LAND USE AND LAND MANAGEMENT IMPACT ON CO_2 EMISSIONS FROM SOILS IN SELECTED SMALLHOLDER FARMING SYSTEMS

By

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PREFACE

The research contained in this thesis was completed by the candidate while based in the Discipline of Soil Sciences, School of Agricultural, Earth and Environmental Sciences of the College of Agriculture, Engineering and Science, University of KwaZulu-Natal, Pietermaritzburg campus, South Africa. The research was financially supported by Water Research Commission (projects K5/2266), South Africa

The contents of this work have not been submitted in any form to another university and, except where the work of others is acknowledged in the text, the results reported are due to investigations by the candidate.

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As the candidate's supervisors, we certify the above statement and have approved this thesis for submission.

Signed: Dr Pauline Chivenge
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Date:

DECLARATION 1: PLAGIARISM

I, ***KHATAB ABDALLA***, declare that:

(i) The research reported in this dissertation, except where otherwise indicated or acknowledged, is my original work;

(ii) This dissertation has not been submitted in full or in part for any degree or examination to any other university;

(iii) This dissertation does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other persons;

(ii) This dissertation does not contain other persons' writing, unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:

a) Their words have been re-written but the general information attributed to them has been referenced;

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(v) Where author have used material for which publications followed, I have indicated in detail my role in the work;

(vi) This dissertation is primarily a collection of material, prepared by myself, published as journal articles or presented as a poster and oral presentations at conferences. In some cases, additional material has been included;

(vii) This dissertation does not contain text, graphics or tables copied and pasted from the Internet, unless specifically acknowledged, and the source being detailed in the dissertation and in the References sections.

Khatab

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DECLARATION 2: PUBLICATIONS

This thesis is written in manuscript format, where each of chapters 2 to 6 contain abstract, introduction, materials and methods, results, discussion and conclusions. Minor editorial differences may exist between the published papers and the thesis chapters. All references cited in the chapters are presented in the final section of this thesis. Details of my contribution to each manuscript that form part of this thesis are indicated below.

Chapter 2

This chapter was published as: **Abdalla, K**., Chivenge, P., Ciais, P., Chaplot, V., 2015. No-tillage lessens CO_2 emissions from soil the most under arid and sandy soil conditions: results from a meta-analysis. **Biogeosciences.** 13, 3619–3633. The research reported is based on data I collected from 46 published research papers worldwide. The author searched for the articles, collected and analysed the data and wrote the paper. Final editing was provided by the co-authors and anonymous reviewers of the manuscript.

Chapter 3

This chapter was based on: **Abdalla, K.,** Chivenge, P., Everson, C., Chaplot, V. No-tillage and mulching with crop residues reduce CO_2 emissions and increase soil organic carbon stocks. This is a paper being submitted to the journal of **Agriculture Ecosystems Environment**. The paper is based on data the researcher collected from different tillage practices under continues maize cultivation at Potshini, KwaZulu-Natal province, South Africa. The author analysed the data and wrote the paper with technical advice and editing from the co-authors.

Chapter 3 was supported by: Chaplot V., **Abdalla K.,** Alexis M., Darboux F., Dlamini P., Everson C., Mchunu C., Muller-Nedebock D., Mutema M., Quenea K., Thenga H., Chivenge P. 2015. Surface organic carbon enrichment to explain greater CO_2 emissions from short-term no-tilled soils, **Agriculture Ecosystems Environment,** 203, 110-118. This paper based on a lab incubation the author contributed with the other co-authors in designing the experiments, collected the bulk soil from the field, applying the treatments in the lab. The author measured CO_2 emissions in the lab jointly with Thenga, and also did the CO_2 emissions data analysis, results presentation (tables and figures), results and discussion section.

Chapter 4

This Chapter was based on; **Abdalla, K.,** Chivenge, P., Everson, C., Chaplot, V. Grassland degradation increases CO_2 emissions relative to soil organic carbon stocks and aboveground produced biomass. This is a paper being submitted to the **Geoderma**. The paper is based on data the author collected from different grassland degradation levels at Potshini, KwaZulu-Natal province, South Africa. The author analysed the data and wrote the paper with technical advice and editing from the co-authors.

Chapter 5

This chapter was based on: **Abdalla, K.,** Chivenge, P., Everson, C., Chaplot, V. Grassland rehabilitation through the shift in cattle management drastically decreases CO_2 emissions relative to soil organic carbon stocks and produced biomass. This is a paper being submitted to the **Geoderma.** The paper is based on data the author collected from different grassland management practices at Potshini, KwaZulu-Natal province, South Africa. The author analysed the data and wrote the paper with technical advice and editing from the co-authors.

Chapter 6

This chapter was published as**: Abdalla, K.**, Chivenge, P., Everson C., Mathieu, O., Thevenot, M., Chaplot, V., 2016. Long-term annual burning of grassland increases CO_2 emissions from soils. **Geoderma**. 282, 80-86. The paper is based on data the author collected from long term grassland trial at Ukulinga research farm, KwaZulu-Natal at Pietermaritzburg, KwaZulu-Natal province, South Africa. The author analysed the data and wrote the paper with technical advice and editing from the co-authors and anonymous reviewers of the manuscript.

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ABSTRACT

The increase in global surface temperature since the 19th century has been attributed to increased atmospheric greenhouse gas accumulation due, mainly, to anthropogenic activities. However, agricultural lands can potentially sequester a lot of atmospheric carbon (C) if properly managed. While agricultural land mismanagement, such as intensive tillage of croplands and overgrazing of grasslands, has been reported to increase soil C losses, the losses are largely linked to increased soil erosion. There is limited knowledge linking land mismanagement to C losses through CO_2 emissions. Therefore, this thesis aimed at evaluating the impact of (1) different tillage and crop residue management practices under continuous maize cultivation, (2) grassland degradation and (3) short- and long-term grassland management practices on CO_2 emissions from soils and the factors of control. As a first step, a meta-analysis, based on 46 studies worldwide, was conducted to provide a quantitative synthesis of the impact of soil tillage on CO_2 emissions from soils. Its results showed that no-tillage can decrease gross CO_2 emissions by an average of 21% compared to tilled soils, with the greatest benefits under sandy soils of the dry tropics. However, climate appeared to mask the effects of other controlling factors. Therefore, field studies were conducted at Potshini in KwaZulu-Natal Province, South Africa, to compare the impact of different land management practices on CO_2 emission from soils. One field study compared CO_2 emission from soil under an innovative technique combing no-tillage with high-density short duration stocking (HDSD: 1200 cows ha^{-1} for three days per year) against no-tillage with different crop residue management practices (free grazing, complete removal and retention). Conventional tillage with free grazing was the control. All these practices were under maize mono-cropping. A second study at Potshini assessed the impact of grassland degradation on CO_2 emissions from the soil. There were three grassland degradation levels; namely non-degraded: 100%, degraded: 25-50%, and highly degraded: 0-5% aerial cover by grass) grassland. A third study at Potshini investigated the impact of different short-term (4 years) grassland management practices; namely high-density short duration stocking (HDSD: 1200 cows ha^{-1} for three days per year); livestock exclosure with NPK fertilization, annual burning and traditional free grazing on CO_2 emissions from the soil. The last study was conducted at a long-term (62 years) grassland management trial consisting of annual burning and mowing compared to no-burning at Ukulinga farm, also in KwaZulu-Natal. All the CO_2 emissions were measured using a LI-COR 6400XT, once a month in winter and twice a month in summer. Results from

the maize trial pointed that no-tillage with HDSD decreased gross CO_2 emissions from the soil by 56% and increased SOCs by 1.4 Mg C ha^{-1} $year^{-1}$ compared to the control, i.e. conventional tillage. Such benefits were explained by lower topsoil (0-0.05 m) temperature and higher compaction which most likely decreased soil biological activity. The highly degraded grassland produced 71 and 81% higher CO_2 emission relative to SOCs and produced biomass, respectively, than the non-degraded grassland. These results could be explained by lower C protection under degraded than non-degraded grassland because SOCs were 86% lower in the highly degraded than the non-degraded grassland. The short-term grasslands management trial results showed that fertilization had the highest CO_2 emission relative to SOCs (1.93 mg CO_2-C g^{-1}C day^{-1}), while traditional free grazing had the lowest (1.22 mg CO_2-C g^{-1}C day^{-1}). The CO_2 emissions relative to produced biomass were lowest under HDSD (0.13±0.02 g CO_2-C kg^{-1} produced biomass day^{-1}). The HDSD practice increased grass production by 74% and SOCs by 34% after three years, suggesting great potential for C sequestration through, for example, C inputs to the soil. Long-term annual burning and mowing had 30 and 34% higher gross CO_2 emission than no-burning, respectively. The higher CO_2 emissions in long-term annual burning imply lower C stability than no-burn grasslands. Overall, the field study results suggest that HDSD has great potential to increase soil C by reducing top soil temperature and increasing compaction which probably reduce decomposition of SOCs. Extrapolating the HDSD result onto South Africa's grasslands could increase mean annual soil C sequestration to 0.17 Gt C $year^{-1}$, which is 90% greater than the current rate of 0.01 Gt C $year^{-1}$. On a global level, HDSD could achieve 41% higher soil C sequestration than current rate. On the basis of these results, farmers are encouraged to integrate HDSD in no-tillage cropland and grasslands; however, the benefits still need to be confirmed by long-term investigations at different sites to cater for climate and soil type variability.

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TABLE OF CONTENTS

Page

LIST OF TABLES

Table **Page**

LIST OF FIGURES

LIST OF ABBREVIATIONS

AB	Annual burn
AM	Annual mowing
ANOVA	Analysis of variance
C	Carbon
CH4	methane
CO2	Carbon dioxide
COP21	Conference of Parties 21
CP	Chisel plow
CT	Conventional tillage
CTFG	Conventional tillage with free grazing
CV	Coefficient of variation
D	Degraded
DO	Disk plough
DT	Deep tillage
g	Gram
GHG	Greenhouse gas
GT	Gigaton
GWP	Global warming potential
ha	Hectare
HD	Highly degraded
HDSD	High-density short duration stocking rate
HO	Disk harrow
IPCC	Intergovernmental panel on climate change
kg	kilogram
LEF	Livestock exclosure with NPK fertilization
m	Meter
MAP	Mean annual precipitation
MAT	Mean annual temperature
Mg	Megagram
mg	milligram
MP	Moldboard plow

MWD	Mean weight diameter
N	Nitrogen
N2O	Nitrous oxide
NB	No burning
ND	Non-degraded
NT	No-tillage
NTFG	No-tillage with free grazing
NTHDSD	No-tillage with high-density short duration stocking
NTNR	No-tillage without crop residue mulching
NTR	No-tillage with crop residue mulching
PCA	Principal components analysis
Pg	Petagram
PVC	Polymerizing vinyl chloride
R	Coefficients of determination
REML	Restricted Maximum Likelihood
ROT	Rotary tillage
RT	Reduced tillage
SE	Standard error of the mean
SOC	Soil organic carbon
SOCs	Soil organic carbon stocks
SOM	Soil organic matter
ST	Soil temperature
ST	Strip tillage
SWC	Soil water content
T	Tillage
TFG	Traditional free grazing
UNEP	United nation environmental programme
USA	United states of America
USDA	United states department of agriculture
WRP	World Reference base

LIST OF SYMBOLS

b	Constant equal to 0.001
C:N	Carbon to nitrogen ratio
cw	Empty core weight (g)
cv	core volume (cm^{-3})
$CO_{2F\ T\text{-}NT}$	CO_2 emission difference between tilled and no-tilled soil
CO_{2NT}	CO_2 emissions from no-tilled soil (g CO_2-C m^{-2} $year^{-1}$)
CO_{2T}	CO_2 emissions from tilled soil (g CO_2-C m^{-2} $year^{-1}$)
ln	Natural log
lnR	Natural log of R
Nc	Nitrogen content (g N kg^{-1})
Ns	Nitrogen stocks (kg N m^{-2})
PF	Proportion of fragments of >2mm in percent
SOCc	Soil organic carbon content (g kg^{-1})
SOCNT	Soil organic carbon from no-tilled soil (g kg^{-1})
SOCs	Soil organic carbon stock (kg C m^{-1})
SOCT	Organic carbon from tilled soil (g kg^{-1})
T	Thickness of the soil layer
ρb	Bulk density (kg m^{-3})
odw	Oven dry weight (g)
rf	Weight of rocks fragments (g)

CHAPTER 1: INTRODUCTION

1.1 Background

Greenhouse gas (GHG) emissions, mainly from fossil fuel combustion and agriculture contribute to global warming and climate change (Lal, 2004; Emanuel, 2005; Le Quéré et al., 2015). GHG emissions from fossil fuel combustion and agriculture have increased rapidly in the last 25 years in response to increasing demand for energy and food caused by the growing human population (Bai et al., 2008). Carbon dioxide (CO_2) is a major GHG, accounting for 72% of the total atmospheric GHGs, while Methane (CH_4), Nitrous oxide (N_2O) and other minor GHGs represent the remainder (IPCC, 2004). Although CO_2 is the major GHGs, its radiative forcing (global warming potential or GWP = 1) is much less than N_2O (GWP = 265) and CH_4 (GWP = 28) (IPCC, 2013). Agricultural activities are important contributors of atmospheric CO_2, representing about 25% of the estimated cumulative CO_2 in atmosphere (Le Quéré et al., 2015). Carbon (C) is mainly stored in the oceanic and soil pools. The soil pool is the second largest, after the oceanic pool, and is estimated to store 2344 Pg C in the top three meters, which is 1.7 times higher than the amount found in vegetation and atmosphere together (Schlesinger, 1997; Jobbagy and Jackson, 2000).

Agricultural land (cropland, grassland and forests), covering 40-50% of the earth's terrestrial surface area (Smith et al., 2007) is considered a large pool of soil C (IPCC, 2001). This soil C pool has already been depleted by 50-70% (Lal, 2003a) due to improper land use and mismanagement, such as deforestation, overgrazing, inappropriate fertilization, and tillage and crop residue management (Lal, 2008a; Bhattacharyya et al. 2015). Improper land uses and mismanagement are responsible for rampant land degradation worldwide (Lal, 2004), which is associated with high C losses to the atmosphere. For example, water erosion exposes soil C to processes that drive CO_2 emissions from soil to the atmosphere, during transportation to stream channels and/or in river networks (Chaplot et al., 2007; Guillaume et al., 2015). Another important soil C loss pathway is through organic matter mineralization (Lal, 2003b; 2008a). Of the 136 Pg terrestrial C pool since 1850, 78 Pg has been lost from world soils consisting of 26 Pg through soil erosion and 52 Pg via mineralization (Lal, 2008b), which makes CO_2 emissions from soil a very essential loss route.

The lost C is potentially recoverable through a series of land management practices such as the abandonment of tillage, mulching using crop residues, introduction of cover crops and addition of organic manure that sequester atmospheric C in the soil. Tillage disturbs soils by breaking down the aggregates and exposing protected C to decomposers, thus increasing C loss from soils through CO_2 emissions and leaching (Six et al. 2000, 2004). In addition, intensive tillage induces soil erosion with consequently high soil C losses (Wilson et al., 2004; Chaplot et al. 2012; Bhattacharyya et al. 2015). Many studies worldwide have shown that tillage abandonment could potentially induce lower CO_2 emissions from soil and higher soil organic C (SOC) (Thiagalingam et al., 1996; Ussiri and Lal, 2009; Olson and Al-Kaisi, 2015). However, there are still many discrepancies concerning the impact of no-till practices on C inputs to soils and controlling soil C losses through CO_2 emissions. For example, Baker et al. (2007), Luo et al. (2010) and Dimassi et al. (2014) reported little to no tillage abandonment benefit on C sequestration. Crop residue mulching has also given contradicting results, with Reicosky (1997) and Al-Kaisi and Yin (2005) reporting increased C sequestration under high residue retention while Ren et al. (2014) observed no increase of the soil C over four years of straw and organic manure addition.

Grasslands represent 70% of the world's agricultural area (Abberton et al., 2010) and store about 10% of the global soil C (Suttie et al., 2005). The grasslands are also subjected to extreme degradation due to poor management practices such as intensive tillage and overgrazing (Lal, 2004). For example, Gang et al. (2014) estimated that 49% of global grasslands are degraded with about 5% of these extremely degraded. Restoring the lost soil C through suitable grassland management practices could benefit agriculture ecosystem and climate (Shekhar, 2012). Grassland management has largely focused on improving biodiversity and livestock production through practices like controlled grazing, fertilization and annual burning (Menke, 1982; SANBI, 2014). The impacts of these management practices on soil properties and CO_2 emissions from soil has been an important area of research (Conant and Paustian, 2002; Xu and Wan, 2008; Du et al., 2014), with some results showing that after three years of grazing abandonment CO_2 emissions increase by 21% compared to grazed grassland (Li et al., 2013); while others reported no significant differences in CO_2 emissions from soil between moderate and heavy grazing (e.g. Liebig et al., 2013). However, McSherry and Ritchie (2013) found that grazing effects on SOC density is highly variable and dependent on climate, soils, grass type, and grazing intensity.

While many grazing studies focused on the impact of different grazing intensities on CO_2 emissions from soil, there is a lack of information regarding the holistic management or "Savory" technique (Savory and Parsons, 1980; Savory, 1983). The Savory technique is based on a shift in cattle management involving high-density short duration stocking of livestocks once a year followed by livestock exclusion. Although, it has been promoted for rehabilitation of degraded grasslands and has been successfully used in Hawaii (Leung and Smith, 1984) and Southern Africa's countries (Skovlin, 1987; Fynn, 2008), but its performance on soil processes that affect SOC stocks (SOCs) and CO_2 emissions is still unclear. The assumption behind this method is that cattle hoof action could till the top-soil, smash soil crusts and trample down grass tufts in the presence of dung and urine, and result in increased nutrient cycling. The long grazing exclusion time would lead to better grass growth because of low top-soil compaction (no trampling) and the available nutrients from the dung and urine. High-density short duration stocking for short duration practice offers promising results in term of soil C sequestration. Chaplot et al. (2016) showed that soil C stock rates increased by 12.4±2.1 g C m² year^{-1} after only two years of grassland rehabilitation through this technique under sandy soil conditions of South Africa.

The current study evaluated total CO_2 emissions from soil surfaces under crop and grassland with different land management practices. The measured CO_2 emissions from soil consist of autotrophic (from plant roots) and heterotrophic (soil microorganisms) respiration (Subke et al., 2006). In addition, associated soil and environmental factors of control was also evaluated.

1.2 Research questions

There are uncertainties on the impact of land management (e.g. crop or grassland) practices on the rate, mechanisms and factors driving CO_2 emissions from agricultural soils. The main research question needing answerers are:

(i) What is the best land management practice that can increase C sequestration and decrease CO_2 emissions from soil?
(ii) What are the main factors controlling CO_2 emissions from soils?

1.3 Aim and Objectives

This dissertation aimed to assess the impacts of selected cropping and grassland management practices on total CO_2 emissions from soil. This was done to better quantify the impacts of agriculture on the global C cycle and to improve the understanding of the mechanisms and factors driving CO_2 emissions from soil. The specific objectives were:

(i) To assess the effects of tillage and different residue management practices on CO_2 emissions from soil and SOCs, in a continuous maize production system at Potshini, KwaZulu-Natal province.

(ii) To quantify the impacts of grassland degradation on CO_2 emissions from soil and identify its main control factors at Potshini, KwaZulu-Natal province.

(iii) To assess short-term (4 years) impacts on grassland rehabilitation on CO_2 emissions from soil and its factors of control at Potshini, KwaZulu-Natal province.

(iv) To quantify the effects of long-term (over 62 years) annual burning and mowing on CO_2 emissions from soil and SOCs at Ukulinga research farm, KwaZulu-Natal province.

1.4 Study area

The study benefited from existing research trials at Potshini smallholder farming community and at Ukulinga Research Farm, both located within a sub-tropical humid climate of KwaZulu-Natal Province of South Africa (Figure 1.1). The locations and main features of the sites are summarised in Table 1.1. The farming systems at Potshini are mixed crop-livestock with the crops being mainly staple food crops such as maize and vegetables, and livestock such as cattle and goats. Soils under cropland are subjected mostly to continuous maize cultivation with post-harvest free communal grazing on the cropping land by farm animals. The poor management of grasslands such as over grazing (defined as extensive consumption of plant exceeding carry capacity according to Allen-Diaz et al., 1995) associated with high anthropogenic activities due to population growth in the area has lead to severe grass and land degradation which in turn has resulted in C losses from soils due to high runoff and erosion (Podwojewski et al., 2011; Dlamini et al., 2011; Mchunu and Chaplot, 2012; Chaplot et al., 2012; Dlamini et al., 2014). Grassland burning is a common practice used in the study area for increasing fodder production, species composition and avoiding bush encroachments (Everson and Tainton, 1984; Tainton, 1999; Fynn et al., 2004). However, it has been known to cause high C losses to the atmosphere as a result of biomass burning (Jain et al., 2006; Van der Werf et al., 2006) but its real impact on soil C loss as CO_2 emissions is still not well understood. This study took advantage of existing annual grassland burning trials at Ukulinga research farm to evaluate the response of long-term grassland burning on CO_2 emissions from soil and SOCs.

Figure 1.1 Location of the study sites in KwaZulu-Natal province, South Africa

Table 1.1 Sites location and soil characteristics at the A horizon

Characteristics	Potshini	Ukulinga
Longitude	29° 21'	30° 24'
Latitude	28° 48'	29° 40'
Elevation (m)	1080-1455	847-838
Mean annual precipitation (mm)	745	694
Mean annual temperature (°C)	13	16
Soil classification (WRB, 2006)	Acrisol	Plinthic Acrisols
Parent material	Sandstone & mudstone	Shale & dolerite
A Horizon depth (m)	0-15	0-35
pH (KCl)	4.9-5.2	3.9-4.4
Sand %	55	20
Silt %	28	13
Clay %	17	37

1.5 Outline of the thesis

This thesis is organized as a series of research papers, with papers addressing each of the objectives listed in Section 1.2. The papers are either published or accepted by peer-reviewed journals or under preparation for submission to named journals. The thesis consists of seven chapters. Chapter 1 is the introduction of the thesis.

Chapter 2 provides a detailed quantitative review of literature worldwide with the main objective of quantifying the impacts of tillage on CO_2 emissions from soil and the main factors of control. A comprehensive meta-analysis was conducted using 46 peer-reviewed publications, totalling 174 paired comparisons of CO_2 emissions from soil over entire seasons/years from tilled and untilled soils across different climates, crop types and soil conditions. This chapter has been published in Biogeosciences as; Abdalla, K., Chivenge, P., Ciais, P., Chaplot, V., 2015. No-tillage lessens soil CO_2 emissions the most under arid and sandy soil conditions: results from a meta-analysis. Biogeosciences 13, 3619–3633.

Chapter 3 investigated *in-situ* CO_2 emissions from continuous maize cultivation under tillage and no-tillage soils and different crop residue management practices at Potshini, KwaZulu-Natal province in South Africa. This chapter based on a paper in preparation for publication in Agriculture, Ecosystem and Environment and supported by a paper has been published in Agriculture, Ecosystem and Environment as; Chaplot V., Abdalla K., Alexis M., Darboux F., Dlamini P., Everson C., Mchunu C., Muller-Nedebock D., Mutema M., Quenea K., Thenga H., Chivenge P. 2015. Surface organic C enrichment to explain greater CO_2 emissions from short-term no-tilled soils. Agriculture Ecosystems Environment, 203, 110-118.

Chapter 4 investigates the effects of grassland degradation on CO_2 emissions at Potshini, KwaZulu-Natal Province in South Africa. CO_2 emissions from soil were measured in the field over two years from a grassland showing different degradation intensities (non-degraded grassland, which exhibited an aerial cover of 100%; degraded grassland, with 25<Cov<50%; and highly degraded grassland, with 0<Cov<5%). This chapter is in preparation for publication in Geoderma.

Chapter 5 examines the effects of grassland rehabilitation on CO_2 emissions from soil under degraded grassland at Potshini, KwaZulu-Natal, South Africa. The main objective of this study was to assess the impact of high-density short duration stocking (1200 cows ha^{-1} for three days per year), an innovative grassland management practice, on CO_2 emissions from soil. High-density short duration stocking practice was compared against two commonly used grassland management practices, namely livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) and short-term annual burning, all being compared to traditional free grazing as a control. This chapter is in preparation for publication in Geoderma.

Chapter 6 evaluates CO_2 emissions from soil over two years, from long-term (over 60 years) annual burning and mowing trials at Ukulinga research farm, University of KwaZulu-Natal, South Africa. The study compared CO_2 emissions from annual burning and mowing to no burning associated with tree encroachment and to annual mowing. This chapter has been published in Geoderma as: Abdalla, K., Chivenge, P., Everson, C., Olivier, M., Mathieu, T., Chaplot, V. 2016. Long-term annual burning of grassland increases CO_2 emissions from soils Geoderma, 282, 80-86.

Finally, chapter 7 is a summary of the conclusions from the main chapters (chapter 2 to 6). In addition, study limitations, recommendations for farmers and policy makers on techniques to promote, as well as suggestions for future research are put forward in this chapter.

One reference list for all chapters is provided at the end of the dissertation.

CHAPTER 2: NO-TILLAGE LESSENS CO_2 EMISSIONS FROM SOIL THE MOST UNDER ARID AND SANDY SOIL CONDITIONS: RESULTS FROM A META-ANALYSIS

2.1 Abstract

The management of agroecosystems plays a crucial role in the global C cycle with soil tillage leading to known organic C redistributions within soils and changes in CO_2 emissions from soil. Yet, discrepancies exist on the impact of tillage on CO_2 emissions from soils and on the main soil and environmental controls. A meta-analysis was conducted using 46 peer-reviewed publications totalling 174 paired observations comparing CO_2 emissions from soil over entire seasons or years from tilled and untilled soils across different climates, crop types and soil conditions with the objective of quantifying tillage impact on CO_2 emissions from soil and assessing the main controls. On average, tilled soils emitted 21% more CO_2 than untilled soils. The difference increased to 29% in sandy soils from arid climates with low soil organic carbon content (SOC_C<1%) and low soil moisture, but tillage had no impact on CO_2 emissions in clayey soils with high background SOC_C (>3%). Finally, nitrogen fertilization and crop residue management had little effect on the CO_2 responses of soils to no-tillage. These results suggest no-tillage is an effective mitigation measure of carbon dioxide losses from dry land soils. They emphasize the importance of including information on soil factors such as texture, aggregate stability and organic C content in global models of the C cycle.

Keywords: *land management, tillage; no-tillage; CO_2 emissions.*

2.2 Introduction

The evidence for climate change is irrefutable and the necessity of mitigating climate change is now accepted. Yet, there are still large uncertainties on the effectiveness of the measures that could be taken to reduce GHG emissions by land-use management (Smith et al., 2008; Ciais et al., 2011). There are several reasons for these uncertainties. While inventories can be made of the different C pools (Bellamy et al., 2005), C pool changes are small and difficult to detect; they require sampling programs with periodic revisits over many years. Thus, the magnitude and variability of CO_2 fluxes, both sinks and sources, between the soil and the atmosphere are difficult to quantify and they may not have been accurately assessed. This is particularly the case for CO_2 fluxes associated with land use and land management, such as deforestation and changes in agricultural practice (Al-Kaisi and Yin, 2005; Alluvione et al., 2009; Dilling and Failey, 2012).

Soils are the largest terrestrial pool of C, storing 2344 Pg C (1 Pg = 1 billion tonnes) of soil organic carbon (SOC) in the top three meters (Jobbágy and Jackson, 2000). Tilling the soil before planting for seedbed preparation and weeding has been a common practice in agriculture since Neolithic times (McKyes, 1985). This technique is energy intensive and also affects SOC stocks (SOCs). Tilling changes the balance between organic C inputs into the soil by plants and rendered available for soil micro-organisms, and C output as greenhouse gases (GHGs) due to organic matter decomposition (Rastogi et al., 2002). Soil tillage may also lead to the vertical and lateral export of particulate and dissolved organic C by leaching and erosion (Jacinthe et al., 2002; Mchunu et al., 2011).

Soil tillage is estimated to have decreased SOCs by two-thirds from pre-deforestation levels (Lal, 2004). But this estimate is highly uncertain, due to the lack of detailed site-level meta-analysis for different climates, soil types and management intensities. Six et al. (2000, 2004) reported that tillage induces soil disturbance and disruption of soil aggregates, exposing the protected SOC to microbial decomposition and thus causing C loss from soils through CO_2 emissions and leaching. Tillage is also responsible for soil compaction, soil erosion and loss of soil biodiversity (Wilson et al., 2004). In some instances, tillage is thought to have caused a net sink of atmospheric CO_2, for instance by displacing SOC to deeper soil horizons or accumulation areas where it decomposes more slowly (Baker et al., 2007; Van Oost et al., 2007). Soil tillage also modifies the mineralization rates of nutrients, which feeds back on soil C input, implying that the effect of tillage on the balance of SOC needs to be considered at ecosystem level (Barré et al., 2010).

Nowadays, tillage is being increasingly abandoned as the use of mechanised direct planters becomes widespread and weed control is performed with herbicides or in a more ecologically friendly way by using cover crops and longer crop rotations.

The consequences of this change in practice on soil properties and soil functioning are numerous. Importantly, it also raises the unsolved question: what is the impact of tillage abandonment on GHG emissions and climate change? Common wisdom is that no-tillage (or zero-tillage) agriculture enhances SOCs (Peterson et al., 1998; Six et al., 2002; West and Post, 2002; Varvel and Wilhelm, 2008) by reducing soil C loss as CO_2 emissions (Paustian et al., 1997; West and Post, 2002; Dawson and Smith, 2007). For instance, Paustian et al. (1997) reviewed 39 paired comparisons and reported that abandonment of tillage increased SOCs in the 0-0.3 m layer by an average of 258 g C m^{-2} (i.e., 8%). Ussiri and Lal (2009) observed a two-fold increase of SOCs in the top 0.03 m of soil (800 versus 453 g C m^{-2}) after 43 years of continuous *Zea mays* (maize) under no-tillage compared to tillage. Virto et al. (2012) in a meta-analysis based on 92 paired comparisons reported that SOCs were 6.7% higher under no-tillage than tillage.

While consensus seems to exist on the potential of no-tillage for C sequestration and climate change mitigation, several scientific studies point to possible flaws in early reports (Royal Society 2001; VandenBygaart and Angers, 2006; Baker et al., 2007; Luo et al., 2010; Dimassi et al., 2014; Powlson et al., 2014). VandenBygaart and Angers (2006) indicated that the entire plow depth had to be considered for not overstating zero-tillage impact on SOC storage. Baker et al. (2007) were the first to point out that the studies concluding on C sequestration under no-tillage management had only considered the top-soil (to a maximum of 0.3 m), while plants allocate SOC to much lower depths. False conclusions may be drawn if only C in the top-soil is measured. Using meta-analysis based on 69 paired-experiments worldwide where soil sampling depth extended to 1.0 m, Luo et al. (2010) found that conversion from tillage to no-tillage resulted in significant top-soil SOC enrichment, but did not increase the total SOCs in the whole soil profile. Furthermore, Dimassi et al. (2014) reported SOC losses over the long term.

Evidence for higher CO_2 emissions from land under tillage than a no-tillage regime has been widely reported (e.g., Reicosky, 1997; Al-Kaisi and Yin, 2005; Bauer et al., 2006; Sainju et al., 2008; Ussiri and Lal, 2009). For instance, in a study performed in the US over an entire year, Ussiri and Lal (2009) found that, tillage emits 11.3% (6.2 versus 5.5 Mg of CO_2-C per hectare per year, CO_2-C ha^{-1} $year^{-1}$) more CO_2 than no-tillage. Similarly, all the field surveys by Alluvione et al. (2009) reported that land under tillage had 14% higher CO_2 emissions than land with no-tillage. Al-Kaisi and Yin (2005) found this difference to be as much as 58%. A few *in situ* studies, however, found CO_2 emissions from no-tillage soils were similar to those from tilled soils (Aslam et al., 2000; Oorts et al., 2007; Li et al., 2010). However, Hendrix et al. (1988) and Oorts et al. (2007) found higher CO_2 emissions from untilled compared to tilled soils, with Oorts et al. (2007) reporting that no-tillage increased CO_2 emissions from soil by 13% compared to tillage.

In a further example, Cheng-Fang et al. (2012) showed that in central China, no-tillage increased CO_2 emissions from soil by 22-40% compared with tillage. Oorts et al. (2007) attributed the larger CO_2 emissons from no-tillage soil compared to tilled soil to increased decompostion of the weathered crop residues lying on the soil surface.Crop residue management has been shown to greatly impact CO_2 emissions from soils under both tillage and no-tillage (Oorts et al., 2007; Dendooven et al., 2012). Jacinthe et al. (2002) reported annual CO_2 emissions to be 43% higher with tillage compared to no-tillage with no mulch, but found a 26% difference for no-tillage with mulch. Some other authors associated the changes in CO_2 emissions from soil following tillage abandonment to shifts in nitrogen fertilization application and in crop rotations (Al-Kaisi and Yin, 2005; Álvaro-Fuentes et al., 2008; Cheng-Fang et al., 2012). Sainju et al., (2008) working in North Dakota pointed to CO_2 flux differences between tilled and untilled soils only for fertilized fields, while other studies pointed to the absence of nitrogen impact (Drury et al., 2006; Cheng-Fang et al., 2012). Crop type and crop rotation may also constitute important controls on the CO_2 efflux differences between tillage and no-tillage, mainly through differences in root biomass and its respiration, and nitrogen availability (Amos et al., 2005; Álvaro-Fuentes et al., 2008). Omonode et al. (2007) found a 16% difference in CO_2 outputs between tillage and no-tillage under continuous maize, while Sainju et al. (2010b) found no difference between continuous barley and barley-pea rotations.

Micro-climatic parameters such as soil temperature and precipitation are other likely controls of the response of CO_2 emissions from soil to tillage (Angers et al., 1997; Flanagan and Johnson, 2005; Lee et al., 2006; Oorts et al., 2007). These controls also need further appraisal.

The existence of research studies from different soil and environmental conditions worldwide opens the way for a more systematic assessment of tillage impact on CO_2 emissions from soil and their controls. Meta-analysis is commonly used for combining research findings from independent studies and offers a quantitative synthesis of the findings (Rosenberg et al., 2000; Borenstein et al., 2011). This method has been used here in order to assess the effects of background climate (arid to humid), soil texture (clayey to sandy), crop types (maize, wheat, barley, paddy rice, rapeseed, fallow and grass), experiment duration, nitrogen fertilization, crop residue management and crop rotations on the CO_2 emissions responses of soils following tillage abandonment. CO_2 emissions from soil with tillage and no-tillage were compared for 174 paired observations across the world.

2.3 Materials and Methods

2.3.1 Database generation

A literature search identified papers considering in situ CO_2 emissions from soil and top-soil (0-0.03 m depth) SOC changes under tillage and no-tillage management regimes. Google, Google scholar, Science Direct, Springerlink and SciFinder were used. In order to make the search process as efficient as possible, a list of topic-related keywords was used such as “soil C losses under tillage compared to no-tillage”, “CO_2 emissions from soil under tillage and no-tillage”, “land management practices and greenhouse gases emissions”, “land management effects on CO_2 emissions or soil respiration”, “effects of tillage versus no-tillage on CO_2 emissions” and “SOC”. Many papers (over 200) were found dealing with CO_2 emissions from soil and SOC under cropland systems, but only (46) those that reported CO_2 emissions from soil measured under field conditions for both tillage and no-tillage from the same crop and period and considered soil respiration as a whole (heterotrophic + belowground autotrophic) were used in the study.

The crops considered in this study were maize, wheat, barley, oats, soybean, paddy rice and fallow. The practices considered as tillage in this review are those that involve physical disturbance of the top-soil layers for seedbed preparation, weed control, or fertilizer application. Consequently, conventional tillage, reduced tillage, standard tillage, minimum tillage and conservation tillage were all considered as tillage. However, only direct seeding and drilling were considered as no-tillage, among different practices reported in the reviewed literature. The studies used in the meta-analysis covered 13 countries (USA, Spain, Brazil, Canada, China, Denmark, France, Finland, New Zealand, Lithuania, Mexico, Argentina and Kenya). A total of 46 peer-reviewed papers with 174 comparisons for CO_2 emissions from soil and 162 for SOC content (SOCC) were identified. Table 2.1 summarizes information on site location, climatic conditions, crop rotation systems, and average CO_2 emissions from soil under tilled and untilled soils. Most of the data (37%) came from USA followed by Canada, China and Spain (11% each), and Brazil (9%). There was only one study from Africa, conducted in Kenya by Baggs et al. (2006).

Several soil variables were considered, as follows: SOC_C (%), soil bulk density (ρb, g cm^{-3}), and soil texture (Clay, Silt, and Sand, %) in the 0-0.03 m layer. In addition, mean annual temperature (MAT, °C) and mean annual precipitation (MAP, mm), crop types, crop rotations, nitrogen fertilization rate, experiment duration and crop residue management were also considered.

Data for CO_2 emissions from soil (n = 46) were obtained for all studies by using open chambers and reported on an area basis. CO_2 emissions from soil were directly extracted from the papers and were standardized to g CO_2-C m^{-2} $year^{-1}$. Thirty eight studies gave SOC_C for both tillage and no-tillage. Four studies (Hovda et al., 2003; Álvaro-Fuentes et al., 2008; Lee et al., 2009; Dendooven et al., 2012) gave SOC_C, in term of the mass of C in the 0-0.03 m layer and per unit area (kg C m^{-2}). Finally, for the remaining studies, SOC_C was extracted from other existing papers describing work at the same site. SOC_C was estimated from the SOC_S (kg C m^{-2}) and bulk density following Eq. (1) by Batjes (1996).

$$SOCs = SOCc \ x \ \rho b \ x \ T\left(1 - \frac{PF}{100}\right) b \qquad (1)$$

where SOC_S is the soil organic C stock (kg C m^{-2}); SOC_C is soil organic C content in the ≤2mm soil material (g C kg^{-1} soil); ρ_b is the bulk density of the soil (kg m^{-3}); T is the thickness of the soil layer (m); PF is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

Although papers reported SOC and SOCs results for different soil profiles, only those papers reporting on the 0-0.03 m or deeper profiles were accepted. However, the analyses used data for the 0-0.03 m soil profiles only because this is the soil depth most affected by tillage operations.

Information on mean annual precipitation (MAP) and men annual temperature (MAT) were extracted from the papers, but were estimated in nine studies where such information was not provided, based on the geographic coordinates of the study site and using the WORLDCLIM climatology (Hijmans et al., 2005) with a spatial resolution of 30 seconds. In eight studies where soil texture was only given as textural class, particle size distribution was estimated using the adapted version of USDA texture classification system by Saxton et al. (1986).

Table 2.2 shows the variables used in categorizing the experimental conditions. The climatic regions were extracted from the papers and then categorized into arid and humid climate according to Köppen, (1936), on the basis of mean annual temperature and precipitation. SOC_C were categorized into three categories following Lal (1994): low (SOC_C <10 g C kg^{-1}), medium (10-30 g C kg^{-1}) and high (>30 g C kg^{-1}). Soil texture was categorized based on the soil textural triangle (Shirazi and Boersma, 1984) into three classes (clay, loam and sand). Particle size distribution was estimated using the adapted version of USDA texture classification system (Saxton et al., 1986)._Fertilization rate for this meta-analysis was classified into the categories defined by Cerrato and Blackmer (1990): low when below 100 kg N ha^{-1} and high when above 100 kg N ha^{-1}.

Table 2.1. References included in database with locations, mean annual precipitation (MAP), mean annual temperature (MAT), climate, land use, no-tillage comparisons and average tillage (T) and no-tillage (NT) CO_2 emissions from soil

SN.	Author (s)	Country	Comparisons	MAP	MAT	Climate	Land use	No-tillage vs.	CO_2-C emissions	
				mm	**°C**				gCO_2-C m^{-2} $year^{-1}$	
									T	NT
1	Ahmad, S. et al (2009)	China	2	2721	17	Humid	Rice-rape	CT	857	888
2	Al-Kaisi & Yin (2005)	USA	4	889	10	Humid	Maize-soybean	ST&DT&CP&MP	292	206
3	Alluvione et al (2009)	USA	2	383	11	Arid	Maize	CT	490	599
4	Almaraz et al (2009a)	Canada	2	979	6	Humid	Soybean	CT	747	523
5	Almaraz et al (2009b)	Canada	4	979	6	Humid	Maize	CT	1269	1374
6	Alvarez et al. (2001)	Argentina	1	1020	17	Humid	Wheat-soybean	CT	2154	1533
7	Álvaro-Fuentes et al (2008)	Spain	24	415	15	Arid	Wheat-barley-fallow-rape	CT&RT	2311	1891
8	Aslam et al (2000)	New Zealand	1	963	13	Humid	Maize	MP	2306	2281
9	Baggs et al. (2006)	Kenya	2	1800	24	Humid	Maize-fallow	CT	171	215
10	Brye et al (2006)	USA	4	1282	16	Humid	Wheat-soybean	CT	3264	2604
11	Carbonell-Bojollo et al (2011)	Spain	3	475	25	Arid	Wheat-pea-sunflower	CT	298	100
12	Chatskikh &Olesen 2007	Denmark	2	704	7	Humid	Barley	CT&RT	117	102
13	Cheng-fang et al (2012)	China	4	1361	17	Humid	Rice-rape	CT	636	699
14	Chevaz et al 2009	Brazil	1	1755	19	Humid	Oots-soybean-wheat-maize	CT	464	573
15	Datta et al, (2013)	USA	1	1016	11	Humid	Maize	CT	438	634
16	Dendooven et al, (2012)	Mexico	2	600	14	Arid	Maize-wheat	CT	100	100
17	Drury et al (2006)	USA	3	876	9	Humid	Wheat-maize-soybean	CT	575	559
18	Elder and Lal (2008)	USA	1	1037	11	Humid	Maize- wheat	MT	225	189
19	Ellert and Janzen (1999)	Canada	5	400	5	Arid	Wheat-fallow	CT&RT	406	186
20	Feizine et al (2010)	Lithuania	24	500	18	Humid	Wheat-rape-barley-pea	CT&RT	302	296

21	Hovda, et al (2003)	Canada	2	979	6	Humid	Maize	CT	1342	1277
22	Jabro et al (2008)	USA	1	373	14	Humid	Sugarcane	CT	3424	2247
23	Lee et al (2009)	USA	3	564	16	Arid	Maize-sunflowers-pea	ST	933	917
24	Li et al (2010)	China	4	1361	17	Humid	Rice-rape	CT	284	328
25	Li et al (2013)	China	2	1361	18	Humid	Rice	CT	2196	1534
26	Liu et al (2011)	China	4	550	13	Humid	Maize	RT &PT	1340	1194
27	López-Garrido et al (2009)	Spain	1	484	17	Arid	Wheat-sunflower -Pea	CT	1080	943
28	López-Garrido et al (2014)	Spain	3	484	17	Humid	Wheat-pea-red clover	CT	1075	887
29	Lupwayi et al (1998)	Canada	1	336	-1	Arid	Wheat-pea-red clover	CT	621	464
30	Morell et al (2010)	Spain	8	430	14	Arid	Barley	CT&MP	300	229
31	Mosier et al (2006)	USA	9	382	11	Arid	Maize	CT	387	351
32	Menéndez et al (2008)	Spain	2	350	16	Arid	Wheat–sunflower	CT	183	214
33	Omonode et al (2007)	USA	4	588	19	Humid	Maize	MP&CP	273	268
34	Oorts et al. (2007)	France	2	650	11	Humid	Maize-wheat	CT	475	620
35	Pes et al. (2011)	Brazil	2	1721	19	Humid	wheat - soybean	CT	1387	1004
36	Regina and Alakukku (2010)	Finland	6	585	4	Humid	Barley-wheat-oats	CT	1856	2009
37	Reicosky and archer (2007)	USA	1	301	5	Humid	Maize-soybean	MP	5807	1545
38	Ruan and Robertson (2013)	USA	1	890	10	Humid	Soybean	CT	1825	1533
39	Sainju et al (2008)	USA	4	368	14	Arid	Barley-pea	CT	6726	4217
40	Sainju et al (2010a)	USA	6	350	16	Humid	Barley-pea	CT	240	208
41	La Scala et al (2001)	Brazil	4	1380	21	Humid	Maize	ROT&CP&DO&HO	1264	657
42	La Scala et al (2005)	Brazil	4	1380	21	Humid	Maize	CT	758	518
43	Scala et al (2006)	Brazil	2	1380	21	Humid	Sugarcane	RT&CT	5435	2604
44	Smith, D. et al (2011)	USA	1	796	17	Humid	Maize-soybean	CT	141	152
45	Smith, K. et al (2012)	USA	4	1370	17	Humid	Maize-soybean	CT	970	935
46	Ussiri and Lal (2009)	USA	2	1037	11	Humid	Maize-soybean	CT&MT	721	500

CT, conventional tillage; ST, strip tillage; DT, deep tillage; CP, chisel plow; MP, moldboard plow, RT, reduced tillage; ROT, rotary tillage; DO, disk plough ; HO, disk harrow.

In addition, no-tillage treatment was classified as short duration when <10 years, or long duration when exceeding 10 years. Crops residues were either left on the soil surface or removed after harvest with no distinction between removal proportions. Crops rotations were divided into two categories: a series of different types of crop in the same area classed as "rotation", or continuous monoculture, classed as "no rotation".

2.3.2 Meta-analysis

The response ratio (R) of CO_2 emissions from soil to SOCc under tillage (T) and no-tillage (NT) was calculated using equation (2) and (3). As common practice, natural log of the R (lnR) has been calculated as an effect size of observation (Hedges et al., 1999)

$$InR = In(CO_{2T} / CO_{2NT}) \tag{2}$$

$$InR = In(SOC_T / SOC_{NT}) \tag{3}$$

Where CO_{2T} is the CO_2 emissions from tilled soil (g CO_2-C m^{-2} $year^{-1}$); CO_{2NT} is the CO_2 emissions from no-tilled soil (g CO_2-C m^{-2} $year^{-1}$); SOC_T is soil organic carbon from tilled soil (g kg^{-1}); $SOCn_T$ is soil organic carbon from no-tilled soil (g kg^{-1}).

The MetaWin 2.1 software (Rosenberg et al., 2000) was used for analyzing the data and generating a bootstrapped (4,999 iterations) to calculate 95% confidence intervals. The means of effect size were considered to be significantly different from each other if their 95% confidence intervals were not overlapping and were significantly different from zero if the 95% level did not overlap zero (Gurevitch and Hedges, 2001).

Table 2.2 Categories used in describing the experimental conditions

Categorical variable	Level 1	Level 2	Level 3
SOC_C	Low	Medium	High
	(<10 g kg^{-1})	(10-30 g kg^{-1})	(>30 g kg^{-1})
Climate	Arid	Humid	
Soil texture	Clay	Loam	Sand
	(>32% clay)	(20-32 clay)	(<20% clay)
Experiment duration	<10 years	≥10 years	
Nitrogen fertilizer	Low	high	
	(<100 kg N ha^{-1})	(≥100 kg N ha^{-1})	
Crop residues	Removed	Returned	
Crop rotation	No rotation	Rotation	

2.4 Results

2.4.1 General statistics of CO_2 emissions from tilled and untilled soils

Overall, average CO_2-C emissions from soil computed from the 174 paired observations was 1152 g CO_2-C m^{-2} $year^{-1}$ from tilled soils compared to 916 g CO_2-C m^{-2} $year^{-1}$ from under no-tillage (Table 2.3), which corresponds to a 21% average difference, significant at $P<0.05$. The highest CO_2-C emissions amongst the considered sites was 9125 g CO_2-C m^{-2} $year^{-1}$ observed under tilled soils with barley in an arid area at Nesson Valley in western North Dakota, USA (Sainju et al., 2008). The lowest CO_2 emissions from soil were 11 g CO_2-C m^{-2} $year^{-1}$ observed under no-tillage wheat in the humid climate of Lithuania (Feiziene et al., 2011).

Table 2.3 summary statistics of mean annual precipitation (MAP), mean annual temperature (MAT), clay, soil bulk density (ρb), soil organic carbon content (SOC_C), soil organic carbon stocks (SOC_S) and CO_2 emissions (g CO_2-C m^{-2} $year^{-1}$ and g CO_2-C g^{-1} C $year^{-1}$) under tilled (T) and untilled (NT) soils

	MAP	MAT	CLAY	ρb		SOC_C		SOC_S		CO_2-C emissions			
				T	NT	T	NT	T	NT	T	NT	T	NT
	mm	°	%	g cm^{-3}		%		kg m^{-2}		g CO_2-C m^{-2} $year^{-1}$		g CO_2-C gC^{-1} $year^{-1}$	
Minimum	301	-1	3	0.5	0.8	0.3	0.6	0.7	1.1	33	11	0.006	0.001
Maximum	2721	25	60	1.9	1.9	8.0	7.8	9.6	10.4	9125	5986	0.823	0.118
Mean	904	15	1.3	1.3	1.3	1.3	2.9	2.9	3.1	1152	916	0.109	0.016
Median	704	16	1.3	1.3	1.3	1.1	2.5	2.5	2.7	587	533	0.071	0.012
SD	570	6	0.2	0.1	0.1	1.0	1.0	1.5	1.5	1482	1054	0.132	0.017
Skewness	1	0	-0.7	0.6	0.6	4.0	3.2	2.0	2.8	2.8	2.4	3.127	3.599
Quartile1	415	11	1.3	1.3	1.3	0.7	0.7	2.2	2.4	287	283	0.037	0.008
Quartile3	1321	18	1.4	1.4	1.4	1.3	1.7	3.3	3.3	1414	1210	0.107	0.020
Kurtosis	2	0	9.9	3.4	3.4	23.3	14.3	6.3	10.7	9.8	6.69	12.48	17.81
CV	63	41	0.1	0.1	0.1	0.8	0.4	0.5	0.5	1.29	1.15	1.214	1.018
SE	48	0	0.01	0.01	0.01	0.08	0.09	0.12	0.13	112	80	0.011	0.001

2.4.2 Controls on the response of CO_2-C emissions from soil to tillage

2.4.2.1 Climate

Tillage soils emitted 27% more CO_2 than no-tillage in arid climates; while in humid climates, tillage emitted 16% more CO_2 than no-tillage. However, the differences in CO_2 emissions from soil between tillage and no-tillage were not statistically significant (at 0.05 confidence interval) between arid and humid climates (Figure 2.1A). When compared across all studies, mean SOC_C under tillage was 10% lower than under no-tillage (Figure 2.1B). In arid climates, SOC_C in tillage was 11% lower than no-tillage, whereas in humid climates SOC_C under tillage was only 8% less than for no-tillage. However, the differences in SOCc between the two climatic zones were found to be non-significant.

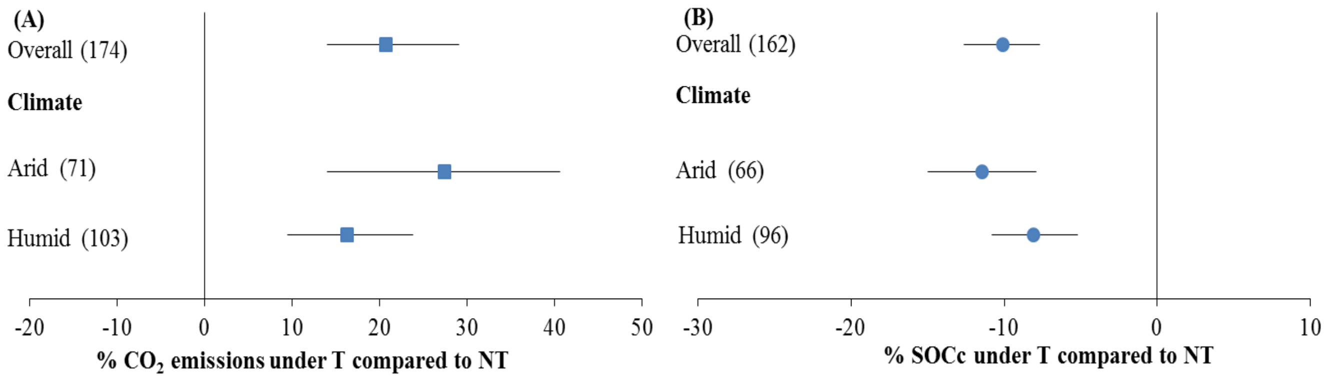

Figure 2.1 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of climate (arid and humid). The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

2.4.2.2 Soil organic carbon content

On average, CO_2 emissions from tilled soils were 25% higher compared to untilled soils with SOC_C lower than 10 g kg^{-1} (Figure 2.2). For SOC_C between 10 and 30 g kg^{-1}, tilled soils emitted an average 17% more CO_2 than untilled ones. In the case of C-rich soils with SOC_C higher than 30 g kg^{-1}, there were no significant differences between tillage and no-tillage CO_2 emissions. Thus, the difference between tillage and no-tillage decreased with increasing background SOC_C. Overall, CO_2 emissions from soil under no-tillage were about five times higher for low compared to high SOC_C.

2.4.2.3 Soil texture

Differences in CO_2 emissions between tilled and untilled soils were largest in sandy soils where tilled soils emitted 29% more CO_2 than untilled soils (Figure 2.3A). In clayey soils, the differences between tillage and no-tillage were much lower, with tilled soils emitting 12% more CO_2 than untilled soils. On the other hand, SOC_C under tillage was significantly lower than under no-tillage: by 17% under sandy soils and 9% in clayey soils (Figure 2.3B). However, there were no differences between clayey and loamy soils.

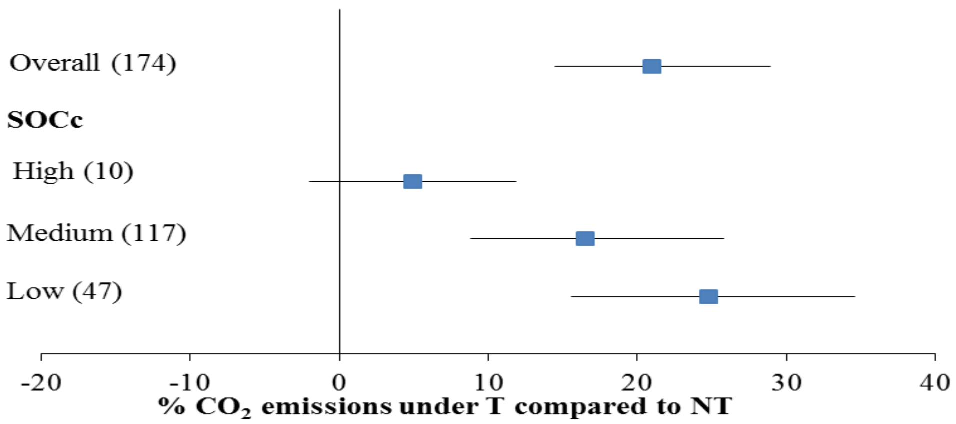

Figure 2.2 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of SOCc (low, <10 g kg^{-1}, medium 10-30 g kg^{-1}, high >30 g kg^{-1}). The numbers in the parentheses indicate the direct comparisons of meta-analysis. Error bars are 95% confidence intervals

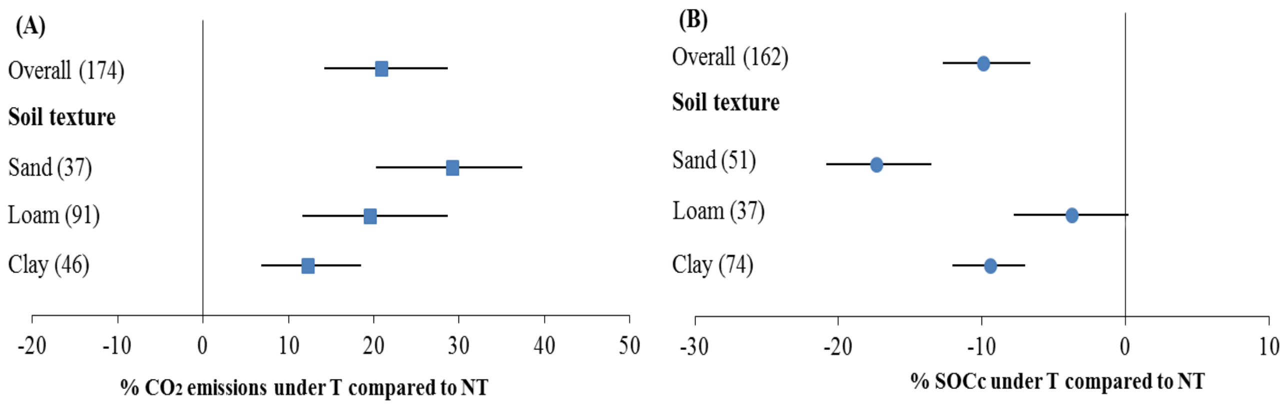

Figure 2.3 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of soil particle distribution (clay, loam and sand). The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

2.4.2.4 Crop type

Grouping all crop types together, SOC_C under tillage was significantly lower than under no-tillage. Among the different crops (rice, maize, soybean, wheat and barley) a significant SOCc difference between tilled and untilled soil was only observed for maize (15%) at one site and for rice (7.5%). The SOCc was under no-tillage was slightly higher than under tillage in soil under fallow, but the difference was not significant (Figure 2.4B). Highest SOC_C differences between tilled and untilled soils were observed for maize where SOC_C was on average 15% lower under tillage compared to no-tillage.

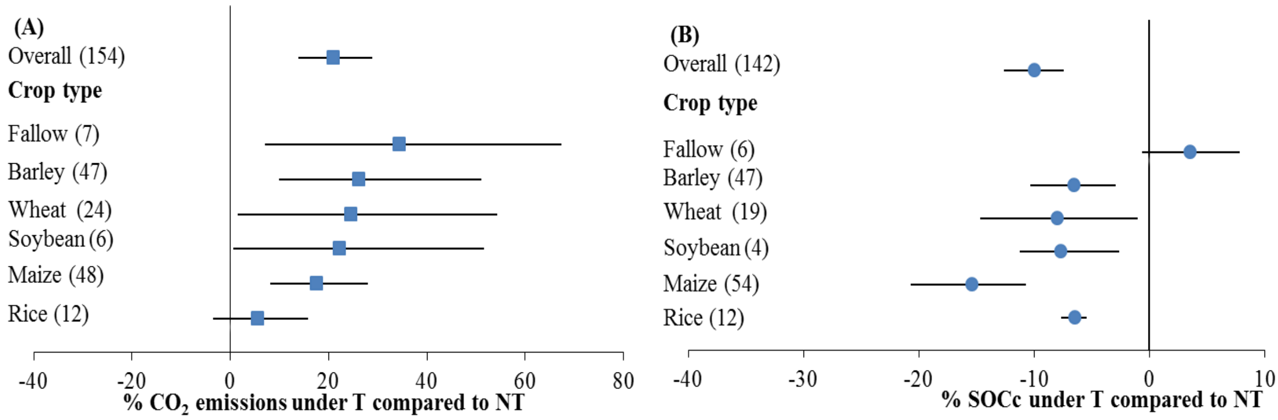

Figure 2.4 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of crop type. The numbers in the parentheses indicate the direct comparisons of meta-analysis. Error bars are 95% confidence intervals

2.4.2.5 Duration of no-tillage

The duration of no-tillage (i.e., time since tillage was abandoned) had no statistical association with CO_2 emissions from soil. However, there was a tendency for the differences between tillage and no-tillage to increase with increasing duration of the no-tillage regime with an average 18% difference for experiments of less than 10 years, and 23% for those longer than 10 years (Figure 2.5A). SOC_C under tillage was 14% lower compared to no-tillage for experiments lasting longer than 10 years, whereas there were no differences in SOC_C between tillage and no-tillage for shorter durations (Figure 2.5B).

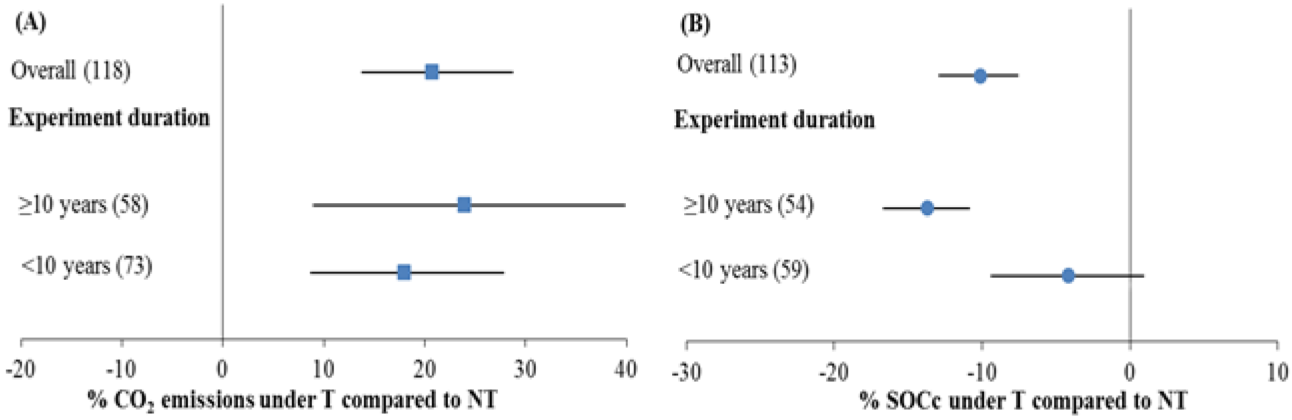

Figure 2.5 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of experiment duration (<10 years and ≥ 10 years). The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

2.4.2.6 Nitrogen fertilization

Nitrogen fertilization did not produce statistically significant differences between CO_2 emissions and SOC_C differences from tilled and untilled soil (Figure 2.6). Compared to tillage, no-tillage decreased CO_2 emissions from soil by an average of 19% when 100 kg N ha^{-1} or more was applied, while at lower fertilization rates, CO_2 emissions decreased by 23%, but owing to the low sample size this difference was not statistically significant.

2.4.2.7 Crop residue management and crop rotation

On average, when crop residues were not exported, no-tillage decreased CO_2 emissions from soil by 23% compared to tillage, which corresponded to a significant difference at $P < 0.05$. On the other hand, crop residue removal resulted in a lower difference of only 18% (Figure 2.7A). SOC_C was 12% lower under tillage than no-tillage in the absence of crop residues, and only 5% lower when crop residues were left on the soil (Figure 2.7B). On the other hand, Soils under a crop rotation regime exhibited a much sharper decrease (i.e., 26%) in

CO_2emissions following tillage abandonment than the soils under continuous monoculture for which the changes were not significant at $P<0.05$ (Figure 2.8).

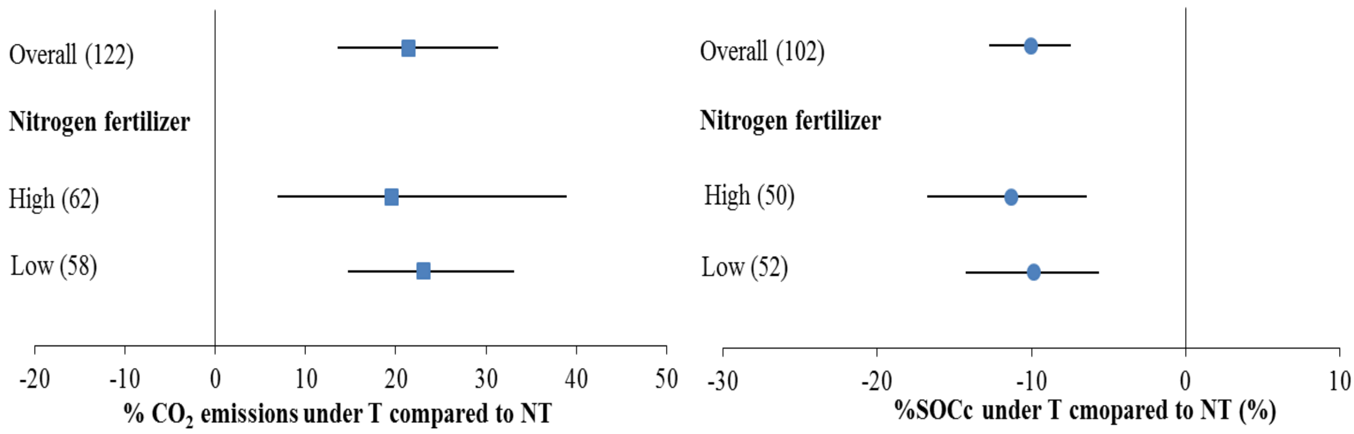

Figure 2.6 Percent change in (A) CO_2 emissions from soil (B) and SOCc in tillage (T) soil compared to no-tillage (NT) as a function of nitrogen fertilization (low <100 kg N ha^{-1} and high ≥100 kg N ha^{-1}). The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

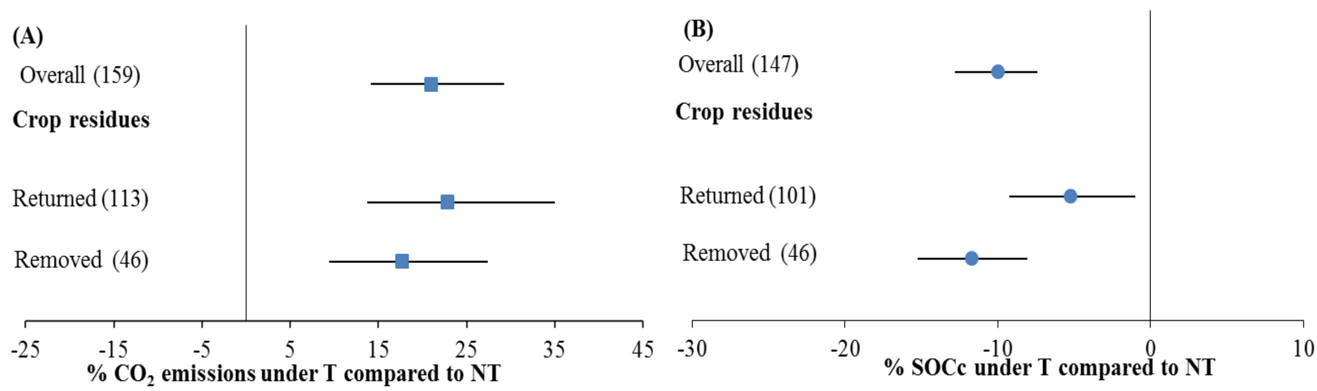

Figure 2.7 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of crop residues (returned and removed). The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

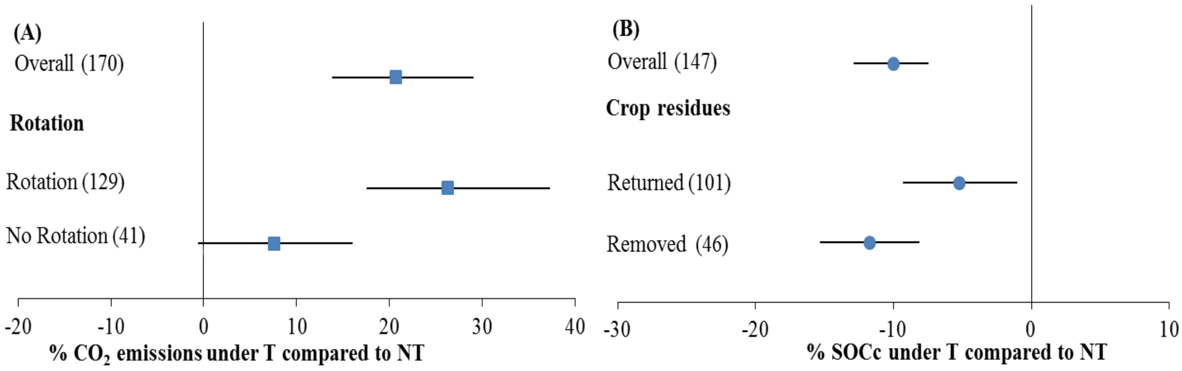

Figure 2.8 Percent change in (A) CO_2 emissions from soil and (B) SOCc in tillage (T) soil compared to no-tillage (NT) as a function of crop rotation. The numbers in the parentheses indicate the direct comparisons of the meta-analysis. Error bars are 95% confidence intervals

2.4.3 Multiple correlations between CO_2 emissions from soil and selected soil variable and environmental factors

Figure 2.9 shows the interaction between the changes in CO_2 emissions from soil following tillage abandonment on one hand and the selected soil and environmental variables on the other. The first two axes of the PCA explained 66% of the entire data variability. The first PCA axis (Axis 1), which described 35% of the total data variance, was highly correlated to latitude (LAT), mean annual temperature (MAT), SOCc, and soil clay content (CLAY). LAT and ρb showed positive coordinates on Axis 1, while the other variables showed negative ones. Axis 1 could, therefore, be regarded as an axis setting clayey organic and warm soils against compacted, sandy soils from a cold climate. The second PCA axis, which explained 21% of the data variance, correlated the most with silt content. The differences in CO_2 fluxes from soil between tillage and no-tillage ($\Delta CO_{2T\text{-}NT}$) showed positive coordinates on Axis 1, which revealed higher CO_2 emissions from soil under tillage compared to no-tillage under cool sandy and dense soils compared to warm clayey and organically rich soil from a warm and humid climate.

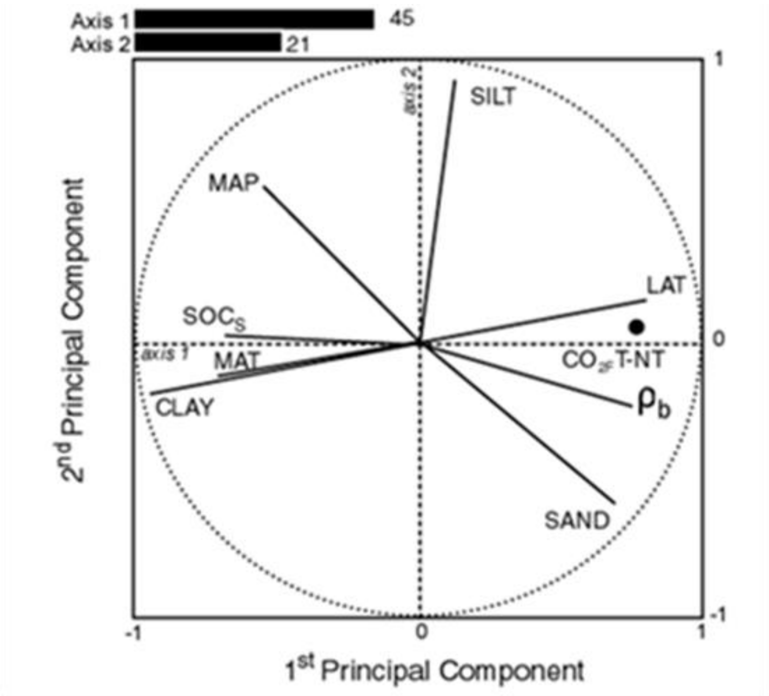

Figure 2.9 Principal components analysis (PCA) using the different environmental factors as active variables and CO_2 emission difference between T and NT (CO_{2F} T-NT) as the supplementary variable.

2.5 Discussion

2.5.1 Overall influence of tillage on SOC_C and CO_2 emissions from soil

The meta-analysis results shows that tillage has a significant impact on decreasing top-soil (0-0.03 m) organic C content (SOC_C) and increasing CO_2 emissions from soil, with 10% lower SOC_C and 21% higher CO_2 emissions in tilled than untilled soils. Lower SOC_C and higher CO_2 emissions under tillage reflect faster organic matter decomposition as a result of higher soil aeration and incorporation of crop residues to the soil, and breakdown of soil aggregates, which all render the organic material more accessible to decomposers (Reicosky, 1997; Six et al., 2002, 2004). However, results from the literature do not always agree with this. In case of soil C, for example, Cheng-Fang et al. (2012) found 7-48% higher SOC_C under tilled rice in China, when Ahmad et al. (2009) observed no significant differences. In case of CO_2

emissions from soil, while for instance Ussiri and Lal (2009) for a 43 years maize monoculture in USA observed 31% higher CO_2 emissions from tilled than from no-tilled soils, Curtin et al. (2000) and Li et al. (2010) found no significant difference in CO_2 emissions from soil between these treatments while Oorts et al. (2007) reported higher CO_2 emissions under no-tillage (4064 kg CO_2-C ha^{-1}) compared to tillage soil (3160 kg CO_2-C ha^{-1}), which they attributed to higher soil moisture content and amount of crop residue on the soil surface.

2.5.2 Influence of climate

Although there was no significant difference between arid and humid climates, CO_2 emissions from soil and SOC_C changes between untilled and tilled soils tended to be higher in arid than in humid climates (Figure 2.1). In support, Álvaro-Fuentes et al. (2008), who investigated tillage impact on CO_2 emissions from soils in a semiarid climate, attributed the observed large difference between tillage and no-tillage to differences in soil water availability. At humid sites high soil moisture favor high decomposition rates resulting in low differences between tilled and untilled soils, while large differences develop in arid climates with much lower soil water content (Fortin et al., 1996; Feiziene et al., 2011). This supports the idea that the soil response to tillage is affected by climate thresholds (Franzluebbers and Arshad, 1996).

2.5.3 Influence of soil properties

2.5.3.1 Soil organic carbon content

The decrease of CO_2 emissions differences between tillage and no-tillage soils with increasing SOC_C (Figure 2.2) is most likely due to diminishing inter-aggregate protection sites as SOCc level increases. Several studies have shown that C inputs into C-rich soils show little or no increase in soil C content with most of the added C being released to the atmosphere, while C inputs in C-depleted soils translate to higher C stocks because of processes that stabilize organic matter (Paustian et al., 1997; Solberg et al., 1997; Six et al., 2002). Another reason, which doesn't involved stabilization, is the fact that soils that have been depleted in C tend to recover and accumulate SOC until equilibrium is reached (Carvalhais et al., 2008). Therefore, abandoning tillage in soils with low SOC_C tends to offer higher protection of SOC than in soils with inherently high SOC_C levels. In support, Lal (1997) reported low SOC_C and aggregation correlations under high SOC_C soils, which suggests that substantial proportions of the SOC were not involved in aggregation. Hence, the higher difference of CO_2 emissions between tilled and untilled soils for C-depleted soils compared to C-rich soils may be due to much higher stabilization of extra SOC delivered to the C-depleted soil by protection in soil aggregates within the top-soil layers (0.0-0.05 m). Tillage of C-depleted soils is likely to lead to the breakdown of more soil aggregates, thus leading to higher decomposition of the residues added under no-tillage, as hypothesized by Madari et al. (2005) and Powlson et al. (2014).

2.5.3.2 Soil texture

Soils under zero tillage emitted less CO_2 than tilled and the CO_2 emission from soil difference was the highest in sandy soils (Figure 2.3). In addition, the higher CO_2 emissions difference under sandy soils correlated with the highest change in SOC_C. Higher SOC_C and then CO_2 differences under sandy soils might be due to the lower resistance of soil aggregates to disaggregation under sandy soil conditions with tillage highly impacting on aggregate breakdown and associated organic matter protection and loss of soil C. Another reason for the greater response of sandy soils to tillage could be the fact that sandy soils when tilled can be highly porous, thus allowing changes in soil management to translate into large variations in the gas fluxes to the atmosphere (Rastogi et al., 2002; Bauer et al., 2006). These suggestions contrast, however, with the results of for instance Chivenge et al. (2007) working in Zimbabwe and where little impact of tillage on C sequestration was found under sandy soils as compared to clayey ones.

2.5.4 Influence of the duration since tillage abandonment

The differences in SOC_C between tilled and untilled soils increased with the time since abandonment of tillage (Figure 2.4B). When abandonment of tillage took place less than 10 years old there were no differences in SOC_C between tillage and no-tillage, but for longer durations tilled soils had 14% less SOC_C than untilled soils. This can be explained by the progressive increase of soil C accumulation with time as a result of the retention of a fraction of the crop residue under no-tillage. This explanation is consistent with the results of Paustian et al. (1997) and Ussiri and Lal (2009). Six et al. (2004) reported that the potential of no-tillage to mitigate global warming is only noticeable a long time after (>10 years) a no-tillage regime has been adopted. This would suggest that shifts in CO_2 emissions differences between tillage and no-tillage will occur over time; this could not be observed in this results (Figure 5a) because the majority of experiments in this study were less than 10 years in length. Furthermore, in some cases no-tillage leads to C loss in the top-soil layer (0-0.3 m) in the first years of adoption (Halvorson et al., 2002; Six et al., 2004) and this can be attributed to slower incorporation of surface residues into the soils by soil fauna. However, several studies produced contrasting results, for instance, the long-term experiments in northern France by Dimassi et al. (2014) showed that SOC increased in the top-soil (0-0.1 m) until 24 years after tillage was abandoned, then plateaued, before continuously decreasing below 0.1 m soil layer. A loss of SOC following tillage abandonment was also suggested by Luo et al. (2010) and Baker et al. (2007).

2.5.5 Crop types, residues management and crop rotation

The no-tillage versus tillage variations of CO_2 emissions from soil and SOC_C were significant amongst the crop types (Figure 2.5) while residue retention appeared to be insignificant (Figure 6A and B). This was a surprising result because crop residues when retained on the soil surface under the no-tillage regime are expected to protect the soil against water and wind erosion (Ussiri and Lal, 2009), and improve soil aggregate stability (Chaplot et al., 2012), thus limiting soil C losses before becoming soil C through the process of decomposition and organic matter incorporation to soils.

Reicosky et al. (1995) and Wilson and Al Kasis (2008) reported on increased SOCc under maize monoculture than maize-soybean rotations because maize returns nearly twice as much residue than soybean, and as soybean residues decompose faster because of a lower C:N ratio. In addition, Van Eerd et al. (2014) using winter wheat in rotations concluded in higher C allocation to soils, which was attributed to greater belowground C inputs by cereals than legumes. Reicosky (1997) and Al-Kaisi and Yin (2005) also reported improved soil C sequestration with subsequent decrease in CO_2 emissions from soil under maize than soybean rotations due to better residue retention. However, several recent studies pointed to the lack of impact of residue management on soil C, with Lemke et al. (2010) showing that crop residue removal in a 50 years experiment did not significantly ($P > 0.05$) reduce soil C, while Ren et al. (2014) showed that inputs through wheat straw and manure up to 22 ton ha^{-1} yr^{-1} couldn't increase soil C over 4 years. De Luca et al., (2008) explained the lack of crop residue impact on soil C by the very low mass of C in residues compared to that in the bulk soil, while Russell et al (2009) having investigated several systems pointed to a concomitant increase of organic matter decomposition with C input rates. The present study tends to confirm the low impact of crop residue retention on the till *vs* no-till differences in CO_2 emissions from soil and SOCs. Crop type and rotation significantly impact on tillage effect on soil C and their role needs further appraisal.

Finally, the present analysis suggests that tilled soils emit significantly higher CO_2 emissions than no-tilled under crop rotation system (Figure 2.7). This is likely because crop rotation increases SOC_C, and microbial activity and diversity. For instance, Lupwayi et al. (1998, 1999) found greater soil microbial biomass under tillage legume-based crop rotations than under no-tillage with tillage increasing the richness and diversity of active soil bacteria by increasing the rate of diffusion of O_2 and the availability of energy sources (Pastorelli et al., 2013). This study showed that continuous monoculture did not result in significantly different CO_2 between tilled and untilled soils (Figure 2.7). Rice is one crop often produced under a continuous monoculture practice, however, in this meta-analysis, paddy rice did not show significant difference of CO_2 emissions between tillage and no-tillage soils. Li et al. (2010) and Pandey et al. (2012) attributed the lack of difference to anaerobic soil conditions occurring under both practices.

2.5.6 Nitrogen fertilization

The differences of CO_2 between tillage and no-tillage soils did not differ with nitrogen fertilizer level (Figure 2.8A), confirming observations by Alluvione et al. (2009) and Almaraz et al. (2009a). This result could be due to the fact that nitrogen fertilization increases productivity and C inputs to the soil under both tilled and untilled systems, which may dominate nitrogen effects on decomposition such as shown by Russell et al. (2009). Increasing SOC as a response to nitrogen fertilization was however expected under no-tillage over a longer period of time (Morell et al., 2010), a result also found long term in a 50 yr experiment by Lemke, et al. (2010). Yet Sainju et al. (2008) reported the opposite: a 14% increase of soil CO_2 flux with nitrogen fertilizer, because fertilizer application stimulated biological activity, thereby producing more CO_2, and potentially a SOC_C decline (Khan et al., 2007; Mulvaney et al., 2009). In contrast, Wilson and Al Kaisi (2008) showed that increasing

N fertilization generally decreased CO_2 emissions from soil, with a maximum decrease of 23% from 0-135 kg N ha^{-1} to 270 kg N ha^{-1} occurring during the growing season, which might be explained by a series of mechanisms from the inhibition of soil enzymes and fungus to the reduction of root activity.

Overall, these results pointed to little benefit of not tilling clayey soils with high SOC_C, with the highest no-tillage benefits occurring under sandy soils with low SOC_C. This can be explained by differences in soil aggregate stability. The stability of soil aggregates shows a positive correlation with clay and organic matter content. Clayey and organic soils produce stable aggregates, which are likely to be more disaggregated by tillage compared to sandy aggregates of low C content. The SOC protected within soil aggregates under no-tillage becomes exposed under tillage because of aggregate dispersion; which explains the greater reduction in CO_2 emissions with no-tillage under sandy soils. Rather, emissions are likely to be reduced under zero tillage as a result of improved soil aggregate stability and the associated protection of decomposed and stable organic matter. Crop management such as fertilization and crop type, or climate are shown to have little effect on aggregation. Our analysis did not include time since cessation of tillage as a specific predictor and classified instead the experiments into two simple categories (short versus long term).

2.6 Conclusion

The aim of this study was to provide a comprehensive quantitative synthesis of the impact of tillage on CO_2 emissions from soil using meta-analysis. Three main conclusions can be drawn. Firstly, tillage systems had 21% higher CO_2 emissions from soil than no-tillage, worldwide. Secondly, the reduction in CO_2 emissions from soil following tillage abandonment was greater in sandy soils with low SOC_C compared to clayey soils with high SOC_C. Thirdly, crop rotation significantly reduced the CO_2 emissions from untilled soil, by 26% compared to tilled soil, while continuous monocultural practice had no significant effect. This is most probably due to the fact that crop rotation can increase SOC_C and more microbial activity under a tilled compared to an untilled treatment. These results emphasize the importance of including soil factors such as texture, aggregate stability and organic C content in global models of the C cycle.

Long-term process studies of the entire soil profile are needed to better quantify the changes in SOC following tillage abandonment and to clarify the changes in the dynamics of C inputs and outputs in relation to changes in microbial activity, soil structure and microclimate. In addition, more research is needed to identify the underlying reasons why, over a long period of time, the abandonment of tillage results in a decrease in integrated CO_2 emissions that appears to be much higher than the observed increase in SOC_S. The goal remains to design agricultural practices that are effective at sequestering C in soils.

Finally, one future application of these data could be to use them to calibrate soil C models. The models could be run with prescribed inputs (from observation sites) used to simulate decomposition and the mass balance of SOC over time for different climates, soil texture and initial SOCc with respect to the theoretical value assuming equilibrium of decomposition and input (Kirk and Bellamy, 2010). Most soil C models developed for generic applications (e.g., RothC, DNDC, and CENTURY) would be suitable tools for exploitation of the data presented here (Adams et al., 2011).

CHAPTER 3: NO-TILLAGE AND MULCHING WITH CROP RESIDUES REDUCE CO_2 EMISSIONS AND INCREASE SOIL ORGANIC CARBON STOCKS

3.1 Abstract

The impact of tillage techniques on CO_2 emission from soils varies with environmental factors and land management practices. The main objective of this study was to compare CO_2 emissions and soil organic carbon (C) stocks (SOCs) from soils under no-tillage with high-density short duration stocking rate (HDSD: 1200 cows for three days per year), no-tillage with free grazing, no-tillage with crop residue mulching, no-tillage without crop residue mulching against conventional tillage with free grazing as commonly used practicein an integrated crop-livestock smallholder farming system of KwaZulu-Natal, South Africa. The CO_2 emissions were measured using LI-COR-6400XT in the last two years of the trial (2013-2014), while baseline SOCs measured in 2012-2013 were compared against values obtained during the period (2014-2015). On average, gross CO_2 emissions were 3.25±0.35 g CO_2-C m^{-2} day^{-1} in no-tillage with HDSD treatment, which was 56 and 29% lower than in conventional tillage with free grazing and no-tillage without crop residue mulching, respectively. CO_2 emissions from the soils relative to SOCs were 86 and 72% lower under no-tillage with HDSD compared to conventional tillage with free grazing (4.36±0.52 mg CO_2-C g^{-1}C day^{-1}) and no-tillage without crop residues mulching (4.02±0.44 mg CO_2-C g^{-1}C day^{-1}), respectively. In addition, CO_2 emissions from soil relative to produced biomass were significantly higher in conventional tillage with free grazing (8.02±0.97 g CO_2-C kg^{-1} biomass day^{-1}) than other treatments. After three years of treatment implication, the greatest C sequestration rate was observed under no-tillage with HDSD practice with 1.4 Mg C ha^{-1} $year^{-1}$. However, no-tillage without crop residue mulching did not show a significant effect on SOCs. The lower gross and CO_2 emission relative to SOCs from no-tillage with HDSD were attributed to a decrease in topsoil (0.05 m) temperature and increased compaction. On the basis of these results, the best land management system to mitigate against climate change would be a combination of no-tillage and high-density short duration stocking because it resulted in the lowest CO_2 emission from soil and highest C sequestration rate. However, a long-term assessment is still required to evaluate their potential to reverse global warming.

Keywords: *Global warming; No-tillage; Crop residue managements; CO_2 emissions*

3.2 Introduction

The dramatic increase in global surface temperatures by 0.8°C since the late nineteenth century is attributed to greenhouse gas (GHG) emissions to the atmosphere. The cumulative emissions of carbon dioxide (CO_2) over years 1870-2014 is estimated of 545 ± 55 Pg C (1 Pg = 10^{15} g = 1 billion tonnes) (Le Quéré et al., 2015), which is of the same order of magnitude of the total amount of C in the biosphere (620 Pg C) and in the atmosphere (720 Pg C). While two third of the past emissions is understood to come from fossil fuel combustion, the other one third is thought to be linked to agricultural activities such as deforestation and land misuse and miss management, often leading to low C input from litter, high erosion and associated C losses from the soil (An et al., 2008; Guillaume et al., 2015). Other agricultural activities leading to massive C release to the atmosphere include biomass burning and drainage of wetlands.

In face of global warming there is a strong interest in stabilizing GHG emissions to the atmosphere, as further emphasized by the Conference of Parties (COP21) held in Paris, France, in December 2015. Since soils have contributed as much as 10% of atmospheric CO_2, (Raich and Potter, 1995), restoring that C into the soil appears a promising strategy to mitigate against climate change. Soils constitute a major C sink through plant photosynthesis and subsequent transfer of the C to soils by live and dead organic matter. Moreover, soil C sequestration offers multiple benefits such as improved soil fertility, soil quality, water holding capacity and soil biodiversity.

Several soil management practices or ecological engineering techniques, such as no-tillage, mulching with crop residues, controlled grazing, nitrogen fertilizer application and crop rotations, have been shown to help in sequestering soil organic C (SOC) into soils over the long term (Lal, 2015a and b; Bhattacharyya et al., 2015). While there is a consensus on the benefits of no-tillage to C sequestration and climate change mitigation, some recent studies seem to indicate limited benefits (e.g. Baker et al., 2007; Geisseler and Horwath, 2009; Luo et al., 2010; Dimassi et al., 2014; Powlson et al., 2014). Likewise, reports on the impact of no-tillage on CO_2 emissions from soil have also been contradictory. For example, several studies reported lower CO_2 emissions under no-tilled than tilled soils (La Scala et al., 2006; Ussiri and Lal, 2009; Chaplot et al., 2015), others reported no significant differences (Aslam et al., 2000; Li et al., 2010), and yet others reported higher CO_2 emissions from no-tilled than tilled soils (Barreto et al., 2009; Smith et al., 2011).

Several studies showed that crop residue mulching in no-tillage systems offered great nutrient cycling and reduced soil erosion by improving the soil physical conditions (Adekalu et al. 2006; Chaplot et al., 2012; Carr et al., 2013). Crop residue retention protects the soil surface from water erosion (Adekalu et al., 2006) and improve soil aggregate stability (Mochizuki et al., 2008; Chaplot et al., 2012), which in turn decrease soil C losses. The benefit of crop residue mulching for SOC sequestration is dependent on the quantity and quality of the retained residues (Lal, 2008b; Wilson and Al Kaisi, 2008). However, Abdalla et al. (2015) in a global meta-analysis of 46 studies reported little crop residue mulching benefit on CO_2 emissions and SOCc in no-tilled compared to tilled soils.

While grazing of crop residues was reported to reduce C input to the soil, which in turn limits organic matter accumulation (Díaz-Zorita and Grove, 1999), integrating high-density short duration livestock stocking rate in no-tillage systems could also help reducing CO_2 emissions from soil through compaction (Chaplot et al., 2015), which could in turn aid soil C sequestration. Several studies showed that animal trampling during crop residue grazing induces significant top-soil compaction (Greenwood and McKenzie, 2001; Díaz-Zorita et al., 2002; Piva et al., 2014). Díaz-Zorita et al. (2002) reported greater soil compaction by animal trampling (in the top 0–0.05 m) of tilled than no-tilled systems in temperate Argentina, but no-significant differences in the lower (0.05-0.15 m) soil layer. However, Silva et al. (2011) reported no significant compaction effect on CO_2 emissions form soil in a laboratory incubation of mechanically compacted Oxisol soil.

Nitrogen (N) fertilier application to soils has been demonstarted to increase SOC stocks (SOCs) in the short (Morell et al., 2010) and long term (Lemke et al., 2010) because of increased biomass production, hence organic matter input to soils. Wilson and Al-Kaisi (2008) argued that SOC accumulation following N fertilization was not solely due to increased C input to the soils, but was also a result of decreased organic matter decomposition because the decomposers would get their N from the artificial N fertilizer applied. In contrast, some studies reported a decrease in SOCs following N fertilization application because of the stimulation of biological activity (Khan et al., 2007; Mulvaney et al., 2009; Sainju et al., 2008).

While shifts in agricultural practices, such as tillage to no-tillage, crop residue burial to retention on soil surfaces, livestock overgrazing to controlled grazing of crop residues, have potential for C sequestration, they are still not adequate methods for climate change mitigation. There is still a need for more research on the mechanisms of C sequestration in the systems to help in designing new soil management practices that foster soil C sequestration (through reduction in CO_2 emissions from soil and increasing SOC without compromising food security.

The main objective of this study was to compare CO_2 emissions from soils under tillage and no-tillage with different crop residue management practices. The trial site was a small scale agriculture area under a temperate climate of South Africa where sandy soils were submitted to continuous maize cultivation with different modalities of tillage, residue retention and post-harvest grazing. The CO_2 measurements were performed in conjunction with the evaluation of SOC (content and stocks), N (content and stocks), soil temperature, water content, penetration resistance and bulk density for improving the understanding of land management impact on soil C dynamics.

3.2 Material and methods

3.2.1 Study site

The study was conducted at the trial site located at Potshini (long: 29˚ 21́; lat: -28˚ 81′, 1305 m a.s.l.), 10 km north of Bergville town, in the upland of KwaZulu-Natal province, South Africa. Based on the Köppen classification, Potshini has sub-tropical humid climate with hot wet summers (October to April) and cold dry winters (May to September). The mean annual temperature and precipitation of the area are 13°C and 684 mm, respectively (Peel et al., 2007). Upslopes with shallow soils are used for cattle grazing, whereas downslopes, with 0-10% slopes, deep soils (>1.5 m) are used for rain-fed crop production (Chaplot et al., 2015). The soil in the study site (downslope) derived from sandstone and mudstone as parent material, classified as acidic Acrisol (WRB, 2006). The top-soil layer (A-horizon) is approximately 0.35 m deep has sandy loamy texture (55–68% sand and 17–19% of clay content) and fine granular structure. The soil pH ranged between 4.9–5.2, low cation exchange capacity (2–4 $cmol_+ kg^{-1}$) and SOCc ranging from 9 to 12 g C kg^{-1}. Maize (*Zea mays*) as a main crop in KwaZulu-Natal, planted around mid-November on lands tilled with draft oxen, but mechanization is becoming more common for tillage. Planting, weeding and harvesting are done manually. Little fertilizer is usually applied due to limited funds.

3.2.2 Experimental design and treatments

Before the trials were set up in 2011/12 season, the soils at the study site had been cultivated traditionally (conventional ox-drawn plough tillage and allowing free cattle grazing of the crop residues after harvesting) for a long time (at least 60 years). Maize (Zea mays) production under rain-fed irrigation was the main cropping system. The experimental site, with total surface area of 450 m^2 was divided into 5 plots of size 30 m^2 each. Five different tillage and crop residue mulching treatments, namely (i) conventional tillage with free grazing; (ii) no-tillage with free grazing; (iii) no-tillage with crop residue mulching; (iv) no-tillage without crop residues mulching and (v) no-tillage with high-density short duration stocking (no-tillage with HDSD) were then applied. Three CO_2 measurement locations were marked in each plot, and were laid out in a completely randomized design.

Conventional tillage with free grazing was the traditional tillage using an ox-drawn plough and allowing cattle to graze freely as commonly practised by the small-holder farmers in the area. The depth of ploughing was approximately 15 cm. No-tillage practice was a manual operation using a wooden wedge to open narrow slots of sufficient width and depth (approx. 5 cm x 5cm) to guarantee proper seed cover with minimum soil disturbance. At planting, Maize seeds were placed manually in the slots and then covered with just enough soil, to achieve adequate protection of the seeds with spacing of 0.3 m. There was one plot of no-tillage with free grazing where cattle were allowed to graze freely from June to September every year.

The other no-tillage plots were fenced to either eliminate or control livestock grazing. No-tillage with crop residue mulching was a no-tillage system where crop residues were retained as soil surface cover (approx. 100% mulch). No-tillage without crop residue mulching was a no-tillage system where crop residues were removed manually Finally, no-tillage with high-density short duration stocking (no-tillage with HDSD) was a no-tillage system where crop residues were retained, but subjected to high density stocking of cattle (1200 cattle ha^{-1}) for a short duration (three days per year).

3.2.3 CO_2 emissions measurements

The total (autotrophic and heterotrophic) CO_2 emissions from soils were monitored from January 2013 to May 2015, once a month in the dry season and twice a month in the wet season. The CO_2 measurements were done using a LI-COR 6400XT gas exchange system (LI-COR, Lincoln, NE, USA) attached with a LI-COR 6400-09 soil respiration chamber which had an internal volume of 991 cm^3 and covering a soil surface area of 71.6 cm^2. At the time of CO_2 measurement, the LI-COR chamber was inserted in PVC collars which were set permanently at three observation positions of each plot. Each PVC collar was inserted 0.02 m into soil leaving another 0.02 above (three collars in each treatment between maize rows). The first CO_2 measurements were done 48 hours after inserting the collars in the soil (Healy et al., 1996). The PVC collars were installed in first week of January 2013, and kept in place permanently except for temporary removal during conventional tillage and high density grazing. The CO_2 measurements were done between 10.00 and 13.00 hours to avoid the effects of diurnal temperature variations. Measuring the CO_2 emissions during this time period was found to represent average daily values in grasslands soils (Rey et al., 2011; Mielnick and Dugas, 2000).The CO_2 fluxes from soil were expressed in three units; (1) g CO_2-C per unit-surface area to evaluate gross CO_2 emissions from soil (g m^{-2} day^{-1}) to the atmosphere, (2) g CO_2-C per gram of soil C (g g^{-1} C day^{-1}) to evaluate CO_2 emissions intensity from soil relative to SOCs, which calculated as follows; CO_2 emissions (g CO_2-C m^{-2} day^{-1})/ SOCs (g C m^{-2}), and (3) g CO_2-C per kg of biomass produced (g kg^{-1} biomass day^{-1}) to evaluate CO_2 emissions intensity from soil relative to produced biomass, which calculated as follows; (CO_2 emissions (g CO_2-C m^{-2} day^{-1})/ dry aboveground biomass (kg m^{-2}).

3.2.4 Soil temperature and water content

Soil temperature was determined concurrently with CO_2 emissions from soil using a thermocouple connected to the LI-COR chamber (LI-COR 6400-09). The thermocouple was inserted 0.05 m into the soil during soil temperature measurement close to the measurement points of CO_2 emissions. Soil water content measurements also were performed as close to the collars as possible using a Hydrosense soil moisture meter (Campbell Scientific, Inc., USA) with because of methodological problem data only reported from December, 2014 to April 2015. The Hydrosense was calibrated by measurement of the meter responses at saturated soil in the study area. Finally, precipitation and air temperature were obtained from a Duncan weather station located about 500 m from the trial and at same altitude.

3.2.5 Soil sampling and analysis

Soil samples were collected in all treatments from the top-soil (0-0.05 m) layer for evaluation of SOC content (SOCc) and soil N content (Nc) at the beginning of the study and after three years of the treatments implementation. Three replicate samples were taken from around the CO_2 measurement collars. The samples were air-dried for 48 hours, ground and sieved through a 2mm sieve. Total C and N were measured using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). The total soil C was considered equivalent to SOC content when no more reaction with HCl was obtained. The SOC and N stocks (SOCs and Ns) were calculated following Batjes (1996):

$$SOCs = SOCc \ x \ \rho b \ x \ T\left(1 - \frac{PF}{100}\right) b \qquad (1)$$

where SOC_S is the soil organic C stock (kg C m^{-2}); SOC_C is soil organic C content in the ≤2mm soil material (g C kg^{-1} soil); ρ_b is the bulk density of the soil (kg m^{-3}); T is the thickness of the soil layer (m); PF is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

Soil bulk density (ρb) was determined by taking undisturbed soil by hammering cutter edge metallic cylinders with 0.075 m diameter and 0.05 m height in each plot. The soil was stored and transported in air-tight plastic bags, and then water content was determined by the gravimetric method. Soil ρb was determined using the ratio of water content corrected mass to volume of the soil following Grossman and Reinsch (2002):

$$\rho b = \frac{odw - rf - cw}{cv - \left(\frac{rf}{pd}\right)} \qquad (2)$$

Where ρb is the bulk density of < 2 mm soil material (g cm^{-3}); odw is oven dry weight (g); rf is the weight of rocks fragments (g); cw is empty core weight (g); cv is core volume (cm^{-3}); pd is the density of the rocks fragments (g cm^{-3}).

3.2.6 Penetration resistance

The penetration resistance of the top-soil (0-0.05 m), a proxy for soil compaction was measured using a cone penetrometer (Herrick and Jones, 2002). The penetration resistance was evaluated once at fifteen randomly selected positions for each treatment during the fourth season (2014-2015) before any grazing and/or tillage. This data was used to complement the bulk density measurements.

3.2.7 Dry maize biomass

In order to determine aboveground biomass, whole maize plants (i.e. grain and vegetative biomass) were harvested from randomly selected 1 m^2 areas (approx. 8 plants per m^2) in each plot. The harvest was weighed fresh in the field and sub samples representing the whole plants without grain taken to the laboratory for drying in an oven dried at 70 C until constant weight was attained. The moisture content of the subsamples was used to determine the total dry biomass for each treatment. All the plant materials used for dry biomass calculation were returned to the respective plots to ensure best application of treatments, except for no-tillage without crop residue mulching treatment.

3.2.8 Data analysis

The data were analysed as a completely randomized design. Soil properties (e.g. SOCs, SOCc, Nc, Ns, penetration resistance and soil bulk density were analysed by one-way general analysis of variance (ANOVA). CO_2 emissions data was tested for normality and homogeneity of variance. Since CO_2 emissions from soil, temperature and water content were repeatedly measured at the same locations, these parameters were statistically analyzed using mixed model Restricted Maximum Likelihood (REML) repeated measures ANOVA. The treatment means were compared using Tukey's for multiple comparisons, a significant threshold defined as $P < 0.05$, unless otherwise specified. In addition, cumulative CO_2

emissions from soil were also analysed using REML repeated measure ANOVA and the final cumulative values of the CO_2 emissions from soil between treatments were compared using Tukey's. All analyses were done using Genstat (version 14, VSN International, UK, 2011). In addition, principal component analysis was carried out to evaluate the multiple relationships between CO_2 emissions from soil and the factors of control. The relationships between CO_2 emissions from soil on hand and soil temperature, moisture and penetration resistance on the other were assessed by linear regressions, using Sigma Plot software (version 10.0, Systat Software Inc., USA).

3.3 Results

3.3.1 Tillage and crop residue management impact on carbon and nitrogen sequestration

Table 3.1 presents the mean ± standard error (SE) of SOC content (SOCc), SOC stocks (SOCs) nitrogen content (Nc), and N stocks (Ns), in the top-soil (0-0.05 m) under the different treatments at the beginning (2012) and after three years (2015) of the trial implementation. SOC and N (content and stocks) were not significantly different amongst the treatments in 2012. However, three years later (2015), SOCc was 40% higher under no-tillage with high-density short duration stocking (no-tillage with HDSD; 30.17 g kg^{-1}) than no-tillage with free grazing (21.59 g kg^{-1}). No-tillage with HDSD and no-tillage with free grazing treatments increased SOCc by 52% and 9%, respectively, compared to conventional tillage with free grazing (19.86±0.3 g kg^{-1}). SOCs increased by 57% from 12±0.4 Mg C ha^{-1} under conventional tillage with free grazing to 18.83±0.4 Mg C ha^{-1} under no-tillage HDSD. Three years after trial implementation, no-tillage with HDSD had increased SOCs by as much as 1.4 Mg C ha^{-1} yr^{-1}, while no-tillage with free grazing had increased by only 0.4 Mg C ha^{-1} yr^{-1}. In contrast, no-tillage without cattle grazing (i.e. with and without crop residue retention) and conventional tillage with free grazing resulted in a reduction of the SOCs over the considered period with a highest loss of 0.4 Mg C ha^{-1} yr^{-1} occurring under no-tillage without crop residue mulching (Table 3.1).

Soil Nc was highest in no-tillage with crop residue mulching (1.93±0.1 g kg^{-1}), followed by no-tillage with HDSD (1.89±0.1 g kg^{-1}) and lowest in conventional tillage with free grazing

(1.56±0.04 g kg^{-1}). Ns did not differ significantly amongst the treatments. However, after three years of implementation no-tillage with HDSD, increased Ns by 10%, corresponding to a sequestration rate of 0.3 Mg N ha^{-1} year^{-1}, while no-tillage with free grazing had a reduction of 0.2 Mg N ha^{-1} year^{-1}.

3.3.2 No-tillage and crop residue management impacts on selected soil physical properties

The treatment effects on soil temperature, soil water content, penetration resistance and bulk density at 0-0.05 m soil layer are presented in Figure 3.1. The soil temperature varied the most under no-tillage with HDSD and least under no-tillage with crop residues mulching (Figure 3.1A). However, conventional tillage with free grazing and no-tillage with HDSD showed high 25th -75th ranges. Overall, average soil temperature was highest under conventional tillage with free grazing with 21.76±0.45°C and lowest in no-tillage with crop residue mulching (19.88±0.41°C).

Soil water content also varied greatly in all treatments, with a general increase from conventional tillage with free grazing to no-tillage with free grazing and no-tillage with crop residue mulching treatments (Figure 3.1B). No-tillage with crop residue mulching and with HDSD treatments showed quite large 25th -75th ranges. The average soil water content was highest in no-tillage with residue mulching and lowest in conventional tillage with free grazing, with 38% difference between the two treatments.

Variability of penetration resistance was much higher under no-tillage systems than conventional tillage with free grazing (Figure 3.1C). The 25th -75th range increased dramatically from conventional tillage with free grazing to no-tillage with HDSD. The mean penetration resistance decreased from no-tillage with HDSD to no-tillage with crop residue mulching treatment. The mean penetration resistance increased by 141% from 0.085±0.8 N cm^{-2} under conventional tillage with free grazing to 0.021±1.9 N cm^{-2} under no-tillage with crop residue mulching treatment; and by 132% to no-tillage with free grazing (198±11 N cm^{-2}).

Table 3.1 Mean ±SE of selected top-soil (0-0.05m) chemical properties for the conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop residue mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high-density short duration stocking (NTHDSD) before (2012) and after three years of the treatments implementation (2015). N=9.

	SOCc (g kg^{-1})		SOCs (Mg C ha^{-1})		Nc (g kg^{-1})		Ns (Mg N ha^{-1})		Sequestration rate per year^{-1}	
Treatments	2012	2015	2012	2015	2012	2015	2012	2015	SOCs (Mg C ha^{-1})	Ns (Mg N ha^{-1})
CTFG	19.88±0.92	19.86±0.29^{c}	12.54±0.71	12.00±0.07^{c}	1.63±0.10	1.56±0.04^{b}	10.40±0.46	9.40±0.21	-0.1	-0.2
NTFG	18.23±0.28	21.59±0.38^{b}	10.99±1.01	12.51±0.11^{b}	1.53±0.07	1.79±0.10^{ab}	10.00±0.39	10.37±0.46	0.4	0.1
NTR	19.22±0.46	19.68±0.28^{c}	11.86±0.45	11.11±0.02^{c}	1.60±0.05	1.93±0.07^{a}	10.00±0.49	10.91±0.22	-0.2	0.2
NTNR	18.84±0.62	18.81±0.48^{c}	12.16±1.12	10.73±0.11^{c}	1.59±0.08	1.72±0.06^{ab}	10.20±0.43	9.83±0.43	-0.4	-0.1
NTHDSD	18.29±0.46	30.17±1.00^{a}	12.22±0.74	18.83±0.40^{a}	1.51±0.04	1.89±0.09^{ab}	10.10±0.41	11.20±0.41	1.4	0.3

Means on the same column followed by different letters are significantly different at $P < 0.05$ level.

Soil organic carbon content (SOCc), soil organic carbon stocks (SOCs), nitrogen content (Nc) and nitrogen stocks (Ns)

Soil bulk density variability was much higher under no-tillage with free grazing and with HDSD than conventional tillage with free grazing (Figure 3.1D). The 25th -75th range of bulk density was lower in conventional tillage with free grazing than no-tillage with free grazing and with crop residue mulching.

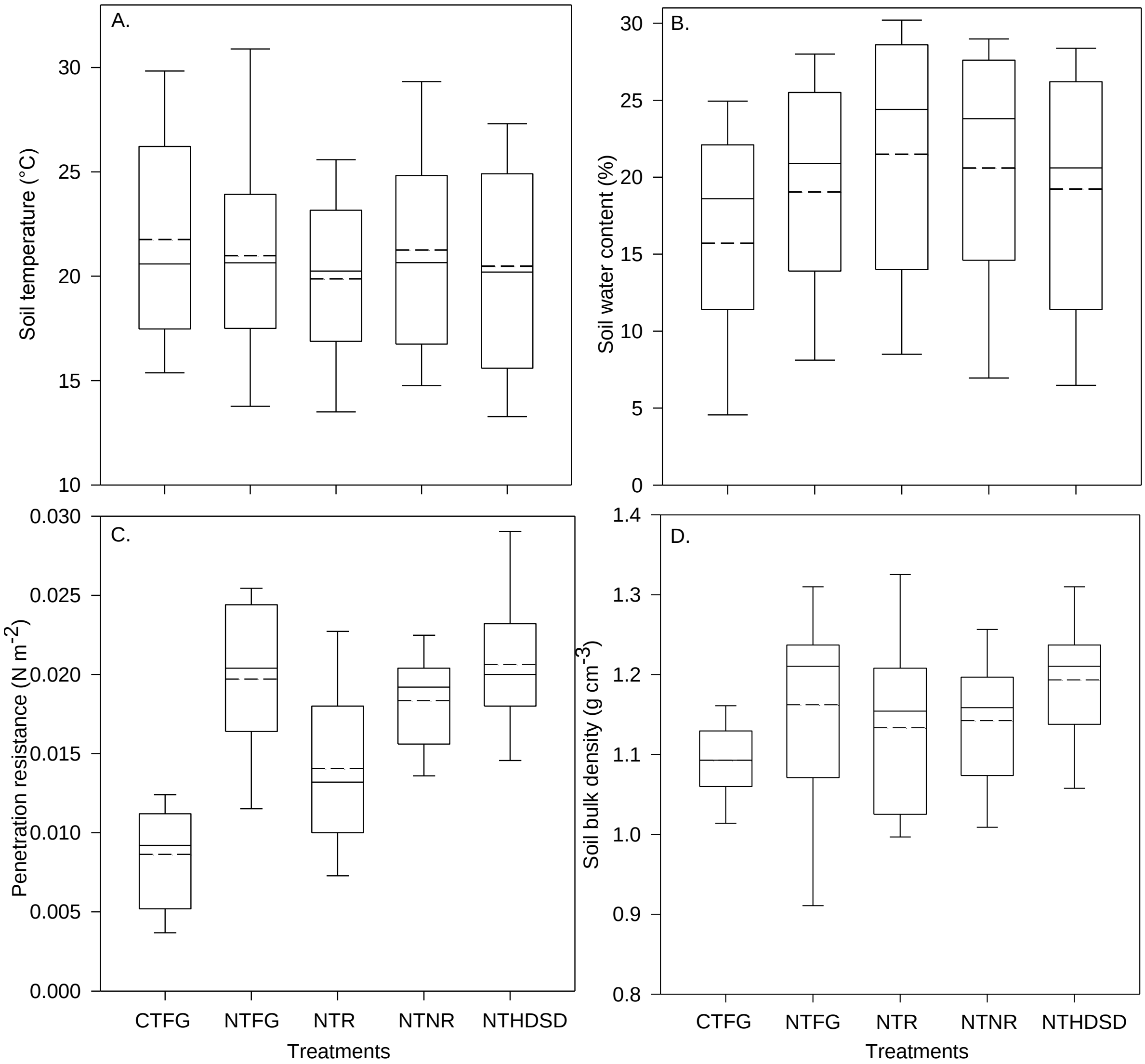

Figure 3.1 Box-whisker-plots for soil temperatures (A), soil water content (B) penetration resistance (C) and soil bulk density at 0.05 m soil depth from conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop residue mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high-density short duration stocking (NTHDSD). Plain lines correspond to 10th, 25th, median, 75th and 90th percentiles; short dash lines to the mean. N = 120, 27, 15, for soil temperatures, soil water content and penetration resistance, respectively

Overall, soil bulk density was higher under no-tillage with HDSD and free grazing by 8% and 6%, respectively, than conventional tillage with free grazing (1.09±0.02 g cm^{-3}); however, these differences were not significant.

3.3.4 Tillage and crop residue management impacts on CO_2 emissions from soil

The repeated-measures analysis showed that treatment, date of CO_2 sampling and the treatment-CO_2 sampling date interactions significantly (P<0.001) affected CO_2 emissions from soil (Table 3.2).

Table 3.2 Repeated-measures ANOVA for the effects of treatments, date of sampling and their interaction on CO_2 emissions from soil, soil temperature (ST) and soil water content (SWC)

		CO_2-C						ST		SWC	
		g m^{-2} day^{-1}		mg g^{-1}C day^{-1}		g kg^{-1} $biomass^{-1}$$day^{-1}$		°C		%	
Source of variations	DF	MS	P	MS	F pr.	MS	F pr.	MS	F pr.	MS	F pr.
Treatment	4	82.0	< 0.001	0.84	< 0.001	343.5	< 0.001	45.3	< 0.001	130.6	< 0.001
Date	1	70.0	< 0.001	0.54	< 0.001	140.9	< 0.001	356.9	< 0.001	870.5	< 0.001
Treatment × date	4	8.7	< 0.001	0.06	< 0.001	20.7	< 0.001	6.0	< 0.001	8.5	< 0.001

The overall gross CO_2 emissions from soil (g m^{-2} d^{-1}) were 21, 49, 55 and 65% higher under conventional tillage with free grazing (5.05±0.57 g m^{-2} d^{-1}) than no-tillage without crop residues mulching, no-tillage with crop residue mulching, no-tillage HDSD and no-tillage with free grazing, respectively (Table 3.3). While gross CO_2 emissions from soil (g m^{-2} d^{-1}) differed among treatments in the wet season, there were no treatment differences in the dry season. Gross CO_2 emissions from soil were up to 36% lower in no-tillage with HDSD than conventional tillage with free grazing treatment in the wet season. On the other hand, CO_2 emissions from soil relative to SOCs (mg g^{-1}C day^{-1}) were 8, 43, 55 and 86% higher in conventional tillage with free grazing (4.36±0.52 mg g^{-1}C day^{-1}) than no-tillage without crop residues mulching, no-tillage with crop residue mulching, no-tillage with free grazing and no-tillage with HDSD, respectively (Table 3.4). On average, CO_2 emissions from soil relative to SOCs decreased by 90% from no-tillage with HDSD to conventional tillage with free grazing in the wet season and only 52% in the dry season. Finally, CO_2 emissions from soil relative to produced biomass was 46, 53, 97 and 116% higher under conventional tillage with free grazing treatment than under no-tillage with HDSD, no-tillage without crop residues mulching, no-tillage with crop residue mulching, no-tillage with free grazing, respectively.

Table 3.3 Mean ± SE of CO_2 emissions from soil under conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop residue mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high-density short duration stocking (NTHDSD). N=120, 90, 30, for overall average, wet and dry period, respectively

Variable	Time	CTFG	NTFG	NTR	NTNR	NTHDSD
Gross CO_2-C ($g\ m^{-2}\ day^{-1}$)	Overall	5.05±0.57^a	3.06±0.27^d	3.38±0.31^c	4.18±0.45^b	3.25±0.35^{cd}
	Wet season	6.04±0.64^a	3.61±0.30^d	4.01±0.43^c	5.06±0.45^b	3.88±0.51^{cd}
	Dry season	1.59±0.20^a	1.18±0.14^a	1.22±0.14^a	1.39±0.17^a	1.27±0.15^a
CO_2-C relative to SOCs ($mg\ g^{-1}C\ day^{-1}$)	Overall	4.36±0.52^a	2.81±0.25^e	3.05±0.28^d	4.02±0.44^b	2.34±0.25^c
	Wet season	5.31±0.61^a	3.31±0.27^b	3.66±0.30^c	4.90±0.45^c	2.8±0.29^c
	Dry season	1.37±0.17^a	1.18±0.14^{ab}	1.11±0.13^{ab}	1.26±0.13^{ab}	0.90±0.14^b
CO_2-C relative to biomass ($g\ kg^{-1}$ biomass day^{-1})	Overall	8.02±0.97^a	3.71±0.33^c	4.07±0.38^c	5.25±0.57^b	5.50±0.59^b
	Wet season	9.76±1.12^a	4.38±0.36^c	4.89±0.41^c	6.39±0.63^b	6.55±0.68^b
	Dry season	2.25±0.32^a	1.64±0.18b	1.48±0.17^b	1.64±0.17^b	2.11±0.32^a

Means on the same row followed by different letters are significantly different at $P < 0.05$ level

3.3.5 Temporal variations of CO_2 emissions from soil

3.3.5.1 Gross CO_2 emissions from soils

The gross CO_2 emissions from soil under tillage treatments changed markedly over time, but were always much higher in conventional tillage with free grazing than the other treatments in the second month (January) after soil tillage (Figure 3.2C). In addition, differences between treatments mostly occurred in the hot and wet period. Differences in cumulative CO_2 between conventional tillage with free grazing and no-tillage without crop residues mulching were not significant from the beginning of the study (January 2013) until April 2014; however for the rest of the study period conventional tillage with free grazing treatment showed much higher cumulative values compared to no-tillage without crop residues mulching (Figure 3.2D). There were no significant differences observed among no-tillage with free grazing, no-tillage with crop residue mulching and no-tillage with HDSD over the entire study period. The final cumulative gross CO_2 emissions from soil were significantly higher in conventional tillage with free grazing (by 36%) and no-tillage without crop residues mulching (by 23%) than the average of no-tillage with free grazing, no-tillage with crop residue mulching and no-tillage with HDSD.

3.3.5.2 CO_2 emissions from soil relative to soil organic carbon stocks

CO_2 emissions from soil relative to SOCs from the treatments also changed over time with generally higher emissions occurring during the wet than dry season (Figure 3.3A). In addition, significant CO_2 emissions from soil relative to SOCs differences between treatments were observed in only 16 of the 30 wet season measurement events. Cumulative CO_2 relative to SOCs exhibited much higher CO_2 in conventional tillage with free grazing and no-tillage without crop residues mulching than no-tillage with crop residue mulching, no-tillage with free grazing and no-tillage with HDSD (Figure 3.3B). The final cumulative CO_2 emissions from soil relative to SOCs were higher in conventional tillage with free grazing treatment (175±4.53 mg g^{-1}C) and no-tillage without crop residues mulching (161±4.49 mg g^{-1}C) compared to the combined average of no-tillage with crop residue mulching, no-tillage with free grazing and no-tillage HDSD.

3.3.5.3 CO_2 emissions from soil relative to produced biomass

CO_2 emissions from soil relative to produced biomass presented in Figure 3.4A showed the same pattern as in gross and CO_2 relative to SOC stocks. Cumulative CO_2 emissions from soil relative to produced biomass were highest in conventional tillage with free grazing treatment over entire study period (Figure 3.4B). No-tillage without crop residues mulching showed much higher CO_2 than no-tillage with HDSD in the first year, while there was no significant difference between the two treatments for the rest of the study period. Finally, CO_2 emissions from soil relative to biomass decreased in the order; conventional tillage with free grazing treatment> no-tillage without crop residues mulching> no-tillage with HDSD> no-tillage with free grazing > no-tillage with crop residue mulching with the respective values of 321±13, 210±6.05, 220±3.55, 163±6.12 and 149±5.5 g CO_2-C kg^{-1}produced biomass.

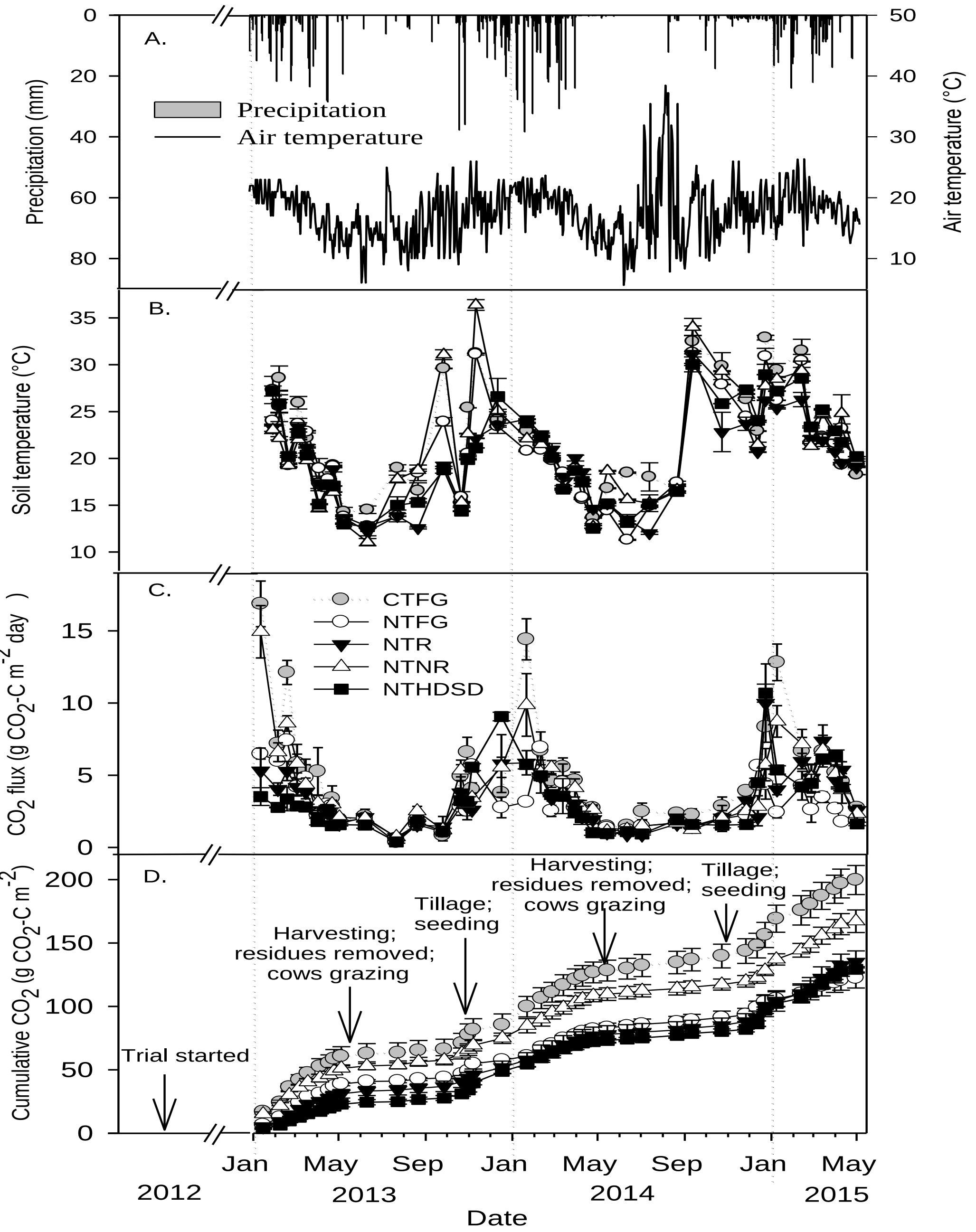

Figure 3.2 Precipitation and air temperature (A), soil temperature (B), daily fluxes (C) and cumulative (D) of gross CO_2 emissions from soil (g CO_2-C m^{-2}) conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop residue mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high density for short duration grazing (NTHDSD). Error bars represent standard error of the mean. N=3

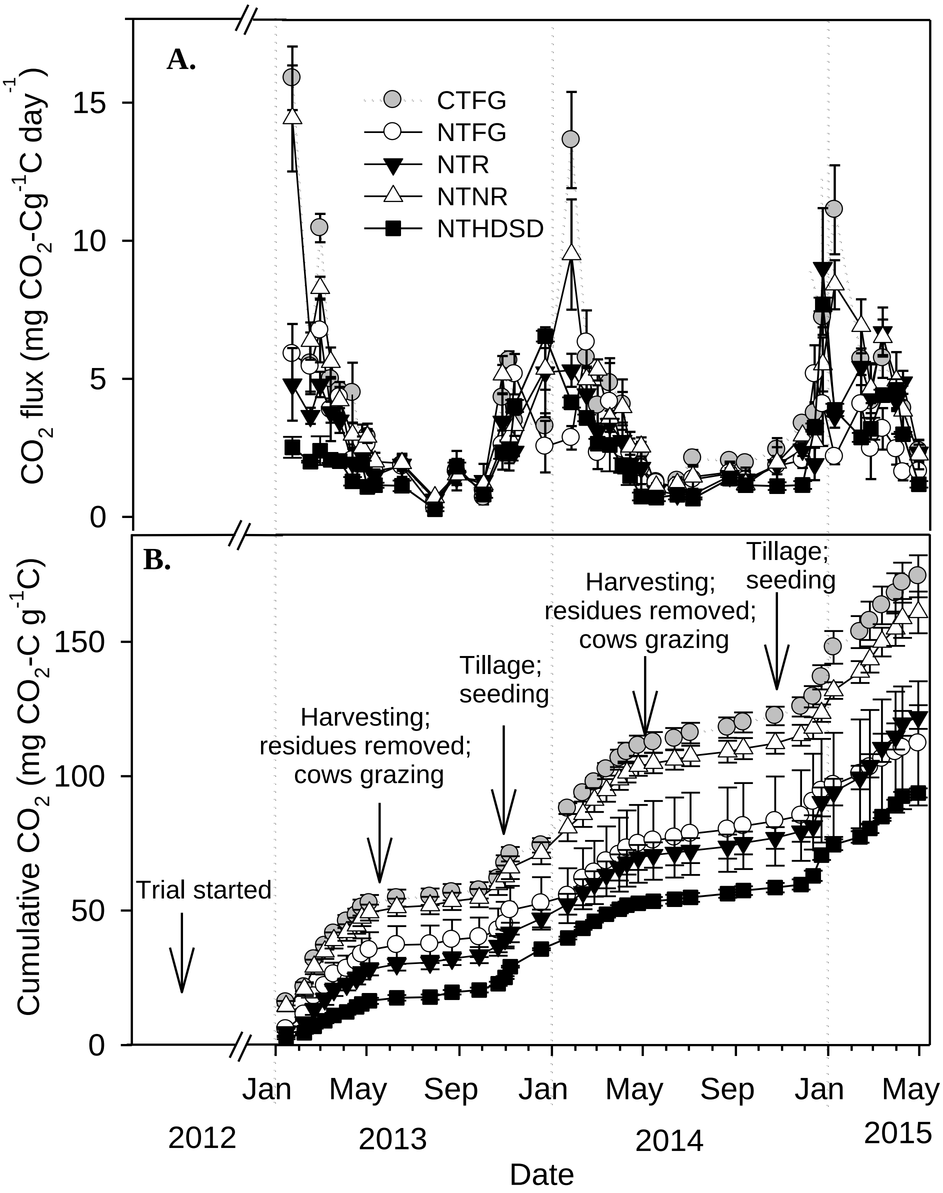

Figure 3.3 Daily fluxes (A) and cumulative (B) of CO_2 emissions from soil relative to soil carbon stocks (mg CO_2-C gC^{-1}) from conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop residue mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high density for short duration grazing (NTHDSD). Error bars represent standard error of the mean. N= 3

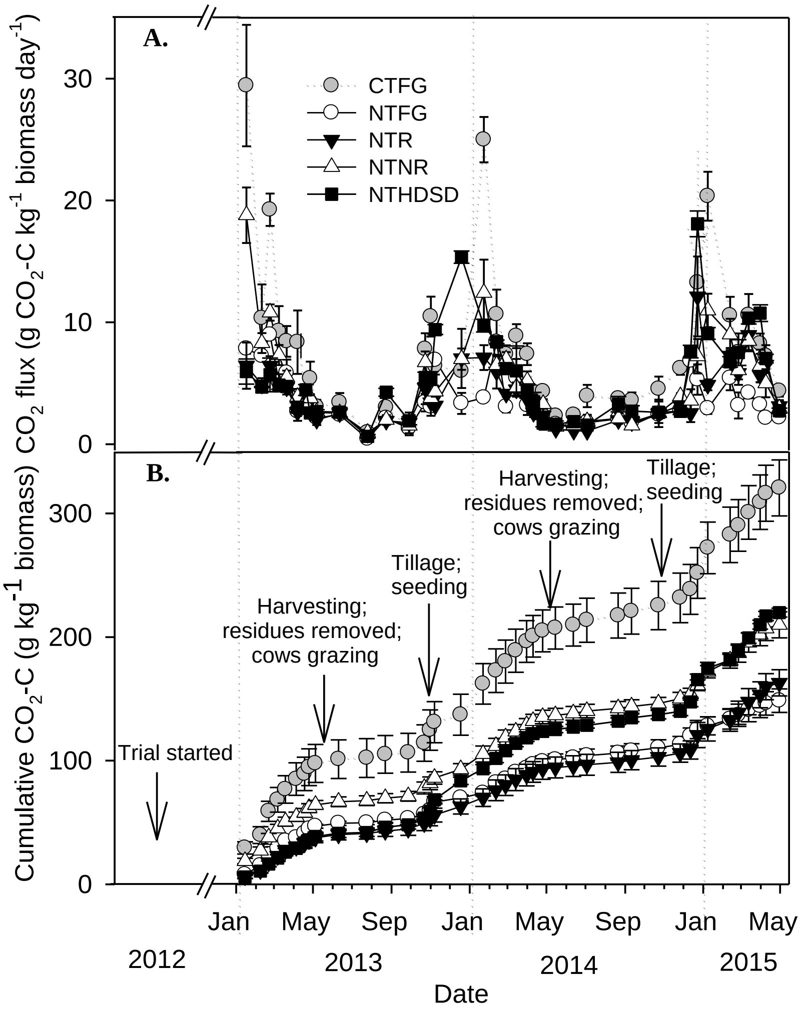

Figure 3.4 Daily fluxes (A) and cumulative (B) of CO_2 emissions from soil relative to produced aboveground biomass (g CO_2-C kg produced biomass^{-1}) from conventional tillage with free grazing (CTFG), no-tillage with free grazing (NTFG), no-tillage with crop reside mulching (NTR), no-tillage without crop residue mulching (NTNR) and no-tillage with high density for short duration stocking (NTHDSD). Error bars represent standard error of the mean. N=3

3.3.6 The main controls of CO_2 emissions from soil

The relationships between CO_2 emissions from soil and soil factors were explored using principal components analysis (PCA) (Figure 3.5). The first two axes (axis 1 and 2) of the PCA, generated using CO_2 emissions from soils and the soil factors, explained 97% of the total variation of CO_2 emissions. The first PCA axis (axis 1), accounted for 89% of the variance, and was positively correlated the most to penetration resistance and soil water content and negatively to soil temperature. The second PCA axis (axis 2), which described only 8% of the variation, was positively correlated with SOC content and stocks. The gross CO_2 emissions from soil were strongly correlated to axis 1 in the negative direction. In addition, the dependence of CO_2 emissions from soil on soil temperature, soil water content and penetration resistance at 0-0.05 m was further investigated by linear functions, which revealed that the relationships were significant for all variables (Figure 3.6). Gross and CO_2 relative to SOCs were correlated to soil temperature in the positive direction (R^2= 0.49 and 0.55, respectively) (Figure 3.6A). However, the CO_2 emissions from soil decreased with increase in soil water content with R^2 of 0.31 and 0.36 for gross and CO_2 relative to SOCs, respectively (Figure 3.6B and C). CO_2 emissions from soil also decreased significantly with the increase of penetration resistance, a proxy of soil compaction at 0.05 m soil layer (Figure 3.6E and F), with R^2 of 0.49 for gross CO_2 emissions from soil and 0.42 for CO_2 relative to SOCs.

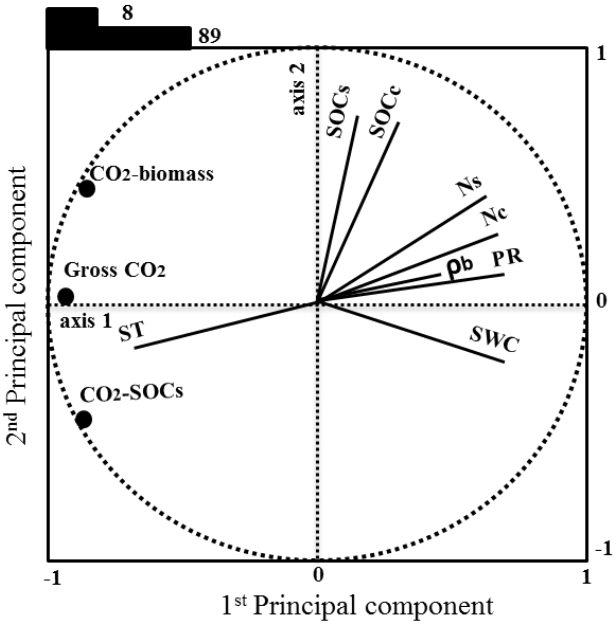

Figure 3.5 Principal components analysis (PCA) scatter diagrams for gross CO_2 emissions from soil (gross CO_2), CO_2 emissions relative to SOC stocks (CO_2-SOCs) and CO_2-C emissions relative to produced aboveground biomass (CO_2-biomass) as supplementary variables and selected soil factors (soil organic carbon content and stocks (SOCc; SOCs), nitrogen content and stocks (Nc; Ns), soil temperature (ST), soil water content (SWC), penetration resistance (PR) and soil bulk density (ρd)) as active variables. N=15

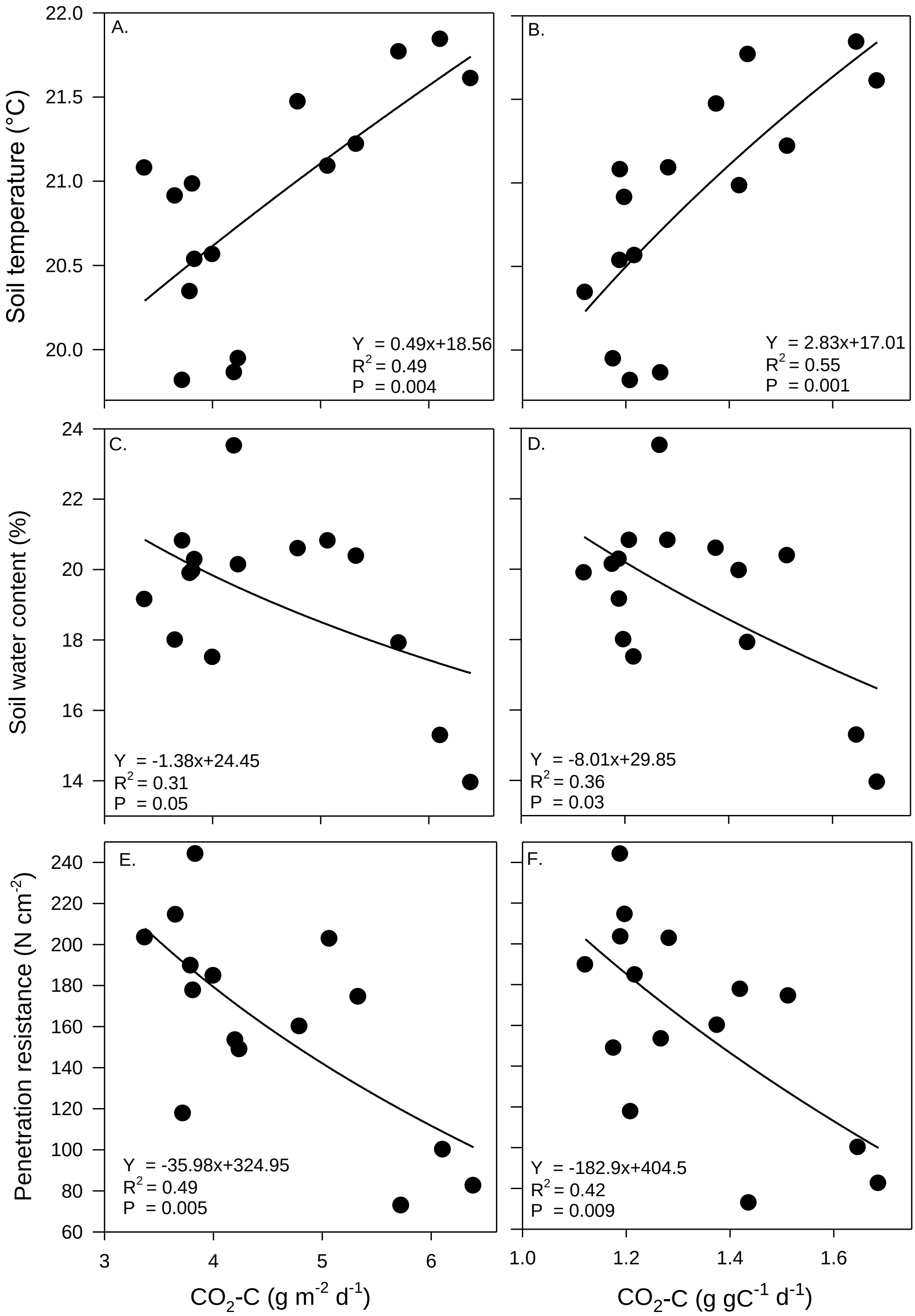

Figure 3.6 CO_2 emissions from soil plotted against soil temperature (A, gross CO_2-C emissions and B, CO_2-C relative to SOCs), soil water content (C, gross CO_2-C emissions and D, CO_2-C relative to SOCs) and penetration resistance (E, gross CO_2-C emissions and F, CO_2-C relative to SOCs). N = 15

3.4 Discussion

3.4.1 Tillage and crop residue management impacts on carbon sequestration

Three years after treatment implementation, no-tillage with free grazing and no-tillage with HDSD treatments showed higher potential to sequester C compared to conventional tillage with free grazing (Table 3.1). The greater C sequestration rate in no-tillage with high-density short duration stocking of cattle was attributed to sufficiently high amount of crop residues left-over after the trampling and soiling by the high number of cattle heads on the plot, which ultimately lead to greater soil organic matter accumulation in comparison to other treatments. In addition, the presence of crop residues and dung on the soil surface (visual observation) buffered against high soil temperature and soil water content changes, while cattle trampling also increased the top-soil (0-0.05 m) compaction (as indicated by the greater penetration resistance and soil bulk density) (Figure 3.1).

No-tillage with HDSD significantly increases SOCs after three year of implementation. This result could be attributed to the increase of C accumulation with time as a result of the crop residue fractions retention. However, this study showed fast effects of no-tillage to increase C sequestration compared to other studies such as e.g. Ussiri and Lal (2009) and Six et al. (2004).Six et al. (2004) reported that no-tillage increase C sequestration during the second decade of the adoption. This study suggests that integrating high-density cattle stocking for short duration increased no-tillage potential for C sequestration. Such results, could be attributed to the the presence of large animals which can positively modify nutrient cycling and increasing soil quality (Carvalho et al. 2010).

Several studies reported that cattle trampling increases soil compaction, which in turn affect soil properties such as penetration resistance, soil bulk density, infiltration rate, soil moisture and soil temperature (Greenwood and McKenzie, 2001; Hamza and Anderson, 2005; Agostini et al., 2012; Piva et al., 2014). For instance, Agostini et al. (2012) reported high stocking rate of cows significantly increased penetration resistance in the top 0.05 m soil depth of a mollisol soil managed for 13 years with no-tillage under crop-livestock systems in Argentina. Hamza and Anderson (2005) reported, in a review of global literature published in the period

1990-2005, that the depth of trampling-induced soil compaction varied between 0.05 to 0.2 m depending on animal weight and soil moisture.

Top-soil compaction have shown to reduce soil C loss as a results of lower organic matter mineralization (Silva et al., 2011) or as dissolved organic C due to low infiltration rate (Agostini et al., 2012). Additionally, the combination of soil compaction and crop residue mulching reduces surface soil loss (C-enriched horizon) by water erosion (Adekalu, et al. 2006). However, some studies reported higher bulk density and penetration resistance due to animal trampling under tilled than no-till soils (Silva et al., 2000; Duarte and Díaz-Zorita, 1999). Silva et al. (2000) argued that soil compaction due to animal trampling could be responsible for the degradation of the physical quality of soils and mainly influences soil structure.

In the current study, SOC (both stocks and content) did not increase after three years of no-tillage with crop residue mulching (Table 3.1), which agreed with several other studies (e.g. Lemke et al., 2010; Singh et al., 2015; Abdalla et al., 2015) suggesting little to no crop residue mulching impact on soil C. For instance, Singh et al. (2015) reported limited increase of top-soil C by reduced tillage and straw management practices in a 30-year experiment on cereal monoculture systems in the boreal region of Southern Finland. Abdalla et al. (2015), in a global meta-analysis, also reported that SOCc was 12% lower under tillage than no-tillage in the absence of crop residues, and only 5% lower when crop residues were left on the soil; but the differences were not significant in both cases. However, several studies pointed to great benefit of no-tillage with crop residues mulching on C sequestration (e.g. West and Post, 2002; Al-Kaisi and Yin, 2005; Ussiri and Lal, 2009; Hou et al., 2012).

3.4.2 Tillage and crop residue management impacts on CO_2 emissions from soils

The result showing significantly higher CO_2 emissions from soil in conventional tillage with free grazing compared to all no-tilled treatments (Table 3.3 and Figure 3.2), implies higher stimulation of CO_2 emissions from soil under tillage practises. As explained by several studies, tillage accelerate SOC oxidation to CO_2 through increased soil aeration, putting crop residues in direct contact with soils and exposing aggregate protected organic matter to decomposers (e.g. Beare et al., 1994; Six et al., 2000). However, differences in CO_2 emissions from soils due to tillage effect vary greatly worldwide. For example, La Scala et al. (2006) reported 73% higher CO_2 emissions in tilled than no-tilled soils in Brazil, while Sainju et al. (2010) found only 2% in USA but Abdalla et al. (2015), in a global meta-analysis found 21%. Some studies pointed to no significant CO_2 differences (Aslam et al., 2000; Oorts et al., 2007; Li et al., 2010), yet others reported on higher CO_2 from no-tilled than tilled soils (e.g. Barreto et al. (2009) and Smith et al. (2011) in Brazil and USA, respectively).

Lower CO_2 emissions from soil under no-tillage with high-density short duration stocking of cattle than tilled soils with free grazing, in the current study, was attributed to lower soil temperature, and soil compaction. This study confirmed that CO_2 emissions from soil increased with soil temperature and also with decreasing soil water content (Figure 3.5A and B). Many studies have also reported significant correlations between CO_2 emissions from soil on one hand and soil temperature and water content on the other (e.g. Fortin et al., 1996; Al-Kaisi and Yin, 2005;Ussiri and Lal, 2009; Mathiba et al., 2015). However, Despite greater CO_2 emissions from tilled than no-tilled soils, Moussadek, et al. (2011) reported low correlations between CO_2 emissions from soil and soil temperature and water content in a Mediterranean Vertisol. The lower soil temperature observed under no-tillage with high-density short duration stocking of cattle and no-tillage with crop residue mulching treatments compared to conventional tillage with free grazing and no-tillage without crop residue mulching, in the current study, was attributed to low thermal conductivity provided by crop residue mulch (Duiker and Lal, 2000). On the other hand, the mulch on soil surface also reduces water losses by evaporation (Mitchell et al., 2012). Greater soil storage capacity is one of major advantages of no-tillage especially under low rainfall climates (Jemai et al., 2013; Abdullah, 2014).

In addition to the importance of soil temperature and water content (Reichstein, et al. 2000; Wang et al., 2010; Guntiñas et al., 2012), organic matter mineralization is also influenced greatly by soil compaction (Tan and Chang, 2007; Silva et al., 2011). This study pointed to lower CO_2 emissions from compacted soils, which agreed with results found by Silva et al. (2011) and Chaplot et al. (2015). The lower CO_2 emissions from the compacted soil layer could be attributed to poor aeration due to low porosity in the compacted soil layer leading to less O_2 and associated low microbial activity, which consequently lowers organic matter mineralization (Silva et al., 2011). Another possible explanation is probably greater physical protection of soil organic matter from decomposers in compacted soils (Silva, et al., 2011; Torbert and Wood, 1992). In support, Silva, et al. (2011) reported 29% lower CO_2 emissions from compacted than less compacted soil, while as much as 65% difference observed by Torbert and Wood (1992).

Overall, the results pointed to greater benefit of no-tillage for C sequestration than the other treatments when it is associated with high-density short duration stocking of cattle. This was explained, mainly, by low soil temperature (as a result of mulching, due to cow trampling on the maize residues) and high soil compaction (due to cattle trampling). This result confirms the finding by Chaplot et al. (2015) in a laboratory incubation of bulk soils from the same experimental site, which were compacted manually to different levels. Their study suggested that C sequestration could be achieved by increasing top-soil bulk density. Moreover, the mean C sequestration rate from no-tillage with HDSD (1.4 Mg C ha^{-1} $year^{-1}$) was 4 fold greater than the amount (0.34 Mg C ha^{-1} $year^{-1}$) reported in other no-tilled soils (direct sown) such as in tropical Brazilian (Bayer et al., 2006) and temperate North American soils (West and Marland, 2002). However, the C sequestration rates for Brazilian and North America soils cannot be compared with results in this study because our results are based on data from 0.05 m soil depth of no-tillage associated with grazing, while Bayer et al. (2006) and West and Marland (2002) considered only 0.02 and 0.03 m layers, respectively. Assuming that C sequestration rate is constant from soil surface to one meter depth, no-tilled soils with high-density short duration stocking could store as much as 28 Mg C ha^{-1} $year^{-1}$. Adoption of no-tilled soils with high-density for short duration stocking would, therefore, be one of the available solutions to offset the annual net CO_2 emission estimated at 11 Pg in 2014 (Lal, 2015a).

3.5 Conclusions

In this study conducted in a smallholder farming system in South Africa, with a main objective to investigate the drivers of CO_2 emissions from soil under different tillage and crop residue management practices. Five tillage practices were compared, namely; conventional tillage with free grazing, no-tillage with free grazing, no-tillage with crop residue mulching, no-tillage without crop residue mulching and no-tillage with high-density short duration stocking (1200 cows ha^{-1} for three days per year). Three main conclusions can be drawn from the study. The first one was that, three year of no-tillage with free grazing resulted in top-soil (0-0.05 m) C sequestration rate of 0.4 Mg C ha^{-1} $year^{-1}$, and as much as 1.4 Mg C $ha^{-}year^{-1}$ in no-tillage with high-density short duration stocking. In addition, short-term (three years) tillage with free grazing, no-tillage with and without crop residue mulching lead to soil C loss. The second one is that, no-tillage with high-density short duration stocking decreased the overall average of gross CO_2 emissions from soil by 56% compared to conventional tillage with free grazing and by 29% compared to no-tillage without crop residue mulching. The third one is that, climate parameters such as temperature and precipitation are an important factors that control CO_2 emissions from soil.The greater SOCs and lower CO_2 emissions from soil under no-tillage with high-density short duration stocking compared to conventional tillage with free grazing could be a result of lower mineralization as suggested by significant correlations between CO_2 emissions from soil (average of gross and CO_2-C relative to SOCs) and soil temperate ($R^2 = 0.52$) and penetration resistance ($R^2 = 0.46$). The integration of high-density short duration stocking within no-tillage practices in monoculture systems could result in soil surface compaction which in turn decreases C export from the soils. These results could improve the understanding of the factors driving CO_2 emissions from soils under different tillage and crop residue management practices. This research results could open new perspectives of cropland management for smallholder farmers to be more efficient in food production while at the same time increasing the potential of their agro-systems to mitigate against climate change. However, crop-livestock system would increase other GHG emission (e.g. N_2O and CH_4) due to, mainly, excreta depositions and the anaerobic soil conditions induced by top-soil compaction. The high C sequestration rate achieved by no-tillage with high-density cattle stocking for short duration would offset these emissions. More studies related to GHG emissions must be carried out in different climate and soil conditions.

CHAPTER 4: GRASSLAND DEGRADATION INCREASES CO_2 EMISSION BASED ON SOIL AND PLANT ORGANIC CARBON STOCKS

4.1 Abstract

Grassland degradation significantly affects plant diversity; primary production and soil fertility, however little has been done to evaluate its consequences on CO_2 emission from soils. The main objective of this study was to quantify the impact of grassland degradation on CO_2 emission from soils and the main controls. A grassland showing different degradation intensities (non-degraded grassland, which exhibited an aerial cover, Cov, of 100%; degraded, with 25<Cov<50%; and highly degraded grassland, with 0<Cov<5%) in KwaZulu-Natal province, South Africa, was used for the study. The CO_2 emissions from soil were measured at three randomly selected locations per grassland degradation intensity, once a month in winter and twice in summer from January 2013 to April 2015 using a LI-COR 6400XT. Non-degraded grassland exhibited the highest gross CO_2 emissions (1.78±0.013 gCO_2-C m^{-2} day^{-1}), which was162% higher than from highly degraded grassland. However, CO_2 emissions relative to soil organic carbon (C) stocks (SOC_S) (0.034±0.01 g CO_2-C g^{-1} C day^{-1}) and relative to produced biomass (1.76±0.2 g CO_2-C kg^{-1} biomass day^{-1}) were lowest in non-degraded grassland. Degraded grassland had the highest CO_2 emission from soil relative to SOCs (0.058±0.02 g CO_2-C g^{-1} C day^{-1}) while highly degraded grassland had the highest CO_2 emissions relative to produced biomass (3.18±0.61 g CO_2-C kg^{-1} biomass day^{-1}). Overall, gross CO_2 emissions from the soils increased significantly with SOC content and stocks (r = 0.83 and 0.82, respectively) and with soil water content (r = 75), but decreased with increasing clay content (r = -0.89). CO_2 emissions relative to SOCs decreased significantly with increasing SOC content (r = -0.50). These results demonstrate that, grassland degradation greatly contribute to the current rise in atmospheric greenhouse gases and global warming. In contrast, grassland rehabilitation might allow sequestration of atmospheric C while improving grass production; however more still need to be done to identify rehabilitation techniques that increase biomass production at the same time limiting CO_2 emissions from soil.

Keywords: *Carbon cycle; Climate change; Grassland degradation; CO_2 emissions*

4.2 Introduction

Increasing greenhouse gas (GHG) concentrations in the earth's atmosphere, as a result of anthropogenic disturbances, and its direct influence on climate change is a matter of great concern. Grasslands play a crucial role in the global C cycle as they cover about 40% of world surface area and store 10% of the soil C stocks of 2400 Pg (1 Pg = 10^{15} g = 1 billion tonnes) (Suttie et al., 2005). Since the grasslands are recently subjected to degradation (Lal, 2004), which is associated with high C losses to the atmosphere.This makes studies of CO_2 emissions from soils under grasslands extremely important.

Land degradation, defined here as a process which lowers current or potential capability of soils to produce food and fodder (FAO, 1979), is generally attributed to the global climate change and human activities (Shang and Long, 2007; Gang et al., 2014; Fassnacht et al., 2015). Grassland degradation can be induced to a noticeable degree by different group of factors, which are used for diagnosis of the degradation severity, such as plant variables e.g. vegetation cover, plant diversity and productivity, which can be estimated from acquired space borne remote sensing imagery (Zhao et al, 2009) or manually. Biophysical variables such as altitude, slope, precipitation, temperature and soil conditions (Li, et al. 2012), factors that induce overgrazing or degradation such as livestock and wildlife (Garibaldi et al., 2007), and variables that describe socioeconomic development and human interference on grassland (Yang, et al. 2004) can also be used". However, most of these methods require long-term data which was not available for the study site. The site was located in a communal area where no studies were done before.

Approximately 50% of global grasslands are thought to be already degraded (Gang et al., 2014). Grassland degradation has well known negative consequences on grass production and biodiversity (UNEP, 2007; Dong et al., 2012). It may also constitute a threat to both the existing soil organic C (SOC) stocks (SOCs) and the capacity to sequester more C into soils (Nair et al., 2011; Zhang et al., 2011).

Grassland degradation in this study, defined as a reduction of soil cover by vegetation, thus exposing the top-soil to water erosion (Podwojewski et al., 2011; Mchunu and Chaplot, 2012;

Dalimini et al., 2014), which has negative consequences on soil physical, chemical and biological properties (Dlamini et al., 2011; Nunes et al., 2012; Yi et al., 2012). For example, the negative consequences of grassland degradation on soil properties include lowering soil water content (Yi et al., 2012) and increasing soil temperature (Mills and Fey, 2004). In a meta-analysis of 131 comparative studies worldwide, which examined the impact of grassland degradation on SOCs, Dlamini et al. (2016) found that grassland degradation reduced SOCs by an average 9%, with 16 and 8% reductions in dry and wet climates respectively. Zhang et al. (2011), in temperate continental semi-arid monsoon climate of eastern Inner Mongolia, China, recorded a reduction in total C of 14%. Dlamini et al. (2014) recorded much greater losses of soil C (89%), in a degraded grassland of sub-tropical climate in South Africa.

Although many studies have investigated the consequences of grassland degradation on soil properties including SOCs, little is known about its impact on CO_2 emissions from soil. Among the few existing studies, Rey et al. (2011) in Southeastern Spain found 25% higher CO_2 emission from soil in non-degraded than degraded grassland. A recent study by Traoré et al. (2015) reported 82% higher average CO_2 emissions from soil under native land cover (non-degraded) than from degraded land cover in a semi-arid climatic region of West Africa. The main reason for higher CO_2 emission from soil in non-degraded than degraded grasslands in both cases was greater contribution of root biomass to soil respiration (Traoré et al. 2015; Rey et al., 2011).

There is a general lack of quantitative *in-situ* evaluation of the impact of grassland degradation level (severity) on CO_2 emissions from soil, not only for gross emissions (i.e. emissions per unit of surface) but also for CO_2 emissions from soil relative to SOCs and relative to produced biomass. In addition, more is still to be known on the factors controlling the changes in CO_2 emissions following grassland degradation. Only a single study has been conducted on South African grasslands to estimate CO_2 emissions from soil in response to grassland degradation in communal grassland of KwaZulu-Natal (Mchunu and Chaplot, 2012). This study, which reported that plant cover reduction (as indictor of grassland degradation) significantly decreased the cumulative CO_2-C emissions by as much as 68% in comparison to non-degraded grasslands, was based on laboratory incubation of bulk soils and thus need to be validated under natural conditions and over time.

The main objective of this study, performed *in-situ* at Potshini in KwaZulu-Natal, was to quantify the consequences of grassland degradation on CO_2 emissions from the soil and the main factors of control. The research site had a degradation gradient showing a grass aerial cover (as indicator for degradation) as high as 100% in non-degraded grassland to highly degraded grassland (with 0-5% grass aerial cover).

4.3 Materials and methods

4.3.1. Study area

The experiment was conducted at Potshini (29° 21' E; 28° 48' S), 10 km south of Bergville in KwaZulu-Natal province, South Africa. The experimental site is located on a sloping land of about 10% gradient with altitude ranging from 1080 to 1455 m.a.s.l. The climate is sub-tropical humid characterised by warm wet summers and cool dry winters. Long-term (30 year) mean annual temperature and precipitation at the study site were 13°C and 684 mm, respectively (Dlamini et al., 2011). Weather data for the study were obtained from a Duncan weather station situated about 500 m from the study site. Soils in the area were classified as Plinthic Acrisols according to World Reference base (WRB, 2006). The parent materials are sandstone and mudstone. The native vegetation of the study area is dominated by the Moist Highveld Sourveld (Camp and Hardy, 1999). More details about the study site and its soil properties can be found in Dlamini et al. (2014), Mchunu and Chaplot (2012) and Podwojewski et al. (2011).

4.3.2 Experimental design

An experiment site with 1500 m^2 (30 m × 50 m) area and homogeneous soils but showing variations in grassland degradation intensity, varied from highly degraded area (bare soils) to non-degraded grasslands (100% grass aerial cover) (Mchunu and Chaplot, 2012; Dlamini et al., 2014). The grass aerial cover means the area of the ground covered by the vertical projection of the aerial portion of the plants (USDA, 1996). Grass aerial cover was measured by demarcate a 1 m × 1 m ground area at fixed intervals along each corresponding arial grass cover degree. The aerial cover of the plants in the demarcated area (1 m × 1 m) was taken as an estimate of the % of total area (Dlamini et al., 2014). CO_2 emissions from soil measurements were performed for three grassland degradation categories: non-degraded grassland, which exhibited an aerial cover, Cov, of 100%; degraded, with $25<Cov<50\%$; and highly degraded grassland, with $0<Cov<5\%$. Within each degradation category, three replicate represented by three plastic (PVC) collars (diameter = 10 cm, height = 4 cm) were installed, two weeks prior to the first CO_2 emissions from soil sampling date to avoid errors associated with soil disturbance (Hui-Mei et al., 2005; Heinemeyer et al., 2011). Each collar was positioned between grass tufts and was inserted 2 cm into the soil. More information about the experimental site, plots description and soil properties of the study site can be found in Dlamini et al. (2014)

4.3.3 CO_2 emissions measurements

CO_2 emissions from soil were measured from Jan 2013 to April 2015, once a month in winter and twice a month in summer. The measurements were done using LI-COR 6400XT gas exchange system (LI-COR, Lincoln, NE, U.S.A.) fitted with a LI-COR 6400-09 soil respiration chamber. The closed chamber system had an internal volume of 991 cm^3 and surface area of 71.6 cm^2 (Healy et al., 1996). The chamber was positioned on each PVC collars (three collars per plot) immediately prior to CO_2 measurement. In order to avoid strong diurnal variations, all the CO_2 measurements were carried out between 10.00 and 13.00 hours. Measuring the CO_2 emissions during this time period was found to represent average daily values in grasslands (Rey et al., 2011; Mielnick and Dugas, 2000). The CO_2 fluxes from soil were expressed in three units: (1) g CO_2-C per unit of surface area (g CO_2-C m^{-2} day^{-1}) to evaluate the gross CO_2 emissions from soil to the atmosphere; (2) g CO_2-C per gram of C in the soil (g CO_2-C g^{-1}C day^{-1}) to evaluate the CO_2 emissions from soil relative to soil organic C stocks; and (3) g CO_2-C per kg of produced biomass (g CO_2-C kg^{-1} day^{-1}) to evaluate the CO_2 emissions from soil relative to aboveground biomass production.

4.3.4 Soil temperature and soil water content

Soil temperature was determined using a thermocouple connected to the soil chamber (LI-COR 6400-09), which was inserted 0.05m into the ground. Soil temperature was measured in conjunction with CO_2 emissions and soil water content and for the same soil layer (0-0.05 m). However, soil water content measurements only took place from December, 2014 to April

2015 (one season) due to methodological issue. Soil water content measurements were performed as close to the PVC collars as possible using a Hydrosense soil moisture meter (Campbell Scientific, Inc., USA), which was calibrated using saturated soil at the study site.

4.3.5 Soil sampling and analysis

Soil samples from the top-soil (0-05 m) layer were collected for the evaluation of soil organic C content (SOCc) and nitrogen content (Nc). Sampling was performed in June 2014, with three replicates taken from near each CO_2 measurement collar. The samples were air-dried for 48 hours, then gently grounded and sieved through a 2 mm sieve. Total C and N were measured using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). The total soil C was considered equivalent to soil organic C content (SOCc) when no reaction could be obtained on addition of HCl. Undisturbed soil samples for bulk density determination were collected by inserting metallic cylinders with 7.5 cm diameter and 5 cm height into the topsoil. The collected samples were stored in hematic plastic cans and later dried in an oven at 105°C for 24 hours. The bulk density was calculated according to Grossman and Reinsch (2002). The pipette method was used to determine the clay content after removal of organic matter with H_2O_2 and dispersion using Na-hexametaphosphate. The SOC stocks (SOCs) and N stocks (Ns) were calculated using the following equation by Batjes (1996):

$$SOCs = SOCc \; x \; \rho b \; x \; T\left(1 - \frac{PF}{100}\right) b \tag{1}$$

where SOC_S is the soil organic carbon stock (kg C m^{-2}); SOC_C is soil organic carbon content in the ≤2mm soil material (g C kg^{-1} soil); ρ_b is the bulk density of the soil (kg m^{-3}); T is the thickness of the soil layer (m); PF is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

4.3.6 Aboveground biomass

The aboveground biomass was evaluated in three randomly placed metallic quadrats (0.5 m × 0.5 m) in each grassland degradation category at peak biomass in June, 2013 and 2014. All shoot material from the soil surface to the crown within the quadrats was clipped. The plant samples were oven-dried at 70°C and then weighed until constant weight to estimate the aboveground biomass (kg m^{-2} $year^{-1}$).

4.3.7 Statistical analysis

Summary statistics were calculated for the CO_2 emissions from soil under non-degraded, degraded and highly degraded grassland during the whole study period. Since the CO_2 emissions measurements were done at regular time intervals from the same points, was statistically analysed using mixed model Restricted Maximum Likelihood (REML) repeated-measures analysis of variance. The grassland degradation catogrey means were compared using Tukey's for multiple comparisons, a significant threshold defined as $P < 0.05$, unless otherwise specified T. In addition, cumulative CO_2 emissions from soil were also analysed using REML repeated measure ANOVA and the final cumulative values of the CO_2 emissions from soil between treatments were compared using Tukey's. All analyses were done using Genstat (version 14, VSN International, UK, 2011). In addition, coefficients of determination (r) and principal component analysis (PCAs) were carried out to evaluate relationships between the CO_2 emissions from soil and the factors of control.

4.4 Results

4.4.1 Impacts of grassland degradation on soil variables

The SOCs, SOCc, Nc, Ns, C: nitrogen ratio (C:N ratio) and soil bulk density for the top-soil (0-0.05m) in non-degraded, degraded and highly degraded grasslands are shown in Figure 4.1. The mean SOCs and SOCc were highest in non-degraded and least in highly degraded grassland. SOCs in non-degraded were 60% higher than in degraded and 86% higher than in highly degraded grassland. The SOCc decreased with increasing grassland degradation as follows: non-degraded (15.72±0.58 g kg^{-1}) > degraded (5.69±0.66 g kg^{-1}) > highly degraded (1.84±0.18 g kg^{-1}). The decrease from non-degraded to degraded grassland corresponded to 64%, while that from non-degraded to highly degraded grassland was 88%. The Ns were lower by 65% in non-degraded compared to highly degraded, while no significant difference existed between degraded and highly degraded grassland. C:N ratio decreased with increasing grasslands degradation, averaging 13±45, 11±0.61 and 5±0.88 for non-degraded, degraded and highly degraded, respectively; the difference observed between non-degraded and degraded grasslands was not significant (P < 0.05). Soils under highly degraded and degraded had higher bulk density (13%) than non-degraded one; however the difference was not significant at P<0.05.

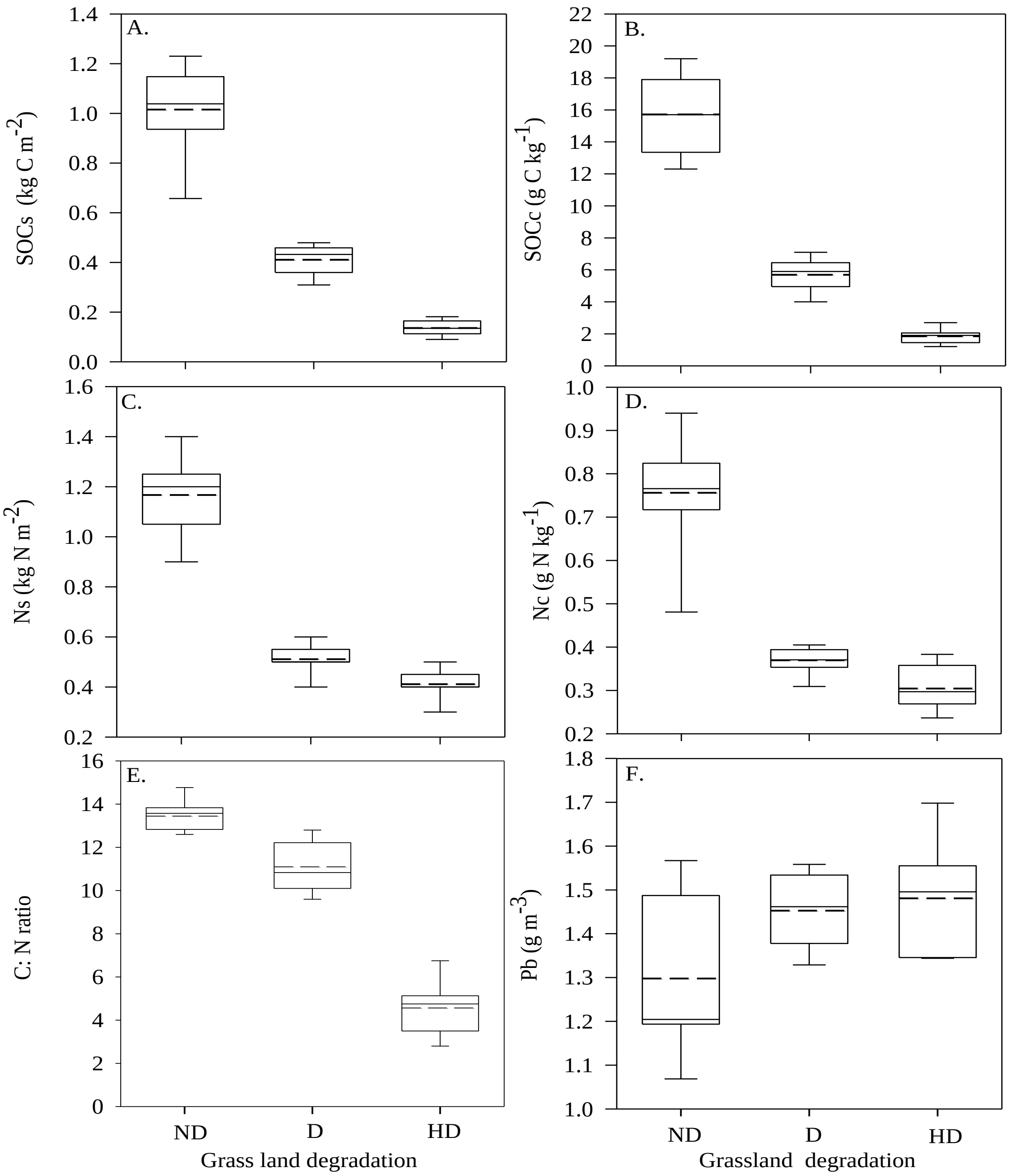

Figure 4.1 Grassland degradation (ND, non-degraded; D, degraded; HD, highly degraded) impact on (A) SOC stocks (SOCs); (B) SOC content (SOCc); (D) nitrogen content (Nc); (C) nitrogen stocks (Ns); (E) carbon to nitrogen ratio (C:N ratio) and (F) soil bulk density (ρb) in 0-0.05 m soil layer Plan lines corresponded to 10th, 25th, median, 75th and 90th percentiles; Medium dashed lines to the mean, N = 6

Soil temperature at the 0-0.05 m depth, over the course of the study period, ranged from 9°C in February 2015 to 45°C in June 2015 in both study years, respectively (Figure 4.2). The highest soil temperature was in degraded and the lowest in highly degraded grassland. Soil temperature in each grassland degradation level changed markedly over time ($P < 0.001$), but in degraded and non-degraded grasslands were always higher than highly degraded. Overall, mean soil temperature was significantly lower in highly degraded (21.8°C) compared to non-degraded (24.8°C) and degraded (25.8°C) plots.

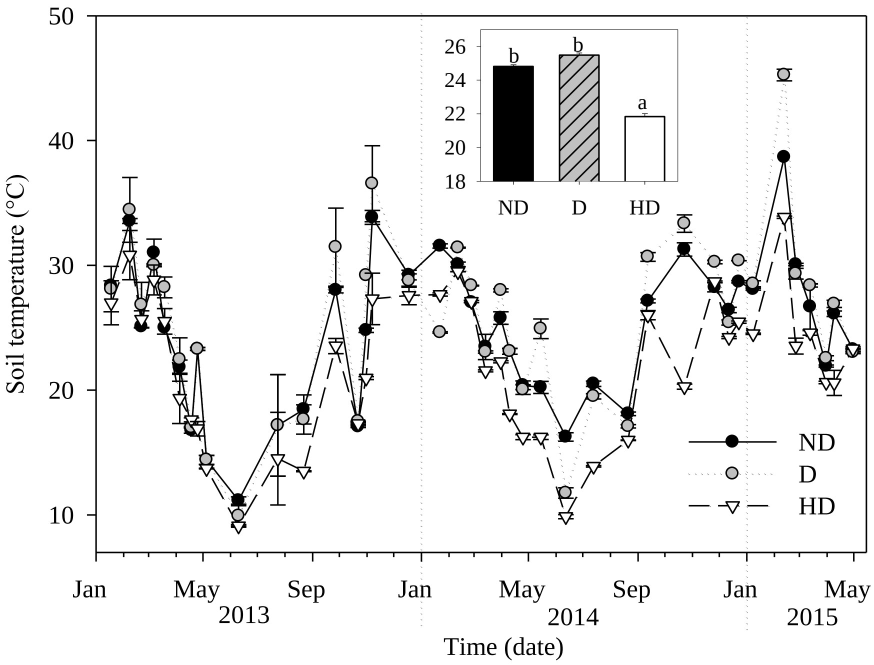

Figure 4.2 Soil temperature (insert: overall mean ±SE) at 0-0.05 m soil layer over the study period from non degreaded (ND), degreded (D) and highly degreded (HD) grassland. Different lower case letter indicates significant different ($P < 0.05$) between the degradation gradients. Error bars represent standard error of the mean .N = 3

4.4.2 Impacts of grassland degradation on CO_2 emissions from soil

Figure 4.3 shows the variations of CO_2 emissions from soil under non-degraded, degraded and highly degraded grasslands. The overall mean daily gross CO_2 emissions from soil for the study period were 11% higher in non-degraded than degraded grassland, and 62% higher in non-degraded than the highly degraded grassland. The overall mean daily CO_2 emissions from soil relative to SOCs increased by 41% from non-degraded to degraded, and by 15% from non-degraded to highly degraded grassland. However, CO_2 emissions from soil relative to produced biomass were 23% lower in non-degraded than degraded, and 81% lower in non-degraded than highly degraded grassland.

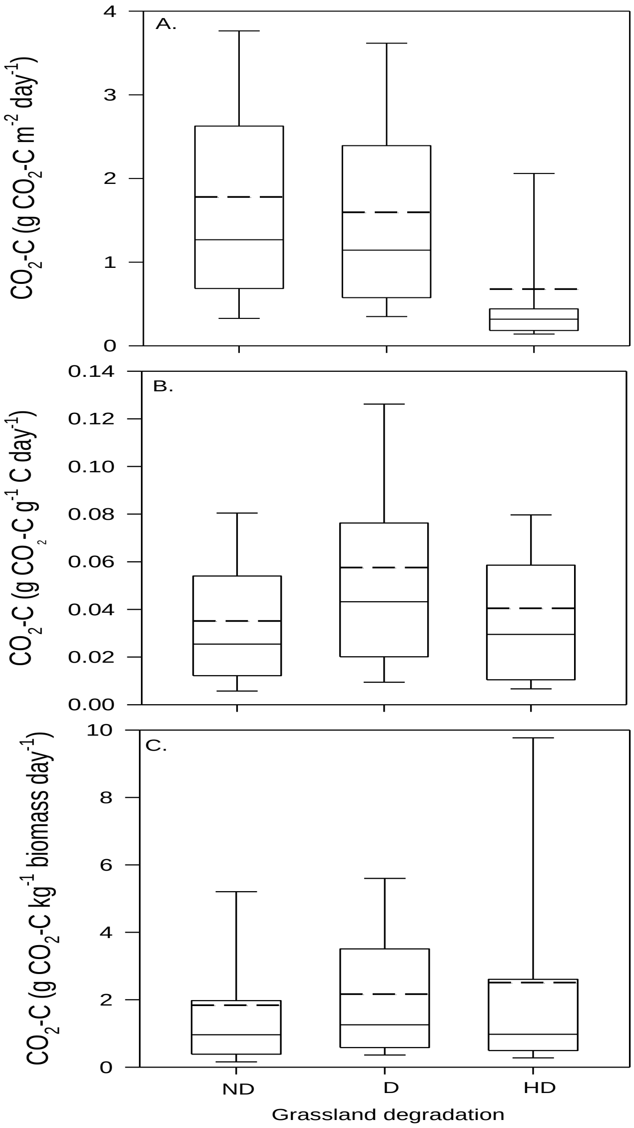

Figure 4.3 Grassland degradation (ND, non-degraded; D, degraded; HD, highly degraded) impact on (A) gross CO_2-C (gCO_2-C m^{-2} day^{-1}); (B) CO_2-C emissions relative to SOCs (g CO_2-C g^{-1} C day^{-1}); (D) CO_2-C emissions relative to biomass production (g CO_2-C kg^{-1} biomass day^{-1}). Plan lines corresponded to 10th, 25th, median, 75th and 90th percentiles; Medium dashed lines to the mean, N = 40

The ANOVA results indicated that grasslands degradation categories, date of CO2 sampling and their interactions had highly significant ($P < 0.001$) effects on CO2 emissions from soils (Table 4.1)

Table 4.1 Repeated-measures ANOVA of the effects of grassland degradation, date of CO_2 sampling and their interaction on CO_2 emissions from soil

	CO_2-C						
		g CO_2-C m^{-2} day^{-1}		g CO_2-C g^{-1}C day^{-1}		g CO_2-C kg^{-1} biomass day^{1}	
Source of variation	DF	MS	P	MS	P	MS	P
Degradation	2	41.80	<0.001	0.018	<0.001	63.87	<0.001
Time	39	11.64	<0.001	0.010	<0.001	82.87	<0.001
Degradation* Time	78	01.36	<0.001	0.002	<0.001	18.38	<0.001

4.4.3 Temporal variations of CO_2 emissions from soil

4.4.3.1 Precipitation and air temperature

The daily precipitation and average daily air temperature, over the study period are shown in Figures 4. 4 A, 4.5 A and 4.6 A. The total annual precipitation was 718 mm in 2013 and 562 mm in 2014, with about 90% of the precipitation in each year occurring in summer (November to April). The mean annual air temperature was 17°C for both 2013 and 2014. Highest daily average air temperature (38°C) was recorded in September 2014 and lowest (13°C) in June for both 2013 and 2014. The CO_2 emissions were higher under wet and hot than dry and cool conditions. However, the daily average temperature for the period from July to September 2014 was unusually high but CO_2 emissions were low.

4.4.3.2 Gross CO_2 emissions from soil

The patterns of gross CO_2 emissions from soil show that CO_2 changed over time, but were consistently higher in non-degraded than degraded and highly degraded grassland in most cases (Figure 4.4B). The differences between non-degraded and highly degraded grasslands were observed to be significant at only 17 out of the 40 sampling events. In addition, significant differences between degraded and highly degraded grasslands were observed only in three events during this period. The significant differences occurred mostly in summer.

There were no significant differences in winter CO_2 emissions among the degradation categories. However, values were significantly lower during winter periods. While the differences in cumulative CO_2 emissions from soil between non-degraded and degraded grasslands were not significant, this was not the case between degraded and highly degraded grasslands (Figure 4.4C). The final cumulative gross CO_2 emissions from soil were significantly lower by 62% under highly degraded than non-degraded and by 58% in degraded than non-degraded grassland.

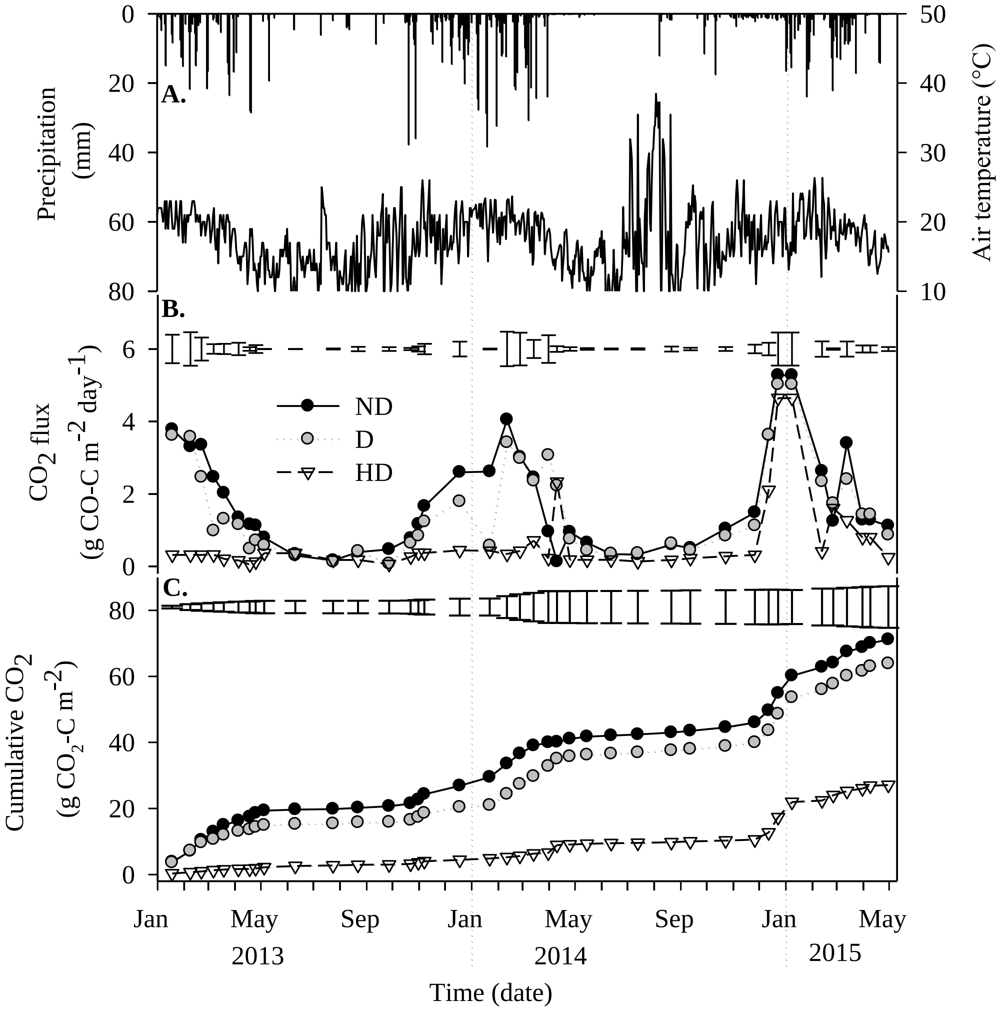

Figure 4.4 Precipitation and air temperature (A), daily gross CO_2 (g CO_2-C m^{-2}) emissions (B) and cumulative CO_2 from soil (C), over the study period from non-degraded (ND), degraded (D) and highly degraded (HD) grassland. Error bars represent ± one standard error of the difference. N = 3

4.4.3.3 CO_2 emissions from soil relative to SOCs

CO_2 emissions from soil relative to SOCs also varied over time, with generally higher CO_2 in degraded than non-degraded and highly degraded grassland during summer months (October to April) (Figure 4.5B). The CO_2 emissions from soil relative to SOCs were higher under highly degraded compared to degraded and non-degraded grasslands in winter 2013 (May to September) only. The differences between degradation categories were only significant for 5 out of the 27 summer sampling events. In addition, significant differences between degraded and non-degraded grasslands were only observed in three events during this period (Figure 4.5B). In general, summer CO_2 emissions from soil relative to SOCs were higher than in winter. There were no significant differences in winter CO_2 emissions from soil relative to SOCs between the degradation categories. The cumulative CO_2 emissions from soil relative to SOCs exhibited much higher cumulative CO_2 emissions in degraded than highly degraded and non-degraded grasslands (Figure 4.5C). There were no significant differences in cumulative CO_2 emissions from soil relative to SOCs between non-degraded and highly degraded grasslands over the study period. At the end of the study period, the cumulative CO_2 emissions from soil relative to SOC_S was highest in degraded (2.3±1.2 g CO_2-C g^{-1} C), followed by highly degraded (1.6±0.18 g CO_2-C g^{-1} C) and lowest in non-degraded (1.4±0.09 g CO_2-C g^{-1} C) grassland. While the difference in final cumulative CO_2 emissions relative to SOCs between degraded on one hand and highly degraded and non-degraded grasslands on the other hand was significant, this was not the case between highly degraded and non-degraded grasslands.

4.4.3.4 CO_2 emissions from soil relative to SOCs

CO_2 emissions from soil relative to produced biomass appeared significantly different amongst the degradation categories only between January and May, each year (Figure 4.6B). For example, degraded grassland had significantly higher CO_2 emissions from soil relative to produced biomass than both non-degraded and highly degraded grasslands in 2013, while highly degraded was much higher than both degraded and non-degraded grasslands in 2015. Cumulative CO_2 emissions from soil relative to produced biomass was slightly higher in degraded followed by non-degraded, but emissions in highly degraded grassland increased sharply in May 2014 to a value much higher than both in non-degraded and degraded grasslands (Figure 4.6C). Thereafter, CO_2 emission from soil relative to produced biomass at highly degraded grassland continued increasing at a faster rate resulting in highest cumulative value (100±32 g CO_2-C kg^{-1} biomass produced) by end of the study. At the end of the study, cumulative CO_2 emissions from soil relative to produced biomass under degraded were lower than highly degraded but still higher than non-degraded grassland. The difference in final cumulative CO_2 emissions from soil relative to produced biomass between highly degraded and degraded grasslands was 13% and between highly degraded and non-degraded was 27%, however, these differences were not significant at $P < 0.05$.

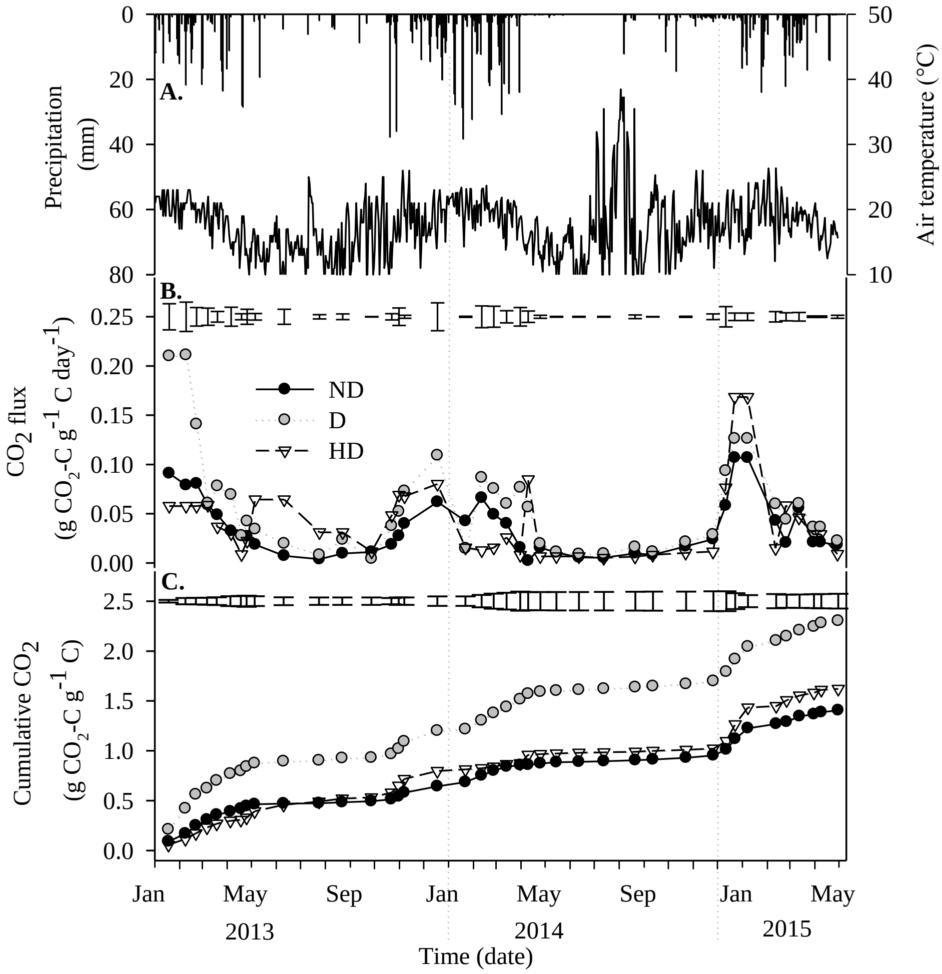

Figure 4.5 Precipitation and air temperature (A), daily CO_2 emissions from soil relative to SOCs (g CO_2-C g^{-1} C) (B) and cumulative CO_2-C (C), over the study period from non-degraded (ND), degraded (D) and highly degraded (HD) grassland. Error bars represent ± one standard error of the difference. N = 3

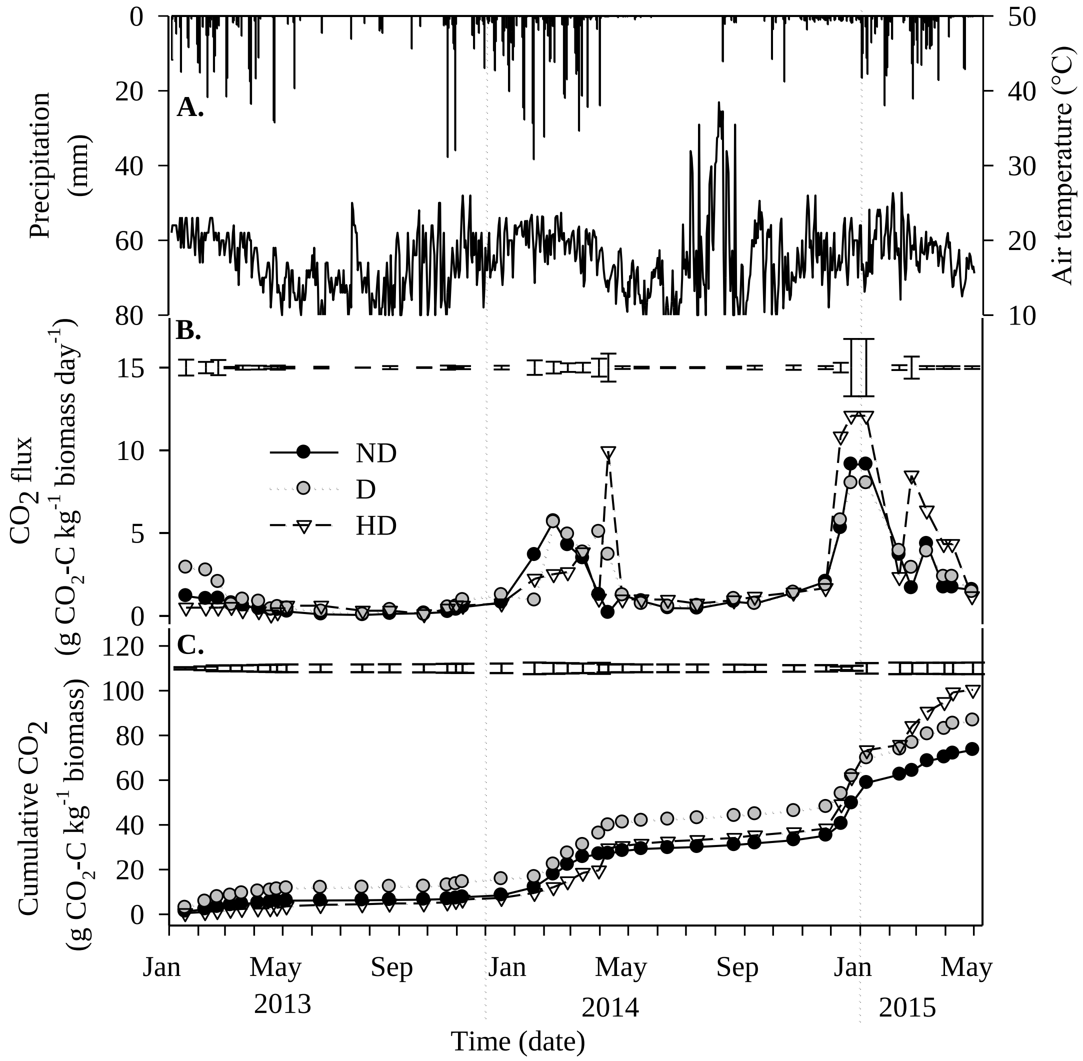

Figure 4.6 Precipitation and air temperature (A), daily CO_2 emissions from soil relative to produced biomass ($g\ CO_2$-$C\ kg^{-1}$ produced biomass) (B) and cumulative CO_2, over the study period from non-degraded (ND), degraded (D) and highly degraded (HD) grassland. Error bars represent ± one standard error of the difference. N = 3

4.4.4 Controls of CO_2 emissions from soil

Gross CO_2 emissions from soil increased significantly with SOCc (r = 0.83), followed by SOCs (r = 0.82), soil water content (r = 0.75) and Nc (r = 0.67), but decreased with soil clay content (r = -0.89) (Table 4.3). On the other hand, CO_2 emissions from soil relative to SOCs were shown to decrease significantly with increasing SOCs and SOCc with r = -0.51 and -0.50, respectively. CO_2 emissions from soil relative to produced biomass increased significantly with C: N ratio (r = 0.78), followed by Ns (r = 0.60), soil temperature (r = 0.59) and soil bulk density (r = 0.58); but decreased significantly with aboveground biomass (r = -0.55).

The relationships between CO_2 emissions from soil and selected factors were investigated further using principal components analysis (PCA) (Figure 4.6). The first two axes (axis 1 and 2) of the PCA in Figure 4.6A, generated using CO_2 emissions from soil and the environmental factors, explained 77% of the total variation of the dataset. The first axis (axis 1), which described 44% of the variance, was positively correlated to SOCc, SOCs, Nc, C:N ratio and Ns. The second axis (axis 2) describing 33% of the variation was negatively correlated with clay content. The gross CO_2-C emissions from soil were increased the most with soil water content and decreased with clay content. The CO_2 emissions relative to produced biomass was strongly correlated to axis 1 in the positive direction. The other PCA in Figure 4.7B shows that axis 2 and axis 3 accounted for 56% of the CO_2 variation. In this PCA, Nc, SOCc, SOCs, aboveground biomass, C: N ratio and Ns correlated negatively to gross CO_2 and positively to CO_2 relative to SOCs.

Table 4.2 Coefficients of determination (r) between gross CO_2 emissions (g CO_2-C m^{-2}), CO_2 emissions relative to soil carbon stocks (g CO_2-C g^{-1} C) and CO_2 emissions relative produced biomass (g CO_2-C kg^{-1} biomass) from soil and multiple factors: soil organic carbon content and Stocks (SOCc and SOCs), nitrogen content and stocks (Nc and Ns), carbon: nitrogen ratio (C:N), soil bulk density (ρb), Clay content, soil water content (SWC), soil temperature (ST) and aboveground biomass (AGB)

CO_2-C	SOCc	SOCs	Nc	Ns	C:N	ρb	Clay	SWC	ST	AGB
g CO_2-C m^{-2}	0.83*	0.82*	0.67	0.53	0.62*	-0.22	-0.89	0.75	-0.12	0.37
g CO_2-C g^{-1} C	-0.50*	-0.51*	-0.45	0.42	-0.29	0.02	-0.18	-0.17	0.22	0.00
g CO_2-C kg^{-1} biomass	0.15	0.35	0.16	0.60*	0.78*	0.58*	0.12	-0.24	0.59*	-0.55*

*Statistically significant determinants at P <0.

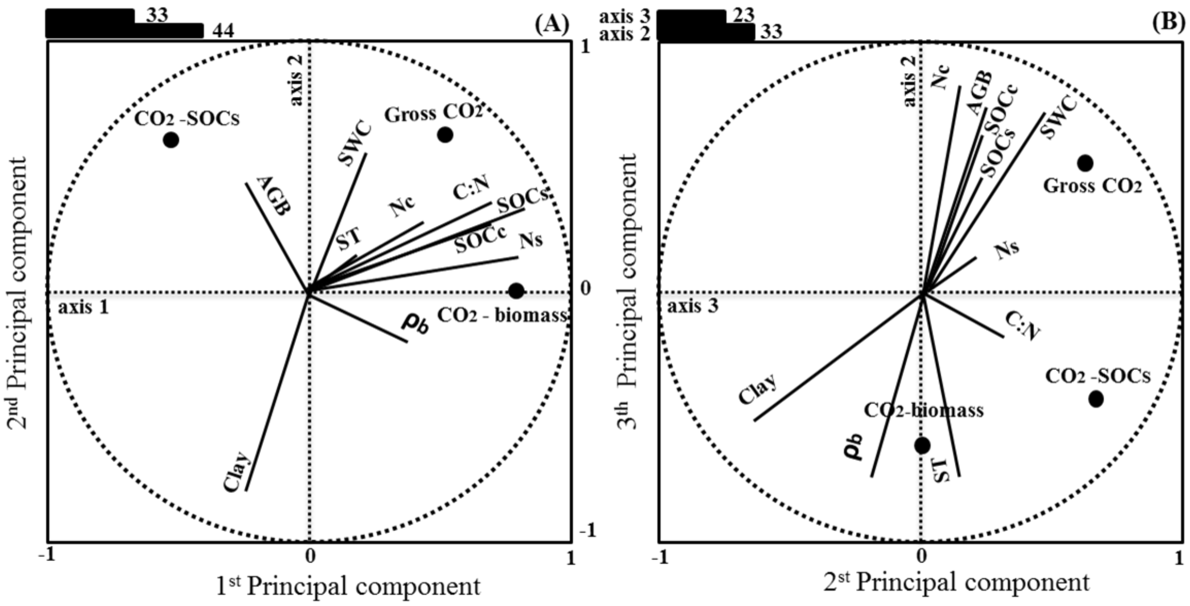

Figure 4.7 Principal components analysis scatter diagrams for gross CO_2 emissions (gross CO_2), CO_2 emissions relative to soil organic carbon stocks (CO_2-SOC$_S$) and CO_2-C emissions relative to produced biomass (CO_2-biomass) as supplementary variables and selected factors as active variables. (A) scatter diagram with the two first PCA axes (axis 1 and 2); (B) scatter diagram with axis 2 and 3.

4.5 Discussion

4.5.1 Grassland degradation impacts on CO_2 emissions from soil

4.5.1.1 Gross CO_2 emissions from soil

Gross CO_2 emissions from soil (CO_2-C emissions per area basis) decreased significantly with grassland degradation (Figure 4.1; non-degraded grassland had 11 and 62% higher CO_2 emission than degraded and highly degraded grassland, respectively), suggesting lower CO_2 stimulation under degraded grasslands. This result confirms findings in previous studies that reported lower CO_2 emissions in degraded than non-degraded grasslands (Wang et al., 2010; Rey et al., 2011; Traoré et al., 2015). The low CO_2 emissions from highly degraded grasslands could largely be explained by low above and below ground biomass and subsequently low root respiration. Reduced vegetation results, indeed, in a reduction of root biomass with direct consequences on CO_2 emissions as root respiration has been reported to contribute between 38 and 78% of total CO_2 emissions from soil (Raich and Tufekciogul, 2000; Wanga et al., 2005). Another reason for the decrease of CO_2 emissions with grassland degradation could be due to lower C inputs from the aboveground biomass to the soil (Zhao et al., 2011; Li et al., 2009). Reduced microbial biomass in soils due to unfavorable conditions for microbial activity (Nunes et al., 2012; Li et al., 2015) might also explain such a decreas. Since SOCs positively correlated (r = 0.82) to CO_2 emissions in the present study, lower SOCs in the degraded grasslands might also be a reason to explain the lower CO_2 emissions from the degraded grassland soils. However, contrary to the above mentioned results, a study in Qinghai-Tibetan Plateau of China by Li et al. (2015) found greater CO_2 emissions in degraded grasslands than non-degraded swampy grassland, which was explained to be a result of greater C mineralization as degraded grasslands exhibit higher soil temperature.

4.5.1.2 CO_2 emissions from soil relative to soil organic carbon stocks

CO_2 emissions from soil relative to SOCs increased significantly with grassland degradation, by 41% when grassland degradation decreased from non-degraded to degraded, and by 15% from non-degraded to highly degraded grassland (Figure 4.1), implying lower soil C protection under degraded grassland. The lower C protection under degraded grassland could be explained by lower soil aggregate stability as a result of soil water erosion, which washes away the organic C rich topsoil exposing the subsoil with less C (Mchunu and Chaplot, 2012). However, the fact that the greatest increase in CO_2 emissions from soil relative to SOCs occurred during the initial stage of grassland degradation was not expected. The results on soil temperature (Figure 4.2) pointed to higher soil temperature which might explain the lowest organic matter protection at the degraded grassland than at non-degraded one (Wang et al. 2010; Davidson and Janssens, 2006). However, the fact that differences in soil temperature between non-degraded and degraded grasslands soils are not significant led us to probably refute that hypothesis. The priming, which is the stimulation of soil organic matter mineralization as a result of incorporation of fresh organic matter in the soils (Fontaine et al, 2003; Bingeman et al., 1953), might play an important role in explaining the highest CO_2 emissions from the intermediate category of grassland degradation.

No such priming effect was observed at the highly degraded sites, where the old organic matter from the out cropping deep B-horizons is not in contact with fresh organic matter either from the eroded A surface horizons or from the vegetation, which have been both removed. Indeed, because of top-soil loss, deep organic matter of high recalcitrance is put in contact with fresh organic matter which increases microbial activity due to the greater availability of energy and nutrients. The availability of energy and nutrients enhances organic matter decomposition and associated CO_2 emissions from soil (Fontaine et al, 2003). In addition, why there is a relatively low increase in emissions at the highly degraded sites? Among the possible explanations these results point to significantly lower soil temperature (average of 21.8 vs 24.8°C at non-degraded) and soil water content (average of 9.4 vs 11.2%) in the highly degraded than non-degraded sites. The later was contrary to what Haddix et al. (2011) found, i.e. high depletion of stable organic matter due to the sensitivity of resistant organic matter to high temperature following grassland degradation.

4.5.1.3 CO_2 emissions from soil relative to produced biomass

Grassland degradation significantly increased CO_2 emissions from soil relative to biomass production by as much as 81% in case of the highest degradation category (Figure 4.1) and at the same time reduced grass biomass production by 82% for non-degraded to highly degraded. Therefore, the increase of CO_2 emissions from soil relative to biomass production from non-degraded to degraded grassland could be explained by the fact that non-degraded systems emit high CO_2 from soil with production of high level of biomass, hence the CO_2 emission from soil will be lower compared to highly degraded grasslands where biomass production is very low.

4.5.2 Keys factors affecting gross CO_2 emissions from soil

The significant and positive correlations between gross CO_2 emissions from the soils, SOC and N were consistent with results from several studies (e.g. Rey et al., 2011; Liu et al., 2014 and Li et al., 2015). These results could be explained by the fact that organic matter is the substrate of CO_2 emissions from soils. Moreover, the high and positive correlation found between gross CO_2 emissions and soil water content (Table 4.3) can be explained to a certain extent by the fact that soil water content enhances microbial activity and thus soil organic matter decomposition (Wu et al., 2010). Other factors such as soil bulk density, which Chaplot et al. (2015) showed to significantly decrease CO_2 emissions from the soil under a maize production system, appeared to be insignificant. In the present study, CO_2 emissions from the soil negatively correlated with clay content, which could be due to the fact that the clay-enriched B horizon was outcropping to the soil surface in the highly degraded grassland. However, contrary to our results, Li et al. (2015) reported significantly positive correlation between CO_2 emissions and clay content in the Qinghai-Tibetan Plateau of China.

4.6 Conclusions

Three main conclusions can be drawn from this study of African grasslands whose aim was to quantify the impact of grassland degradation on CO_2 emission from soil. The first conclusion was that grassland degradation significantly decreased gross CO_2 emissions (per area basis) from soil by 11% in degraded (25<Cov<50%) and 62% in highly degraded (0<Cov<5%) compared to non-degraded grasslands (i.e. grassland with an aerial cover, Cov of 100%). This correlated with a decrease in soil organic C stocks and grass primary production. The second conclusion was that grassland degradation decreased soil organic matter protection from decomposers as shown by a significant increase in CO_2 emissions relative to soil organic C stocks by 15-41%, depending on degradation intensity. The third conclusion was that grassland degradation increased CO_2 emissions relative to produced biomass by as much as 81% because it lessened grass productivity.

While, the decrease in CO_2 emissions from soil following grassland degradation was expected, several possible reasons may explain the associated lower organic matter protection from decomposers. Loss of soil cover is associated with increased soil C loses by water erosion and lower production of fresh and easily decomposable organic matter (Mchunu and Chaplot, 2012). Other possible explanations, which also require further appraisal, are the changes of surface albedo and soil temperature, modification of organic matter quality and/or microbial biomass, its activity and diversity. There is a need for further multidisciplinary research that not only investigates CO_2 emissions from soils, but also includes the C losses through water erosion and their associated impact on organic matter quality and decomposers. Further research is also needed to find effective grassland rehabilitation technique to improve fodder production while limiting soil C losses, especially those through GHGs.

CHAPTER 5: GRASSLAND REHABILITATION THROUGH A SHIFT IN CATTLE MANAGEMENT DECREASES CO_2 EMISSION BASED ON SOIL AND PLANT CARBON STOCKS

5.1 Abstract

Rehabilitation of degraded grasslands can potentially sequester atmospheric carbon (C) and thus mitigate against climate change. While numerous grassland rehabilitation strategies are available, little is known about their consequences on C outputs from soils to the atmosphere. The main objective of this study was to assess the impact of high-density short duration stocking (HDSD: 1200 cows ha^{-1} for three days per year), an innovative grassland management practice, on CO_2 emissions from soils. The study was conducted in communal grassland of South Africa. HDSD was compared against two commonly used grassland management practices, namely livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) and annual burning, all being compared to traditional free grazing as a control. CO_2 emissions from soil were measured *in-situ* with three replicates per treatment regularly from January 2013 to April 2015. Overall, gross CO_2 emissions (i.e. emissions per unit of surface area) from soil were with 2.07±0.16 g CO_2-C m^{-2} day^{-1} the highest under HDSD and decreased by 45% form the average of annual burning and traditional free grazing. However, CO_2 emissions from soil relative to soil organic C stocks (SOCs) were the lowest under traditional free grazing (1.22±0.16 g CO_2-C g^{-1} C day^{-1}) and HDSD (1.36±0.10 g CO_2-C g^{-1} C day^{-1}). Finally, CO_2 emissions from soil relative to yearly produced biomass were lowest under HDSD (0.13±0.02 g CO_2-C kg^{-1} produced biomass day^{-1}) and were 2.3 and 5.7 times higher under traditional free grazing and annual burning, respectively. Grassland rehabilitation using HDSD may afford greater C protection from decomposers while yielding the highest grass biomass production. Gross CO_2 emissions from soil significantly increased with soil water content and decreased with soil temperature whereas CO_2 emissions from soil relative to SOCs decreased the most with increasing soil bulk density. HDSD, which increased grass production by 74% and SOCs by 34% after only three years of implementation on a degraded grasslands, is expected to have greater potential for C sequestration in the long term.

Keywords: *Grassland, Degradation, Rehabilitation, Grazing, CO_2 emissions, Climate change*

5.2 Introduction

Soils play a crucial role in the storage of atmospheric carbon (C), with 2344 Pg C (1 Pg=1 billion tonnes) being estimated to be stored in the top three meters, which is higher than the amount found in vegetation (560 Pg C) and the atmosphere (800 Pg C) (Schlesinger, 1997; Jobbagy and Jackson, 2000). However, soils have lost two thirds of their C with 300 Pg C being lost from grasslands (Lal, 2004). Since grasslands are a major component of terrestrial ecosystems, representing about 70% of the world's agricultural area (Abberton et al., 2010), they are considered as a potentially significant C sink if suitable management practices are implemented (Lal, 2004; Conant et al., 2001; Dlamini et al., 2014). Indeed, there is urgent need for innovative grassland management practices that can increase fodder production while at the same time enhancing C sequestration in the soils.

Grassland management practices such as intensive grazing, fertilization and burning have been successfully used to increase grass production (Ojima et al., 1994; Conant et al., 2001; Castellano and Valone, 2007) and are thus expected to affect the C exchanges between soils and the atmosphere (Conant and Paustian, 2002; Jones and Donnelly, 2004; Li et al., 2013). For instance, Li et al. (2013) who investigated grazing impact under semi-arid climate in China reported a decrease in soil respiration (microbial and roots respiration) by 26% following grazing. The decrease in CO_2 emissions from soil reached 50% when the grazing intensity was doubled at Tibetan plateau in China (Cao et al., 2004). However, Liebig et al. (2013) under a semiarid continental climate in the USA reported no significant differences in CO_2 emissions between moderate and heavy grazing. In grassland fertilization experiments, Du et al. (2014) reported a 30% increase in CO_2 emissions from soil after application of a compound fertilizer to natural grasslands in Southern China. In support, Peng et al. (2011) observed that nitrogen fertilization levels (0, 50, 100 and 200 kg N ha^{-1} $year^{-1}$) enhanced CO_2 emissions from soil, but did not change the seasonal emissions pattern. However, several studies have shown no significant effect of nitrogen fertilization on CO_2 emissions from soil (e.g. Micks et al., 2004; Gong et al., 2014). However, grass burning is a traditional grassland management practice, but can have detrimental influence on the grass itself, and litter and soil organic matter (O'Neill et al., 2002). Xu and Wan (2008) in an *in-situ* study over two growing seasons found that annual burning significantly increased mean CO_2 emissions from soil by 24%. Jia et al. (2012) reported an increase of 11% at the northern Loess Plateau of China. As pointed by Hogue and Inglett (2012), the combustion of plant material and residues releases soil nutrients, thus providing suitable conditions for microbial growth (Guénon et al., 2013), which in turn increases CO_2 emissions from soil (Du et al., 2014).

In general, grassland management practices affect CO_2 emissions from soil by altering the soil physical, chemical and biological properties (Lal, 2001; Du, et al., 2014). For instance, soil temperature and moisture content have been shown to control CO_2 emissions due to its impact on soil microbial activity (Xu and Wan, 2008; Guntiñas et al., 2013). Soil organic matter content is also an important parameter affected by grassland management practice with subsequent effect on CO_2 emissions from soil. Several studies found positive correlations between soil organic C (SOC) and CO_2 emissions from soil (Xu and Qi, 2001; Brito et al., 2009; Panosso et al., 2009).

While many studies focused on the impact of grassland management such as over grazing, fertilization and burning on soil properties and CO_2 emissions from soil, little is known on the impact of innovative grassland management such as shift in cattle management involving high density stocking rate but for short duration, a technique suggested by Savory and Parsons (1980). Communal rangelands on the uplands of KwaZulu-Natal Province, South Africa provide an important asset to the wellbeing of smallholder farmers. However, the rangelands are degraded because they are exposed to free grazing, which is poorly managed. Overgrazing combined with low soil productivity have led to grassland degradation with a drastic increase in soil erosion (Dlamini et al., 2011; Mchunu Chaplot, 2012) and consequent depletion of SOC stocks (SOCs) by as much as 90% (Dlamini et al., 2014). This study aimed to evaluate the impact of high-density short duration stocking on CO_2 emisssions from soil and its main controls.

5.3 Materials and methods

5.3.1 Study site

The site is located at Potshini catchment (29°21' E; 28°48' S, 1305 m a.s.l.), 10 km south of Bergville in KwaZulu-Natal province, South Africa (Figure 5.1). Potshini has a subtropical humid climate with hot wet summers and cold dry winters, with a mean annual temperature of

13°C (Schulze, 1997) and mean annual precipitation of 745 mm $year^{-1}$, calculated from 1945 to 2010 (Grellier et al., 2012). The site is on a sandy loam soil derived from sandstone and mudstone as a parent material and is classified as Acrisol (WRB, 2006). The soil was characterized by a dark brown (7.5YR 4/4) A horizon, with a weak sub-angular blocky structure. The soil is acidic (pH 3.784–3.94) in the top layer (0-0.05m), with acid saturation ranging from 26 to 78% (Dlamini et al., 2014). The most common vegetation species include *Hyparrhenia hirta* and *Sporobolus africanus* (Camp and Hardy, 1999).

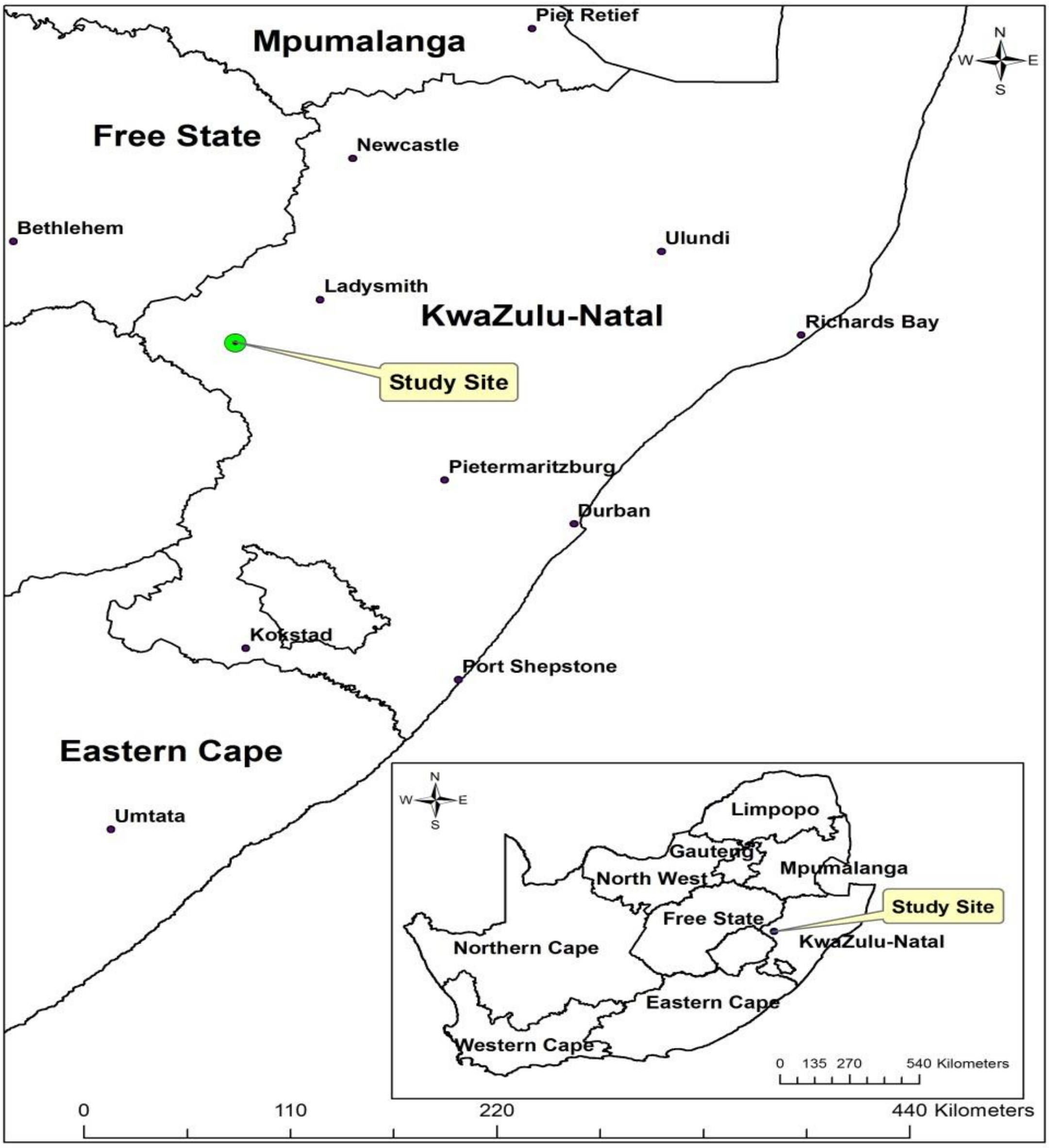

Figure 5.1 Location of the Potshini study site in South Africa

5.3.2 Experimental design and treatments

The experimental site was established in July 2011 with the aim of rehabilitating a degraded communal grassland. The grassland exhibited clear shifts in grass aerial cover from non-degraded, represented by 100% grass aerial cover to heavily degraded, where the grass aerial cover was as low as 0%. A total surface area of 1500 m^2 (30×50 m^2), with homogeneous soils, was demarcated by Dlamini et al. (2011). Two degradation levels, namely non-degraded (100% aerial grass cover) and degraded (25-50%) grassland were considered for the current study. Four treatments consisting of (1) high density cattle stocking rate (1200 cows ha^{-1}) for short duration (3 days) followed by livestock exclusion for 362 days (HDSD); (2) livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}); (3) annual burning and (4) traditional free grazing as a control were applied to each degradation level. HDSD treatment is based on the idea of a holistic management method (Savory and Parsons, 1980; Savory, 1983) involving the management of livestock grazing in such a way that rehabilitation of degraded grassland is achieved. The rational behind this method is that cattle hooves break soil crusts and crumple down grass tufts in the presence of dung and urine, thus increasing nutrients cycling and congruently grass growth. In this study, 38 Nguni cattle from the local community were left overnight for 3 days in each of the seasons 2011-2012, 2012-2013 and 2013-2014 once a year in June. Livestock were excluded from the fertilization treatment, where a compound fertilizer (NPK, 2:3:3, 22 at 0.2 t ha^{-1}) was added every year. The fertilizer amount was determined according to commonly used in the community. The annual burning treatment involved burning the grass once every year in June. Finally, the traditional free grazing is a common practice in the community where livestock are allowed to graze freely throughout the year.

5.3.3 Measurements of CO_2 emissions from soil

CO_2 emissions from soil were measured from January 2013 to April 2015, once a month in winter and twice a month in summer. The measurements were done using a LI-COR 6400XT gas exchange system (LI-COR, Lincoln, NE, U.S.A.) attached with a LI-COR 6400-09 soil respiration chamber which had an internal volume of 991 cm^3 and surface area of 71.6 cm^2.

Six plastic (PVC) collars per treatment (three for degraded and three for non-degraded grassland) were inserted into soil three days before starting CO_2 measurement in all treatments. In case of annual burning and HDSD treatments the collars were removed every year before the burning and grazing and inserted again. The LI-COR 6400-09 chamber was inserted 0.02 m into the PVC collars. All CO_2 measurements were carried out between 1000 and 1300 h to avoid impact of diurnal variations (Castaldi et al., 2012). The CO_2 emissions from soil were expressed in three units: (1) g CO_2-C per unit of surface area to evaluate the gross CO_2-C emissions (g CO_2-C m^{-2} day^{-1}) from soil to the atmosphere; (2) g CO_2-C per gram of C in the (g CO_2-C g^{-1}C day^{-1}) to evaluate the CO_2-C emissions from soil relative to SOCs; and (3) g CO_2-C per kg of biomass produced (g CO_2-C kg^{-1} biomass day^{-1}) to evaluate the CO_2-C emissions from soil relative to produced aboveground biomass.

5.3.4 Soil temperature and water content measurements

Soil temperature was determined by a thermocouple connected to the soil chamber (LI-COR 6400-09) inserted 0.05 m into the ground. It was measured at the same time as CO_2 emissions from soil and for the top-soil layer (0-0.05 m). Soil water content measurements were performed as close to the collars as possible using a Hydrosense soil moisture meter (Campbell Scientific, Inc., USA), simultaneously with soil temperature and CO_2 measurements; but because of methodological issues, the soil water content measurements are only reported from December, 2014 to April 2015. Finally, precipitation and air temperature data for the study period (Figure 5.2) were obtained from a Duncan weather station located about 500 m from the trial.

5.3.5 Soil sampling and analysis

Soil samples from the top-soil (0-0.05 m) layer were collected for evaluation of soil organic C content (SOCc) and nitrogen (N) content (Nc). Three replicate samples were taken as close as possible to the CO_2 measurements collars. The samples were air-dried for 48 hours, ground and sieved through a 2mm sieve. Total C and N were measured using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). The total soil C was considered equivalent to SOCc when no reaction could be obtained on addition of HCl. Additional soil samples were taken from the same points for bulk density determination using metallic cylinders with 7.5 cm diameter and 5 cm height (Grossman and Reinsch, 2002). The soil samples were immediately placed in air-tight plastic bags and later dried in an oven at 105°C for 24 hours. The SOC and N stocks (SOCs and Ns) were calculated following Batjes (1996):

$$SOCs = SOCc \ x \ \rho b \ x \ T \ (1 - \frac{PF}{100})b$$

where SOC_S is the soil organic C stock (kg C m^{-2}); SOC_C is soil organic C content in the ≤2mm soil material (g C kg^{-1} soil); ρ_b is the bulk density of the soil (kg m^{-3}); T is the thickness of the soil layer (m); PF is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

5.3.6 Aboveground biomass

The aboveground biomass was measured in six randomly placed metallic quadrats (0.5× 0.5 m) placed on soil surface in each treatment of the two degradation levels (three in each degradation intensity), once a year in June just before the annual burn and cattle grazing treatments. All shoot material from the soil surface to the crown within the quadrats was clipped. The plant samples were oven-dried at 65°C for 48 h and then weighed to estimate the aboveground biomass (kg m^{-2} $year^{-1}$).

5.3.7 Data analysis

Summary statistics were calculated for CO_2 emissions from soil for the treatments in each grassland degradation level. Since the CO_2 emissions from soil were repeatedly measured at the same locations, a repeated measure analysis of variance (ANOVA) was used to determine differences in CO_2 emissions from soil between the treatments, the grassland degradation levels and the measurement dates as the repeated measure. The treatment means were compared using Tukey's for multiple comparisons, a significant threshold defined as $P < 0.05$, unless otherwise specified. Finally, the final cumulative values of the CO_2 emissions from soil between treatments for each grassland degradation level were also compared. In addition, principal component analysis (PCA) was carried out to evaluate the multiple relationships between CO_2 emissions from soil and the factors of control. All analyses were done using Genstat (version 14, VSN International, UK, 2011).

5.4 Results

5.4.1 Precipitation air and soil temperature

Figure 5.2 shows the seasonal pattern of precipitation, air and soil temperature in non-degraded and degraded grasslands from January 2013 to April 2015. The annual precipitation was 718 and 562 mm for 2013 and 2014, respectively, with most of the precipitation occurring in summer (about 90% between months of November to April each year). Mean annual air temperature was 17°C for both 2013 and 2014. The lowest daily average temperature was recorded (13°C) in June both years and greatest (38°C) in September 2014. Daily average temperature of 33°C for the period July-September 2014 was much higher than 20°C for the same period during the previous year (Figure 5.2A and B).

Soil temperature at the 0-0.05 m depth changed significantly ($P < 0.001$) with change of season during the study, with a lowest winter value of about 9°C in HDSD occurred in winter and a highest of 38°C in annual burn recorded in summer for both non-degraded and degraded grasslands (Figure 5.2C and D). However, soil temperature was consistently higher in annual burning and traditional free grazing compared to livestock exclosure with fertilization and HDSD treatments. The highest and lowest soil temperature values were recorded in June and September 2013, respectively. Overall, mean soil temperature under non-degraded grassland was 24, 23, 21 and 20 °C for the annual burning, traditional free grazing, HDSD and livestock exclosure with fertilization, respectively. Under degraded grasslands, overall mean was highest in annual burning and least in HDSD.

5.4.2 Grassland management impacts on CO_2 emissions from soil

<u>5.4.2.1 Summary statistics of CO_2 outputs</u>

The summary statistics of CO_2 emissions from soil (gross CO_2-C emissions (g CO_2-C m^{-2} day^{-1}), relative to SOCs (g CO_2-C g^{-1}C day^{-1}) and to produced biomass (g CO_2-C kg^{-1}produced biomassday^{-1}) from the treatments under non-degraded and degraded grasslands are given in

Table 5.1. Under non-degraded grassland, the overall mean daily gross CO_2 emissions from soil was 26% higher under HDSD than livestock exclosure with fertilization treatment, and 36% higher than annual burning and traditional free grazing. There were no significant differences between annual burning and traditional free grazing treatment. HDSD under degraded grassland emitted 53% higher CO_2 than annual burning and traditional free grazing. However, average daily CO_2 emissions from soil relative to SOCs under non-degraded grasslands, decreased in the order annual burning, HDSD, livestock exclosure with fertilization and traditional free grazing. CO_2 emissions relative to SOCs were lower in traditional free grazing compared to the other treatments. In degraded grasslands, CO_2 emissions from soil relative to SOCs were higher under livestock exclosure with fertilization (2.60±0.03 mg CO_2-C g^{-1}C day^{-1}) than annual burning, traditional free grazing and HDSD. CO_2-C emissions from soil relative to produced biomass under non-degraded grasslands were 60% higher in traditional free grazing than livestock exclosure with fertilization and HDSD. No significant differences were observed between annual burning, livestock exclosure with fertilization and HDSD. On the other hand, under the degraded grasslands, annual burning and traditional free grazing treatments emitted 82% and 57% higher CO_2 emissions relative to produced biomass, respectively, than HDSD treatment.

The repeated-measures analysis of variance results indicated that grasslands rehabilitation treatment, degradation levels, date of CO_2 sampling and their interactions highly affected ($P < 0.001$) CO_2 emissions (Table 5.2).

5.4.2.2 Temporal variations of CO_2 emissions from soil

The trend of gross CO_2 emissions from soil under non-degraded and degraded grasslands showed that CO_2 changed greatly over time, but were mostly higher in livestock exclosure with fertilization and HDSD than annual burning and traditional free grazing (Figure 5.2E and F). In addition, significant differences between treatments mostly occurred in hot and wet period (summer), while there were no treatment differences in winter gross CO_2 emissions from soil among the treatments. Additionally, gross CO_2 emissions were significantly higher during summer periods than winter. Differences in cumulative gross CO_2 emissions from soil between treatments under non-degraded were not significant in the first year (2013); however for the rest of the study period showed much higher cumulative values for HDSD (Figure 5.2G).

Table 5.1 Summary statistics of daily CO_2 emissions from soil for the rehabilitation treatments (annual burn (AB), traditional free grazing (TFG), livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density grazing (1200 cows ha^{-1}) for short duration (3 days per year) (HDSD) in non-degraded and degraded grasslands. N =120

	CO_2-C fluxes											
	Gross CO_2 (g CO_2-C m^{-2} day^{-1})				CO_2 relative to SOC_S (mg CO_2-C g^{-1}C day^{-1})				CO_2 relative to biomass (g CO_2-C kg^{-1}biomass day^{-1})			
	AB	TFG	LEF	HDSD	AB	TFG	LEF	HDSD	AB	TFG	LEF	HDSD
						Non-degraded						
Mean	1.29c	1.30c	1.51b	2.04a	1.60a	0.96b	1.25a	1.46a	0.12b	0.22a	0.09b	0.09b
SD	0.98	1.05	1.03	1.47	1.43	0.80	0.83	1.00	0.10	0.25	0.06	0.07
SE	0.15	0.17	0.16	0.23	0.23	0.13	0.13	0.16	0.02	0.04	0.01	0.01
Min	0.10	0.17	0.21	0.17	0.17	0.14	0.21	0.15	0.01	0.02	0.01	0.01
Median	1.06	1.05	1.26	2.04	1.24	0.70	1.14	1.36	0.10	0.10	0.09	0.09
Max.	4.18	5.41	5.66	6.69	3.40	3.32	4.06	4.17	0.33	0.69	0.28	0.26
						Degraded						
Mean	0.97c	1.02c	1.50b	2.10a	1.56b	1.48b	2.60a	1.26c	0.74a	0.30b	0.11c	0.13c
SD	0.57	0.70	0.98	1.35	1.09	1.28	1.88	0.81	0.85	0.24	0.08	0.10
SE	0.09	0.11	0.15	0.21	0.17	0.20	0.30	0.13	0.13	0.04	0.01	0.02
Min	0.10	0.14	0.23	0.22	0.22	0.17	0.47	0.15	0.02	0.04	0.02	0.02
Median	0.83	0.94	1.29	1.77	1.41	1.08	1.81	1.21	0.19	0.23	0.08	0.09
Max.	2.04	2.86	4.24	4.84	3.46	3.83	5.98	2.48	2.65	0.72	0.25	0.40
						Average						
Mean	1.13c	1.16c	1.51b	2.07a	1.58b	1.22c	1.93a	1.36c	0.43a	0.26b	0.10c	0.11c
SE	0.09	0.10	0.11	0.16	0.14	0.16	0.18	0.10	0.08	0.03	0.01	0.01

*Means in each units followed by the same letter are not significant at P<

Table 5.2 Repeated-measures ANOVA for the effects of grassland rehabilitation treatments (Treatments) degradation intensity (Degradation), time of CO_2 sampling (Date) and their interaction on CO_2 emissions

		CO_2-C					
		Gross CO_2		CO_2 relative o SOCs		CO_2 relative to biomass	
Source of variation	DF	MS	P	Ms	P	Ms	P
Treatments	3	45.33	<.001	0.053	<.001	5.70	<.001
Degradation	1	4.55	<.001	0.193	<.001	8.12	<.001
Date	39	18.74	<.001	0.029	<.001	0.84	<.001
Treatments*date	117	1.36	<.001	0.002	<.001	0.25	<.001
Treatments*degradation	3	2.15	<.001	0.035	<.001	4.97	<.001
Degradation*date	39	1.46	<.001	0.004	<.001	0.25	<.001
Treatments*degradation*date	117	0.83	<.001	0.002	<.001	0.26	<.001

The final cumulative gross CO_2 emissions from soil under non-degraded grasslands were significantly highest in HDSD (2.45±0.04 kg CO_2-C m^{-2}), followed by livestock exclosure with fertilization, traditional free grazing and annual burning. Under degraded grasslands, the cumulative gross CO_2 emissions from soil were significantly higher in HDSD and livestock exclosure with fertilization than traditional free grazing and annual burning (Figure 5.2H).

There was a significant difference between livestock exclosure with fertilization and HDSD in most cases, cumulative gross CO_2 emissions from soil was not significantly different between traditional free grazing and annual burning for the entire study period. At the end of the study duration, the final cumulative gross CO_2 from soil was significantly higher under high density grazing for short duration (by 36%) and livestock exclosure with fertilization (by 14%) than the average of annual burning and traditional free grazing.

CO_2 emissions relative to SOCs from treatments under non-degraded and degraded also changed over time with generally higher emissions during summer months (October to April) and lower in winter (Figure 5.3A and B). In addition, significant differences between treatments under non-degraded were only observed in four events during 2013 (Figure 5.3A) with highest emissions occurred under annual burning compared to other treatments. CO_2 emissions from soil relative to SOCs under degraded grasslands tended to be higher under livestock exclosure with fertilization in most cases (Figure 5.3B). Cumulative CO_2 emissions from soil relative to SOCs from the treatments under non-degraded grasslands exhibited much higher CO_2 emissions from soil relative to SOCs in livestock exclosure with fertilization than annual burning, traditional free grazing and HDSD (Figure 5.3C). However, under degraded grasslands, significant differences in CO_2-C emissions relative to SOCs between treatments were only observed in four events during 2013 (Fig. 3A), with greatest emissions occurring under annual burning. The annual cumulative CO_2-C emissions relative to SOCs exhibited much greater values in annual burning and HDSD than traditional free grazing and HDSD (Figure 5.3D). The final cumulative CO_2-C emissions from soil relative to SOCs was greater in annual burning (1.92±0.20 g CO_2-C g^{-1} C) than HDSD (1.75±0.19 g CO_2-C g^{-1} C), livestock exclosure with fertilization (1.5±0.09 g CO_2-C g^{-1} C) and traditional free grazing (1.10±0.09 g CO_2-C g^{-1} C).

CO_2 emissions from soil relative to produced biomass under non-degraded and degraded appeared to differ amongst the treatments mostly in the first year (2013) of CO_2 measurements (Figure 5.4A and B). For instance, traditional free grazing exhibited significantly higher CO_2 emissions from soil relative to produced biomass than the other treatments under non-degraded grasslands, while CO_2 emissions from soil relative to produced biomass under annual burning were much higher than under traditional free grazing, HDSD and livestock exclosure with fertilization. The cumulative CO_2 emissions from soil relative to produced biomass from non-degraded grasslands were highest under traditional free grazing followed by annual burning and least in livestock exclosure with fertilization and HDSD (Figure 5.4C). However, degraded grasslands showed highest emissions under annual burning (29.45±0.59 g CO_2-C kg^{-1} produced biomass), which corresponded to significant difference (at $p < 0.001$ level) in comparison to HDSD (5.10±0.31 g CO_2-C kg^{-1} produced biomass) and livestock exclosure with fertilization (4.31±0.034 g CO_2-C kg^{-1} produced biomass) (Figure 5.4D).

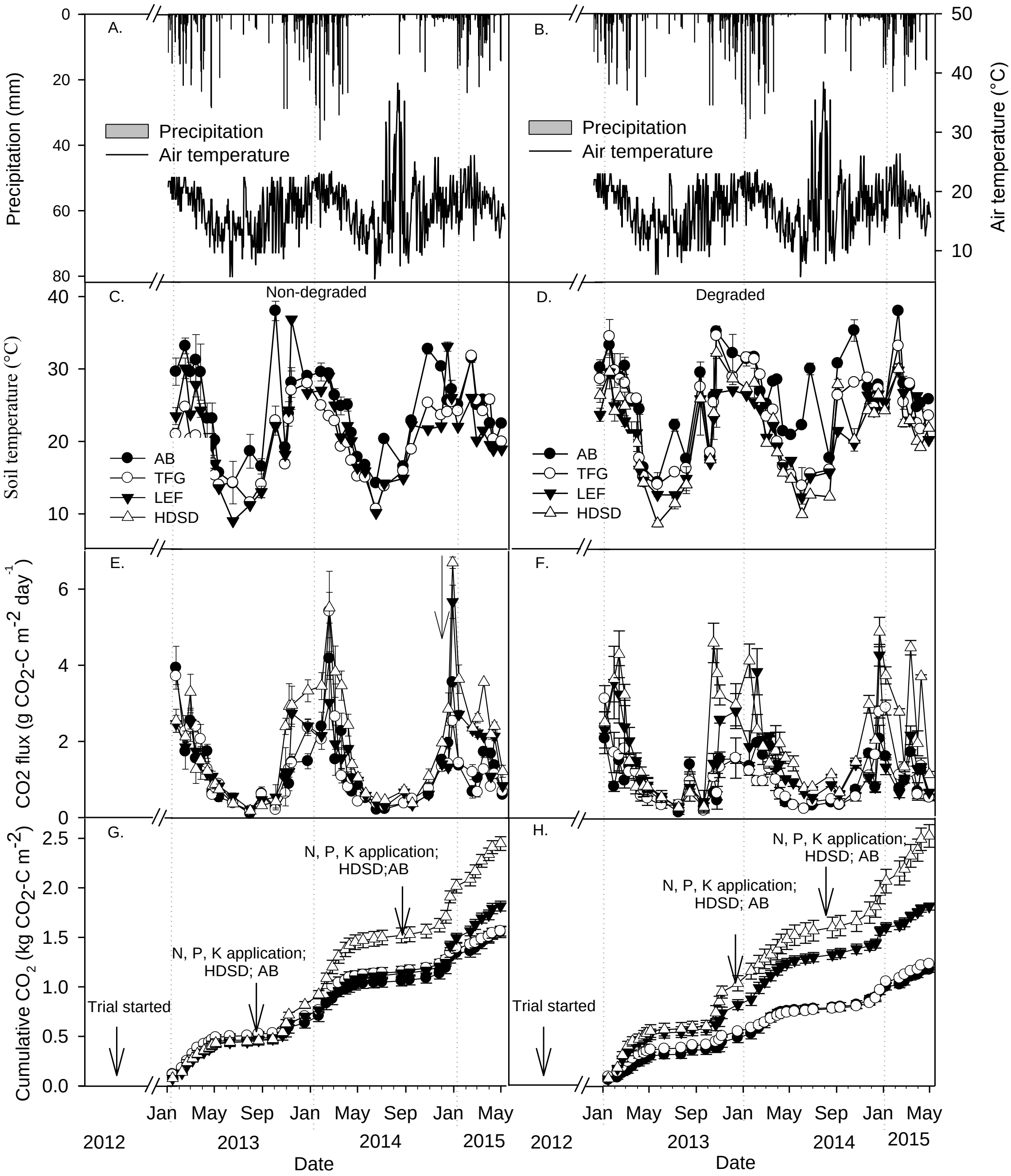

Figure 5.2 Precipitation, air temperature and soil temperature (mean ± SE) at the top-soil (0-0.05 m) layer and daily and cumulative gross CO_2 (g CO_2-C m^{-2} day^{-1}) emissions from soil for annual burn (AB), traditional free grazing (TFG), livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density stocking (1200 cows ha^{-1}) for short duration (3 days per year) (HDSD), under non-degraded and degraded grassland. Error bars represent standard error of the mean. N=3.

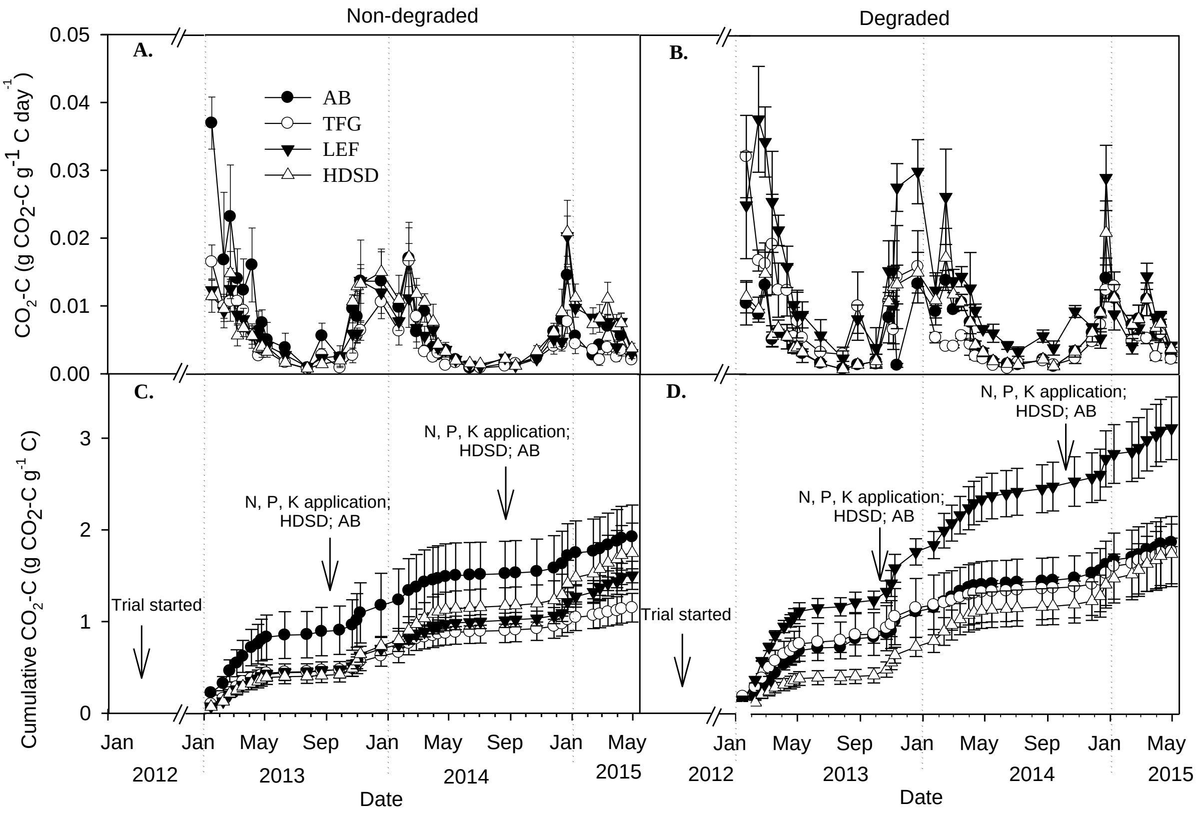

Figure 5.3 Daily and cumulative of CO_2 emissions from soil relative to soil carbon stocks (g CO_2-C g^{-1} C) from the rehabilitation treatments (annual burn (AB), traditional free grazing (TFG), livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density stocking (1200 cows ha^{-1}) for short duration (3 days per year) (HDSD); for non-degraded (A and B) and degraded (C and D) grasslands. Error bars represent standard error of the mean. N=3

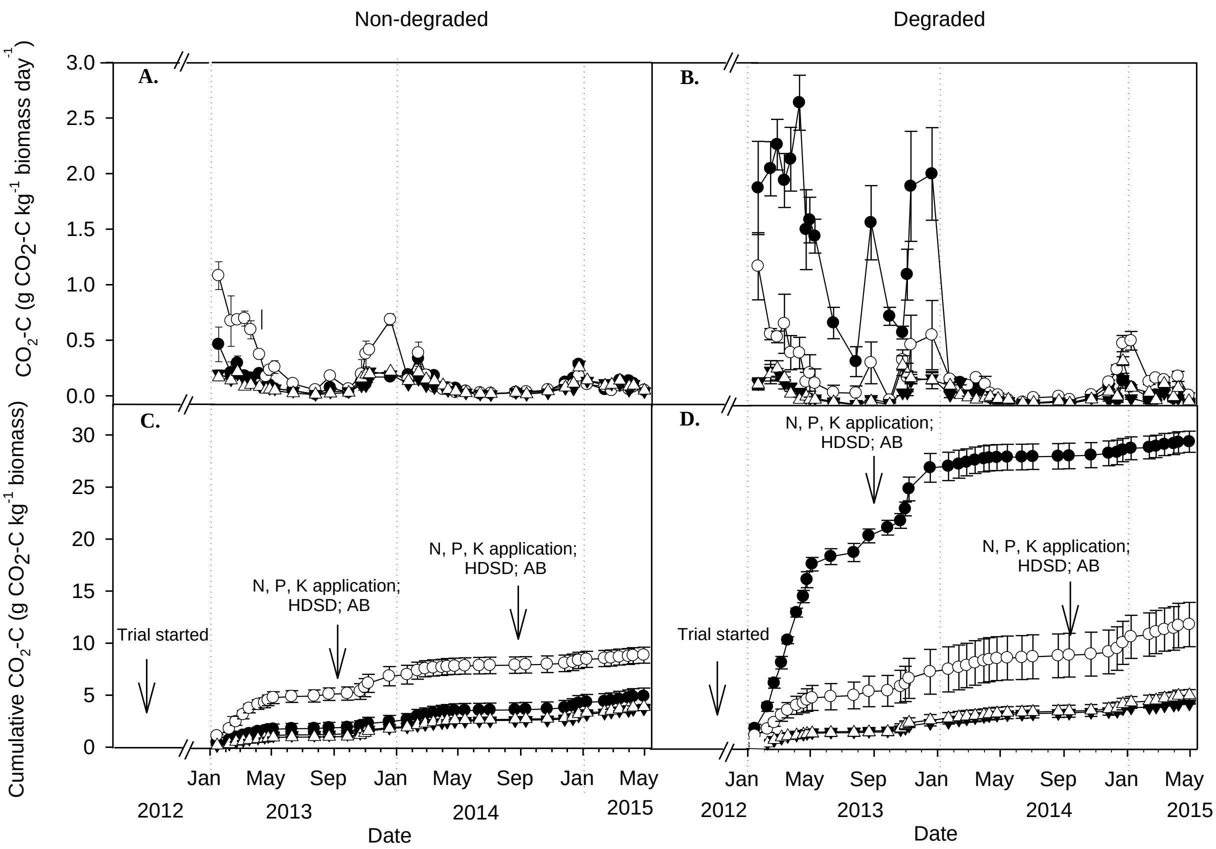

Figure 5.4 Daily and cumulative of CO_2 emissions from soil relative to produced biomass (g CO_2-C kg^{-1} biomass) from the rehabilitation treatments (annual burn (AB), traditional free grazing (TFG), livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density stocking (1200 cows ha^{-1}) for short duration (3 days per year) (HDSD); for non-degraded (A and B) and degraded (C and D) grasslands. Error bars represent standard error of the mean. N=3

5.4.3 Changes in CO_2 emissions from soil under degraded grassland

The percent change in CO_2 emissions from soil under traditional free grazing (as a reference) to annual burning, livestock exclosure with fertilization and HDSD treatments in the first month of the study (i.e. after one year of trial implementation) and last three months of the study (i.e. after three years of trial implementation) are shown in Figure 5.5. One year after trial implementation, HDSD treatment had increased gross CO_2 emissions from soil by an average of 30% and livestock exclosure with fertilization by 23%, while annual burning decreased gross CO_2 emissions from soil by 55% compared to traditional free grazing (Figure 5.5A and B). After three years of experimentation, the difference in gross CO_2 emissions from soil under the control increased to 69% for HDSD, which corresponded to 120% increase.

In the meantime, gross CO_2 emissions from soil decreased by 43% (from 23 to 13%) in case of livestock exclosure with fertilization. However, annual burning emitted less gross CO_2 than traditional free grazing in both occasions (by 55% and 9%, respectively). Livestock exclosure with fertilization and HDSD emitted 60 and 95% higher CO_2 emissions from soil relative to SOCs, than traditional free grazing after one year while annual burning was 40% lower (Figure 5.5C). After three years, the differences with traditional free grazing had changed to 10, 35 and 69% for annual burning, livestock exclosure with fertilization and HDSD, respectively (Figure 5.5D). The differences in CO_2 emissions from soil relative to produced biomass between the treatments and traditional free grazing were all negative in after the first year, which implied higher CO_2-C emissions from soil relative to biomass production by traditional free grazing compared to the other treatments (Figure 5.5E). However, after three years annual burning treatment emitted (average 85%) higher CO_2 relative produced biomass than traditional free grazing, while livestock exclosure with fertilization (by 242%) and HDSD (by 311%) were still lower (Figure 5.5F). The difference between HDSD and traditional free grazing increased from -103 to -311%, meaning that it produced even lower CO_2 emissions from soil relative to produced biomass in three years after start of the experiment.

5.4.4 Changes in biomass production and SOCs for the treatments under degraded grasslands

HDSD had the highest amount and most significant impact on aboveground biomass three years after implementation (Table 5.3). After three (2014), the percent change in aboveground biomass was highest in HDSD (by 74 %) and lowest in traditional free grazing (12%). The changes from 2012 to 2014 in annual burning and traditional free grazing treatments were not significant, while the aboveground biomass changed greatly in livestock exclosure with fertilization and HDSD treatments. The adoption of HDSD and livestock exclosure with fertilization from 2012 to 2014 resulted in significant increase of SOCs, but annual burning and traditional free grazing treatments resulted in decreases, which were not significant. The highest increase in SOCs was recorded under livestock exclosure with fertilization (37%).

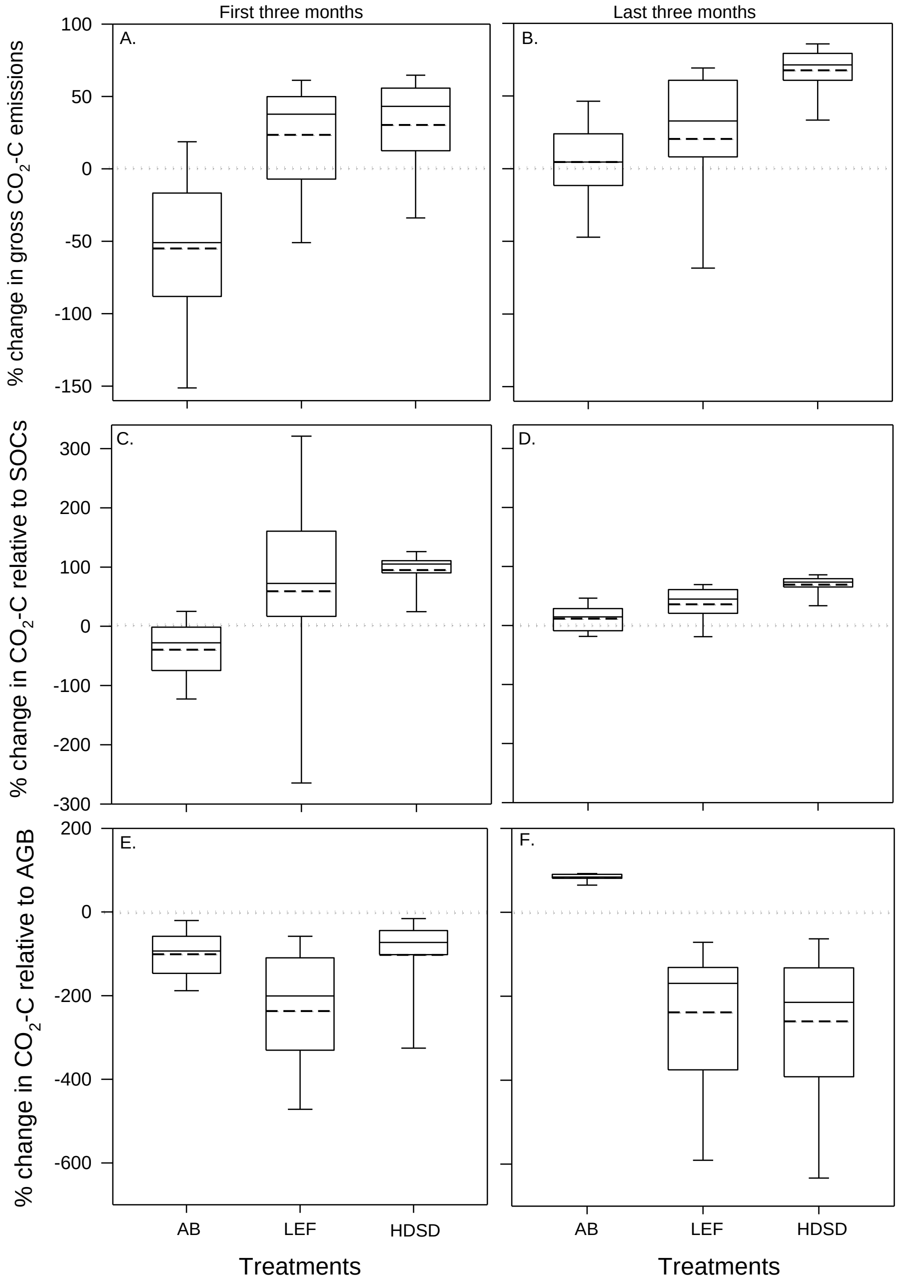

Figure 5.5 Percent change in CO_2 emissions from soil for the first three months (January, February and March, 2012) and last three months (February, March and April 2014) of the experiment from traditional free grazing (TFG) as a reference to annual burn (AB) , livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density stocking (1200 cows ha^{-1}) for short duration (3 days per year) f (HDSD); under degraded grasslands.

Table 5.3 Changes in aboveground biomass and soil organic carbon stocks (SOCs) in the topsoil (0-0.05m) for the treatments (annual burn (AB), traditional free grazing (TFG), livestock exclosure with NPK fertilization (2:3:3, 22 at 0.2 t ha^{-1}) (LEF), high density stocking (1200 cows ha^{-1}) for short duration (3 days per years) (HDSD)) in degraded grasslands from 2012 to 2014. N=3

Treatments	Aboveground biomass (kg m^{-2})					SOCs (kg m^{-2})				
	2012		2014		% change	2012		2014		% change
	mean	SE	mean	SE		mean	SE	mean	SE	
AB	0.03	0.02	0.04	0.03	13.90	0.44	0.03	0.43	0.02	-1.96
TFG	0.15	0.03	0.17	0.01	11.53	0.50	0.04	0.50	0.03	-0.55
LEF	1.29	0.31	4.31	0.66	70.06	0.46	0.03	0.74	0.04	37.18
HDSD	1.23	0.17	4.71	0.44	73.90	0.50	0.04	0.75	0.10	33.50

5.4.5 Relationship between CO_2 emissions from soil and factors of control

Gross CO_2 emissions from soil under non-degraded grasslands appeared to significantly increase the most with soil water content (r = 0.53) but decreased with increasing soil temperature (r = -0.64) (Table 5.4). CO_2 emissions from soil relative to SOC_S were also shown to increase significantly the most with soil water content (r = 0.73) and decreased significantly with increasing of soil bulk density (r = -0.68). While CO_2 emissions from soil relative to produced biomass increased significantly with Nc (r = 0.63), Ns and soil temperature (r = 0.54 each); it decreased significantly with increasing soil water content (r = -0.88). However, under degraded grasslands gross CO_2 emissions from soil increased significantly with C: N ratio (r = 0.74) and soil water content (r =0.55) and decreased the most with soil temperature (r = -0.84), followed by Nc, SOCc and SOCs. In addition, CO_2 emissions from soil relative to SOCs under degraded grassland decreased the most with increasing soil bulk density (r = -0.84), while CO_2 emissions from soil relative to produced biomass increased significantly with soil temperature (r = 0.96) only (Table 5.4).

Table 5.4 Coefficients of determination (r) between CO_2 emissions from soil under non-degraded (ND), degraded (D) grassland and soil factors: soil organic carbon content and Stocks (SOCc and SOCs), nitrogen content and stocks (Nc and Ns), carbon: nitrogen ratio (C:N), soil bulk density (ρb), Clay content, soil water content (SWC) and soil temperature (ST)

	SOCc	SOCs	Nc	Ns	C/N	ρb	SWC	ST
	Non-degraded (ND)							
g CO_2-C m^{-2}	0.28	0.28	0.1	0.19	0.12	0.05	0.53*	-0.64*
g CO_2-C g^{-1} C	-0.34	-0.6	-0.27	-0.48	-0.12	-0.68*	0.73*	0.06
g CO_2-C kg^{-1} biomass	0.31	0.32	0.63*	0.54*	-0.41	0.02	-0.88*	0.54*
	Degraded (D)							
g CO_2-C m^{-2}	-0.63*	-0.63*	-0.76*	-0.7	0.74*	-0.32	0.55*	-0.84*
g CO_2-C g^{-1} C	-0.42	-0.57*	-0.29	-0.45	-0.02	-0.84*	0.04	-0.4
g CO_2-C kg^{-1} biomass	0.27	0.33	0.42	0.35	-0.47	0.39	-0.08	0.96*
	Average							
g CO_2-C m^{-2}	0.11	0.06	0.02	0.01	0.38	-0.22	0.57*	-0.78*
g CO_2-C g^{-1} C	-0.55*	-0.65	-0.51*	-0.56*	0.12	-0.74*	0.11	-0.10
g CO_2-C kg^{-1} biomass	-0.31	-0.23	-0.23	-0.24	-0.16	0.47*	-0.21	0.81*

*Statistically significant determinants at $P < 0.05$

The relationships between CO_2 emissions from soil and the selected soil factors were also investigated further using PCAs (Figure 5.6). The first two axes (axis 1 and 2) of the PCA between the CO_2 emissions and the soil factors under non-degraded grassland shown in Figure 5.6A accounted for 89% of the total variation of the CO_2 emissions from soil. The first PCA axis (axis 1), which described 64% of the variance was negatively correlated the most soil water content. The second PCA axis (axis 2), which described 25% of the variation, was negatively correlated with soil temperature. CO_2 emissions from soil relative to produced biomass were strongly correlated to axis 1 in the positive direction.

The other PCA in Figure 5.6B, produced using CO_2 emissions from soil for the treatments under degraded grasslands shows that axis 1 and axis 2 explained 92% of the data variation. In this PCA, Ns, SOCc, Nc, SOCs and soil temperature correlated negatively to gross CO_2 emissions from soil and positively to CO_2 emissions relative to produced biomass.

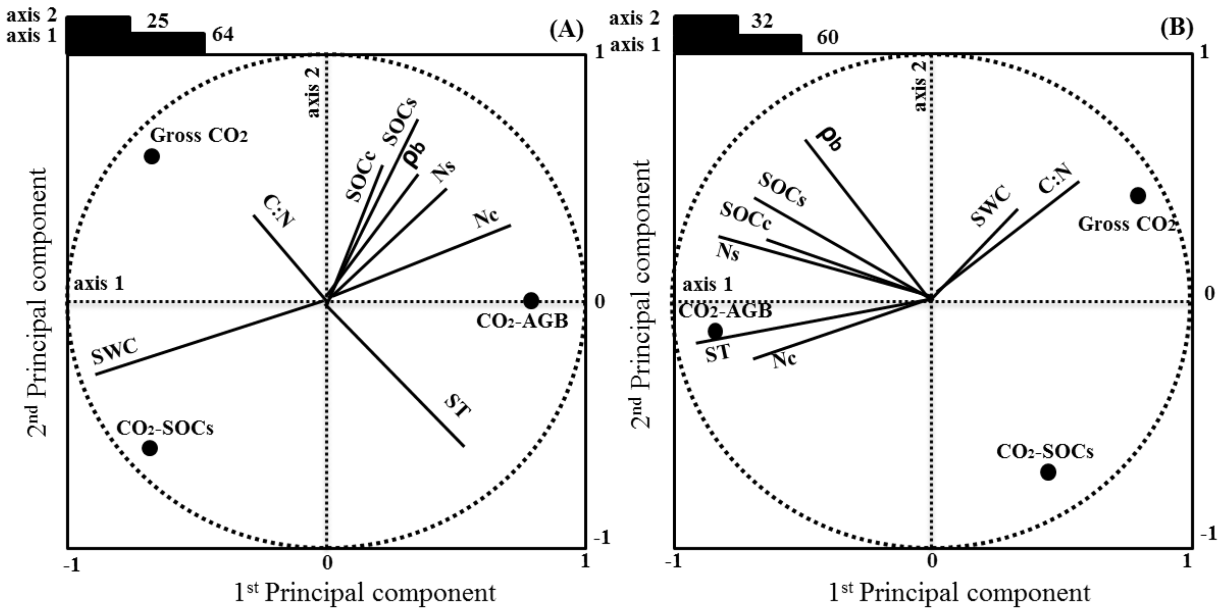

Figure 5.6 Principal components analysis (PCA) scatter diagrams for gross CO_2 emissions from soil, CO_2 emissions from soil relative to SOCs (CO_2-SOCs) and relative to produced biomass (CO_2-biomass) as supplementary variables and selected soil properties as active variables. Scatter diagram with the two first PCA axes (axis 1 and 2) in (A) non-degraded and (B) degraded grassland. N=24.

5.5 Discussion

5.5.1 High density short duration stocking rate impacts on CO_2-C emissions from soil

5.5.1.1 Gross CO_2 emissions from soil

This study pointed to significantly higher gross CO_2 emissions from soil in high density grazing for short durtion and livestock exclosure with fertilization treatments compared to traditonal free grazing and annual burning (Table 5.1 and Figure 5.2). The high density for short durtion stocking (HDSD) increased gross CO_2 emissions from soil by as much as 51% compared to traditonal free grazing under degraded grasslands, suggesting significant stimulation of CO_2 emissions from soil when grasslands management changes from traditonal free grazing to high density grazing for short durtion. The higher CO_2 emissions from soil in HDSD and fertilization treatments could be explained by the higher biomass production. In the present study, implementation of HDSD and livestock exclosure with fertilization for three years led to an increase of aboveground biomass by an average of about 72% (Table

5.3). In support, Bahn et al. (2008) found that annual CO_2 emissions from soil were closely related to total biomass production ($R^2 = 0.88$). Frank and Dugas (2001) found that maximum CO_2 emissions from soil coincided with the period of maximum aboveground biomass. The increase in aboveground biomass under HDSD and livestock exclosure with fertilization grasslands could be attributed to inputs of nutrients (N, P, K) from organic (cattle dung) and inorganic fertilizer. However, Acharya et al. (2012) found that application of slurry increased above and belowground biomass but with no significant effect on CO_2 emissions from grasslands soil in Denmark. The increase in aboveground biomass is usually associated with increase in root biomass (Yang et al., 2010), resulting in greater CO_2 emissions from the soils.

5.5.1.2 CO_2 emissions from soil relative to soil carbon stocks

CO_2 emissions from soil relative to SOCs (per g of C in the soil) were higher in HDSD and annual burning than to traditional free grazing under non-degraded grasslands (Table 5.1 and Figure 5.3A and C), suggesting greater C mineralization in HDSD and annual burning treatments. The high C mineralization under HDSD treatment could be attributed to the impact of cattle hooves on the soils. Cattle hooves could till the surface soil and breaking soil aggregate, thus exposing soil organic matter and placing the C in direct contact with decomposer microorganisms, which speeds mineralization (Menke, 1992). Another possible explanation could be related to the significantly higher microbial biomass and activities in the grazed grasslands compared to non-grazed (Devi et al., 2014). CO_2 emissions from soil rate, due to microbial respiration, depends on organic matter content, O_2 supply, temperature, soil water content, and available nutrient (Rowell, 1994). In the case of annual burning, the greater mineralization could also be attributed to the fact that burning lowers soil aggregate stability and consequently protection of C from decomposers. In addition, the release of nutrients as a results of grass burning have been shown to enhance microbial growth and activity (Guénon et al., 2013; Du et al., 2014), thus increase C mineralization.

In support of this, several studies have shown that SOC stability is strongly related to stability of soil aggregates (e.g. Six et al., 2002; Liu, et al., 2014). Contrary to non-degraded grasslands, HDSD reduced CO_2 emissions from soil relative to SOCs under degraded grassland compared to the other treatments (Table 5.1 and Figure 5.3D), suggesting greater SOC stabilization when HDSD is applied to a degraded grassland. This result might be explained by the fact that, cattle do not stay for a long time in less vegetated grasslands, which was confirmed by field observations. Therefore, it suggests that soil disturbance by hooves on surface soils in degraded grasslands would be lower than under non-degraded grasslands. The addition of cattle dung with little soil disturbance for a long time can enhance soil aggregation via soil fauna activities, which protects SOC from decomposer. Consequently, HDSD treatment increased SOCs by 33.5% after three year of implementation (Table 5.3), suggesting great potential to increase SOCs in degraded grasslands under long-term HDSD adoptions.

Fertilization (NPK) increased the CO_2 emissions relative to SOCs by an average of 42% compared to the average of annual burning and traditional free grazing treatments (Table 5.1 and Figure 5.4D), which implies greater stimulation of C mineralization after the addition of chemical fertilization to a degraded grasslands. Mineral nitrogen input has a 'priming effect' on microbial decomposition of soil organic matter (Fontaine et al, 2003; Chen et al., 2014), and this might explain the higher CO_2 emissions from soil under fertilization treatment in comparison to other treatments under degraded grasslands. Without a doubt, addition of chemical fertilizer (N.P.K) can increase microbial activity due to the greater availability of nutrients released, which enhance organic matter decomposition and associated CO_2 emissions from soil (Chen et al., 2014).

5.5.1.3 CO_2 emissions from soil relative to produced biomass

CO_2 emissions from soil relative to produced biomass (g CO_2-C per kg produced biomass) under non-degraded and degraded grasslands were significantly higher in annual burning and traditional free grazing than livestock exclosure with fertilization and HDSD treatments (Table 5.3 and Figure 5.5). This could be explained by the fact that biomass production under annual burning was very low, and hence CO_2 emissions from soil relative to produced biomass would be higher compared to other systems (e.g. livestock exclosure with fertilization and HDSD) where biomass production is very high. The same reason may also explain the higher CO_2 emissions from soil relative to produced biomass in traditional free grazing treatment compared to livestock exclosure with fertilization and HDSD. However, CO_2 emissions from soil relative to produced biomass in traditional free grazing was lower than in annual burning despite a lack of significant difference in their produced biomass (Table 5.3). Therefore, the difference in CO_2 could be explained by higher soil temperature under annual burning than traditional free grazing (Figure 5.2). Moreover, CO_2 emissions from soil relative to produced biomass correlated most significantly with soil temperature (Table 5.4).

5.5.2 Factors controlling CO_2 emissions from soil

Gross CO_2 emissions from soil under both non-degraded and degraded grasslands correlated positively with soil water content and negatively with the soil temperature (Table 5.4), which was consistent with several studies (e.g. Qi et al. (2002), Suseela et al. (2012) and Wang et al. (2014)). In addition, significantly negative correlations were found with N and SOC (content and stocks) under degraded grasslands only, implying that CO_2-C emissions from soil was controlled more by the presence of soil organic matter than root respiration. In general, studies have reported root respiration contribution to the total CO_2 emissions from soils ranging from 25% to 91 % in grasslands ecosystems (Dugas et al., 1999; Wanga et al., 2005: Wang et al., 2007). However, CO_2 emissions from soil relative to SOC_S increased with decreasing soil bulk density in both non-degraded and degraded grasslands, indicating higher SOC stability under high bulk density soils. This was likely to come from decreasing microbial activity, porosity and gaseous exchanges between the soil and the atmosphere. In support, several studies have reported negative correlations between CO_2 emissions from soil and soil bulk density (e.g. Saiz et al. (2006) and Moyano et al. (2012)).

5.6 Conclusions

The main objective of this study performed on a grassland site in South Africa was to investigate the impact of grassland rehabilitation strategies on CO_2 emissions from soil. High density for short duration cattle stocking, livestock exclosure with fertilization and annual burning were compared against the traditional free grazing treatment. Three main conclusions can be drawn from the results of the study.

The first conclusion is that soils submitted to high density for a short duration stocking rate and livestock exclosure with fertilization emitted higher amounts of gross CO_2-C from soil than under annual burning and traditional free grazing treatments, with CO_2-C emissions from soil increasing positively with soil water content and negatively with soil temperature. Interestingly, after three years high density grazing for a short duration increased aboveground biomass production and SOCs by 70% and 37%, respectively. The second conclusion is that CO_2 emissions from soil relative to SOCs (per gram of C in the soil) under non-degraded grasslands were significantly higher in annual burning and high density grazing for a short duration than livestock exclosure with fertilization and traditional free grazing treatment, while under degraded grasslands the lowest emissions were recorded under high density grazing for a short duration. This was correlated negatively the most with the soil bulk density. The third conclusion is that, overall CO_2 emissions from soil relative to aboveground biomass production (per kg produced biomass per year) were 74 and 58% higher in traditional free grazing treatment and annual burning, respectively, than high density for short duration stocking treatment. These results suggest high density for short duration stocking treatment to have a potential for grassland rehabilitation with greater C sequestration and grass production however, a longer term study is recommended.

CHAPTER 6: LONG-TERM ANNUAL BURNING OF GRASSLAND INCREASES CO_2 EMISSIONS FROM SOILS

6.1 Abstract

Grasslands have potential to mitigate against climate change because of their large capacity to store soil organic C (SOC). However, the long-term impact of grassland management such as burning, which is still common in many areas of the world, on SOC is still a matter of debate. The objective of this study was to quantify the long-term effects of annual burning on CO_2 output from soils and SOC stocks (SOCs). The study was performed on a 62 years old field trial comparing annual burning to no burning associated with tree encroachment, and to annual mowing with all treatments laid out in randomized block design with three replicates per treatment. CO_2 emissions from soils were continuously measured over two years and were correlated to soil chemical and physical properties. Annual burning and annual mowing produced 30 and 34% higher gross CO_2 emissions from soils (1.80±0.13 *vs.* 2.34±0.18 and 2.41±0.17 g CO_2-C m^{-2} day^{-1}) than no burning, respectively. Annual burning and annual mowing also produced about 26% higher CO_2 emissions from soil relative to SOCs than no burning (1.32 ± 0.1 *vs.* 1.05 ± 0.07 mg CO_2-C g^{-1} C day^{-1}), suggesting lower SOC stability in annual burning and mowing grasslands. Overall, gross CO_2 emissions from soil correlated mostly to soil water content (r = 0.72) followed by SOCs (r = 0.59), SOC content (r = 0.50), soil bulk density (r = 49), soil temperature (r = 0.47), C:N ratio (r = 0.46) and mean weight diameter (r = 0.38). These findings suggest that long-term annual burning increases CO_2-C output from soils. Additional greenhouse gases emissions from burning itself were finally discussed.

Keywords; *Grassland management; Burning; Soil organic carbon; CO_2 emissions; Carbon cycle*

6.2 Introduction

Grasslands cover approximately 40% of the earth's terrestrial surface area and play an important role in the global carbon (C) cycle by storing about 10% of the global soil C stocks (Suttie et al., 2005). Additionally, in grasslands soil organic C (SOC) concentration is higher at the soil surface, which may turn the C to the external factors such as climate and land management. Fire is the most common anthropogenic grassland management practice, used since the early Holocene (Behling and Pillar, 2007), because of easy application in difficult terrains and on large areas. However, other practices like mowing, grazing and fertilization are also in use (Blüthgen et al., 2012; Peng et al., 2011). All these grassland management practices have potential to influence soil C stocks and eventually CO_2 emissions from the soil to the atmosphere (Peng et al., 2011; Granged et al., 2011; Jia et al., 2012).

Burning is a common practice used for increasing fodder production and quality, whilst avoiding bush encroachments (Tainton, 1999). It results in increased biomass growth period and biomass production (Ojima et al., 1994), while at the same time improving grass cover and biodiversity (Boakye et al., 2013). This grassland management practice can have negative consequences on soil chemical, physical and biological properties (Andersson et al., 2004; Granged et al., 2011). Burning causes a general decline of SOC through combustion of soil organic matter (SOM) in the upper soil layer (Granged et al., 2011). For example, Granged et al. (2011) reported a 35% reduction in SOM after three years of burning with significant changes in soil physical properties, leading to increased water repellence. In addition, the high soil temperature and low soil moisture content, during fires, causes a sharp decrease in topsoil (0-0.05 m) biological biomass and its activity (Nardoto and Bustamante, 2003; D'Ascoli et al., 2005). Moreover, because of greater root and tuff density at the soil surface frequently-burned grasslands have the tendency to show detritus accumulation on the topsoil compared to non-burned shrubby grasslands and trees (Ansley et al., 2002). This, together with an enhanced mineralization of SOM (Singh et al., 1991), highly affects CO_2 effluxes from soil.

Several studies have reported lower CO_2 emissions from soils under no burn (NB) grasslands than burned (e.g. Knapp et al., 1998; Rutigliano et al., 2007; Ward et al., 2007; Xu and Wan, 2008). For instance, Knapp et al. (1998) reported that annual burn (AB) for 17 years in eastern

Kansas region, USA, resulted in 55% higher monthly CO_2 emission from the soil than in NB treatment. Similarly, Xu and Wan (2008) reported 23.8% more CO_2 emission from the soil in AB than NB on sandy soils of semiarid Northern China over two growing seasons, whereas Jia, et al. (2012) reported 11% lower emissions with NB compared to AB for one growing season in the same region. Castaldi et al. (2012) also reported less CO_2 emissions from unburned compared to burned plots in central Africa. Bushes often encroach into grasslands where neither burning nor mowing are applied (Trollope, 1980; Tainton, 1999; Montané, et al. (2007). Montané et al. (2007) reported a slight increase of soil C stocks in the upper soil layers (top 15 cm depth) following shrubs encroachment into grasslands. Wang, et al. (2013) also found an increase of soil C storage by shifting from grassland to woody plants.

Mowing is also regarded an improved grassland management practice (Zhou et al., 2007; Hamilton, et al., 2008), which reduce CO_2 emissions from soils by 20–50% compared to burning (Wan and Luo, 2003). The reasons for such a decrease are unclear. Wan and Luo (2003) explained the reduction of CO_2 in terms of the sensitivity of roots and microorganisms to a decrease in photosynthetic C supply from aboveground biomass. However, others authors (e.g. Bahn et al. 2006) suggested that it could be a result of the depletion of easily available C substrates for the microflora. Yet others (e.g. Zhou et al. 2007; Hamilton et al. 2008) stated that mowing results, precisely, in an increase of rhizodeposition, soil microbial biomass and labile C, which might suggest increased CO_2 emission from the soil.

While numerus studies exist on the impact of grassland burning on CO_2 emissions from soil and SOCs, the existence of discrepancies between these limits decision making on grassland management. These studies show inherent limitations, related to their short duration, which long-term experiments might allow to overcome. In this study, 62-year annual burn and mow were compared against no burn treatment in an African Savanna. The no burn treatment was characterized by encroachment of large trees. The main objective of this study was to evaluate the impact of annual grassland burn management on soil organic matter dynamics (C-stocks and CO_2 emissions from soils) and their factors of control.

6.3 Material and methods

6.3.1 Study area

The experiment was conducted at Ukulinga Farm, the training and research farm of the University of KwaZulu-Natal, Pietermaritzburg, South Africa (24° 24'E, 30° 24'S) (Figure 6.1). The experimental site is located on top of a small sloping plateau ranging in altitude from 847 to 838 m (Fynn et al., 2004). Soil depths vary from 0.05 m in the upslope to 0.20 m in midslope and 0.6 m at the footslope, and were classified as Plinthic Acrisols (WRB, 2006). The parent material is colluvium shale with intrusions of dolerite. The soil is acidic with a pH (KCl) of 5.5 at the top-soil and its texture is silty clay loam (37% clay, 43% silt and 20% sand).

The climate is sub-tropical humid and characterised by warm and wet summers (October - April), and cool and dry winters (May-September). Long-term (30 years) mean annual temperature and precipitation at the farm were 16°C and 694 mm, respectively.

The native vegetation of the study area is dominated by the southern tall grassveld, which produces dense vegetation with plant heights ranging between 0.5 and 0.75 m (Fynn et al., 2004). Depending on the grassland management, some scattered trees, for instance *Acacia sieberiana* and some grass species such as *Themeda triandra* and *Tristachya leucothrix* are also found (Fynn et al., 2004). The native grass species (e.g. *Themeda triandra* and *Tristachya leucothrix)* all use the C4 photosynthetic path (Fynn et al., 2005).

6.3.2 Experimental design

The experiment involved 3 treatments; namely no burn, annual burning and annual mowing. There has been neither burning nor mowing in the no burn plots since 1950, and these plots are now encroached by densely spread trees of *Acacia sieberiana* species. Long-term annual burning involves the burning of grass in the 1st week of August every year since 1950. At the time of study, the annual burning plots were dominated by sparse *Themeda triandra* grass. In the annual mowing plots, the grass is cut at the same time as burning and the material is removed from the treatment plots. All treatments are replicated three times by slope position (up, mid and footslope) in completed randomized block design and the plots sizes are 18.3 x 13.7 m spaced by 4 m sidewalks. The three treatments were represented once in each slope position. There was no grazing at the experimental site since it was established in 1950. More details about the experimental site and design can be found in Tainton et al. (1978).

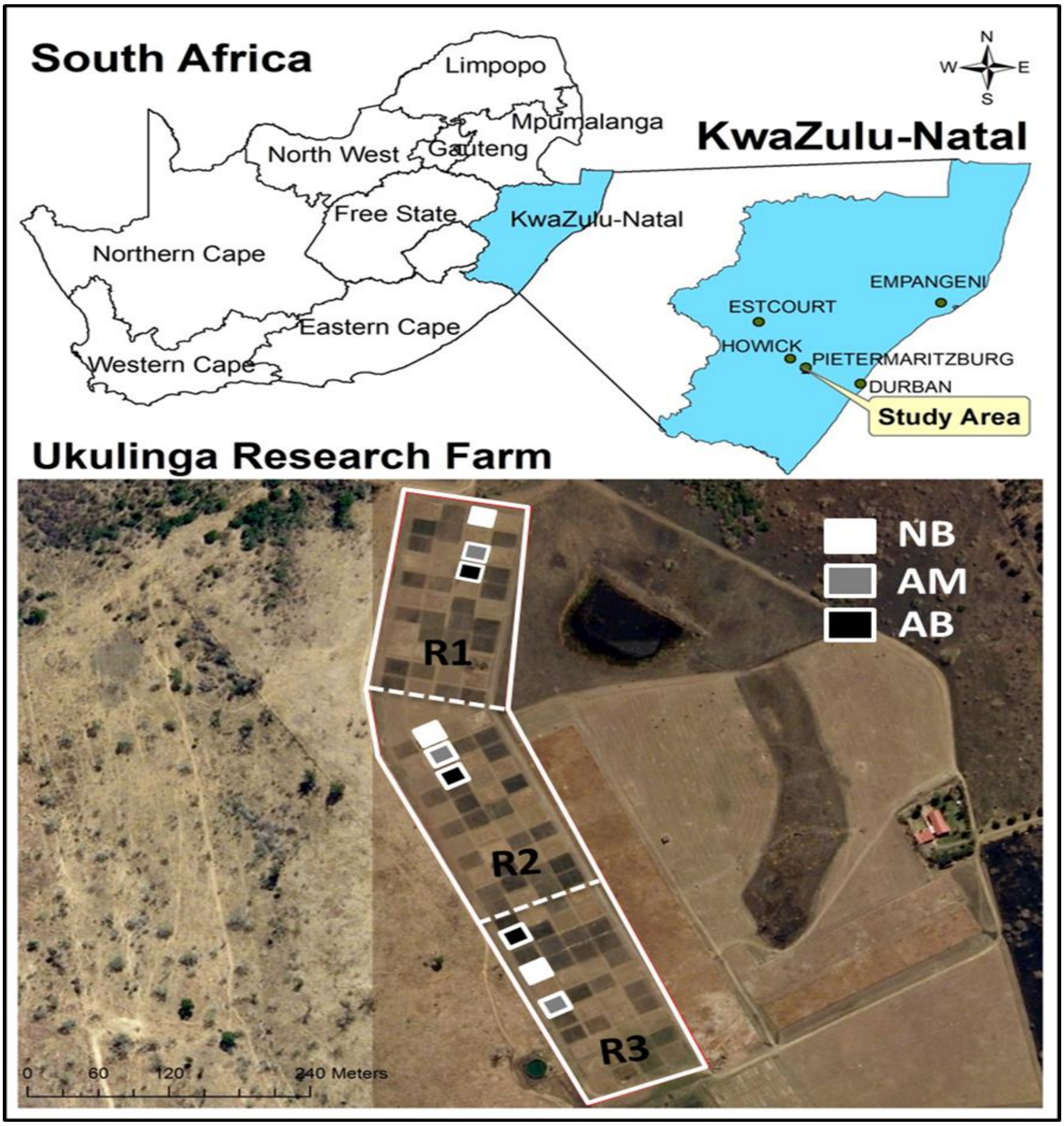

Figure 6.1 Location of the study site in South Africa. Satellite imagery was taken in 2009 during the winter season. The selected treatments (NB; no- burning, AB; annual burning and AM; annual mowing) and their replicate (R) in each plot (R1: upslope, R2: midslope and R3: downslope position)

6.3.3 Soil sampling and analysis

Soil samples for evaluation of SOC content (SOCc) and nitrogen content (Nc) were collected once (at the beginning of the second year) in each plot at three randomly selected pits (0-0.05 m deep). The samples were air-dried for 48 hours, then gently ground and sieved through a 2 mm sieve. Total C and N were measured in the soil samples using LECO CNS-2000 Dumas dry matter combustion analyzer (LECO Corp., St. Joseph, MI). The total soil C was considered equivalent to SOCc when no reaction could be obtained on addition of HCl. On the same day, additional soil samples for bulk density (ρb) determination were also collected from the middle pit of each plot within the 0-0.2 m soil layer using 7.5 cm diameter metallic cylinder core with the height of 5 cm. The ρb for each sample was determined using the core method where the ratio of water content corrected mass to volume was computed (Grossman and Reinsch, 2002). Soil organic C and nitrogen stocks (SOCs and Ns) were calculated using the following equation (Batjes, 1996):

$$SOCs = SOCc \ x \ \rho b \ x \ T\left(1 - \frac{PF}{100}\right) b \qquad (1)$$

where SOC_S is SOC stock (kg C m^{-2}); SOC_C is soil organic carbon content in the ≤2 mm soil material (g C kg^{-1} soil); ρb is the bulk density of the soil (kg m^{-3}); T is the thickness of the soil layer (m); PF is the proportion of fragments of >2mm in percent; and b is a constant equal to 0.001.

Water stable soil aggregates were separated using wet sieving methods described by Elliott (1986). Field moist soil samples were sieved through an 8 mm sieve and air-dried. A subsample of 80 g was placed on a 2 mm sieve and submerged in water for 5 mins followed by wet sieving for 2 mins. The wet sieving process involved moving the sieve up and down 50 times. The materials remaining on the 2 mm sieve were collected by backwashing the sieve into a pre-weighted drying pan. Eventually, four aggregate size classes were collected from each treatment (2, 0.25-2, 0.053-0.25, and > 0.053 mm), by repeating the wet sieving procedure using 0.25 mm, and 0.053 mm sized sieves. The mean weight diameter (MWD) for the water stable aggregate for each treatment was calculated using the following equation (Kemper and Rosenau, 1986):

$$\text{MWD} = \sum_{i=1}^{n} Xi\, Wi \quad (2)$$

Where Xi is the mean diameter for each fraction size, Wi is the proportional weight the fraction from the total dry weight of soil used, and n is the number of aggregate classes separated.

6.3.4 CO_2 emissions measurements

CO_2 emissions from soil were measured once a month from March 2013 to March 2015 with ten measurements per plot using LI-COR 6400 gas exchange system (LI-COR, Lincoln, NE, U.S.A) fitted with the LI-COR 6400-09 soil respiration chamber. The closed chamber system has an internal volume of 991 cm^3 and soil area of 71.6 cm^2 (Healy et al., 1996). We evaluated the sample size necessary to estimate CO_2 emissions points within the plot with a standard error of ±10% of the mean as in equation 3 below:

$$n = (CV/10)^2 \quad (3)$$

Where n is the sample size and CV is the average coefficient of variation within the plot.

All the measurements were carried out during daylight hours, starting at 10:00 and finishing around 13:00 hr. This time period was determined by a pre-experiment, which compared CO_2 emissions from soil during the day and found that emissions between 10:00 and 13.00 closely represented the average daily CO_2 emissions from soil. CO_2 fluxes from soil were expressed in two units: (1) in g CO_2-C in unit of surface per day (g CO_2-C m^{-2} day^{-1}) to evaluate the CO_2 emissions from soil to the atmosphere; and (2) in g CO_2-C per gram of soil C per day (mg CO_2-C g^{-1} C day^{-1}) as a mean to evaluate the CO_2-C emissions from soil relative to SOCs.

6.3.5 Soil temperature and water content

Soil temperature and soil water content were determined in conjunction with CO_2 emissions at the 10 data points per plot. Soil temperature was evaluated by a sensor connected to the soil chamber (LI-COR 6400-09) by inserting the thermocouple close to the measurement points of

CO_2 emissions at a depth of 0-0.05 m. Soil water content was measured for one season at the closest point to the CO_2 chamber using a Hydrosense soil moisture meter (Campbell Scientific, Inc., USA), calibrated by measurement of the meter responses at saturated soil.

6.3.6 Statistical analysis

Summary statistics were done for CO_2 emissions from soil under no burning, annual burning and annual mowing during the whole study period, and summer and winter periods were separated (Tables 2). A coefficient of variation was carried out using all the data together. The CO_2 emissions from soil and soil properties data were analysed as a complete randomized block design using the GENSTAT 14th Edition software. Since CO_2 emissions from soil were measured 24 times during the study period repeated analysis of variance was performed. Treatment means were compared using Tukey corrections for multiple comparisons, with significant differences defined at $P < 0.05$, unless specified otherwise. Finally, the study period and seasonal cumulative CO_2 emissions from soil for the treatment were compared.

6.4. Results

6.4.1 Impact of treatments on soil properties

The mean ± SE of the soil bulk density, SOCc, Nc, SOCs, Ns, C:N ratio and mean weight diameter in the top-soil (0-0.05 m) for no burning, annual burning and annual mowing are reported in Table 6.1. Annual mowing had the highest soil bulk density, while no burning had the least. Annual burning soils were 6.5% denser than no burn and 14% denser than annual mowing. SOCc was 19% higher in annual burning than no burning, while annual mowing was intermediate with average value of 27.2 g kg^{-1}. SOCs show the same trend with 13% higher SOCs in annual burning than no burning. Nc was highest in annual burning, followed by annual mowing and least in no burning. Finally, soil aggregates stability, as measured by mean weight diameter, was highest in no burning and lowest in annual mowing. All these differences were not significant probably due to low number of soil samples used.

Table 6.1 Selected properties of top-soils (0-0.05 m) under grassland and subjected to no burning, annual burning and annual mowing. The values are means ± standard error (SE). N=9

Treatments	ρ_b (g cm^{-3})	SOCc (g kg^{-1})	Nc (g kg^{-1})	SOCs (kg m^{-2})	Ns (kg m^{-2})	C:N ratio	MWD (mm)
No burning	1.0±0.2	25.7±2.2	1.9±0.1	1.4±0.2	0.1±0.0	13.7±1.1	2.5±0.3
Annual burning	1.1±0.0	30.5±0.9	2.1±0.13	1.6±0.1	0.1±0.0	14.4±0.8	2.3±0.3
Annual mowing	1.2±0.0	27.2±1.1	1.92±0.1	1.1±0.5	0.1±0.0	14.2±0.5	1.8±0.8

Soil bulk density (ρb), soil organic carbon content and Stocks (SOCc and SOCs), nitrogen content and stocks (Nc and Ns), carbon: nitrogen ratio (C:N) and Mean weight dimeter (MWD).

6.4.2 Precipitation, air and soil temperature during the study period

The precipitation, average monthly air temperature and soil temperature for the study period March 2013 to March 2015 are presented in Figure 6.2. The total annual precipitations were 631 mm in 2013 and 480 mm in 2014, with about 90% of the precipitation occurring in summer. The highest mean air temperature of 23°C was recorded in January and December 2014 and lowest (8°C) in July 2013. Average soil temperature in annual burning of 23.4°C was significantly higher than annual mowing (21.4°C) and no burning (20.7°C). Overall, the average summer soil temperature of 24 °C was significantly higher than that of winter which was 19 °C. While all treatments had similar soil temperature in winters, the summer period soil temperature was significantly higher in annual burning with average of 25.5 °C compared to no burn with 22.5 °C, which was not significantly different from that of annual mowing (23.3 °C).

6.4.3 Seasonal variation in CO_2 emissions from soil

Table 6.2 shows summary statistics of gross CO_2 emissions from soil for all seasons (summer and winter) and summer and winter, independently. The lowest mean gross CO_2 emissions from soil was 0.87 g CO_2-C m^{-2} day^{-1} in no burning during winter and the maximum was 3.34 g CO_2-C m^{-2} day^{-1} in annual mowing during summer. The average daily gross CO_2 emissions from soil for the study period were significantly lower in no burning than annual burning and annual mowing, by 30% and 34%, respectively. The average CO_2 from soil relative to SOCs was 26% and 29% lower in no burning than annual burning and annual mowing, respectively. In both cases there was no significant difference between annual burning and annual mowing. The highest differences in gross CO_2 emissions from soil and CO_2 relative to SOCs occurred during summer period, while there were no significant differences among the treatments in winter.

The results of CO_2 emissions from soil either gross (g CO_2-C m^{-2} day^{-1}) or relative to SOCs (mg CO_2-C g^{-1} C day^{-1}) are shown in Figure 6.2 C and D, respectively. The patterns of CO_2 fluxes from soil were similar in all treatments (no burning, annual burning and annual mowing) during the study period with highest fluxes observed in summer from October to April, and lowest in winter from May to September. Regardless of treatment, on average, 65% of CO_2 emissions from soil occurred during summer, which coincided with higher precipitation (90% of total annual rainfall) and highest air temperature (average 19°C) (Figure 6.2). During summer periods, CO_2 fluxes from soil were generally lower under no burning than annual burning and annual mowing, with higher differences at 12 out of 14 sampling events. However, there were no significant differences between annual burning and annual mowing during this period. During the winter periods, CO_2 emissions from soil for all treatments were statistically similar. In addition, annual burning did not show significant effect on CO_2 emissions from soil during the winter period (Figure 6.3).

Table 6.2 Summary statistics of gross CO_2 emissions (g CO_2-C m^{-2} day^{-1}) and CO_2 emissions relative to soil organic carbon stocks (mg CO_2-C g^{-1} C day^{-1}) from soil under no burning (NB) annual burning, (AB) and annual mowing (AM) grasslands during the whole study period (n=24), summer (n=14) and winter (n=10).

	CO_2-C						
	CO_2-C (g CO_2-C m^{-2} day^{-1})				CO_2-C relative to SOCs (mg CO_2-C g^{-1} C day^{-1})		
	NB	AB	AM	All seasons	NB	AB	AM
Mean	1.80^{b}	2.34^{a}	2.41^{a}		1.05^{b}	1.32^{a}	1.35^{a}
SD	1.08	1.57	1.48		0.63	0.89	0.83
Min	0.25	0.08	0.09		0.15	0.05	0.05
Median	1.84	2.11	2.41		1.07	1.19	1.31
Max	4.15	5.27	5.22		2.40	3.04	2.73
SE	0.13	0.19	0.17		0.07	0.11	0.10
CV	0.60	0.67	0.61	Summer	0.60	0.68	0.61
Mean	2.47^{b}	3.21^{a}	3.34^{a}		1.44^{b}	1.81^{a}	1.87^{a}
SD	0.81	1.37	1.06		0.48	0.78	0.59
Min	0.88	0.84	1.54		0.52	0.46	0.89
Median	2.51	3.23	3.33		1.45	1.85	1.89
Max	4.15	5.27	5.22		2.40	3.04	2.73
SE	0.13	0.21	0.17		0.07	0.12	0.09
CV	0.33	0.43	0.32	Winter	0.33	0.43	0.32
Mean	0.87^{a}	1.12^{a}	1.12^{a}		0.51^{a}	0.63^{a}	0.63^{a}
SD	3.16	3.16	3.16		3.16	3.16	3.16
Min	0.25	0.08	0.09		0.15	0.05	0.05
Median	0.62	0.65	0.64		0.36	0.37	0.37
Max	2.19	2.76	3.06		1.27	1.61	1.60
SE	0.59	0.59	0.59		0.59	0.59	0.59
CV	3.62	2.83	2.82		6.18	5.02	5.03

Means followed by different superscript letters ($^{a-b}$) in the same row are significantly different ($P < 0.05$)

6.4.4 Controls of SOCs and CO_2 emissions from soil

For all data sets, gross CO_2 emissions from soil increased significantly with the increase in soil water content, SOCs, SOCc, soil bulk density, soil temperature and C: N ratio, but decreased with the increasing of Ns (Table 6.3). CO_2 emissions relative to SOCs have shown the same trend, increasing with soil water content followed by SOCc and SOCs, C:N, soil bulk density and mean weight dimeter but decreased with increasing Ns (Table 6.3

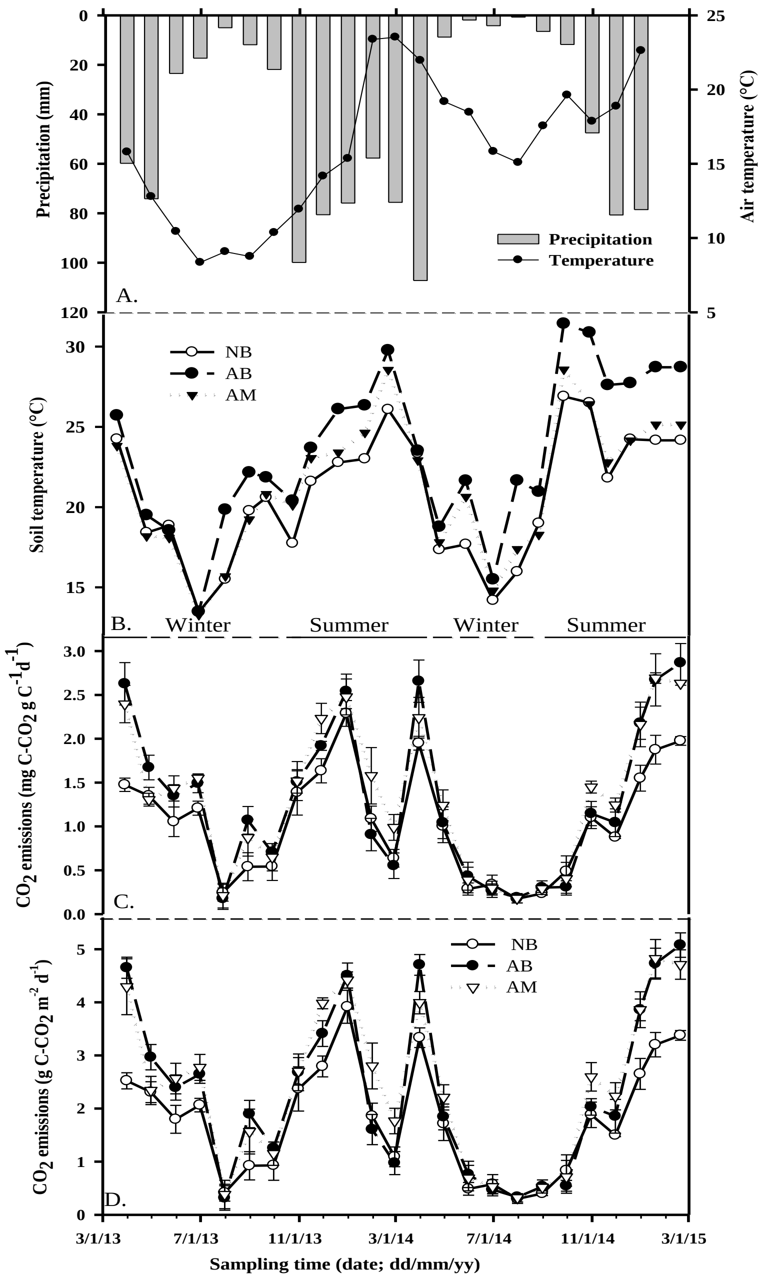

Figure 6.2 Monthly rainfall and average monthly air temperature (A), soil temperature (n=30) at 0-0.05 m depth (B), CO_2 emissions relative to soil organic carbon stocks (g CO_2-C g^{-1}C day^1) (C) gross CO_2 emissions (g CO_2-C m^{-2} day^{-1}) (D), form no burning (NB), annual burning (AB) and annual mowing (AM) grasslands. Error bars represent standard error of the mean. N=30.

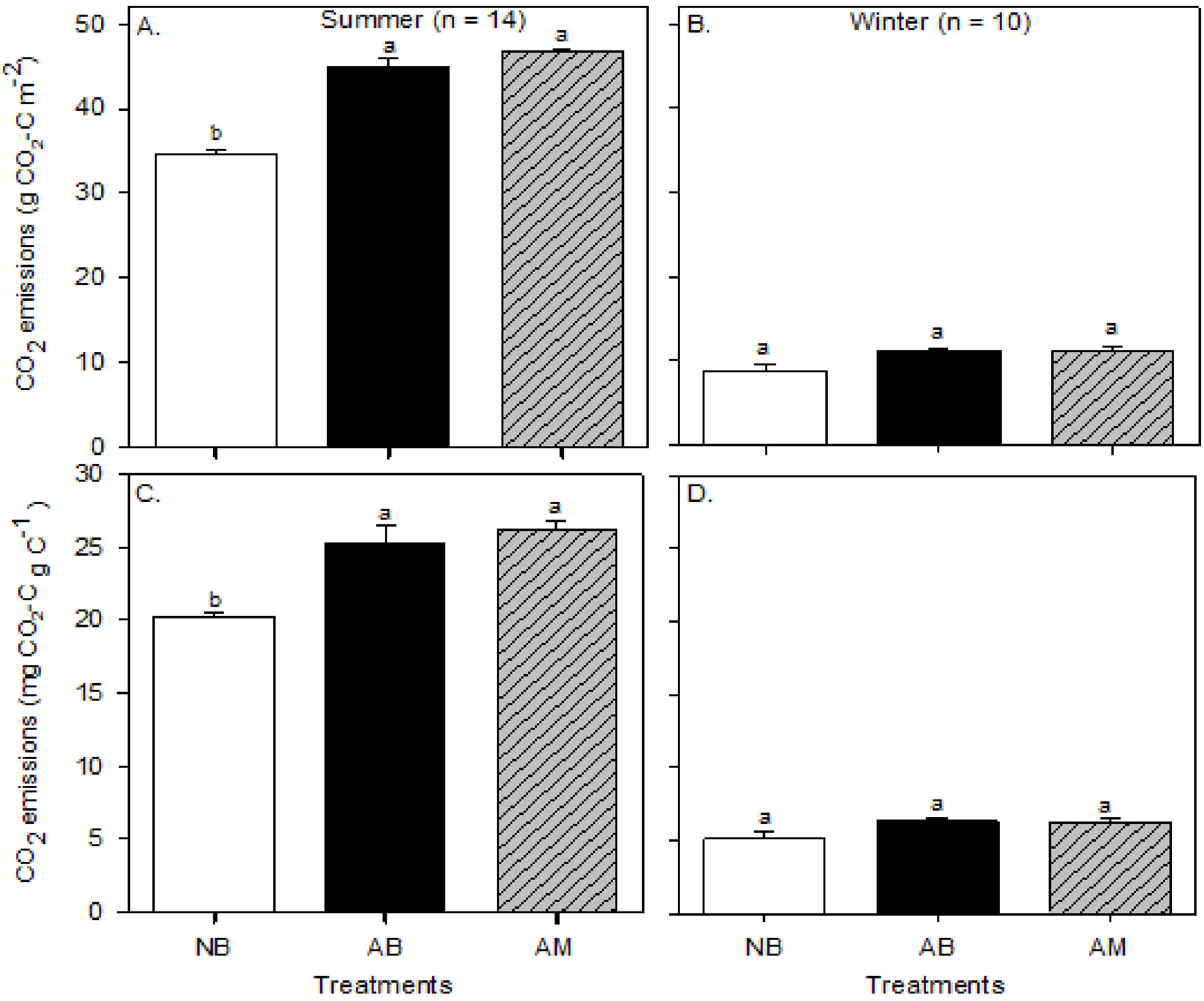

Figure 6.3 Cumulative means (±SE) of daily gross CO_2 emissions (g CO_2-C m^{-2}) from no burning (NB), annual burning (AB) and annual mowing (AM) grasslands, for (A) summer and (B) winter and daily CO_2 emissions relative to soil organic carbon stocks (mg CO_2-C g^{-1} C) for (C) summer and (D) winter. Within the same season and CO_2 emissions unit, different lower case letter indicates significant differences between the treatments. Error bars represent standard error of the mean. N=9.

Table 6.3 Coefficients of determination (r) between gross CO_2 emissions (g CO_2-C m^{-2}) and CO_2 relative to soil organic carbon stocks (g CO_2-C g^{-1} C) from soil and multiple soil factors: soil organic carbon and nitrogen content (SOCc; Nc), SOC and nitrogen stocks (SOCs; Ns), carbon to nitrogen ratio (C:N), soil bulk density (ρb); Mean weight diameter (MWD); soil temperature (ST); and soil water content (SWC)

	SOCc	Nc	SOCs	Ns	C:N	ρb	MWD	ST	SWC
g CO_2-C m^{-2}	0.50*	0.16	0.59*	-0.15	0.46*	0.49*	0.38*	0.47*	0.67*
g CO_2-C g^{-1}C	0.55*	0.15	0.55*	-0.23	0.54*	0.50*	0.47*	0.34*	0.72*

*Statistically significant determinants at $P < 0.05$

6.5 Discussion

6.5.1 Long-term burning and mowing impacts on CO_2 emissions from soil

The overall 30% higher CO_2 emissions from soil in annual burning than no burning grasslands (Table 6.2) implies significant stimulation of CO_2 emissions from soil when grassland management changes from no burn to annual burn systems. This finding is consistent with several studies worldwide which reported higher CO_2 emissions in "burn" compared to "no burn" treatment (e.g. Knapp et al., 1998; Rutigliano et al., 2007; Ward et al., 2007; Jia, et al., 2012). The higher CO_2 efflux from soil has been attributed to an increase in nitrogen availability from burning which enhance microbial respiration. Another explanation could be a change in organic matter quality consecutive to burning. Assuming that there is a greater proportion of charcoal with low aggregation potential, the charcoal lowers the soil aggregate stability and consequently the protection of C from the decomposers. There was, therefore, likely lower C protection under long-term annual burning at the study site which resulted in no burning having 26% lower CO_2 from soil relative to SOCs than in annual burning (Table 6.2). In support of this, some literature showed that the SOC stability is strongly related to the stabilization of soil aggregates (e.g. Singh et al., 2009, Carrizo, et al., 2015). However, there are studies who observed reductions (e.g. Ma et al., 2004) or no changes of CO_2 from soil in response to fire (Castaldi et al., 2012), for example, laboratory experiments by Guerrero et al. (2001) reported an increase in aggregate stability after fire despite a decrease in soil organic matter.

Similar to annual burning, annual mowing also stimulated CO_2 emissions from soil than no burning plots during the study period which agreed with the results by Antonsen and Olsson (2005) and Li and Sun (2011). Mowing increases soil temperature due to the removal of vegetation and exposure of the top-soil to direct sunlight, which enhances microbial activity and plant-roots growth. Another explanation for stimulation of CO_2 emissions from soil by mowing was given by Antonsen and Olsson (2005) who indicated the stimulation of arbuscular mycorrhizal fungi growth after mowing However, contrary to our results, some studies (e.g. Bremer et al., 1998; Han et al., 2012; Jia, et al., 2012), found mowing to decrease CO_2 from soil emissions in comparison to no burning grasslands and they attributed it to the reduction in canopy photosynthesis.

In the present study, lower CO_2 emissions from the soil under no-burning with tree encroachment than annual burning and annual mowing was explained by lower soil temperature under no-burning. Tree coverage can significantly influence CO_2 emissions from soils due do its effect on soil microclimate (Smith and Johnson, 2004; McCulley et al., 2007). For example, Smith and Johnson (2004) reported that soil respiration rate in woodlands was 38% less compared to grasslands, which they explained by the change in the soil microclimate (moisture and temperature). Carbone et al. (2008) also reported 86% lower CO_2 emissions from soil under NB encroached by shrubs than pure grass over five months (910 vs 126 g C m^{-2}), which was attributed to differences in soil water availability and below ground C allocation and plant productivity. However, Smith and Johnson (2004) found no difference in fine root biomass between grasslands and woodlands.

6.5.2 Seasonal change in CO_2 emissions from soil

In the present study, significant differences in gross and CO_2 emissions from soil relative to SOCs between treatments were only observed in summer (rainy) seasons (Table 6.2 and Figure 6.2), coinciding with high precipitation, air and soil temperature, which was consistent with what Chen et al. (2002) found in relatively similar conditions in the tropical savannah of northern Australia. They found that 70% of CO_2 was emitted from soil during rainy seasons, where 95% of the precipitation occurred during this period. The higher CO_2-C emissions during the summer seasons were attributed to higher temperature and precipitation. In this study, CO_2 emissions from soil were positively correlated to soil temperature (r = 0.47 and 0.34 for gross and CO_2 emissions from soil relative to SOCs, respectively) and soil water content (r = 0.67 and 0.72 for gross and CO_2 emissions from soil relative to SOCs, respectively). It is well known that soil temperature and water content are the most essential microclimatic factors controlling CO_2 emissions from soils. Thus, the absence of significant differences in CO_2 emissions from soil among the treatments over the dry period can be explained by lower soil moisture in the root zone, which reduces fine root growth and soil microbial activity (Chen et al., 2002).

6.5.3 Relevance of grassland burning for carbon emissions in Africa

Since burning will also produce CO_2 and other greenhouse gases (GHG) such as CO, NO_x and non-methane hydrocarbons (Jain et al., 2006) and the fact that burned area represents up to 80% of the total grassland area in some region (Csiszar et al., 2005), the implications on GHG to the atmosphere through annual burning at global scale are huge. Annual burning and respiration have already been reported to induce significant C losses into the atmosphere (Van der Werf et al., 2006), it suggests more than 95% of the 58 Pg C year^{-1} fixed by plant through net primary production would eventually be emitted into the atmosphere. Van der werf et al. (2006) estimated the highest emissions as a result of biomass burning to come from Africa (49%), South America (13%), equatorial Asia (11%), boreal regions (9%), and Australia (6%).

6.6 Conclusions

In this study performed on long-term (62-years) grassland management trial in South Africa CO_2 emissions from soil under no burned grasslands were compared to annual burning and mowing treatments. Annual burning and mowing resulted in 30% and 34%, respectively, higher gross CO_2 emissions from soil than no-burning. These differences could be explained by lower stability of soil organic carbon in annual burning treatment. Since in this study the higher the aggregation (mean weight dimeter), the lower the CO_2 fluxes from which propose a decrease in soil organic matter physical protection occurs upon annual burning and mowing and that is likely to be one of the causes of the increased CO_2 emissions from soil compared to no-burned grasslands.

They are several implications of these results. The first one is that burning, which is a common practice in grasslands of the developing world, should be avoided because of a significant increase in CO_2 emissions from soil. Greenhouse gases other than CO_2 are also emitted during burning and these need to be further investigated. This result directly implies that alternative grassland management practices have to be found. While burning abandonment appeared to lessen CO_2 emissions from, it poses a major threat to forage production as grass species get replaced by woody ones. The third implication is that grass mowing is beneficial for avoiding the release of fire-derived GHGs emissions and maintaining the grass sward in good conditions. Mowing, however, slightly decrease grass palatability and this might constitute a major limitation for its broad adoption.

There is a need to find grassland management emitting low amount of CO_2 emissions from soil while sustaining high grass productivity and diversity. Following these results, a combination of mowing and burning and/or controlled grazing such as the high-density short duration stocking rate one should be tested. Further research needs also to be performed on the underlying reasons of the variation in CO_2 emissions from soil between different treatments.

CHAPTER 7: CONCLUSIONS, RECOMMENDATIONS AND AREAS FOR FURTHER RESEARCH

7.1 Conclusions

This dissertation aimed to quantify and compare CO_2 emissions from soils under different cropland and grassland management practices in KwaZulu-Natal province of South Africa with an overall goal to identify potential options for farmers worldwide geared at climate change attenuation without compromising on food security. The main conclusions drawn from the results of this dissertation are:

A quantitative meta-analysis of 46 studies worldwide performed before the field studies showed that tilled soils emit, on average, 21% more CO_2 than no-tilled soils at global scale. The highest decrease in C outputs from soils following tillage abandonment occurred on sandy soils (SOC content: SOCc <10 g C kg^{-1}) of arid climates. The meta-analysis results also showed that maize cropping systems had the highest SOCc difference between tilled and no-tilled soils, with no-tilled systems having 15% higher SOCc than the tilled systems. In the same meta-analysis, crop rotations emitted 26% lower CO_2 than monocultures. These results suggested that soil and climate variables could be the main drivers of CO_2 emissions from soils. The results also showed that tillage abandonment and crop rotations can significantly reduce CO_2 emissions from soils, and thus increase soil C sequestration. The CO_2 emission results from all systems that were analysed in the quantitative literature review depended strongly on soil factors such as soil texture, aggregate stability and organic C content, which showed that they are very important in global models of the C cycle.

The tillage abandonment trials, under a maize monoculture, in KwaZulu-Natal showed that, on average, no-tillage with high-density short duration stocking (1200 cattle heads ha^{-1} for three days per year) and no-tillage with free grazing over three years resulted in 60 and 31% lower gross CO_2 emission than conventional tillage with free grazing, which confirmed the fact that tilled soils emit higher CO_2 than no-tilled soils. No-tillage with high-density short duration stocking also increased SOC stocks (SOCs) sequestration rate by average 1.4 Mg C ha^{-1} $year^{-1}$ within a three year period, which was significant at $p<0.05$. C sequestration in the no-till soils under high-density short duration stocking correlated positively with increased soil compaction and decreased soil temperature. These results suggested higher C sequestration benefits from integrating high-density short duration cattle stocking in no-tillage maize systems than conventional practices and/or no-tillage systems where crop residues are removed. The cropping system field study results suggested that land management techniques which lead to an increase of top soil bulk density while decreasing the temperature could result in increased soil C sequestration.

At a grassland site in KwaZulu-Natal, grassland degradation resulted in an increase of CO_2 emission from the soils. The CO_2 emission relative to SOCs (g CO_2-C g^{-1} C day^{-1}) was 41% higher compared to a non-degraded grassland, while that relative to produced biomass (g CO_2-C kg^{-1} produced biomass day^{-1}) was 81% higher. The SOCs were 86% higher in non-degraded than highly degraded grassland, implying greater SOCs depletion under highly degraded than non-degraded grassland. However, short-term (4 years) rehabilitation of the degraded grassland by high-density short duration stocking reduced CO_2 emissions relative to SOCs and CO_2 emissions relative to produced biomass than the tradition free grazing system and annual burning. Interestingly, high-density short duration stocking increased aboveground biomass production by 70% and SOCs by 37% on the degraded grassland over three years of implementation. The results from grassland rehabilitation were similar to those from the tillage abandonment trials in that high-density short duration stocking resulted in lower CO_2 emission from the soil than the other treatments. This further supports the notion that higher top soil bulk density and reduced temperature could be very important drivers of soil C sequestration even on grasslands.

Annual burning of grasslands as a management technique appeared to have negative impact on global warming because CO_2 measurements at Ukulinga Farm in KwaZulu-Natal showed that long-term (60 years) burning of grass once every year resulted in 30% higher gross CO_2 emission than a no-burning system where trees encroached on the trial plots. The annual burning treatment also emitted 26% higher CO_2 relative to SOCs than a no burning system (1.32 ± 0.1 *vs.* 1.05 ± 0.07 mg CO_2-C g^{-1}C day^{-1}). These results could be explained by lower stability of SOC in the annual burning treatment because the higher CO_2 emission was associated with lower aggregation (measured by mean weight diameter method), which suggest lower physical protection of SOC in annual burn soils than those not subjected to annual burning. This implies that although annual burning is promoted for grassland management to improve aboveground biomass production, the resultant weak top soil aggregation gives rise to high CO_2 emissions from the soil, which has negative implications on climate change.

7.2 Overall thesis discussion and contributions to new knowledge

The information generated from the field studies could help identify the main factors driving CO_2 emissions from the soils. The CO_2 emissions correlated negatively with soil temperature and bulk density, which was consistent with results from several studies (e.g. Silva et al., 2000; Qi et al., 2002; Suseela et al., 2012). This suggest that any land management technique which results in increased soil bulk density and reduced topsoil temperature could reduce soil C losses through CO_2 emissions. The increased soil bulk density and decreased soil temperature might lead to lower microbial activity, hence less SOM decomposition. In addition, the results of this dissertation point to a need to include more soil factors such as soil aggregate stability and organic C stocks and content in global C cycle models. This would help to improve the understanding of the mechanisms driving C sequestration, and better estimation of the C pools dynamics. For example, a modelling study by Wang et al. (2016) showed that critical C inputs could be estimated more effectively by using current SOC levels, mean annual temperature, precipitation and soil clay content. However, it is important to highlight that C pool dynamics do not always respond linearly to the increase in C inputs (Gulde et al., 2007).

Overall, the summary of gains and losses of SOC_S, N stocks, water erosion and other evaluated parameters due to different land management techniques are expressed in percentage changes from reference land management systems, i.e. conventional tillage in case of croplands and degraded grassland elsewhere (Table 7.1). No tillage with free grazing after harvest was the most efficient cropland management technique in terms of reducing runoff, soil loss and C losses through particulate and dissolved C as well as CO_2 emission. However, the technique did not induce a significant increase in soil carbon stocks, pointing to low efficiency in terms of atmospheric C sequestration. In contrast, no-tillage with HDSD was the most efficient for atmospheric C sequestration and reducing sediment concentrations onsite, but the high amount of surface runoff generated pose potential environmental challenges, for instance higher occurrence of gullying and/or flooding. Finally, removing crop residues from the top-soil increased soil erosion, particulate and dissolved soil C losses with subsequent reduction in CO_2 emission; while residue retention had no significant reductions on soil erosion and soil C stocks. On the basis of these results, the best land management technique for croplands would be no tillage with free grazing after harvest, because it decreased surface runoff, soil and C losses.

Table 7.1 Summary of change results of land management impacts in grassland and croplands on carbon dynamics on plot levels with results compiled from thesis chapters (3, 4, 5 and 6) unpublished data (Mchunu et al. submitted to Geoderma) at the cropland site (erosion variables; R, SC, SL, POC_L and DOC_L) and published data (Mchunu and Chaplot, 2012) based on grass cover at the grassland site. The values are expressed in percentage change from the reference land management i.e. conventional tillage with free grazing and degraded grassland as a references for cropland and grasslands, respectively.

	SOCs	Ns	Bio	CO_2-C	R	SC	SL	POC_L	DOC_L
% change from reference									
Croplands (N=12)									
NTFG	4	10	15	-39	-36	0	-75	-42	-56
NTR	-7	16	4	-33	236	-56	-13	0	78
NTNR	-11	5	2	-17	57	22	36	222	224
NTHDSD	57	19	7	-36	236	-56	-13	0	77
Grasslands (N=12)									
AB	8	-15	-80	-19	126	-41	-60	-36	68
AM	-25	0	0	6	NA	NA	NA	NA	NA
HDSD	88	153	720	28	-100	-77	-88	-60	-60
LEF	85	53	760	-6	-100	-77	-88	-60	-60
NB	13	0	6.5	-23	NA	NA	NA	NA	NA
TFG	25	44	0	-19	126	-41	-59	-36	68

NTFG = no-tillage with free grazing; NTR = no-tillage crop residue mulching; NTNR = no-tillage without crop residue mulching; NTHDSD = no-tillage with high-density short duration stocking; AB = annual burn; TFG, traditional free grazing; LEF, livestock exclosure with fertilization; HDSD, high density short duration stocking; NB, annual burning and AM annual mowing; NA=Not assessed; SOCs, Soil organic carbon stocks; Ns, nitrogen stocks; bio, biomass production; CO_2-C, CO_2 emissions from soil; R, runoff; SC, sediment concentration in the runoff; SL, soil losses; POC_L and DOC_L, particulate and dissolve organic matter losses.

Rehabilitating grasslands by using fertilization and HDSD resulted in high soil C sequestration and dramatic decrease in soil erosion while enhancing aboveground biomass production (Table 7.1). The other techniques were less efficient in respect of C sequestration and biomass production, especially in case of annual mowing which also decreased soil C stocks. Though equally beneficial, fertilization and HDSD differ in terms of the costs of their implementation; chemical fertilization is more costly due to the need to purchase mineral NPK fertilizers. Therefore, HDSD was the best bet practice in grassland management and smallholder farmers in South Africa and anywhere else in the world are encouraged to adopt it to help in offsetting the annual emissions of CO_2 to the atmosphere, estimated at 11 Pg in 2014 (Lal, 2015a). For example, South Africa could sequester 0.17 Gt C year-1 at a C sequestration rate of 1.4 Mg C ha-1 year-1 achieved by managing degraded grassland using HDSD, which is much greater than 0.011 Gt C year-1 estimated by Knowles, et al. (2014) using a sequestration rate of 0.09 Mg C ha-1 year-1. South African grassland occupies 70% (12.19×107 ha) of the land surface (Snyman, 2003) and store about three-quarters of her terrestrial C stocks, which accounts for 90% of the gross primary production (Knowles, et al. 2014). When applied to the global level, where grasslands cover 30% (59.32×108 ha) with SOCs amounting to 1500 Gt (Suttie et al., 2005), then the amount of C sequestered in the top 1 m would be 8.3 Gt C year−1, which is 41% higher than 5.87 Gt C year−1 (Chaplot et al., 2016). Given that grassland soils have already lost 300 Gt C (Lal, 2004), it would take 36 years for HDSD to restore this C stock at the rate of 8.3 Gt C year-1. However, high stocking rate of animals has also been reported to increase other GHG emissions (e.g. N_2O and CH_4) through, mainly, excreta depositions and anaerobic soil conditions induced by top-soil compaction (e.g. Piva, et al., 2014).

The results in this dissertation contribute to new knowledge and quantitative information to advance the understanding of factors controlling CO_2 emissions from soils under different cropland and grassland management practices. The specific contributions may be enumerated as follows;

(i) Quantitative information on CO_2 emissions from soils under field conditions using modern equipment, providing the first estimates in South African smallholder farming systems for both grassland and cropland.

(ii) Integration of livestock grazing at high stocking rate for short duration within no-tillage practices in monoculture systems provides a means by which smallholder farmers can use their agro-systems to mitigate against climate change without compromising on food security.
(iii) Livestock grazing at high stocking rate for short duration also provides a means through which smallholder farmers can rehabilitate degraded grassland, to achieve high soil C sequestration and grass production.

7.4 Recommendations

- Smallholder farmers at Potshini are encouraged to abandon tillage in favour of no-tillage practices not only because of the demonstrated benefits in reducing CO_2 emissions and increasing C sequestration rate, but also for soil and water conservation purposes as well as for reducing crop production costs (Erenstein et al., 2008). However, weed control is a major disadvantage of no-tillage especially for farmers with limited resources (Opara-Nadi, 1993). The farmers are also encouraged to integrate cattle grazing into their continuous maize cropping systems instead of removing the residues; but high-density short duration cattle stocking in no-tillage continuous maize cropping systems would be more beneficial in terms of reducing CO_2 emissions and increasing C sequestration rate.

- Chemical fertilizer application and high-density short duration stocking were better techniques for grassland rehabilitation which reduced CO_2 emissions from soil, and increased C sequestration and biomass production, but smallholder farmers at Potshini are encouraged to adopt the high-density short duration stocking because it is a cheaper method, for example, it does not require money to buy artificial fertilizers. However, this high-density short duration stocking rate technique still requires a longer term assessment in area to evaluate its impact, for example, on the emissions of other greenhouse gases (e.g. Methane and nitrous oxide) and on grass diversity.

- Annual burning of grasslands, which is a common practice in grassland management in the developing world, need to be avoided because of the significant increase in CO_2 emission from the soils and also other GHG released during the burning process.

7.5 Areas for future research

The results presented in this thesis fill an important gap of knowledge and of quantitative information on CO_2 emissions form agricultural soils, especially the impact of different cropland and grassland management practices. In addition, this work also helped in increasing our understanding of the factors driving CO_2 emissions from soils to the atmosphere, which might be important in planning land management that mitigate against climate change without compromising on food security. While these results provide important recommendations for farmers, there are several limitations that call for improvements in future studies.

- The short-term duration of the experiments in cropland (tillage and no-tillage practices) and grassland rehabilitation was probably not adequate for better understanding of the impact of the many interacting factors. For example, in as much as this study offered promising results such as low CO_2 emissions, high grass biomass and soil SOCs when tillage is abandoned in favour of no-tillage and/or a degraded grassland is rehabilitated by the high density grazing for short duration (over three years); several studies reported that significant differences in soil C stocks only become significant after a long time e.g. at least 10 years of implementing tillage and no-tillage in the case of cropping systems (e. g. Six et al., 2000; Abdalla et al., 2015).

- Future research could lie on long-term assessments and integration of other factors such as microbial biomass and activity, belowground biomass, C allocation by plant roots and temperature sensitivity of recalcitrant soil organic matter decomposition in both the cropping and grassland trials.

- While annual burning proved to be an important grassland management practices in terms of promoting high grass diversity, removal of old and lignified parts and protecting grasslands against invasive and woody species (DiTomaso and Johnson, 2006; Bowles and Jones, 2013), its implementation in a rural set up like Potshini could be problematic. For example, grassland burning requires firefighting skills and equipment which are not normally available in many rural areas. Moreover, in this study long-term annual burning showed higher CO_2 emissions in comparison to no burn with tree encroachment.

Therefore, the high density grazing for short duration appears a better technique for managing grasslands in a rural area.

- The study investigated the impact of high-density short duration stocking on CO_2 emissions and SOCs at Potshini, all of which were within a subtropical climate high-density short duration stocking practice has been shown to be a success in terms of rehabilitating grasslands by increasing grass biomass production and C sequestration rate at Potshini in this study, a finding similar to the one by Chaplot et al. (2016). These results suggested greater potential of such a practice to increase the C inputs to the soil. However, still there is a need for replicating the high-density short duration stocking practice at different sites to capture it is responses under different environmental conditions.

- Unfortunately, due to time and cost limitations, this study only managed to evaluate CO_2 emissions from soils under different agricultural practices, which contribute about 60% of the total greenhouse gases effect (Rastogi et al., 2002). The other greenhouse gas (e.g. methane and nitrous oxide) emissions from agricultural soils at the study site need to be quantified in future research for better decision making regarding the best management practices to mitigate against climate change.

- The closed-chamber method of measuring CO_2 emissions from soils used in the study has its own shortcomings. For example, the measured CO_2 emissions values may have been biased by disturbing air pressure, which has potential to alter the CO_2 concentration in the soil. Also, by measuring the accumulation of CO_2 released from the soil surface, the method does not provide information about emissions from different soil profiles, which might help in understanding the individual contributions from different soil depths (Healy et al., 1996; Tang et al., 2003). This is important for understanding soil C mechanisms. Alternative methods (e.g. small solid-state sensors: GMT222, Vaisala Inc., Finland) need to be used to monitor CO_2 emissions from different horizons continuously in future studies.

- Future studies may need to separate between autotropic (respiration of plant roots and related microorganisms) and heterotrophic (respiration of decomposing dead root biomass, litter, soil organic matter and soil fauna) CO_2 emissions, which require controlled environments, expensive equipment and consumables (e.g. stable isotopes). This is extremely important to enable the evaluation through particulate dissolve organic C also needs to be considered for better estimates of total C outputs.

- The study results may be used to test and improve some models such as DAYCENT ecosystem model, which is the daily time step version of CENTURY model (Parton et al., 1994). The DAYCENT model, which was developed to provide more accurate analyses of greenhouse gases (CO_2, CH_4 oxidation, N_2O, and NO) exchange between the soil and the atmosphere on a daily basis (Del Grosso et al., 2005), could be tested in the study area before trying to find ways to extrapolate the results on a larger scale.

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