sub-tropical humid climate, characterized by cold dry winters and warm rainy summers (October to March), with a mean annual precipitation of 684 mm, a mean annual potential evaporation of 1600 mm and a mean annual temperature of 13 °C (Schulze, 1997). The altitude ranges from 1080 to 1455 m.a.s.l and the average slope gradient is 8%. The underlying geology is sandstone and mudstone, and the soils are classified as Acrisols (WRB, 2006). The vegetation in this area is dominated by Moist Highveld Sourveld (Camp and Hardy, 1999). The dominant vegetation species of the Moist Highveld Sourveld include *Hyparrhenia hirta* and *Sporobolus africanus*.

2.2. Experimental design and sampling strategy

A degraded grassland site with a surface area of 1500 m^2 (30 m × 50 m) and homogeneous soils was selected in the uplands of the Drakensburg region of South Africa (Fig. 1). This site was selected because it exhibited a land degradation gradient varying from highly degraded areas with bare soils in the north to areas fully covered by grass in the south. Such areas are a common feature of many communal rangelands in this part of South Africa. For soil sampling, four categories of grass aerial cover were identified and evaluated in the site, i.e. 75–100% (Cov100, corresponding to non-degraded land), 50–75% (Cov75), 25–50% (Cov50), and 0–5% (Cov5, corresponding to heavily degraded land). In this study, grass aerial cover is defined as the area of the ground covered by the vertical projection of the aerial portion of the plants (USDA, 1996). Aerial cover was measured by placing a 1 m × 1 m plot frame at fixed intervals along each corresponding aerial cover category, while the aerial cover of the plants in the plot was recorded as an estimate of the % of total area (Daubenmire, 1959). At each cover category, three sampling points were randomly selected. For each selected sampling position, four replicate soil samples were collected in the 0–0.05 m soil layer 1 m apart in a radial basis sampling strategy to yield twelve samples per category. The sampling resulted in a total of 48 soil samples. Furthermore, for each category, additional soil samples for bulk density were sampled using a 0.075 m diameter metallic cylindrical core (height, 0.05 m) following similar sampling strategy. The surface layer was intensively sampled because the effects of land degradation on SOC and nutrient stocks have been shown to be more pronounced in this soil layer (Dong et al., 2012; Snyman and du Preez, 2005). For the analysis of SOC and N stocks, with depth in each grass cover category, additional soil samples were collected by horizon at depth increments of 0–0.05 m, 0.05–0.15 m, then every 0.15 m down to 1.2 m using a hand shovel from the face of a 1 m × 1 m × 1.2 m soil pit. Triplicate soil bulk density samples were also collected in the different depth increments of the soil profiles using 220.89 cm^{-3} metal cylindrical cores (height 0.05 m, diameter 0.75 m). Soil samples for bulk density were taken to the laboratory, immediately oven-dried at 105 °C to determine the oven dry weight using the gravimetric method (Blake and Hartge, 1986). Once in the laboratory, the field moist samples were passed through an 8-mm sieve by gently breaking apart the soil. The remaining soil samples were air-dried and ground to pass through a 2-mm sieve for further soil analysis.

2.3. Soil physical and chemical analysis

The particle size distribution was determined by the sieve and pipette method (Gee and Bauder, 1986). The penetration resistance (PR) of the soil, a proxy for soil compaction was measured in the field using a hand-held cone penetrometer (Herrick and Jones, 2002). The PR was evaluated by randomly selecting fifteen positions in each grass aerial cover category for penetration readings of the soil surface. The PR measurements were taken before the soil surface was disturbed for soil sample collection from a 0.05 m soil layer. The soil pH was measured in a 1:2.5 (10 g) to 1 M KCl (25 ml) suspension using a Calimatic pHM766 pH meter. Exchangeable Ca, Mg and acidity were determined by extraction in 1 M KCl while P, K, Zn, Mn and Cu were determined by extraction in Ambic 2-extract containing 0.25 M NH_4HCO_3, with detection by atomic absorption spectrometry (Manson and Roberts, 2000). The concentration of P and K was determined by inductively-coupled plasma optical emission spectrometry (ICP-OES).

Effective cation exchange capacity (ECEC) was calculated as the sum of extractable cations, with the percentage acid saturation calculated as the exchangeable acidity × 100 / (Ca + Mg + K + exchangeable acidity). Total C and N were measured in the bulk soil using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI).

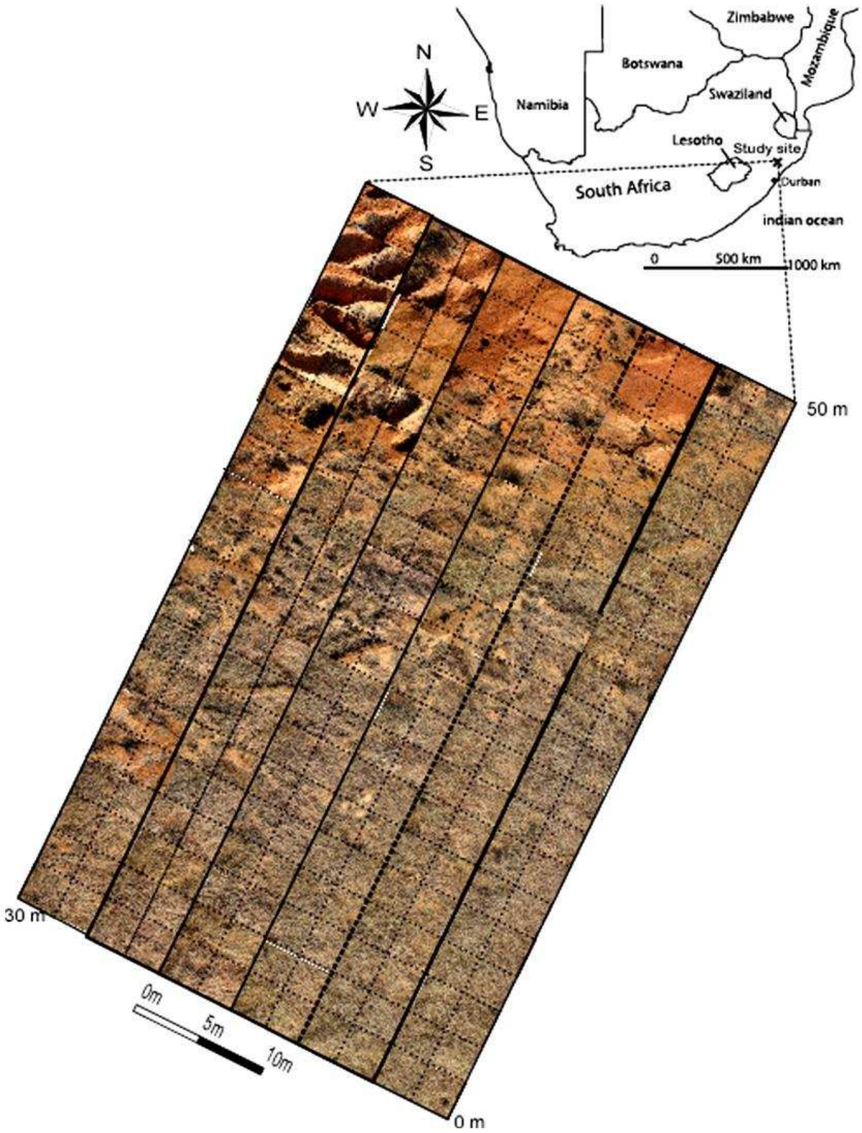

Fig. 1. Mosaic of pictures showing the different land degradation intensities or grass aerial cover from 0–5% in the north to 100% in the south along a degraded grassland site Potshini, South Africa.

2.4. Particle size fractionation

Physical fractionation was applied to the soil samples as a proxy of soil organic matter mineralization potential (Feller and Beare, 1997). Soil samples were separated into two size fractions; sand (20–2000 μm) and clay + silt fraction (2–20 μm) by combining wet sieving and sedimentation, following Schmidt et al. (1999). Twenty grams of soil was dispersed in 25 ml of distilled water using ultrasound (LABSONIC B1510), with a power output of 600 W and an output-energy of 22 J ml^{-1}, which was below the energy threshold that may disrupt coarse sand sized SOM (Amelung and Zech, 1999). The dispersed suspension was then wet sieved to obtain the sand fraction (53–2000 μm), while the remaining material containing the silt and clay fractions (<53 μm) was separated by sedimentation following the standard pipette method (Gee and Bauder, 1986). Thereafter, the fractions were oven-dried at 40 °C, ground to <0.5 mm and analyzed for total C and N in triplicate using a LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI).

Particular attention was undertaken to ensure that the potential losses during the particle-size fractionation procedure did not induce bias. To determine the recovery efficiency, the mass and organic C content of whole soil were compared with the sums of mass and C content in the two fractions to (%). The recovery efficiency obtained was on average 105%.

2.5. Aggregate stability

Aggregates of 3–8 mm in size were obtained by breaking soil aggregates by hand along lines of weakness after air-drying them in the laboratory at room temperature. The stability of soil aggregates was measured on the 3–8 mm aggregates following the ISO standard method (ISO/DIS 10930, 2012) described by Le Bissonnais (1996). The fast wetting, slow wetting and stirring of pre-wetted aggregate tests were applied for each soil sample analyzed. They correspond to a specific disaggregation mechanism, viz, slaking, differential clay swelling and mechanical breakdown, respectively. For the fast wetting test, aggregates (10 g) were immersed in 50 ml distilled water for 10 min. For the slow wetting test, aggregates were placed on top of foam humidified with water for 1 h. For the stirring test, aggregates were first immersed in ethanol, then in water and gently shaken up and down 10 times. The weight of aggregates collected on each sieve (sizes: 2 mm, 1 mm, 0.5 mm, 0.2 mm, 0.1 mm and 0.05 mm) was subsequently measured and expressed as the percentage of the initial sample dry mass to compute the mean weight diameter (MWD), which was calculated as follows:

$$\mathrm{MWD} = \frac{\sum(x_i \times w_i)}{100} \quad (1)$$

where x is the mean inter-sieve size and w_i is the percentage of fragments retained by sieve i. The greater the MWD the more resistant to disaggregation the aggregates are.

2.6. Calculation of SOC and N stocks

The SOC and N stocks were calculated using the following equation by (Batjes, 1996):

$$C_s = x_1 x_2 x_3 \left(1 - \frac{x_4}{100}\right) \times b \quad (2)$$

where C_s is the C stock (kg C m^{-2}); x_1 is the C concentration in the <2 mm soil material (g C kg^{-1} soil); x_2 is the soil bulk density (kg m^{-3}); x_3 is the thickness of the soil layer (m); x_4 is the proportion (%) of fragments of >2 mm; and b is a constant equal to 0.001.

2.7. Calculation of changes in SOC and N stocks induced by land degradation

The effect of land degradation on SOC stocks was determined by using the change in the SOC stock along the degradation gradient relative to an initial value of the SOC stock under Cov100. We assumed that 100% aerial cover represented a "non-degraded land". The change in SOC stock was calculated as follows:

$$\mathrm{SOC_{SC}} = \frac{\mathrm{SOC_{S\text{-}ND}} - \mathrm{SOC_{S\text{-}D}}}{\mathrm{SOC_{S\text{-}ND}}} \times 100 \quad (3)$$

where $\mathrm{SOC_{SC}}$ is the change in SOC stocks ($\mathrm{SOC_S}$), $\mathrm{SOC_{S\text{-}ND}}$ is $\mathrm{SOC_S}$ in the non-degraded soils and $\mathrm{SOC_{S\text{-}D}}$ is $\mathrm{SOC_S}$ in degraded soils. The same equation was used for N stocks. The change in SOC stocks induced by degradation is based on the premise that the loss of SOC stocks from non-degraded soil is of a less consequence than the loss of the same amount of stocks from a soil already depleted of SOC. Thus, the more a soil is depleted of SOC stocks the more difficult it is to rehabilitate (Blair et al., 1995).

2.8. Statistical analysis

A correlation matrix was generated to determine the univariate relationship between changes in SOC and N stocks and selected soil properties. The data was analyzed using the software packages Sigma Plot 8.0 (Systat Software Inc., Richmond, California, USA) and STATISTICA 7.0 (StatSoft, Inc., Tulsa, OK). Differences between means were tested using the DIFF option of the LSMEANS statement, with a significance level of $P < 0.05$ (SAS Institute, 2003).

3. Results

3.1. General soil characteristics of the degraded grassland site

The soils (Acrisols) are characterized by a dark brown (7.5YR 4/4) 0–0.3 m thick A horizon, with a weak sub-angular blocky structure. This horizon is underlain by a reddish (5YR 4/6) B-horizon 0.3–0.7 m. Underlying this horizon is the C horizon 0.7–1.2 m characterized by sandy saprolite showing signs of wetness. Sand content ranged between 49% in Cov5 and 73% in Cov75, silt content ranged between 13% in Cov75 and 17% in Cov5 and clay content ranged between 14% in Cov75 and 34% in

Table 1
Characterization of soils with the different grass covers in the 0.05 m soil layer of a degraded grassland in Potshini, South Africa. Data are means (±SE; n = 48).

Soil property	Grass aerial cover (%)			
	100	50–75	25–50	0–5
Sand, %	71 ± 1.43b	73 ± 2.81b	72 ± 1.10b	49 ± 0.87a
Silt, %	14 ± 0.93ab	13 ± 1.71a	14 ± 0.58ab	17 ± 0.38b
Clay, %	15 ± 0.74a	14 ± 1.15a	15 ± 0.75a	34 ± 0.95b
SWC, %	14.72 ± 0.62a	12.31 ± 4.10a	9.38 ± 0.99a	10.32 ± 0.29a
P, mg kg^{-1}	5.25 ± 0.22b	5.08 ± 0.29b	1.92 ± 0.15a	2.17 ± 0.32a
K, mg kg^{-1}	143.17 ± 6.97c	167.33 ± 8.26d	88.17 ± 11.95b	62.42 ± 2.14a
Ca, mg kg^{-1}	225.83 ± 10.61c	170.17 ± 9.76b	82.67 ± 8.71a	75.25 ± 5.27a
Mg, mg kg^{-1}	103.67 ± 5.02c	67.08 ± 4.79b	30.83 ± 4.32a	79.83 ± 7.45b
Exch acidity, cmol$_c$ kg^{-1}	0.83 ± 0.06a	0.65 ± 0.06a	1.47 ± 0.11b	4.16 ± 0.12c
ECEC, cmol$_c$ kg^{-1}	3.18 ± 0.11b	2.48 ± 0.13a	2.36 ± 0.08a	5.35 ± 0.11c
Acid sat, %	25.83 ± 1.51a	26.33 ± 2.35a	62.25 ± 4.43b	77.58 ± 0.96c
pH (KCl)	3.81 ± 0.02b	3.94 ± 0.02d	3.88 ± 0.02c	3.74 ± 0.00a
Zn, mg kg^{-1}	1.67 ± 0.20c	0.77 ± 0.09b	0.22 ± 0.06a	0.29 ± 0.06a
Mn, mg kg^{-1}	14.33 ± 1.12c	10.58 ± 1.00b	4.08 ± 0.43a	3.25 ± 0.68a
Cu, mg kg^{-1}	1.34 ± 0.10b	0.58 ± 0.03a	0.98 ± 0.23b	0.45 ± 0.07a
MWD, mm	1.36 ± 0.06b	1.35 ± 0.10b	0.89 ± 0.06a	0.71 ± 0.08a
PR, kg cm^{-2}	16.77 ± 0.81b	11.30 ± 0.72b	18.63 ± 1.22b	19.47 ± 1.33a

Values indicate mean ± standard error. sand content (Sand), silt content (Silt), clay content (Clay), soil water content (SWC), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), exchangeable acidity (Exch acidity), effective cation exchange capacity (ECEC), acid saturation (Acid sat), soil pH in KCl (pH KCl), zinc (Zn), manganese (Mn), copper (Cu), mean weight diameter (MWD) and penetrative resistance (PR). Statistical analyses were performed for comparisons between the different aerial grass covers. Within each grass cover, values followed by a different letter are significantly different at $P < 0.05$.

Cov5 in the 0.05 m soil layer (Table 1). The soil pH is acidic with values in the 0–0.5 m layer as low as 3.74 in Cov5 to 3.94 in Cov75. Effective cation exchange capacity (ECEC) ranged between 2.36 $cmol_c\ kg^{-1}$ in Cov50 and 5.35 $cmol_c\ kg^{-1}$ in Cov5, while acid saturation ranged between 26% in Cov100 and 77% in Cov5 (Table 1).

3.2. Impact of aerial cover on SOC and N stocks

The SOC content and stocks in the 0.05 m soil layer decreased with decreasing grass aerial cover in the following order: Cov100 (19.87 g kg^{-1}; 1.39 kg m^{-2}) > Cov75 (11.19 g kg^{-1}; 0.79 kg m^{-2}) > Cov50 (5.17 g kg^{-1}; 0.40 kg m^{-2}) > Cov5 (1.73 g kg^{-1}; 0.14 kg m^{-2}), with differences significant at $P < 0.05$ level (Fig. 2). Similarly, N content and stocks decreased with decreasing grass aerial cover as follows: Cov100 (1.53 g kg^{-1}; 0.106 kg m^{-2}) > Cov75 (0.95 g kg^{-1}; 0.068 kg m^{-2}) > Cov50 (0.48 g kg^{-1}; 0.037 kg m^{-2}) > Cov5A (0.39 g kg^{-1}; 0.032 kg m^{-2}). SOC and N stocks varied greatly along the degradation gradient decreasing sigmoidally in non-degraded grassland soils (Cov100) to heavily degraded grassland soils (Cov5). Thus, the C:N ratios decreased with decreasing grass cover, from an average of 13.0 at Cov100 to 11.8 at Cov75 to 10.6 at Cov50 and 4.4 at Cov5.

On average, land degradation resulted to a decrease in SOC stocks of 79% for Cov5, 42% for Cov50 and a negligible effect for Cov75 (Fig. 3). Similarly, degradation led to a decrease in N stocks of 48% for Cov5 and 39% for Cov50. This result suggests that the critical grass aerial cover threshold for which degradation greatly affects SOC and N stocks is 50%.

While there were significant differences in SOC and N stocks in the 0.05 m soil layer, no significant differences were found with depth.

3.3. SOC and N distribution in the particle-size fractions

Results of land degradation impact on SOC changes in the different soil fractions are presented in Fig. 4A. Soil organic C in the sand fraction decreased from around 6.09 g C kg^{-1} fraction in non-degraded soils (Cov100) to 0.37 g C kg^{-1} fraction in degraded soils (Cov5), which

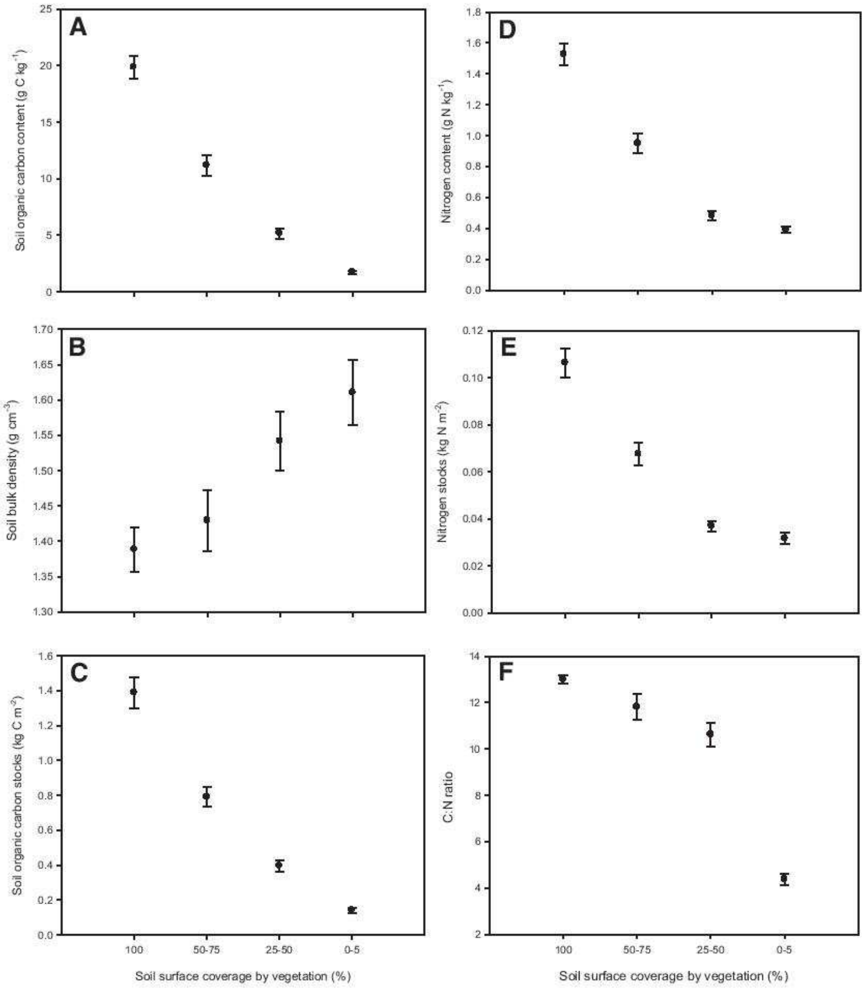

Fig. 2. Mean ± standard error of (A) soil organic carbon content (SOC_C); (B) soil bulk density (ρ_b); (C) soil organic carbon stocks (SOC_S); (D) nitrogen (N) content; (E) nitrogen stocks (N_S) and (F) carbon to nitrogen ratio. Values are the mean ± standard error of four replicate soil samples along the four categories of grass aerial cover (n = 48).

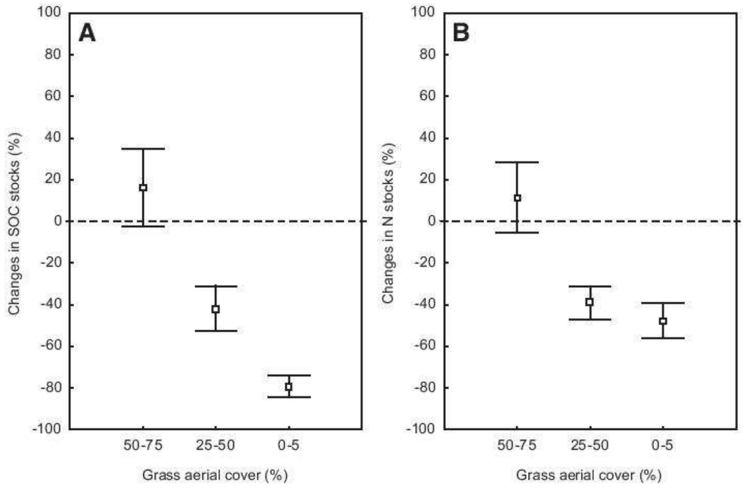

Fig. 3. Influence of land degradation on changes in (A) soil organic stocks (SOC_S) and (B) N stocks (N_S) in the upper 0.05 m soil layer along the degradation gradient. Values for SOC and N are the mean ± standard error of four replicate soil samples from 50 to 75, from 25 to 50 and from 0 to 5 grass aerial covers, relative to 100% grass aerial cover values.

corresponded to a 94% decrease, significant at $P < 0.05$. Concentrations of SOC in the silt + clay size fraction decreased from 61.4 g C kg^{-1} fraction in non-degraded soils to 2.82 g C kg^{-1} fraction in degraded soils, which corresponded to a 95% decrease, significant at $P < 0.05$. As a proportion of total soil C, SOC in the sand fraction decreased from 19% in non-degraded soils to 11% in heavily degraded soils (Fig. 4B). In contrast, SOC in the silt + clay fractions varied little from 84% in non-degraded soils to 78% in heavily degraded soils.

Nitrogen in the sand fraction decreased from around 0.43 g N kg^{-1} fraction in non-degraded soils to 0.26 g N kg^{-1} fraction in degraded soils, which correspond to a 40% decrease, significant at $P < 0.05$ (Fig. 5A). Concentrations of N in the silt + clay sized fraction decreased from 5.31 g N kg^{-1} fraction in non-degraded soils to 0.91 g N kg^{-1} fraction in degraded soils, which corresponded to an 83% decrease, significant at $P < 0.05$. As a proportion of total N, the N in the sand fraction varied relatively little from 17% in non-degraded soils to 22% in heavily degraded soils (Fig. 5B). Nitrogen in the silt + clay sized fraction varied from 84% in non-degraded soils to 78% in heavily degraded soils.

3.4. Impact of grass aerial cover decrease on soil physical and chemical properties

Changes in soil physical properties were observed along the land degradation gradient (Table 1). The sand distribution was similar, with 71%, 73%, and 72% for Cov100, Cov75 and Cov50, respectively, while, the sand content for Cov5 was 49%. The silt distribution was almost similar along the degradation gradient. The clay content was higher (34%) in Cov5 and similar for Cov100 (15%), Cov75 (14%) and Cov50 (15%). The aggregate stability decreased with decreasing grass aerial cover from 1.36 mm in Cov100 to 0.71 mm in Cov5. This pattern was consistent for penetrometer resistance (PR), with 16.8 kg cm^{-2} in Cov100, 18.63 kg cm^{-2} in Cov50 and 19.47 kg cm^{-2} in Cov5.

Potassium varied along the degradation gradient, with concentrations ranging between 143 and 167 mg kg^{-1} in Cov100 and Cov75, then declined to about 62 mg kg^{-1} in Cov5. Calcium decreased from around 226 mg kg^{-1} in non-degraded soils to 75 mg kg^{-1} in degraded soils (Table 1). Magnesium decreased from about 104 mg kg^{-1} in Cov100 to 31 mg kg^{-1} in Cov50, then increased to 80 mg kg^{-1} in Cov5. Phosphorus decreased from 5.25 mg kg^{-1} in non-degraded soils to 2.17 in degraded soils. Zn and Mn, respectively significantly decreased with decreasing grass aerial cover in the following order: Cov100 (1.67 mg kg^{-1}; 14.3 mg kg^{-1}) > Cov75 (0.77 mg kg^{-1}; 10.6 mg kg^{-1}) > Cov50 (0.22 mg kg^{-1}; 4.08 mg kg^{-1}) > (0.29 mg kg^{-1}; 3.25 mg kg^{-1}). In contrast, acid saturation increased with decreasing grass aerial cover from around 26% in non-degraded soils to 78% in degraded soils. ECEC also varied relatively little along the degradation gradient from 3.18 $cmol_c$ kg^{-1} in non-degraded soils to 5.35 $cmol_c$ kg^{-1} in degraded soils.

3.5. Other environmental factors controlling SOC and SON stocks

A correlation matrix (Table 2) revealed that changes in SOC and N stocks induced by degradation were significantly positively correlated to mean weight diameter (MWD), a measure of soil aggregate stability ($r^2 = 0.67$; $P < 0.05$) and negatively correlated with clay ($r^2 = -0.54$; $P < 0.05$), soil bulk density (ρ_b) ($r^2 = -0.40$; $P < 0.05$), penetration resistance (PR), a proxy of soil compaction ($r^2 = -0.29$; $P < 0.05$) acid saturation (Acid sat) ($r^2 = -0.74$; $P < 0.05$), ECEC ($r^2 = -0.36$; $P < 0.05$), and exchangeable acidity (Exch acid) ($r^2 = -0.66$; $P < 0.05$).

4. Discussion

4.1. Impact of land degradation on the depletion of SOC and SON stocks

In this study, the decrease in grass aerial cover resulted in a greater depletion in SOC stocks of 89% in the surface layer of degraded soils. This depletion of SOC stocks is relatively high compared to what has been reported in other studies. For example, Snyman and du Preez (2005) found that degradation of the rangeland from good to poor condition, with species composition and basal cover used to characterize grassland condition decreased SOC and N stocks by 22% and 13%, respectively in fine sandy loamy grassland soils under a semi-arid climate in Bloemfontein, South Africa. Wu and Tiessen (2002) found that land degradation reduced SOC stocks by 33% and N stocks by 28% in a Chinese alpine grassland. Interestingly, the higher SOC depletion rate of 89% found at our study site is of similar magnitude to those reported by Dong et al. (2012), who found that land degradation significantly reduced SOC stocks by 90% in loamy soils under a continental climate in China. Similarly, Wen

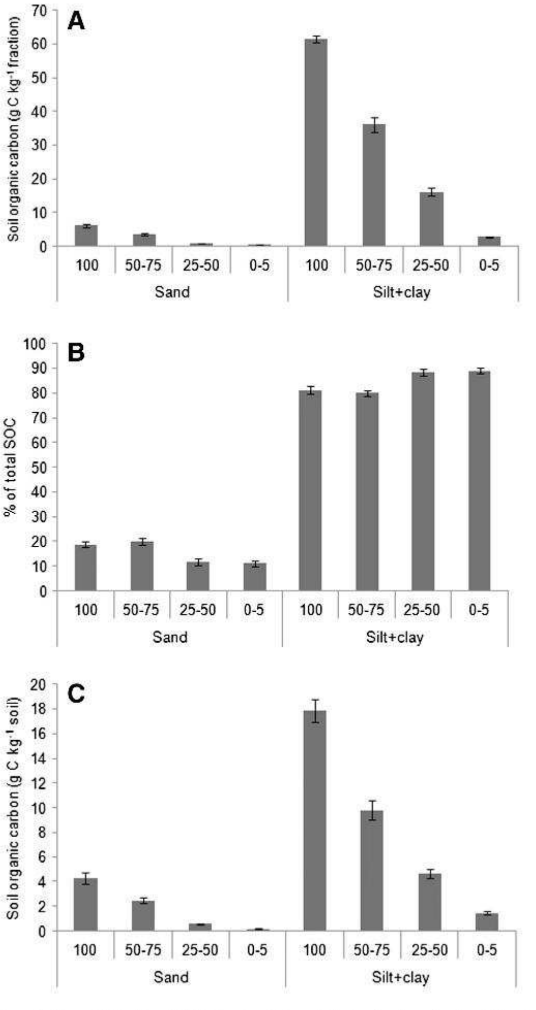

Fig. 4. Relative distribution of SOC stored within particle size fractions for the different grass aerial covers, showing (A) concentrations (g C kg^{-1}) in particle size fractions, (B) proportions (%) of the total C content and (C) concentrations (g C kg^{-1}) in bulk soil. Bars represent mean ± standard error of four replicate soil samples.

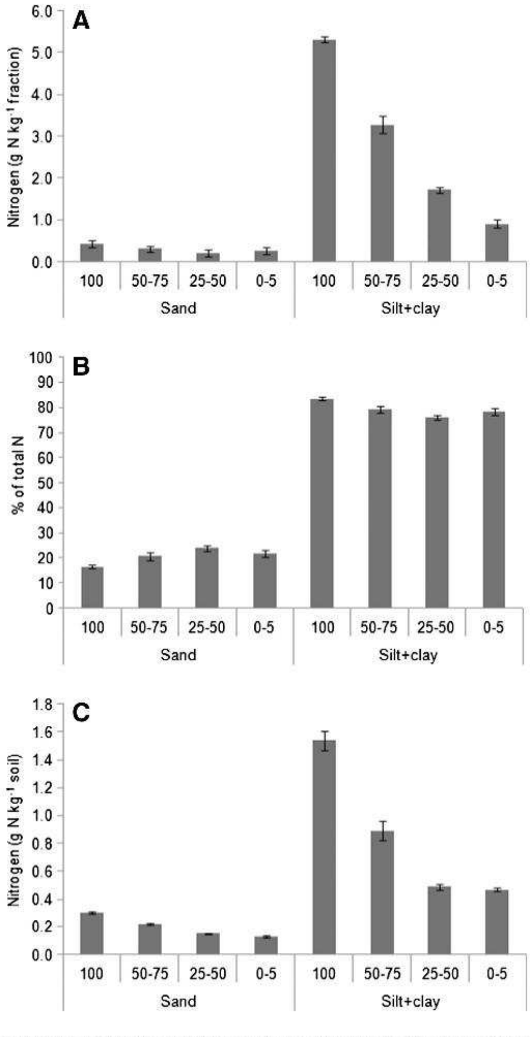

Fig. 5. Relative distribution of SON stored within particle size fractions, for the different grass aerial covers, showing (A) concentrations (g N kg^{-1}) in particle size fractions, (B) proportions (%) of the total N content and (C) concentrations (g N kg^{-1}) in bulk soil. Bars represent mean ± standard error of four replicate soil samples.

et al. (2012) found that land degradation led to an 89% decline in SOC stocks for sandy grassland soils in China.

The greater depletion of SOC and N stocks in the heavily degraded soils may be due to a number of reasons. First, the soils at our site are characterized by a coarse texture (up to 73% sand). The greater SOC loss in such soils is due to the lack of the physical protection of organic matter (Feller and Beare, 1997). Secondly, the soils are acidic (pH < 3.9). Previous studies have shown that in acidic soils, base cations such as Ca, K and Mg are weakly bound to the soil (Berthrong et al., 2009), causing weak interactions with soil organic matter in the soil. In this study, it was found that in the heavily degraded soils, Ca was reduced by 67%, K by 56% and Mg 23%. The loss of SOC lowers nutrient availability and cation exchange capacity, and this then lowers biomass production, which over-time may lower organic inputs thereby lowering SOC. The third reason involves the removal of the nutrient rich A horizon by water erosion. Indeed, as shown by Dlamini et al. (2011) in the same study site, water erosion potentiated soil losses at rates up to 13 t ha^{-1} yr^{-1}.

4.2. Impact of grass aerial cover decrease on soil organic matter quality

The decrease in grass aerial cover was accompanied by a decrease in the C:N ratio and in the proportion of the organic matter present in the sand fraction of the soil. Such a shift in organic matter quality towards less proportion of fresh and easily decomposable compounds can be explained by the decrease in biomass production and residue inputs to the soil. The fact that sorption is the main process that preserves organic matter to mineral surfaces (Christensen, 2001; Kaiser and Guggenberger, 2003) in the silt + clay fraction of the soil suggests that the SOC remaining in the soil at the greatest degradation intensity is likely to be preserved from biological decomposition.

Table 2
Correlation matrix between changes in SOC and N stocks induced by land degradation and selected soil properties.

		N change	Cov	Clay	P	K	Ca	Mg	Ex acidity	ECEC	Acid sat	pH	Zn	Mn	Cu	ρ_b	SOC	SON	MWD	PR
SOC change	1.00*																			
SON change	0.97	1.00																		
Cov	0.67	0.60	1.00																	
Clay	−0.54	−0.43	−0.88	1.00																
P	0.77	0.77	0.68	−0.43	1.00															
K	0.55	0.56	0.78	−0.61	0.73	1.00														
Ca	0.87	0.88	0.66	−0.47	0.81	0.73	1.00													
Mg	0.47	0.54	0.00	0.17	0.55	0.30	0.61	1.00												
Ex acidity	−0.66	−0.57	−0.96	0.96	−0.60	−0.76	−0.63	0.00	1.00											
ECEC	−0.36	−0.24	−0.84	0.94	−0.28	−0.51	−0.24	0.41	0.90	1.00										
Acid sat	−0.74	−0.71	−0.85	0.73	−0.81	−0.91	−0.86	−0.40	0.85	0.57	1.00									
pH	0.05	−0.03	0.65	−0.72	0.22	0.57	0.11	−0.34	−0.69	−0.77	−0.54	1.00								
Zn	0.81	0.80	0.45	−0.34	0.67	0.49	0.84	0.62	−0.46	−0.11	−0.65	−0.03	1.00							
Mn	0.88	0.88	0.64	−0.44	0.80	0.65	0.92	0.52	−0.59	−0.25	−0.76	0.04	0.81	1.00						
Cu	0.48	0.46	0.27	−0.33	0.24	0.15	0.38	0.12	−0.35	−0.25	−0.26	−0.01	0.50	0.37	1.00					
ρ_b	−0.40	−0.33	−0.48	0.39	−0.49	−0.52	−0.48	−0.31	0.49	0.32	0.53	−0.20	−0.42	−0.45	−0.13	1.00				
SOC	0.99	0.95	0.68	−0.53	0.80	0.59	0.89	0.50	−0.66	−0.35	−0.76	0.05	0.83	0.89	0.48	−0.50	1.00			
N	0.96	0.98	0.63	−0.45	0.80	0.60	0.91	0.57	−0.60	−0.25	−0.75	−0.01	0.83	0.90	0.47	−0.48	0.97	1.00		
MWD	0.67	0.64	0.68	−0.49	0.71	0.55	0.68	0.15	−0.60	−0.42	−0.67	0.17	0.58	0.71	0.22	−0.33	0.69	0.66	1.00	
PR	−0.29	−0.30	−0.55	0.34	−0.51	−0.57	−0.40	−0.13	0.45	0.32	0.53	−0.36	−0.11	−0.35	0.25	0.35	−0.30	−0.31	−0.30	1.00

SOC change, changes in soil organic carbon; SON change, changes in soil organic nitrogen; Cov, grass cover; Clay, clay content; P, phosphorus; K, potassium; Mg, magnesium; Ex acidity, exchangeable acidity; Tot cat, total cations; Acid sat, acid saturation; pH; Zn, zinc; Mn, manganese; Cu, copper; ρ_b, soil bulk density, SOC, soil carbon content, N, nitrogen content, MWD, mean weight diameter; PR, penetrative resistance.
* Significant correlation at $P < 0.05$.

4.3. Mechanism of SOC and SON stock depletion

Several mechanisms are likely to explain the depletion of SOC and SON stocks consecutive to the decrease in grass aerial cover. From the available literature, the reduction in grass aerial cover and associated decline in biomass production are likely to have a direct effect on soil stocks through the decline of organic C inputs to soils. The reduction in grass aerial cover can impact soil stocks not only directly, but also indirectly. The loss of grass aerial cover can indeed influence the dynamic nature of the plant–soil interactions through, for instance, the modification of the water cycle and the fluxes of other elements. Hiltbrunner et al. (2012) in Swiss sub-alpine grassland observed an increase in soil bulk density by as much as 20%, with associated changes in soil functions such as biomass production. The associated changes in soil porosity tend to decrease soil infiltration by water, thus potentiating SOC and SON losses by water erosion. Podwojewski et al. (2011) and Mchunu and Chaplot (2012) by using rainfall simulation at the same site found that the decrease of grass aerial cover from 100 to 5% decreased soil infiltration from 21.6 to 6 mm h^{-1}, with an associated increase in SOC losses by 213% from Cov100 to Cov5, lost via erosion processes in particulate forms.

The disruption of soil aggregates through the process of either water erosion or trampling by the livestock constitutes another likely mechanism of SOC and SON stock depletion. Soil aggregates undeniably serve as physical protection for organic matter through a range of interactions from inclusion to sorption (Baldock and Skjemstad, 2000; Jastrow, 1996; Masiello, 2004; Mikutta et al., 2006; Tisdall and Oades, 1982; Torn et al., 1997). The disaggregation process results in SOC losses from soils either through organic matter decomposition and associated CO_2 emissions to the atmosphere as more SOC becomes unprotected, or through preferential SOC erosion, owing to the light nature of soil organic matter. Similarly, the release of the hitherto encapsulated organic material can follow the mechanical breakdown of aggregates during trampling by cattle.

According to the conceptual model of Kinnell (2001), water erosion can breakdown soil aggregates through (1) the physical action of raindrop impact (splash); (2) the combined action of splash and shallow runoff on the soil surface, which increases the disaggregation efficiency of raindrops; (3) and the action of the overland flow, while previous studies performed under clayey soil conditions, for instance, by Brunet et al. (2006) pointed to the absence of preferential SOC erosion. Studies under sandy soil conditions by Boegling et al. (2005) and by Mchunu and Chaplot (2012) at the same site demonstrated preferential SOC erosion, with an SOC enrichment of the exported soil material relative to the bulk soil, increasing by a factor greater than 5.

Another potential mechanism of SOC and SON stock depletion consecutive to the decrease in grass aerial cover lies into the alteration of the local micro-climate. Under similar grassland conditions in South Africa, Mills and Fey (2004) showed that land degradation is accompanied by an increase in soil temperature, which in turn increases the rate of organic matter mineralization.

While the depletion of SOC and SON stocks can be consequential to the decrease in grass aerial cover, whose origin might be either overgrazing or other land mismanagement, there remains the possibility for internal soil processes to induce the depletion of stocks. Disaggregation can not only be physical, but also chemical and biological. Tisdall and Oades (1982) postulated that primary particles (<20 μm) can be agglomerated to form micro-aggregates (20–250 μm) by persistent binding agents such as oxides, polyvalent metal cation complexes and aluminosilicates; these micro-aggregates in turn, agglomerated into macro-aggregates (>250 μm) by biological binding agents such as hyphae and roots. The biological agents tend to rapidly decompose, and when not replaced as the root density decreases, aggregates tend to breakdown, thus releasing the encapsulated SOC and SON. Leaching of oxides, metal cations and aluminosilicates could constitute a natural disaggregation mechanism in acidic sandy soils (Rienks et al., 2000), leading to SOC and SON losses with consequences and feedbacks to soil fertility and land degradation.

The present study by pointing to a sharp decrease in Zn concentration at the initial stages of land degradation highlights the importance of micronutrients in the overall grassland ecosystem functioning. Because Zn is an essential micronutrient required by plants, any depletion below a certain threshold leads to a considerable decrease in plant productivity (Alloway, 2009), potentially explaining the decrease in grass aerial cover. What could then explain the sharp initial decrease in Zn in the study soils? The decrease in nutrients could be caused by high grazing intensity and its accompanied depletion of plant cover and litter input as well as trampling of the soil (Kotzé et al., 2013; Zhou et al., 2010). The nutrient losses could also be the result of erosion and low nutrient input of plants (Snyman and du Preez, 2005; Tefera et al., 2010). In the case of zinc ions, which exist in the soil primarily as stable complexes with proteins and nucleic acids (Alloway, 2009). Soil acidification, a natural process in the region (Rienks et al., 2000) has been suggested to be the main mechanism promoting breakdown of Zn complexes, leading to Zn leaching (Cakmak et al., 1997). Therefore, the depletion of SOC and nutrient stocks could be the result of the initial and internal leaching of micronutrients, resulting in the decrease of both the protective grass cover and the soil aggregate stability, which in turn potentiates soil erosion by water and its known consequences on SOC and SON stock depletion as previously shown by Mchunu and Chaplot (2012) at the same site.

5. Conclusions

In this study of a degraded land in South Africa, our main objectives were to quantify the impact of land degradation on the depletion of SOC and SON stocks and to evaluate some of the associated mechanisms for remediation purposes.

The present study revealed a sigmoidal decrease in topsoil SOC and SON stocks with the linear decrease in grass aerial cover, with depletion rates up to 89 and 76%, respectively The stock depletion was accompanied by an increase in soil bulk density, a decrease in soil aggregate stability, a preferential enrichment in stabilized organic matter and a decrease in chemical elements such as Ca, Mg, K, Mn, Cu, and Zn essential for soil aggregate stability and plant growth.

While little evidence exists on the mechanisms responsible for the decrease in SOC and SON stocks, the sharp decline in micronutrients such as Zn, at the initial stages of land degradation, suggests that leaching of the essential aggregate binding and plant growth elements is the main cause of land degradation as well as the associated losses of SOC and SON. Further research is needed to elucidate such mechanistic linkages in order to establish effective grassland rehabilitation strategies that could replenish the depleted SOC and nutrient stocks. One limitation and therefore research priority is the consideration of other study sites to further improve understanding of the degradation mechanisms.

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On the interpolation of volumetric water content in research catchments

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ABSTRACT

Digital Soil Mapping (DSM) is widely used in the environmental sciences because of its accuracy and efficiency in producing soil maps compared to the traditional soil mapping. Numerous studies have investigated how the sampling density and the interpolation process of data points affect the prediction quality. While, the interpolation process is straight forward for primary attributes such as soil gravimetric water content (θ_g) and soil bulk density (ρ_b), the DSM of volumetric water content (θ_v), the product of θ_g by ρ_b, may either involve direct interpolations of θ_v (approach 1) or independent interpolation of ρ_b and θ_g data points and subsequent multiplication of ρ_b and θ_g maps (approach 2). The main objective of this study was to compare the accuracy of these two mapping approaches for θ_v. A 23 ha grassland catchment in KwaZulu-Natal, South Africa was selected for this study. A total of 317 data points were randomly selected and sampled during the dry season in the topsoil (0–0.05 m) for θ_g by ρ_b estimation. Data points were interpolated following approaches 1 and 2, and using inverse distance weighting with 3 or 12 neighboring points (IDW3; IDW12), regular spline with tension (RST) and ordinary kriging (OK). Based on an independent validation set of 70 data points, OK was the best interpolator for ρ_b (mean absolute error, MAE of 0.081 g cm^{-3}), while θ_g was best estimated using IDW12 (MAE = 1.697%) and θ_v by IDW3 (MAE = 1.814%). It was found that approach 1 underestimated θ_v. Approach 2 tended to overestimate θ_v, but reduced the prediction bias by an average of 37% and only improved the prediction accuracy by 1.3% compared to approach 1. Such a great benefit of approach 2 (i.e., the subsequent multiplication of interpolated maps of primary variables) was unexpected considering that a higher sampling density (~14 data point ha^{-1} in the present study) tends to minimize the differences between interpolations techniques and approaches. In the context of much lower sampling densities, as generally encountered in environmental studies, one can thus expect approach 2 to yield significantly greater accuracy than approach 1. This approach 2 seems promising and can be further tested for DSM of other secondary variables.

1. Introduction

Volumetric soil water content (θ_v) is a key hydrological soil physical variable that influences the hydrological response of catchments (Hawley et al., 1983; Jackson, 2006; Robinson et al., 2008). θ_v influences the partition between runoff and soil infiltration. It also controls some of the energy and gases fluxes between the pedosphere, the biosphere and the atmosphere, which greatly influence land processes such as soil water denitrification (Grimaldi and Chaplot, 2000), carbon sequestration, associated climate change and biodiversity (Asbjornsen et al., 2011). Information on θ_v is thus essential for a better understanding of ecosystem functioning and for modeling purposes.

There is a wide range of direct and indirect techniques available for assessing the spatial variability of θ_v (Robinson et al., 2008; Busscher, 2009). Direct methods are those that directly measure soil water content (θ), and the gravimetric method is the only direct technique for estimating θ. Even though it is destructive, it remains the most accurate and reliable standard method of determining soil water content (Reynolds, 1970a,b,c; Jackson, 2006; Evett et al., 2008; Busscher, 2009). In this study, we focused on the gravimetric method which basically involves taking a soil sample, weighing it before any water is lost, then oven-drying at 105 °C and reweighing it. The amount of water lost on drying is a direct measure of the soil water content. The amount of water in the soil is then expressed into two different ways either on a gravimetric basis (θ_g), as a percentage of the oven dry weight of the soil or on a volumetric basis (θ_v), by multiplying by the bulk density of the soil (Reynolds, 1970a,b,c; Evett et al., 2008). Both of these variables are related by combining the gravimetric water content (θ_g, g g^{-1}) which is calculated using

$$\theta_g = M_w/M_s \quad (1)$$

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then multiplying by the soil bulk density (ρ_b, g cm^{-3}) which is calculated using

$$\rho_b = M_s/V_s \tag{2}$$

to give the volumetric water content (θ_v, cm^3 cm^{-3}) using this relation

$$\theta_v = \theta_g \times \rho_b \tag{3}$$

where M_w (g), the mass of water lost upon oven drying; M_s (g) the mass of the dry soil; and V_s (cm^{-3}) the known volume of the soil samples. Indirect methods are those that do not directly measure θ, but rely on simple proxies as a surrogate to θ_v (Evett et al., 2008). These include, *in situ* portable measurement techniques such as time domain reflectometry (TDR) (Topp and Reynolds, 1998), ground penetrating radar (GPR) (Davis and Annan, 2002), electrical resistivity (Chaplot et al., 2001; Evett and Parkin, 2005) remote sensing (Schmugge et al., 2002) and wireless sensor networks which provide time-varying estimates of soil water content (Bogena et al., 2010).

Traditionally, the spatial variability of soil properties in the landscape was displayed on hand drawn maps by interpolating continuous surfaces through the observation points of soil properties (Burrough and McDonnell, 1998). Advances in computer technology and geographic information systems (GISs) have led to the development of digital interpolation methods such as inverse distance weighting (IDW), regular spline with tension (RST) and geostatistical methods which are proving to be powerful for interpolation of soil properties.

Much effort has been expended to test the accuracy of these techniques to spatially estimate θ. Bádossy and Lehmann (1998) compared the performance of five different geostatistical techniques: ordinary kriging (OK), external drift kriging (EDK), indicator kriging (IK), external drift indicator kriging (EDIK) and Bayes–Markov updating (BMU) to estimate the spatial variability of soil water content in a 6.3 km^2 catchment in southwest Germany. They found that the largest prediction errors were obtained for OK and IK and BMU performed reasonably better for θ. Snepvangers et al. (2003) compared two interpolation techniques: (1) a method that only uses observations of θ (spatio-temporal ordinary kriging (ST-OK); and (2) a method that combines soil water content data with auxiliary information like precipitation (spatio-temporal kriging with external drift (ST-KED) to spatially predict 229 observations of θ across 120 m^2 grassland in the south of Netherlands. They found that ST-KED method performed better than the ST-OK method. However, the ST-OK method was reported to be advantageous over ST-KED method because it requires less data and is simple to use. It is clear from the literature that there is no ideal interpolation technique applicable to each soil property, hence both traditional and geostatistical interpolation methods were evaluated here.

Even though numerous interpolation techniques have been tested to spatially predict θ_v, the best way to estimate such a secondary soil variable still needs to be determined. What is unclear is whether maps of secondary variables should be generated from direct interpolations between the data points of the secondary variable (e.g., volumetric water content, θ_v) or from the interpolated maps of its primary variables (i.e., soil gravimetric water content, θ_g and soil bulk density, ρ_b).

This study aimed to digitally map soil volumetric water content (θ_v), the product of soil gravimetric water content (θ_g) by soil bulk density (ρ_b) using two mapping approaches: one that directly interpolates the observation points of θ_v and the other one that independently interpolates observation points of ρ_b and θ_g and then multiplies the interpolated maps of ρ_b and θ_g to generate θ_v map. It was hypothesized that approach 2 would yield better results because interpolation techniques such as geostatistics are more likely to capture the spatial pattern of primary variables than of secondary variables that are by nature a combination of patterns. Four interpolation techniques were evaluated here: inverse distance weighting with 3 or 12 neighboring points (IDW_3; IDW_{12}), regularized spline with tension (RST) and ordinary kriging (OK) at a high sampling density. A 23 ha research catchment used for livestock grazing in KwaZulu-Natal, South Africa, was considered for this study.

2. Methodology

2.1. Site description

The Potshini Catchment is situated in the Bergville district of the KwaZulu-Natal Province in South Africa (Fig. 1). The mean annual precipitation over the past 30 years has been 684 mm per annum, with a potential evaporation of 1600 mm per annum and a mean annual temperature of 13 °C (Schulze, 1997). A 23 ha grassland catchment was selected within the communal grazing areas of the Potshini catchment. The geomorphology of the catchment is characterized by a relatively gentle relief with a mean slope gradient of about 15.7% and an altitude ranging from about 1381 to 1492 m.a.s.l. Gully erosion incises these sloping slopes by up to 6 m in the central part of the catchment. Bedrock consists of sandstone and mudstone of the Tarkastad Formation, Beaufort Group, shale and sandstone of the Estcourt Formation. Many dykes and sills of Karoo Dolerite are intruded in these horizontal layers giving specific weathering features of rounded boulders. The soils vary in the catchment from Gleyic Luvisols and Gleysols in lower hillslope positions (valley bottom), to red Acrisols in mid hillslope positions and Yellow Acrisols and Luvisols in upper hillslope positions (crest) (WRB, 1998). The site is characterized by grasslands classified as Northern KwaZulu-Natal moist Grassland (Mucina and Rutherford, 2006), and is predominantly used for livestock grazing.

2.2. Sampling strategy for soil water content

The soil samples for gravimetric water content analyses were randomly selected and collected in the dry season in July 2009 from the 23 ha grassland catchment. The samples were taken to a sampling depth of 0.05 m from the soil surface with 221 cm^3 metallic cylindrical cores (Blake and Hartge, 1986). This gave 317 observations, the positions of which are shown in Fig. 1. In the field, the soil samples were prepared following procedures described by Reynolds (1970a). There were limitations to the sampling strategy due to heterogeneity of the landscape and the presence of features such as gullies or dongas and rocks which rendered some of the potential sampling areas inaccessible. The bulk density samples were analyzed in the laboratory, where they were oven-dried at 105 °C for 24 h to determine the gravimetric water content.

In this study, the topographic control of soil surface moisture distribution across the 23 ha catchment was investigated. The following terrain attributes were derived from the 5-m grid-based digital elevation model (DEM) (Fig. 1): mean slope gradient (S); aspect (Asp); stream power index (SPI); compound topographic index (CTI); slope length factor (SLF); plan curvature ($Curv_{pl}$) and profile curvature ($Curv_{pr}$).

2.3. Description of spatial interpolation techniques

There are a wide variety of interpolation techniques available for interpolation and mapping of soil properties and they have been widely reported. A more detailed review of the various interpolation techniques is given by Goovaerts (1999) and Mitas and Mitasova (1999). The interpolation techniques that were used in this study possible (Mitas and Mitasova, 1999). It is therefore, flexible through the choice of the tension parameter which controls the properties of the interpolation function and the smoothing parameter which enables noise filtering (Mitas and Mitasova, 1999). A value of 0.1 and 12 points was chosen to determine the weight from the distance of the neighboring data points.

2.3.3. Kriging

Kriging is a general term that embraces several kriging approaches. The kriging method utilizes the concept of the regionalized variable theory developed by Matheron (1963). Oliver et al. (1989) provide an in-depth discussion of the theory of regionalized variable underpinning geostatistics. The kriging principle entails the determination of the weights of the variable values in neighboring data points, to estimate the soil property at a specific point (Gassner and Schnug, 2006). The kriging approach takes into account both the distance and the degree of variation between known data points (Mitas and Mitasova, 1999). The weight of each point is determined from the variogram. The great merit of the kriging method compared to other spatial interpolation methods is that its estimates are unbiased and have minimum variances (it is an exact interpolator), i.e., the kriged value at a sampling point is the measured value there and the variance is zero (Burgess and Webster, 1980; Oliver et al., 1989). The strength of kriging lies in the statistical quality of its predictions and the ability to predict the spatial distribution of uncertainty (Mitas and Mitasova, 1999). In contrast to spline, kriging is flexible and also adapts to the quantity and quality of spatial dependence exhibited by the data (Matheron, 1963). The reliability of kriging depends on how accurately the variation is represented by the selected spatial model (Webster and Oliver, 2001). In this study, ordinary kriging was evaluated. The choice of a variogram model is a prerequisite for ordinary kriging (Isaaks and Srivastava, 1989). Ordinary kriging (OK) uses a spatial continuity model to describe the statistical distance between sampled points and relies on the spatial correlation structure of the data to determine the weighting values (variogram). It takes into account the correlation between data points to determine the estimated value at an unsampled location.

Variograms were computed to determine the spatial scale and strength of θ_g, ρ_b, and θ_v pattern across the 23 ha catchment. A variogram is a geostatistical characteristic used to describe the spatial structure or spatial dependence of a data set (Atkinson and Tate, 2000; Kravchenko, 2003). The variogram is an integral geostatistical tool because it reveals the nature or pattern of topographic variation of a soil property across a landscape, by quantifying the scale and intensity of the spatial variation. Furthermore, it provides the essential spatial information for optimal estimation and interpolation, and can be used for optimizing sampling schemes (Burgess and Webster, 1980; Oliver, 1987; Oliver et al., 1989). Using the variogram, the spatial structure can be described by fitting observation data to a model. The shape of the variogram gives an indication of the spatial structure of the soil property (Warrick and Nielsen, 1980).

The first step in assessing the spatial distribution for volumetric water content was to determine whether there was a significant spatial structure (Goovaerts, 1999). In order to detect the random function that caused the spatial structure of the observations, an appropriate model was fitted to the experimental variograms using routines from a standard computer program, GS + 7.0 geostatistical software (Robertson, 2007). The data was fitted to permissible variogram models commonly used in practice such as the spherical, exponential, Gaussian which show the spatial correlation of the data, and the linear variogram model which generally indicate drift in the data (Gassner and Schnug, 2006). An isotropic distribution was selected and the parameters of the directional variogram were determined. The best fitted model from the geostatistical analysis of the data was used. Because soil properties do not vary isotropically in the landscape, to detect anisotropy, variograms were analysed for four directions of 0°, 45°, 90° and 135°. Oliver et al. (1989) recommends that in order to detect directional differences or anisotropy the variogram should be estimated in at least three directions.

The variogram parameters for characterizing the spatial structure are the nugget variance, sill, nugget/sill (N/S) ratio and spatial correlation range. It is important to differentiate between these parameters. The nugget variance represents the random variable of the data. Firstly, from the measurement error, and secondly from the spatial variability at distances smaller than the shortest sampling interval. The sill is the plateau which every bounded model will reach. The N/S ratio is defined as the proportion of short-range variability that cannot be described by a geostatistical model based on the isotropic variogram (Kravchenko, 2003). The ratio of nugget semivariance to sill semivariance (N/S) was calculated from the sill and nugget values to determine the spatial dependence within the data. If N/S < 25%, the variable is considered strongly spatially dependent; if 25% < N/S < 75%, the variable is considered moderately spatially dependent and if N/S > 75%, the variable is considered weakly spatially dependent (Kravchenko, 2003). The range is the distance of spatial autocorrelation at which the model reaches the sill or plateau (Gassner and Schnug, 2006). It is an indication of the distance over which soil property data points are spatially depended on each other (Kravchenko, 2003; Gassner and Schnug, 2006). Generally, a large spatial correlation range and small N/S ratios indicate that great accuracy can be achieved in mapping the variable of interest (Isaaks and Srivastava, 1989). The spatial interpolation techniques described above were used to map the spatial distribution of θ_v, θ_g and ρ_b across the 23 ha grassland catchment. Maps showing the spatial distribution of soil water content were generated using ArcView GIS 3.2 (ESRI, 2004).

2.4. Statistical analysis

Descriptive statistics including mean, median, variance, standard deviation (SD), skewness, coefficient of variation (CV) and standard error (SE) were computed for θ_g, ρ_b, and θ_v using 317 data points. Correlation matrix relating the soil surface variation of θ_v to terrain attributes was computed using STATISTICA 7.0 (StatSoft, Inc., Tulsa, OK).

2.5. Mapping procedure for soil water content

The mapping procedure was done in four steps (Fig. 2). Estimation of the observation points following two approaches (step 1). Interpolation of the observation points using the different interpolation techniques and following the two approaches (step 2): Map generation following the two approaches: approach 1 that entailed directly interpolating between the 247 observations of θ_v and the second approach which involved independently interpolating the 247 observations points of ρ_b and θ_g and then multiplying the interpolated maps of ρ_b and θ_g to generate a map of θ_v (step 3). Validation of the maps was done to produce final product (step 4).

2.6. Accuracy assessment

The sampled 317 data points were split into two subsets: (i) 247 data points which was used for map generation, and (ii) the remaining 70 data points were used to independently validate the maps. Validation was performed to evaluate the prediction accuracy of the interpolation techniques in spatially estimating θ_g, ρ_b, and θ_v. The performance of the different interpolation techniques was assessed by comparing the mean error (ME) and the mean absolute error (MAE) values propagated during the testing

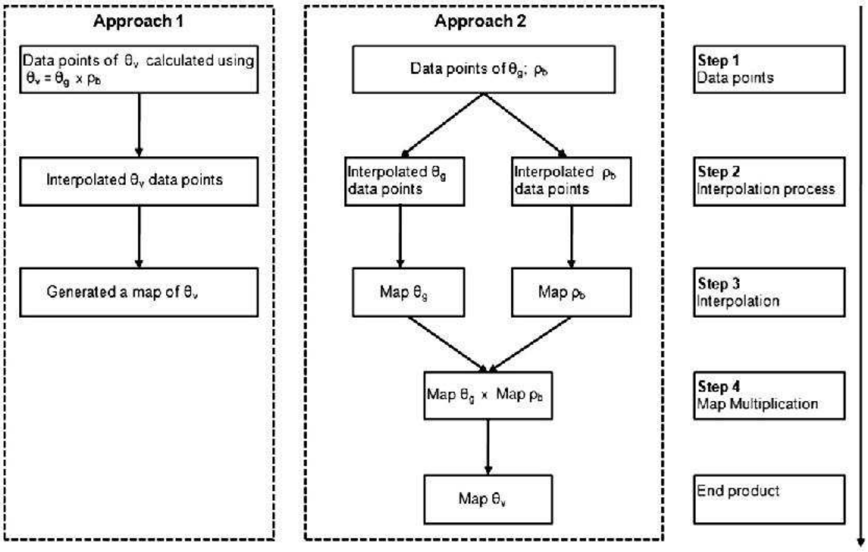

Fig. 2. Overview of mapping approaches used for estimating volumetric water content in the 23 ha Catchment.

procedure. The ME which is a bias estimator and the mean absolute error which is an accuracy predictor were used to determine how well estimated values obtained using the interpolation techniques compared to the observed values. The spatial interpolation techniques were then ranked to find which technique performed well in interpolating data for the primary variables θ_g, ρ_b and the secondary soil variable θ_v. The ME and MAE values were computed as follows:

$$\mathrm{ME} = \frac{1}{n}\sum_{i=1}^{n}\{\hat{z}(x_i) - z(x_i)\} \quad (4)$$

$$\mathrm{MAE} = \frac{1}{n}\sum_{i=1}^{n}\mathrm{ABS}\{\hat{z}(x_i) - z(x_i)\} \quad (5)$$

where $\hat{z}(x_i)$ is the predicted value at location i, $z(x_i)$ is the measured value and n is the number of samples. Notably, ME values closer to zero indicate that the interpolation technique is unbiased. The nearer the MAE values to zero the more accurate the interpolation technique.

3. Results and discussion

3.1. Descriptive statistics of the soil variables

The summary statistics of the soil variables are presented in Table 1 and Fig. 3 shows the distributions for ρ_b, θ_g and θ_v computed from the 317 observations. The mean soil bulk density (ρ_b) in the 23 ha grassland catchment was 0.83 g cm^{-3}. The mean gravimetric water content (θ_g) and volumetric water content (θ_v) were 0.16 g g^{-1} and 0.13 cm^3 cm^{-3}, respectively. Soil bulk density exhibited a coefficient of variation (CV) of 15% and θ_g and θ_v had a CV of about 28%, two times higher than that of ρ_b. Bulk density had a standard deviation (SD) of 0.12 g cm^{-3} while θ_g and θ_v had an SD of 0.05 g g^{-1} and 0.04 cm^3 cm^{-3} respectively. Additionally, ρ_b was found to be negatively skewed with a skewness coefficient of −0.25 and both θ_g and θ_v were found to be positively skewed with almost similar skewness coefficients of 0.64 and 0.61, respectively.

Table 1
Summary statistics of soil bulk density (ρ_b), gravimetric soil water content (θ_g), and volumetric soil water content (θ_v) calculated using 317 data points.

	ρ_b	θ_g		θ_v	
	g cm^{-3}	g g^{-1}	%	cm^3 cm^{-3}	%
Mean	0.83	0.16	16	0.13	13
Median	0.84	0.16	16	0.13	13
Variance	0.01	0.00	20	0.00	13
Standard deviation	0.12	0.05	4	0.04	4
Skewness	−0.24	0.64	1	0.61	1
Kurtosis	−0.45	1.94	2	2.16	2
CV	15	28	28	28	28
SE	0.01	0.00	0.25	0.00	0.21

3.2. Interpretation of spatial structure

To determine the spatial pattern of ρ_b, θ_g and θ_v across the 23 ha grassland catchment directional variograms of ρ_b, θ_g and θ_v were computed using the map generation set of 247 data points (Fig. 4). Table 2 summarizes the main parameters of the variograms. The isotropic variogram for ρ_b displays an increase in semivariance from 150 to 200 and shows a steady increase from lag distance 25 to about 230 m. It plateaus after 230 m, which marks the limit of the spatial dependence. There was no marked anisotropy between 0°, 45°, 90° and 135° direction. The N/S ratio was 50.1% which indicates a moderate spatially dependent structure. The pattern of the variograms for θ_g and θ_v is almost similar. The isotropic variogram for θ_g depicts an increase in semivariance from 12 to 22, with an increasing lag distance from 25 to up to 150 m. It plateaus after 150 m which depicts the limit of the spatial dependence. Furthermore, there was marked anisotropy which was greatest between the 90° east–west directions which correspond to steep slopes. The N/S ratio was 26.8 indicating a moderate spatially dependent structure. The isotropic variogram for θ_v exhibits an increase in semivariance from 13 to 22 with an increasing lag distance from 25 to up 150 m. It flattens after 150 m, which marks the limit of the spatial dependence. The N/S ratio was 29%

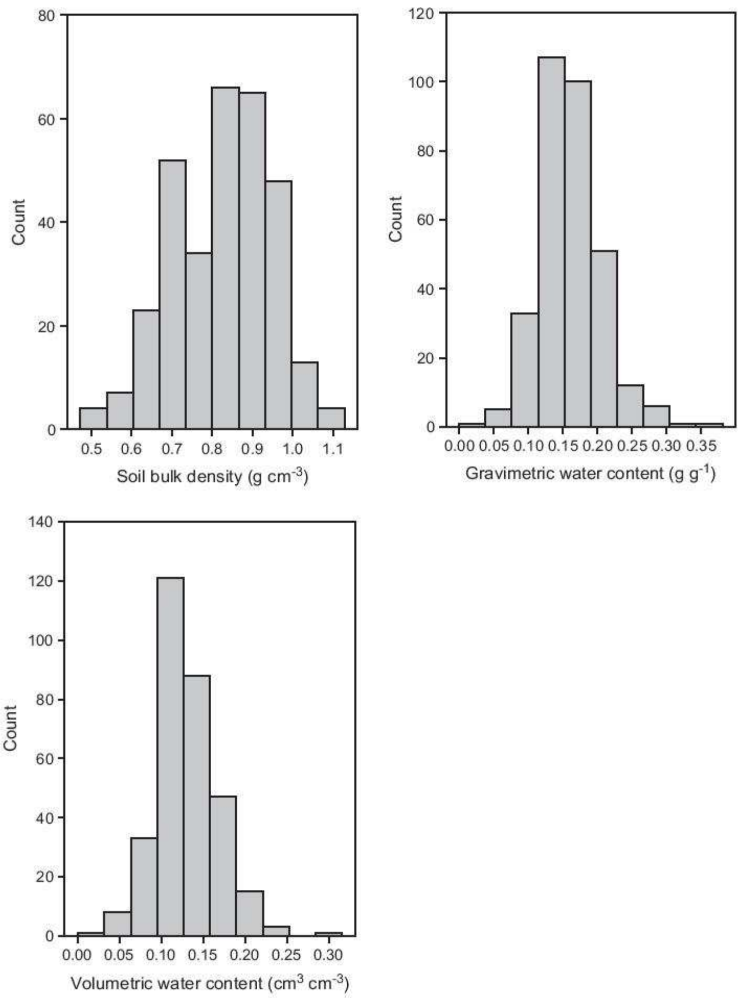

Fig. 3. Histograms depicting the distribution of soil bulk density (ρ_b), gravimetric soil water content (θ_g) and volumetric soil water content (θ_v) computed using 317 data points.

indicating a moderately spatially dependent structure. This implies that volumetric water content under driest conditions in the catchment has a significant spatial autocorrelation or dependency (the tendency or likelihood that sampled points at neighboring locations in space are much more similar to one another than those further apart) up to a distance of 150 m, and beyond this distance θ_v does not depict a significant spatial autocorrelation.

3.3. Performance of interpolation techniques in spatially estimating mapping ρ_b, θ_g and θ_v

The ME and MEA values used to evaluate the performance of the interpolation techniques in spatially estimating ρ_b, θ_g and θ_v are shown in Table 3. The lowest prediction errors for ρ_b were obtained by ordinary kriging (ME = −0.004; MAE = 0.081), inverse distance weighting for θ_g (ME = 0.377; MAE = 1. 697) and θ_v (ME = 0.180; MAE = 1.814), respectively.

While significant improvement in the interpolation of densely disposed observations can be achieved by using an optimal interpolation technique and a high sampling density, a question remains: how accurate are the maps generated either using low density networks or using models? One response that came from this study is that any attempt of mapping such a soil hydro-pedometric property (i.e., volumetric water content) should consider a minimum sampling density to draw pragmatic conclusions.

3.4. Comparison of the two approaches for mapping volumetric water content

The results of estimating the volumetric water content using both approaches 1 and 2 are shown in Fig. 6. As evident in Fig. 6 by the deviation of the estimated values from the 1:1 line, it was found that the volumetric water content values estimated using approach 2 were generally underestimated by all the interpolation

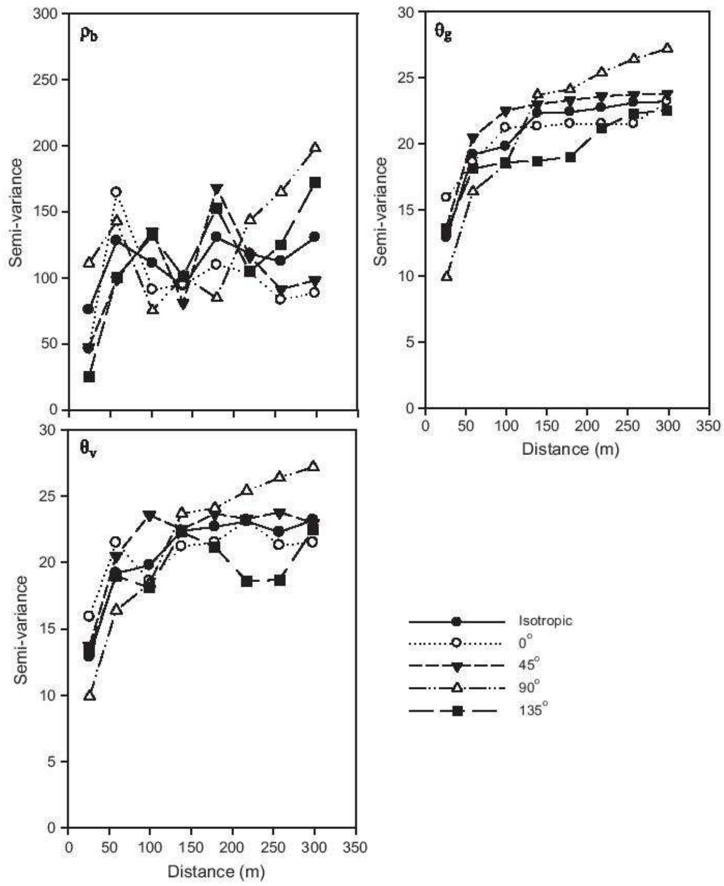

Fig. 4. Directional variograms of gravimetric soil water content (θ_g), soil bulk density (ρ_b), and volumetric soil water content (θ_v) computed using 247 data points.

Table 2
Variograms parameters, nugget, sill and range of fitted variogram models for soil bulk density (ρ_b), soil gravimetric soil water content (θ_g), and volumetric soil water content (θ_v), computed using 247 data points.

Variable	n	Model	Nugget	Sill	Nugget/sill ratio	Range	r^2
ρ_b	247	Exponential	144.8	289	50.1	1368	0.89
θ_g	247	Exponential	6.1	22.8	26.8	138	0.96
θ_v	247	Exponential	3.9	13.7	28.5	93.7	0.94

techniques and those estimated using approach 1 were generally overestimated by all four interpolation techniques. To further assess the performance of the two mapping approaches two sampling densities were investigated (20 and 50 data points were randomly selected from the 317 initial observation points). By using approach 2, the prediction bias was decreased by an average of 37% compared to approach 1 and the prediction accuracy was only improved by 1.3% compared to approach 1.

3.5. Spatial variation of soil water content

Using the best interpolation technique for each variable and following the two mapping approaches, maps showing the distribution of ρ_b and θ_v across the 23 ha catchment were produced (Fig. 5). It is evident from Fig. 4 that the spatial dependence in the variogram arises because there are patches in the catchment

Table 3
Comparison of the mean error (ME, bias estimator) and mean absolute error (MAE, accuracy predictor) for ρ_b, θ_g and θ_{v1} calculated using the different interpolation techniques and mapping approaches (θ_{v1}; approach 1 and θ_{v2}; approach 2). The ME and MAE values were computed from the 70 validation data points.

		OK	IDW_{12}	IDW_3	RST
ρ_b	ME	−0.004	−0.006	−0.006	−0.003
	MAE	0.081	0.081	0.081	0.088
	Rank	1	2	2	3
θ_g	ME	0.528	0.377	0.490	0.937
	MAE	1.810	1.697	1.654	1.982
	Rank	3	1	2	4
θ_{v1}	ME	0.469	0.182	0.180	0.540
	MAE	1.908	1.931	1.814	1.853
	Rank	3	2	1	4
θ_{v2}	ME	0.469	0.317	0.3339	0.707
	MAE	1.908	1.882	1.757	1.892
	Rank	4	2	1	3

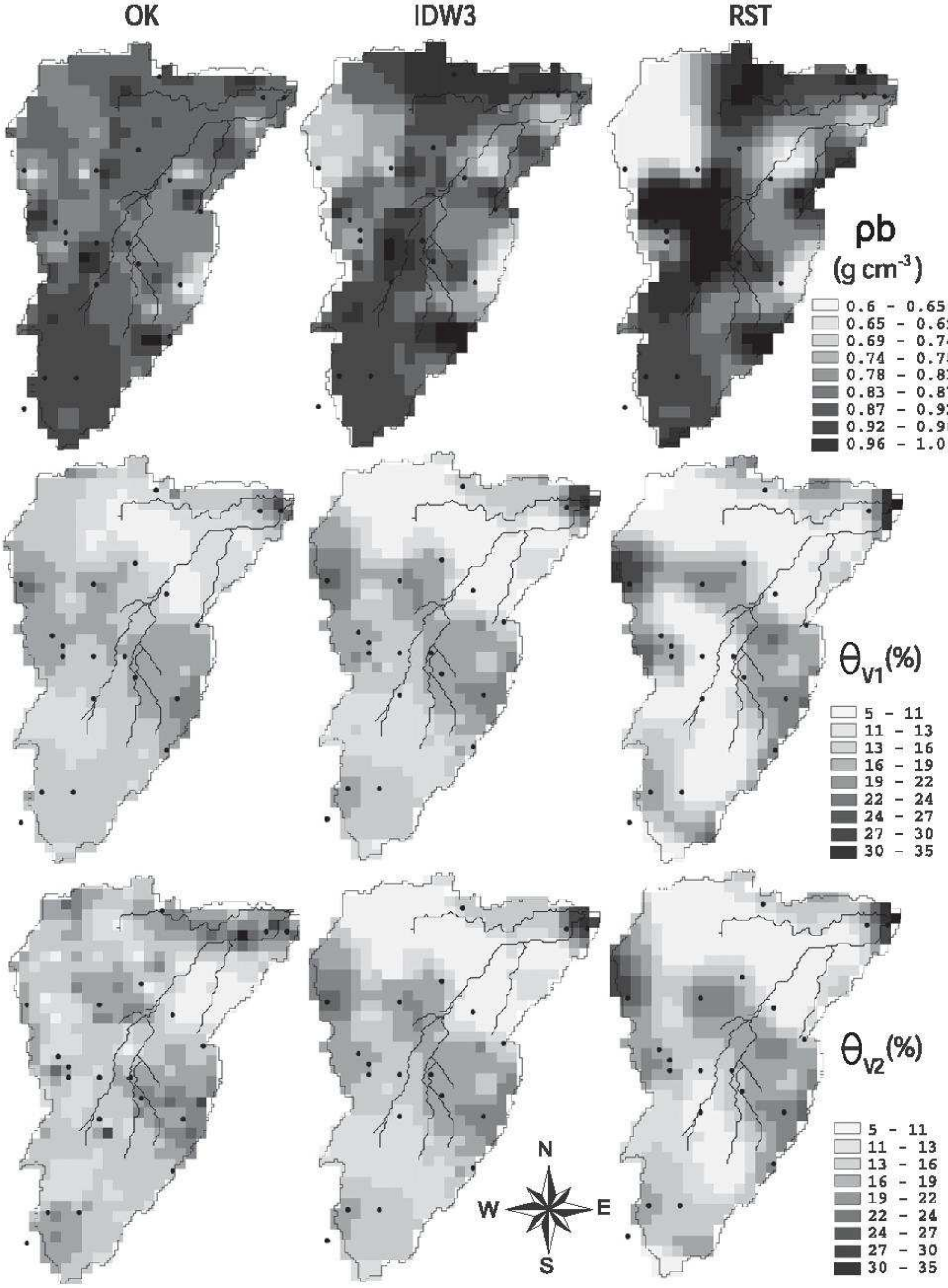

Fig. 5. Spatial distribution of soil bulk density and volumetric water content in the 23 ha grassland catchment. Maps computed using the different interpolation techniques and the two mapping approaches (θ_{v1} approach 1 and θ_{v2} approach 2).

where θ_v is high and other areas where it is low. It seems that that the variation of θ_v in the 23 ha catchment is not purely random. Indeed, volumetric water content was spatially highly variable across the catchment even under dry conditions (Fig. 5). High θ_v were also found in the lower positions of the catchment (valley bottom) where waterlogged conditions are strongly expressed

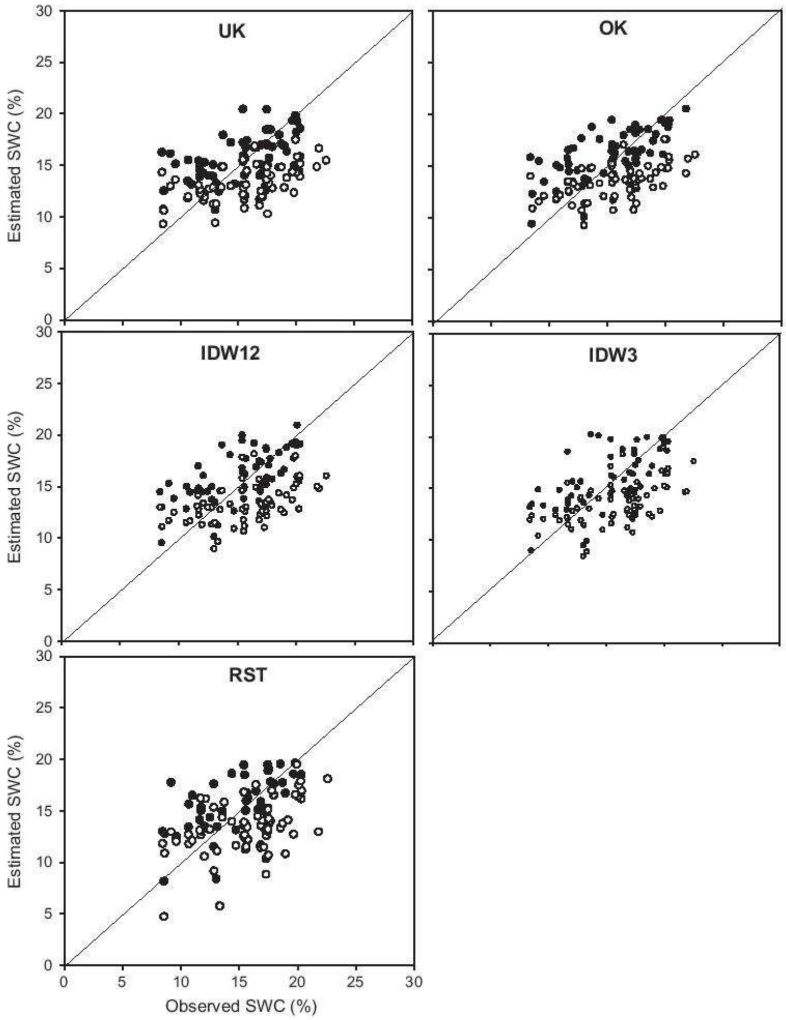

Fig. 6. Scatter plots of estimated versus observed volumetric soil water content. The plots refer to prediction by the different interpolation techniques using both approach 1 (denoted by black dots) and approach 2 (denoted by white dots) using a validation set of 70 data points. Solid lines refer to perfect estimation (1:1 lines).

and also restricted to crest locations at the mid-slope and upslope positions of the catchment. Low θ_v occurred in steep slope positions in the east–west direction of the catchment (Fig. 5). The spatial distribution of θ_v was related to topography seeking possible explanations of the variation. Topography plays a key role in the hydrological response of a catchment and has a major impact on the hydrological processes active in that landscape (Moore et al., 1991). Among the different topographical attributes, only plan curvature which is a measure of the convergence or divergence and hence the concentration of water in the landscape was found to be significantly correlated to volumetric water content (Table 4). It seems that the effect of topography on the spatial variation of θ_v is fairly weak under drier conditions. This finding is in accordance to Burt and Butcher (1985) who found topography to be an important component describing the spatial variation *i* θ_v when the soil is wet and somewhat poorly correlated during drier conditions.

Table 4
Correlation matrix of θ_v and topographical variables.

Variable	S %	Asp	SPI	CTI	SLF	$Curv_{pl}$ m^{-1}	$Curv_{pr}$
θ_v	−0.02	0.01	−0.04	0.02	−0.01	−0.18*	0.11

S, slope gradient; Asp, aspect; SPI, stream power index; CTI, compound topographic index; $Curv_{pl}$, plan curvature; $Curv_{pr}$, profile curvature.
* Marked differences significant at $p < 0.05$.

4. Conclusions

In this study of a 23 ha research catchment in South Africa, the objective was to assess the spatial variation of θ_v, a secondary variable mainly used in hydrological modeling ($\theta_v = \rho_b \times \theta_g$), using commonly used interpolation techniques, and by evaluating two mapping approaches: one that directly interpolates observation points of θ_v, as is common in practice (approach 1) and the other that independently interpolates observation points of ρ_b and θ_g and then multiplies the interpolated maps of ρ_b and θ_g to generate θ_v map (approach 2). It was hypothesized that approach 2 would yield better results because interpolation techniques such as geostatistics are more likely to capture the spatial pattern of primary variables than of secondary variables that are by nature combinations of patterns. Two main conclusions can be drawn from this study.

Ordinary kriging was found to be the better interpolation technique for soil bulk density and inverse distance weighting was the better interpolator for gravimetric water content and volumetric water content, respectively.

It was found that approach 1 underestimated θ_v. Approach 2 tended to overestimate θ_v, but reduced the prediction bias by an average of 37%, and only improved the prediction accuracy by 1.3% compared to Approach 1.

Although previous studies have shown that the prediction quality of soil properties can be obtained by (1) using high sampling densities and selecting (2) an optimal interpolation technique. What is unclear in the case of secondary variables is whether maps should be generated from direct interpolations between the data points of the secondary variable (e.g., volumetric water content (θ_v) or from the interpolated maps of its primary variables (i.e., gravimetric water content (θ_g) and soil bulk density (ρ_b). Approach 2 seems very promising alternative for improved DSM of θ_v and will need to be tested further for other secondary variables.

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Controlling factors of sheet erosion under degraded grasslands in the sloping lands of KwaZulu-Natal, South Africa

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ABSTRACT

The current increase in the global demand for food and fresh water and the associated land use changes or misuses exacerbate water erosion which has become a major threat to the sustainability of the soil and water resources. Soil erosion by rainfall and runoff is a natural and geologic phenomenon, and one of the most important components of the global geochemical cycle.

Despite numerous studies on crop lands, there is still a need to quantify soil sheet erosion (an erosion form that uniformly removes fertile upper soil horizons) under grasslands and to assess the factors of the environment that control its spatial variation. For that purpose, fifteen 1 m^2 micro-plots installed within a 23 ha catchment under pasture in the sloping lands of KwaZulu-Natal (South Africa) were monitored during the 2007–2008 rainy season to evaluate runoff (R) and sediment losses (SL). Soil losses computed from the 37 rainfall events with soil erosion averaged 6.45 ton ha^{-1} $year^{-1}$ with values from 3 to 13 ton ha^{-1} $year^{-1}$. SL were significantly correlated with the proportion of soil surface coverage by the vegetation ($P < 0.01$) whereas the slope gradient, and soil characteristics such as bulk density or clay content were not correlated. R and SL increased as the proportion of soil surface coverage decreased and this trend was used to predict the spatial variations of sheet erosion over the 23 ha catchment. Greater sheet erosion occurred at the catchment plateau and at the vicinity of gully head cuts probably in relation to regressive erosion. Mitigating sheet erosion would require an appropriate management of the soil cover through appropriate management of cattle grazing, especially at places where "natural" erosion is likely to occur.

1. Introduction

Soil erosion by water is one of the main mechanisms of land degradation worldwide (Wischmeier and Smith, 1978). Sheet erosion, through the action of raindrops and shallow running water induces the uniform removal of a thin layer or sheet' of the fertile upper soil horizon. This erosion form, long been thought to be very slow acting, is now recognised to be a major threat to the sustainability of natural ecosystems (UNEP, 1994).

Sheet erosion is thus responsible for direct and indirect consequences on ecosystems, threatening soil functions such as (1) the production of food; (2) the regulation of water flow in the landscape; (3) the function of chemical buffering and transformations of inorganic and biochemical compounds; (4) the habitat for soil organisms such as micro-organisms. Sheet erosion may also promote a greater occurrence of floods, a decrease of groundwater recharge, an increase in eutrophication of surface waters, water pollution by heavy metals and pesticides, and sedimentation in valleys and reservoirs. Overall, these consequences may potentially impact negatively on the economic development of society.

Sheet erosion involves the combined interaction of two major processes, namely, detachment of soil material by the impact of raindrops and transport of the resulting sediments by overland flow (e.g., Lado and Ben-Hur, 2004). Along with rainfall characteristics, sheet erosion is influenced by topographic conditions, soil properties and vegetation cover. For a given rainfall event, slope gradient, surface cover, vegetation canopy and vegetation density directly affect the detachment of soil material (Chaplot and Le Bissonnais, 2003; Chaplot et al., 2007; Le Bissonnais et al., 2005). The main soil properties that affect sheet erosion are soil structure, texture, clay mineralogy and organic matter content. They affect both soil

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detachability by distributing erosive forces, and water infiltration in soil. The way soil properties affect water movement and infiltrability is mainly through a control of crust formation and development (Sharma, 1996; Mermut et al., 1997; Bryan, 2000; Terrence et al., 2002).

Understanding the link between sheet erosion, its spatial variations and the environmental factors of control is a key issue for soil erosion remediation. But so far, despite numerous studies, quantitative approaches on the relationship between spatial aspects of sheet erosion and the factors of the environment are still to be developed (Lal, 2003). This is especially true at large scale (from hundreds of m^2 to several hectares) where the high spatial variability of sheet erosion cannot be explained by varying rainfall conditions.

In South Africa it is estimated that 85% of the country is threatened by land degradation and desertification (Archer, 1994). Furthermore, annual soil losses by water erosion have been estimated at approximately 400 million tons (Whyte, 1995). The focus of much erosion research in South Africa particularly in KwaZulu-Natal has been on linear erosion (Beckedahl, 1996; Rapp, 1998), the second form of water erosion that involves soil detachment by concentrated runoff. Gullies or *dongas* (South African term for gullies) the result of linear erosion, have been extensively studied because they are predominant features of erosion in the hilly areas of KwaZulu-Natal. The extensive occurrence of *dongas* generally associated with overgrazing poses serious problems on agricultural land by rendering arable and grazing land inaccessible and unusable (Rienks et al., 2000). But less attention has been given to what is perceived as less a severe mechanism of erosion, namely sheet erosion (Le Roux et al., 2007).

In South Africa, as in many areas of the world, the evaluation and prediction of soil erosion has often been conducted using the Universal Soil Loss Equation (USLE) method (Fox and Bryan, 1999). It is the most widely used method and often described as the classical soil erosion equation. In terms of the effect of slope on early USLE formulations, soil erosion was predicted as a power function of slope gradient (Fox and Bryan, 1999). In general, there is no apparent distinction between the different forms of water erosion, between sheet erosion and rill erosion. This is highlighted by Torri (1996), who explains that in the past, studies on water erosion were grouped together, which led to incorrect conclusions. Moreover, the USLE was generated in the United States of America and in other countries it has been found that the USLE over-predicts the soil loss from an area (Torri, 1996). Such is the case in areas which have differing climates characterized by sloping lands of varying gradients, which is especially true for South Africa.

Thus, it is important to evaluate the relationship between sheet erosion and the factors affecting it. Furthermore, various interactions exist between the rainfall intensity, the slope gradient, the slope length, land use and soils and these affect the runoff features for sheet erosion.

Only few studies in South Africa gathered quantitative data on sheet erosion (Snyman and Van Rensburg, 1986; Stern, 1990; McPhee et al., 1983) and more is to be done on this field of research and especially on the understanding of the environmental factors of control of sheet erosion (Laker, 2004; Le Roux et al., 2007, 2008).

In the sloping lands of the KwaZulu-Natal, cattle constitute an important part of the rural smallholders' livelihood and because it is important cultural asset, the quantity of cattle heads nowadays tends to sharply increase (Dovie et al., 2006). From preliminary investigations, this combined with highly acidic soils of low productivity, seem to have serious consequences for soil degradation (e.g., Mills and Fey, 2003; Rienks et al., 2000) and sheet erosion.

In this context, the main objectives of this study were to (1) quantify soil sheet erosion; and (2) assess the impact of some selected environmental factors on sheet erosion. This study was conducted within a small agricultural catchment of 23 ha characterized by variations of topography, soils and land management but assuming no spatial variations of the characteristics of rainfall. Fifteen 1 m^2 runoff plots installed on different soils, topographic locations and proportions of soil surface coverage by the vegetation were surveyed over an entire rainy season. By quantifying sheet erosion and its correlations to some environmental factors, this study is intended to contribute towards prioritizing and focusing intervention options that can help address important soil degradation concerns at the catchment level. Results were obtained at the Potshini catchment in the KwaZulu-Natal province, South Africa under the umbrella of the Smallholder System Innovations (SSI) project (Rockström et al., 2002) which mainly aims at improving management of soil and water resources for small holder farmers.

2. Materials and methods

2.1. Climate, geology and soils

The study area is located within the KwaZulu-Natal province, South Africa (Fig. 1). The area is a 23 ha catchment (longitude: 29.36°; latitude: 28.82°) localised in the north sloping lands of the upper Thukela basin (30,000 km^2). The climate is sub-tropical, humid and with a summer rainfall pattern (October–March) (Schulze, 1997). At Bergville, located 10 km to the east of the study site, the mean annual precipitation over the past 30 years has been 684 mm per annum, with a potential evaporation of 1600 mm per annum and a mean annual temperature of 13 °C (Schulze, 1997).

At the Potshini study site, altitude ranges from1381 to 1492 m asl. The relief is relatively gentle with a mean slope gradient of about 15.7% but with steep slopes of 50–70% found in the lower part of the catchment whereas in the vicinity of the catchment outlet and on the plateau, the topography is relatively flat. Soils are formed from the Karoo Supergroup and Beaufort Group parent materials. The geology exhibits a horizontal, alternating, succession of fine-grained sandstone, shale, siltstone and mudstone (King, 2002). A main dyke from the Karoo Dolerite is intruded in these horizontal layers in the upper catchment giving specific weathering features of rounded boulders.

Soils developed from sandstones and dolerites are Acrisols (ISSS Working Group, 1998) and Inanda soil form (Soil Classification Working Group, 1991). Within hillslopes, deep Acrisols (~2 m) characterize footslopes and bottomlands. Bottomlands exhibit features of waterlogging such as a surface dark grey (2.5YR4/1) A

Fig. 1. Landscape morphology at the Potshini catchment of the KwaZulu-Natal province (South Africa) and water erosion features.

Table 1

Characteristics of soil horizons from soil profiles at different landscape position ID: identification of soil profile and number of horizon from soil surface to depth; depth (cm): depths of above and below horizon limit; clay: soil clay content (%); fine silt content (%); coarse silt content; sand content; pH KCl; pH water; C: soil organic carbon content; N: nitrogen content; S: sulphur content; ρ_b: soil bulk density.

ID	Depth (cm)	Clay (%)	Fine silt (%)	Coarse silt (%)	Sand (%)	pH KCl	pH water	C (%)	N (%)	C/N	S (%)	ρ_b
F1	0–10	27.9	15.3	6.9	50.0	4.5	5.5	1.48	0.11	13.5	0.01	1.23
F2	10–25	29.9	14.8	7.3	48.0	4.3	5.6	1.20	0.09	13.3	0.00	1.27
F3	25–40	31.9	13.9	6.6	47.7	4.3	5.7	0.89	0.08	11.1	0.00	1.3
F4	40–55	35.3	12.0	5.9	46.8	4.3	5.7	0.49	0.04	12.3	0.00	
F5	55–90	36.6	10.5	6.5	46.3	4.1	5.6	0.26	0.03	8.7	0.00	
F6	90–170	40.2	12.1	6.2	41.5	4.0	5.7	0.00	0.01		0.00	
F7	170–220	44.3	7.5	7.0	41.2	4.0	5.7	0.00	0.01		0.00	
F8	220–285	46.7	7.2	6.5	39.6	4.3	6.1	0.00	0.00		0.00	
F9	285–300	28.9	14.5	8.7	48.0	4.6	5.9	0.00	0.00		0.00	
M1	0–14	27.4	13.6	6.4	52.6	4.5	5.7	1.21	0.10	12.1	0.01	1.19
M2	14–30	34.2	12.4	4.6	48.8	4.2	5.5	1.00	0.06	16.7	0.00	1.27
M3	30–46	36.9	10.4	5.8	46.9	4.2	5.5	1.04	0.08	13.0	0.18	1.21
M4	46–60	39.8	10.6	5.5	44.1	4.1	5.4	0.48	0.06	8.0	0.00	1.23
M5	60–90	43.4	12.4	3.8	40.5	4.0	5.4	0.16	0.05	3.2	0.00	1.23
M6	90–120	42.1	12.7	4.9	40.3	4.0	5.4	0.02	0.01	2.0	0.00	
T1	0–10	39.8	24.8	3.9	31.5	4.9	5.9	2.84	0.21	13.5	0.02	0.96
T2	10–20	42.7	19.6	4.9	32.9	5.0	6.1	2.26	0.15	15.1	0.01	1.1
T3	20–45	40.2	22.0	4.0	33.8	4.7	5.9	1.73	0.12	14.4	0.01	
T4	45–170	54.5	13.5	3.5	28.6	4.8	6.0	0.08	0.04	2.0	0.00	
T5	170+	15.3	15.5	4.5	64.8	4.3	6.5	0.00	0.00		0.00	
SD1	0–10	53.9	22.1	6.4	17.6	4.1	5.4	3.67	0.23	16.0	0.01	0.96
SD2	10–25	56.4	17.2	7.2	19.1	4.2	5.8	2.32	0.14	17.7	0.02	0.94
SD3	25–50	60.3	14.3	6.2	19.2	4.2	5.1	0.97	0.05	19.4	0.00	0.88
SD4	50–140	64.4	15.2	7.0	13.4	4.1	5.2	0.10	0.02	5.0	0.00	0.89
SS1	0–10	30.5	12.1	7.3	50.2	4.0	4.9	1.77	0.12	14.8	0.01	1.15
SS2	10–25	35.0	10.6	8.1	46.3	4.1	4.9	1.26	0.08	15.8	0.00	1.18
SS3	25–40	35.1	8.9	7.5	48.5	4.2	5.0	0.79	0.05	15.8	0.00	1.09
SS4	40–60	38.7	10.5	4.9	45.9	4.1	5.0	0.37	0.08	4.6	0.00	
SS5	60–85	38.5	11.6	8.5	41.4	4.0	5.0	0.16	0.08	2.0	0.00	

horizon, enriched in organic matter and the presence of redoximorphic features (Soil Survey Staff, 1999) from the soil surface to 2 m depth (Table 1). These soils show a massive structure. Horizons are compact (1.4 < BD < 16), except the A horizon with a BD of 0.8. At the footslope position, the soils are well drained. The humiferous A horizon is dark reddish brown (5YR 3/3), blocky and friable. The Bw horizon, from 0.4 to 0.9 m is dark reddish brown (2.5YR3/4), massive and clayey. A sandy saprolite is reached at about 1.7 m. The midslope position exhibits a similar soil profile but much shorter, the Bw being found between 0.3 and 0.6 m and the saprolite from 0.9 m. The soils at the terrace (T) and the shoulder (SD) developed from dolerites. Both soil profiles show a dark reddish brown (5YR 3/3) A horizon with a clear fine angular blocky structure. The Bw is red (2.5YR 4/6) and is 0.9 m thick (0.5–1.4 m) at T and 1.25 m thick (0.45–1.7 m) at SD. A sandy red (10R4/8) saprolite was reached below followed by a brownish yellow (10YR 6/8) at about 2 m deep. The T situation differs from the SD by a high proportion of blocks from the A horizon to the saprolite. Finally, the SD situation shows a 0.1 m brown (7.5YR 4/4) friable humiferous horizon with a substructure fine angular blocky. The Bw horizon found from 0.5 to 1.4 m is yellowish red (5YR5/7), friable and massive. It exhibits a sharp limit with a cohesive saprolite of sandstones.

Determination of soil bulk density (ρ_b) was performed just before the rainy season. The sampling was performed adjacent to each of the 15 micro-plots in order to avoid disturbance of the micro-plot soil surface. Three soil sample replicates per micro-plot consisting of undisturbed soil cores (250 ml) were collected within a radius of less than 1 m. Samples were oven-dried at 105 °C for 24 h for SW and BD determination while a composite sample was collected and sieved to obtain the clay fractions (<2 μm; AFNOR, 1983). The particle size distribution was determined by the pipette method (Gee and Bauder, 1986). The pH was measured in both distilled water and KCl using a Calimatic PHM766 pH meter, using a solution ratio of 1:2.5 (10 g soil: 25 ml solution). Finally, the cation exchange capacity (CEC) and the exchangeable cations (Ca^{++}, Mg^{++}, Na^{+}, K^{+} and Al^{3+}) were determined by exchange with cobalthexamine cation at natural soil pH (AFNOR, 1983).

2.2. Evaluation of topographic conditions and soil surface features

The 15 micro-plots were installed at different positions within the study catchment. The topographic information (altitude, mean slope gradient) was derived from a DEM with a 5-m resolution with a vertical precision of 0.05 m. This DEM has been generated using 20,000 data points gathered within the 23 ha catchment with a differential GPS and interpolated by using the inverse distance weighting function of ArcView3.2 (ESRI, 2004). From this DEM the altitude above sea level of each micro-plot was extracted using the DEMAT function of ArcView3.2. The mean slope gradient of the eight cells surrounding each plot was estimated using the same ArcView function.

The study catchment is predominantly grazed grassland. Cattle are seen as an important cultural asset and in good times, community members, both those living in the catchment and those who work and live in the major cities, invest in cattle, leading to increasing herd sizes. This, combined with the highly acidic low productive soils, rapidly leads to overgrazing with decreasing proportion of soil surface coverage by the vegetation and associated increase of bare soils.

The proportion of the soil surface covered by the vegetation at each micro-plot has been characterized visually in the field by qualitative assessment using a 1 m^2 grid with 100 cm^2 cells (e.g., Auzet et al., 2004). The proportion of the soil surface with crusts has been also considered because of its recognised impact on runoff generation and sheet erosion (e.g., Casenave and Valentin, 1992). Casenave and Valentin (1992) using studies from semi-arid and arid Africa indicated that surface crusts significantly decrease water infiltration in soils. From the nine main types of crust they identified only three types, i.e., structural (*rough surface made of coalescing partially slaked aggregates*), erosion (*smooth surface made of a single seal of fine cemented particles*) and sedimentary (*laminated with layers of different texture*) types were observed at the study site. Following the field methodology of Casenave and Valentin (1992) the type of crust has been recognised in the field using a knife and a magnifying glass and its proportion on the soil surface was evaluated using the 1 m^2 grid. The proportion of each crust type was afterwards summed to obtain the proportion of the soil surface with crusts.

A total of 200 field observations were performed throughout the catchment. Information was obtained at the 15 micro-plots and at 185 randomly selected points whose location was captured in the field using the differential GPS with a 0.2 m lateral accuracy. GIS layers for the proportion of soil surface coverage and crusting were afterwards interpolated using ArcMap and ordinary kriging which is well adapted for lower sampling densities (McBratney and Webster, 1983).

2.3. Evaluation of in situ sheet erosion

Fifteen 1 m^2 runoff plots were installed within the catchment at different topographical positions, soil type and intensity of overgrazing with three micro-plot replicates per sample site (Fig. 2). These included deep Acrisols at footslope (F); shallow Acrisols at midslope (M); shallow Acrisols at terrace (T); deep reddish Acrisols at shoulder under dolerite (SD); and deep yellowish Acrisols at shoulder under sandstone (SS).

The metal borders surrounding the micro-plots were inserted to a depth of 0.1 m in the soil. Field measurements were carried out from September 15th 2007 (the first rain of the season) to April 1st 2008 (the last seasoned rainfall event). Measurements were assumed to occur under conditions of steady-state soil losses because no significant soil cracking or features of linear erosion were observed within the micro-plots. After each rainfall event, the total runoff volume (*R*) from each micro-plot replicate and collected in a collector (Fig. 2) was measured with a measuring cylinder. The precision was ±10 ml for total runoff volumes between 10 and 2000 ml and ±10 ml at higher runoff volumes. A runoff sample of 800 ml was taken for each event and oven-dried to estimate sediment concentration (SC) in the runoff. The total sediment losses (SL) from the micro-plot was calculated as the product of *R* by SC.

During this study, a total of 555 samples were collected from 37 rainstorm events. Rainfall event characteristics such as rainfall amount and maximum and average rainfall intensity were estimated using an automatic rain gauge with a 6-min step counter located at the study site.

3. Results

3.1. Quantification of sheet erosion

The rainy season was characterized by 37 rainfall events causing sheet erosion with a cumulative amount of 766 mm of the total yearly amount of 850 mm.

The average runoff coefficient (*R*) computed from the 37 erosive rainfall events and the 15 micro-plots was 23%. *R* which exhibit an average between 16.5 and 32% significantly (*P* level < 0.05) varied between micro-plots (Table 2). Lower *R* occurred at the terrace (T), at the footslope (F), and at the shoulder on dolerite (SD), whereas greater *R* occurred at the midslope (M) and at the shoulder on sandstones (SS).

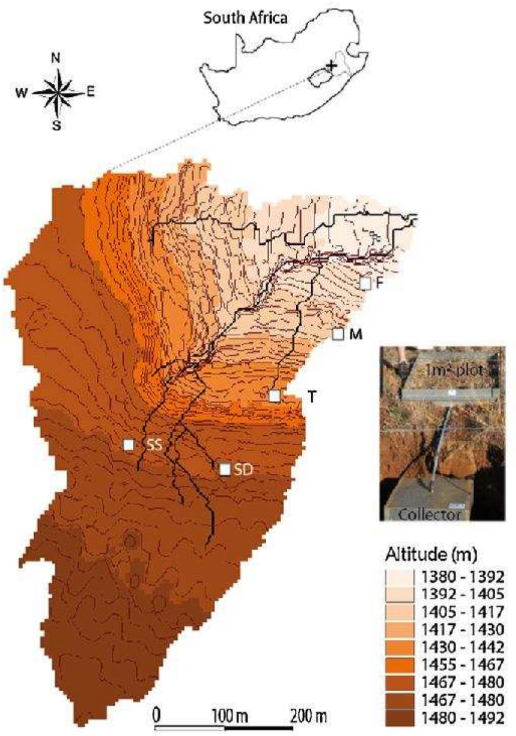

Fig. 2. Digital elevation model (DEM) with a 5-m mesh of the study catchment showing linear erosion features (in black lines) and location of 1 m^2 micro-plots (in white squares) for the evaluation of sheet erosion. Location of the five sites under study—F: deep Acrisol at footslope; M: shallow Acrisols at midslope; T: shallow Acrisols at terrace; SD: deep reddish Acrisols on dolerite at shoulder; SS: deep yellowish Acrisols on sandstones at shoulder. (For interpretation of the references to colour in this figure caption, the reader is referred to the web version of the article.)

The sediment concentration in runoff (SC) varied between 2.3 g l^{-1} at SD to 5.6 g l^{-1} at M with an average at 4.1 g l^{-1}. Significantly higher SC occurred as well at T (5.0 g l^{-1}). These were followed by SS (4.0 g l^{-1}) and F (3.4 g l^{-1}). It is interesting to note that greater SC corresponded to high *R*, except at position T where a high SC was found despite low *R* (Table 2).

Average soil losses (SL) were on average 20.8 g m^{-2} event^{-1} with values between 9.4 and 35.7 g m^{-2} event^{-1}. SL were the greatest at M position followed by SS (26.9 g m^{-2} event^{-1}), T and F (respec-

Table 2
Average runoff (*R*), sediment concentration (SC), and soil losses (SL) for the different sites (F: deep Acrisol at footslope; M: shallow Acrisols at midslope; T: shallow Acrisols at terrace; SD: deep Acrisols at shoulder under dolerite; SS: deep Acrisols at shoulder under sandstones). Data computed from the 37 rainfall events of the 2007–2008 rainy season.

Treatment	*R*		SC (g l^{-1})	SL (g m^{-2})
	ml	%		
F	4242	17.7	3.4	14.4
M	6328	30.4	5.6	35.7
T	3547	16.5	5.0	17.7
SD	4050	17.8	2.3	9.4
SS	6770	32.6	4.0	26.9
Average	4988	23	4.1	20.8

Table 3
Mean ($n=3$) slope gradient (S), proportion of crust ($Crust$), soil surface coverage (Cov), 0–05 m soil clay content ($Clay$), and 0–05 m soil bulk density (ρ_b).

Treatment	S (%)	$Crust$ ($Mg\,m^{-3}$)	Cov	$Clay$	ρ_b
F	25	11.7	88	27.8	1.24
M	26	12.3	74	27.4	1.32
T	23	1.0	85	39.8	1.13
SD	18	8.0	87	53.9	1.02
SS	19	15.0	83	30.5	1.12
Average	22	9.6	84	35.9	1.17

tively 17.7 and 14.4 $g\,m^{-2}$ $event^{-1}$). Finally, with an average of 9.4 $g\,m^{-2}$ $event^{-1}$, SL was 3.8 times lower at SD than at M, which was significant at the $P<0.05$ level.

The resulting yearly losses of sediments would range between 335 and 1300 $g\,m^{-2}\,year^{-1}$, for SD and M, respectively with an average at 645 $g\,m^{-2}\,year^{-1}$ (i.e., 6.45 $ton\,ha^{-1}\,year^{-1}$ or 0.64 $mm\,year^{-1}$).

3.2. Relationship between soil sheet erosion and environmental factors

The average slope gradient of micro-plots was 22% with values between 18 and 26% (Table 3). M plots were on the steepest slope and plots on the shoulder were on more gentle slopes. The average proportion of the soil surface under crusting (*Crust*) was 9.6%. *Crust* varied between 1% at T to 15% at SS. Lower *Crust* values occurred on dolerite, both at the shoulder and terrace positions whereas a much higher occurrence of soil surface crusting characterized soils developed from sandstones at F, M, and SS positions (Table 3).

The average soil surface coverage by the vegetation (*Cov*) was 84%. Greater *Cov* occurred at F (88%) and SD (87%). With 85 and 83%, T and SS were of intermediary *Cov* values. Finally, M was the situation with the lowest *Cov* (74%).

SD plots showed highest clay content (*Clay*) but lower soil bulk density (ρ_b) whereas M and F plots were significantly more compacted (ρ_b of respectively 1.32 and 1.24 $Mg\,m^{-3}$) and exhibited the lowest clay content among micro-plots (Table 3).

Table 4 revealed that R significantly correlated with Cov and ρ_b. With a Pearson r coefficient of −0.68 with Cov and −0.71 with ρ_b, R increased as Cov decreased and ρ_b increased. In contrast, the correlation between R and S, *Crust* and *Clay* was not significant. Consideration of the relationship between SL and the environmental factors under study indicated that Cov and ρ_b were significantly correlated with SL. But S, *Clay*, and *Crust* had no significant impact on SL. *Cov* appeared to be more highly correlated to SL ($r=-0.87$) than it was to R ($r=-0.68$). Such a strong relationship between SL and *Cov* modeled by Eq. (1) is displayed in Fig. 3:

$$SL = -45.84 \times Cov + 4613, \quad r^2 = 0.76 \tag{1}$$

Table 4
Correlation matrix between erosion variable and environmental factors with R: runoff coefficient; SL: soil losses; S: slope gradient; $Crust$: proportion of crusting on the soil surface; Cov: proportion of soil surface coverage by the vegetation; $Clay$: soil clay content; and ρ_b: soil bulk density ($n=37$). Significant correlations at $P<0.05$.

	R	SL	S	$Crust$	Cov	$Clay$	ρ_b
R	1.00						
SL	0.81	1.00					
S	0.49	0.45	1.00				
$Crust$	0.44	0.43	0.24	1.00			
Cov	−0.68	−0.87	−0.34	−0.87	1.00		
$Clay$	−0.50	−0.52	−0.88	−0.36	0.40	1.00	
ρ_b	0.71	0.70	0.89	0.45	−0.62	0.16	1.00

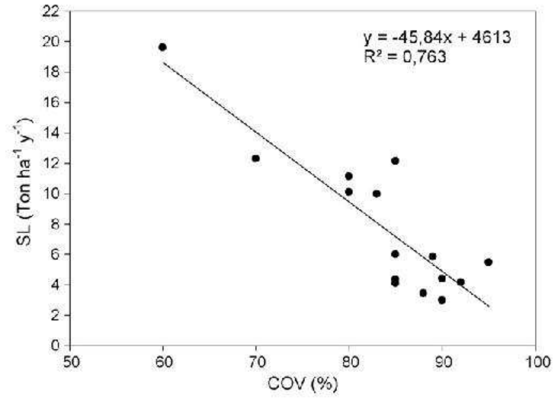

Fig. 3. Relationship between annual soil losses due to sheet erosion (SL) and the proportion of soil surface coverage (Cov) from the 15 micro-plots.

The observed data revealed that SL significantly decreased from about 10 $ton\,ha^{-1}\,year^{-1}$ for $Cov=60\%$ soil coverage to less than 6 $ton\,ha^{-1}\,year^{-1}$ at $Cov=93\%$.

The two first axes of the principal components analysis (PCA) between S, $Crust$, Cov, $Clay$, and ρ_b as variables for the analysis, explained 98% of the variability in data (respectively 75% for axe 1 and 23% for axe 1). From this PCA displayed in Fig. 4 it is noted that the two variables most highly correlated to axis 1 are Cov (contribution to this axis of 0.91) ρ_b ($r=-0.83$) and $Crust$ ($r=-0.64$). From this axis with a trend associated with soil surface degradation and soil compaction it is concluded that as Cov decreases and $Crust$ increases, SC and SL increase. R is correlated with axis 1 ($r=0.68$) but also with the second PCA axis ($r=0.73$) with a trend associated with $Clay$. Finally as shown by the PCA, S had only a small contribution to the two first axes.

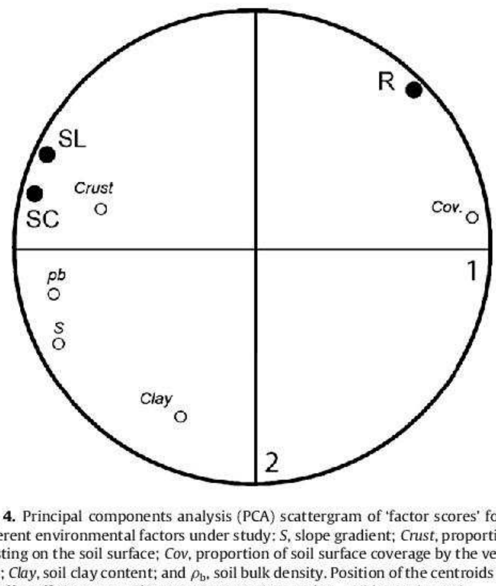

Fig. 4. Principal components analysis (PCA) scattergram of 'factor scores' for the different environmental factors under study: S, slope gradient; $Crust$, proportion of crusting on the soil surface; Cov, proportion of soil surface coverage by the vegetation; $Clay$, soil clay content; and ρ_b, soil bulk density. Position of the centroids for R, runoff coefficient; SC, sediment concentration; and SL, soil losses ($n=37$).

4. Discussion

4.1. Rate of soil material displaced by sheet erosion

The average soil losses computed from the micro-plots was 6.45 ton ha^{-1} $year^{-1}$. It is important to compare data of sheet erosion from the sloping lands of the KwaZulu-Natal region with those obtained countrywide. Losses at our study site were much higher that the 3 ton ha^{-1} $year^{-1}$ estimated by Whyte (1995).

Moreover, this rate was lower than those observed in the sloping lands of Asia, where losses of between 6 and 24 ton ha^{-1} $year^{-1}$ were measured in Laos (Chaplot et al., 2007) and between 6 and 33 ton ha^{-1} $year^{-1}$ in Thailand (Janeau et al., 2003). However, it must be noted that these results were obtained under generally steeper slope conditions with an average slope gradient of 45% in Laos (Chaplot et al., 2007).

Considering that an erosion rate of 6.45 ton ha^{-1} $year^{-1}$ corresponds to an ablation rate of 0.64 mm $year^{-1}$ – is such an ablation rate going to significantly change soil thickness over time? To respond to that question, the changes in soil thickness over time depend not only on the ablation rate but also on the production of soil from the weathering of bedrock and soil replacement through natural surface erosion. The available literature suggests that rock weathering rates for similar bedrocks may range from 0.0036 to 0.077 mm $year^{-1}$ for sandstones of the Tennesse valley (Heimsath et al., 1997) and the pacific North West of the US (Dethier, 1986), respectively. Weathering rates for basalts, a parent material with similar characteristics than dolerites might average 0.0003 mm $year^{-1}$ (Pillans, 1997). Considering that and assuming soil replacement to be close to zero at the landscape level, the net average yearly soil ablation would be of 0.49 mm $year^{-1}$ for F, 1.28 mm $year^{-1}$ for M, 0.65 mm $year^{-1}$ for T, 0.34 mm $year^{-1}$ for SD and 0.95 mm $year^{-1}$ for SS. Thus, net ablation due to sheet erosion ranges from 13×10^2 to 21×10^4 times higher than the natural soil formation. After 100 years the thickness of the soil would be reduced by 0.03–0.13 m. This amount may seem relatively low but should be considered in that the context that the few centimetres of the soil at the surface contain most of the soil organic matter and nutrients.

4.2. Controlling factors of sheet erosion

In this study, we showed the predominant impact of soil surface coverage by vegetation on soil losses, and found no correlation between sheet erosion and slope gradient and some of the other soil characteristics under study.

As expected, sheet erosion decreased with increasing soil surface coverage. This is because as root density increases, soil infiltration increases. In addition, as the vegetation density increases on the soil surface, the protection of soil aggregates from slaking and transport may increase, thus reducing sheet erosion (Wischmeier and Smith, 1978). The results of this study thus confirm previous work by Gafur et al. (2003) in Bangladesh who recorded a sheet erosion rate of 3 ton ha^{-1} $year^{-1}$ under fallow and 18 ton ha^{-1} $year^{-1}$ under bare soil conditions.

This investigation found no correlation between slope gradient and soil loss. Generally, the relationships were either positive or negative, but it is seldom reported that slope gradient has no effect. Experiments where sheet erosion increased with increasing slope gradient are numerous. For instance, De Ploey et al. (1976), Sharma et al. (1983) and Djorovic (1980) attributed such an increase to a decrease in depressional storage and ponding depth. Very few studies have shown that sheet erosion decreases with increasing slope. For instance, Govers (1990) reported that slope had a significant negative effect which was attributed to differential soil cracking, Poesen (1984) to a thinning of the crust, Bryan and Poesen (1989), and Slattery and Bryan (1992) to the erosion of low conductivity crusts, and Janeau et al. (2003) to differential soil crusting.

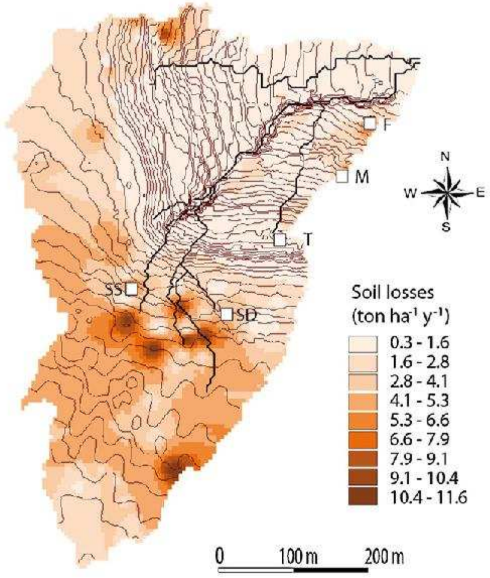

Fig. 5. Spatial variations of annual soil losses due to sheet erosion in 2007–2008 estimated using Eq. (1). Location of linear erosion features (in black lines) and location of 1 m^2 micro-plots (in white squares).

In line with Lal (1976) and Mah et al. (1992) this study did not find any significant effect of slope gradient on runoff and this is probably best explained by the low range of slope gradients investigated (about 10%).

The apparent lack of impact of soil clay content on sheet erosion was also unexpected.

Soils with low clay content generally generate more runoff due to weaker aggregates leading to the formation of a surface crust of lower infiltration (Valentin and Bresson, 1992). Although crusted surfaces induce more runoff, they usually reduce soil sheet erosion due a lower particle detachment associated with an increased hardness (e.g., Valentin and Bresson, 1992). This crust may also protect the soil from further soil detachment as result of lower surface roughness and greater bulk density (Valentin and Bresson, 1992). In this context, soil surface coverage by vegetation plays a key role in the control of sheet erosion and thus this factor overrides the effect of the other environmental factors.

4.3. Spatial variations of sheet erosion

Considering soil surface coverage (*Cov*) explained 76% of the variance in sheet erosion. The map of *Cov* generated using the 200 field observations was used to evaluate the areas of the catchment where erosion might be the more severe and thus where remediation efforts should be focused. Lower SL in 2007–2008 are likely to have occurred in the lower catchment, within the bottomland area where almost 100% of soil surface coverage by the vegetation is found (Fig. 5). In contrast, greater SL (>4 ton ha^{-1} $year^{-1}$) might have occurred on the upper slopes of the catchment, and especially in the vicinity of gully head cuts, probably in relation to regressive erosion. Overall, the soil erosion over the catchment estimated from Eq. (1) would amount to 64.17 tonnes in 2008 which corresponds to 2.8 ton ha^{-1} $year^{-1}$. This amount is significantly lower than that obtained from the micro-plot average and might be explained by an over sampling of micro-plots in degraded areas. Extrapolating the result of relatively few 1 m^2 micro-plots to an entire catchment should thus be performed with caution.

5. Summary and conclusions

In this study, our main objective was to quantify water sheet erosion (i.e., erosion by splash and shallow runoff) within a grassland catchment subjected to overgrazing and to evaluate the main controlling factors. The results obtained on 1 m^2 micro-plots installed at different topographical positions, soils and intensity of overgrazing showed an average soil loss of 0.645 kg m^{-2} that would correspond to 6.45 ton ha^{-1} $year^{-1}$. When extrapolating SL obtained at the m^2 level to the whole catchment using a regression model that explained 76% of the variability in data, SL would reduce to 2.8 ± 0.67 ton ha^{-1} $year^{-1}$.

This amount is almost similar to the USDA tolerable soil loss rate in cropping systems of 2.5 ton ha^{-1} $year^{-1}$ (USDA, 1995). However, the net ablation due to sheet erosion is from 13×10^2 to 21×10^4 times higher than the estimated natural soil formation through bedrock weathering. Moreover, because soils are shallow and are under pasture, most of the nutrients and organic matter accumulate in the surface layer of the soil. Projections for the next 100 years, indicate that from 3 to 13 cm of the fertile-A horizons will be removed and this constitutes a major threat to the soil resource and the sustainability of agriculture.

The second conclusion is that sheet erosion was shown to be highly correlated to the proportion of the soil surface covered by vegetation. Mitigating sheet erosion might thus be obtained by a greater soil surface coverage. By contrast, any further decrease of soil surface coverage would dramatically increase sheet erosion. For instance, the decrease of the soil surface coverage from an actual observed average value of 94% by 10% would increase sheet erosion by 280% (7.7 ton ha^{-1} $year^{-1}$ vs. 2.8 ton ha^{-1} $year^{-1}$). Decreasing soil surface coverage to 50% would result in soil losses of 23 ton ha^{-1} $year^{-1}$, i.e., a 900% increase. The direct consequences of on-site soil degradation can easily be predicted, as can off-site impacts, such as, increased linear erosion, floods, water pollution and silting in aquatic ecosystems.

These results suggest that a proper control of overgrazing that is common on these sloping land areas typical of the headwaters of rivers in KwaZulu-Natal by appropriate management of cattle grazing is crucial and critical to protect soil and water resources. These results are thus particularly important to consider in the development of grazing and livestock management strategies in these areas and similar areas of South Africa. Appropriate mitigation measures are needed where soil degradation and erosion are a threat to sustained agricultural production.

Finally, further research studies need to be performed. Firstly, processes of sheet erosion besides splash and shallow runoff that take place on micro-plots, such as rain-impacted flow (Kinnel, 2001), which longer runoff plots may be used to quantify, have to be considered. Secondly, longer term experiments need to be performed with a special focus on other environmental factors such as cattle density and parent material, and thirdly the statistical relationships obtained in this study could be used to validate existing erosion models or integrated with existing predictive tools of sheet erosion that consider rainfall characteristics.

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