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Khatab Abdalla, Macdex Mutema, Pauline Chivenge, Colin Everson, Vincent Chaplot. Grassland rehabilitation significantly increases soil carbon stocks by reducing net soil CO₂ emissions. *Soil Use and Management*, 2022, 38 (2), pp.1250-1265. 10.1111/sum.12790 . hal-03646193

HAL Id: hal-03646193

<https://hal.science/hal-03646193>


Submitted on 22 Mar 2023

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RESEARCH ARTICLE

Grassland rehabilitation significantly increases soil carbon stocks by reducing net soil CO₂ emissions

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Funding information

Alexander von Humboldt-Stiftung; Water Research Commission, South Africa, Grant/Award Number: K5/2266

Abstract

Restoration of degraded grasslands through improved management is among the possible sustainable solutions to compensate for anthropogenic soil carbon (C) emissions. While several studies have shown a positive effect of rehabilitation on soil C, the impact on soil CO₂ emissions is still uncertain. Therefore, this study aimed at quantifying the impact of grassland rehabilitation on soil CO₂ emissions in a degraded grassland, South Africa. Commonly used rehabilitation practices were considered, that is rotational grazing (RG), livestock enclosure with fertilizer application (EF) and annual burning (AB), all being compared with traditional free grazing (FG). A total of 2880 in situ measurements of CO₂ emissions were performed over 2.5 years under field conditions simultaneously with aboveground biomass, soil temperature, water content and soil organic C (SOC) to understand the changes in C fluxes. The RG performed the best under degraded grasslands by decreasing net CO₂ emissions (per g of C) by 17% compared to FG, while EF increased emissions by 76% and AB had similar emissions to FG. The lower net emission under RG is associated with an increase in SOC stocks by 50% and aboveground biomass by 93%, after three years of implementation. Soil CO₂ emissions were correlated positively to aboveground biomass and topsoil temperature ($r = 0.91$ and 0.60 , respectively), implying a high effect of grass cover on soil microclimate and microbial activity. These results suggested RG as a potential cost-effective nature-based soil management strategy to increase SOC stocks into degraded grassland. However, long-term trials replicated in different environments are still required.

KEYWORDS

climate change, grassland management, grazing, soil carbon, soil respiration, South Africa

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1 | INTRODUCTION

Soils play a crucial role in the global carbon (C) cycle as they store 1700 GtC (1 Gt = 10^9 tonnes), an amount much greater than the C stored in vegetation (450 GtC) and atmosphere (875 GtC) (Friedlingstein et al., 2021). However, inappropriate land uses and land mismanagement, for example deforestation, overgrazing and intensive tillage, depleted soil C stocks and cause widespread land degradation (Lal, 2004a). Most agricultural soils have lost 20%–50% of their original soil organic C stocks (SOCs) during the 20th century (Lal et al., 2007), and increasing evidence points to the export of large amounts of nutrients from soils and consumed plant material as the main reason (e.g. Chaplot, 2021).

Grasslands, which account for about 70% of the global agricultural area (Abberton et al., 2010), have lost almost 300 GtC from the top one-metre depth of their soils (Lal, 2004a, 2004b) because of mainly land degradation caused by inappropriate management practices such as overgrazing, biomass burning and land use change (Abdalla et al., 2021; Dlamini et al., 2016; Lal, 2004a; Lu et al., 2017). Grassland degradation increases C output from the soils leading to drastic SOC depletion (Abdalla et al., 2018; Chaplot et al., 2016; Dlamini et al., 2014). However, SOC depletion under degraded grassland soils is varied worldwide based on soil types and climatic conditions. In a global meta-analysis, Dlamini et al. (2016) reported average SOC losses of ~16% in arid environments compared with ~8% in wetter ones, with Asia being the most affected continent (–27%). The lost C from grassland soils could be restored and that grassland soils may constitute a net sink for atmospheric C if appropriate management practices are adopted (Conant et al., 2001, 2017; Tessema et al., 2020).

Grassland management strategies such as grass burning, fertilizer application and controlled/rotational grazing have traditionally been used to increase grass production and subsequently increased soil C sequestration (Conant et al., 2017; Conant & Paustian, 2002; Fynn et al., 2004). Burning of grass is a traditional grassland management practice that has long been used to enhance grass productivity and prevent bush encroachment (Montané et al., 2007; Trollope, 1980). However, burning of grasslands was found to increase soil C losses (Abdalla, Chivenge, et al., 2021; Abdalla et al., 2016; Xu & Wan, 2008). Abdalla et al. (2016) found that long-term (62-year) annual burning increased soil CO₂ by 30% compared with unburned grass in South Africa. Such an increase in CO₂ emissions from burned grassland soil was explained by soil aggregate instability (Abdalla, Chivenge, et al., 2021). Relatively, low differences in soil CO₂ emissions between burned and unburned grasslands were reported in the northern Loess Plateau of China (Jia et al., 2012; Xu & Wan, 2008).

On the contrary, fertilizer application was proposed as a promising practice for grassland rehabilitation with a direct impact on soil C sequestration (Chaplot et al., 2016). However, Du et al. (2014) reported a 30% increase in soil CO₂ emission after compound fertilizer application in southern China.

In their latest global synthesis study, Conant et al. (2017) reported an increase in topsoil SOC_s by 0.11–1.00 Mg C ha⁻¹ year⁻¹ following improved grazing management practices. Their study indicated that rotational grazing could increase grassland productivity and potentially soil C sequestration. The available studies demonstrated the high potential of controlled rotational grazing or short-duration grazing to increase forage production as a proxy of C inputs, promoting high SOC_s (Chaplot et al., 2016; Díaz de Otálora et al., 2021; Jacobo et al., 2006; Oates et al., 2011). Considering the potential of non-degraded grasslands to store a large amount of C and thus mitigate climate change (Keller et al., 2021; Mbaabu et al., 2020), more research to restore degraded grasslands by identifying and applying sustainable management practices is required. Therefore, improved grazing practices such as rotational grazing (RG) may quickly restore the degraded grassland and subsequently replenish depleted SOC_s.

The RG practice in this study was modified from Savory and Parsons (1980), which has rarely been investigated in terms of soil CO₂ emissions so far. This practice involves a significant shift in livestock management from the traditional grazing approach to reproduce the natural grazing behaviour of herds (Savory & Butterfield, 2016; Savory & Parsons, 1980). The idea behind the practice was that adjusting the livestock number and grazing period to match available forage amounts, followed by a long recovery period, can significantly increase ecosystem services in terms of soil and vegetation qualities (Fynn, 2008; Hillenbrand et al., 2019; Ritchie, 2020; Savory & Butterfield, 2016). For example, Peel and Stalmans (2018) reported that RG had significantly higher soil stability and nutrient cycling in the savannah biome near Victoria Falls town, Zimbabwe. Hillenbrand et al. (2019) found that the RG approach using long-term adoptive multi-paddocks grazing effectively limited overgrazing and increased SOC_s compared with heavy continuous grazing (34.2 kg m⁻² vs. 16.2 kg m⁻²; $p < 0.0001$) in a silty clay loam soil at South Dakota shortgrass prairie, USA.

Despite the existing evidence that RG increased soil C sequestration, its performance in rehabilitating degraded grasslands to simultaneously increase biomass production and soil SOC_s has rarely been investigated. Addressing such a knowledge gap is essential for achieving goals 13 and 17 of the United Nations sustainable development goals in Southern African countries and the whole sub-Saharan African region.

In the current study site, Chaplot et al. (2016) showed that RG (high-density short-duration grazing) increased soil SOC by an average of $12.4 \text{ g C m}^{-2} \text{ yr}^{-1}$ only after two years of implementation in a homogeneous sandy soil under highly degraded grasslands. The high biomass production and subsequent C inputs explained such an increase in SOC, particularly under degraded grasslands. Degraded soils are usually characterized by hard soil crusts (Dlamini et al., 2011); hence, it was postulated that livestock hooves would break the soil crusts and also trample grass tufts in the presence of animal excrement, thus facilitating soil microbial activities. The enhanced microbial activities will be associated with high soil heterotrophic respiration, an important component of the total CO_2 emissions. Therefore, the impacts of RG as an efficient practice to rehabilitate degraded grasslands on soil CO_2 emissions as the C outputs and the underlying factors need to be quantified. The current study hypothesized that high aboveground biomass production and SOC induced by RG (Abdalla, Mutema, et al., 2021; Chaplot et al., 2016) would result in high gross soil CO_2 emissions ($\text{CO}_2\text{-C per m}^2$), but low net soil CO_2 emissions ($\text{CO}_2\text{-C per g C}$). As the net soil CO_2 emissions refer to the difference between the absorbed and released C, the low net CO_2 emissions imply an enhanced topsoil C stocks of the degraded grasslands.

2 | MATERIALS AND METHODS

2.1 | Study site

The study site located ($29^\circ 21' 39.87''\text{E}$; $28^\circ 48' 40.48''\text{S}$, 1305 m.a.s.l.) at Potshini village, 10 km south of Bergville town, KwaZulu-Natal Province, South Africa (Figure 1). The area experiences a temperate climate characterized by hot-wet summers and cold-dry winters. The long-term (30-year) mean annual temperature and precipitation were 13°C and 745 mm year^{-1} , respectively (Dlamini et al., 2011; Schulze, 1997). The soils were classified as *Acrisols* (IUSS-WRB, 2014) developed from sandstones and mudstones (>64% sand content in the top 0.9 m) with kaolinite as the dominant clay mineral (Dlamini et al., 2011, 2014). They were characterized by a dark brown (7.5YR 4/4) A horizon, a weak subangular blocky structure and acidic conditions (pH 3.8–4.2) with a cation exchange capacity range of $1.86\text{--}5.86 \text{ cmol}_c \text{ kg}^{-1}$ (Chaplot et al., 2016; Dlamini et al., 2014). The grass at the study site was classified as Moist Highveld Sourveld with *Hyparrhenia hirta* and *Sporobolus africanus* as the most common species (Mucina & Rutherford, 2006).

Grassland degradation was defined as the reduction in the capacity of grasslands to deliver their key ecosystem

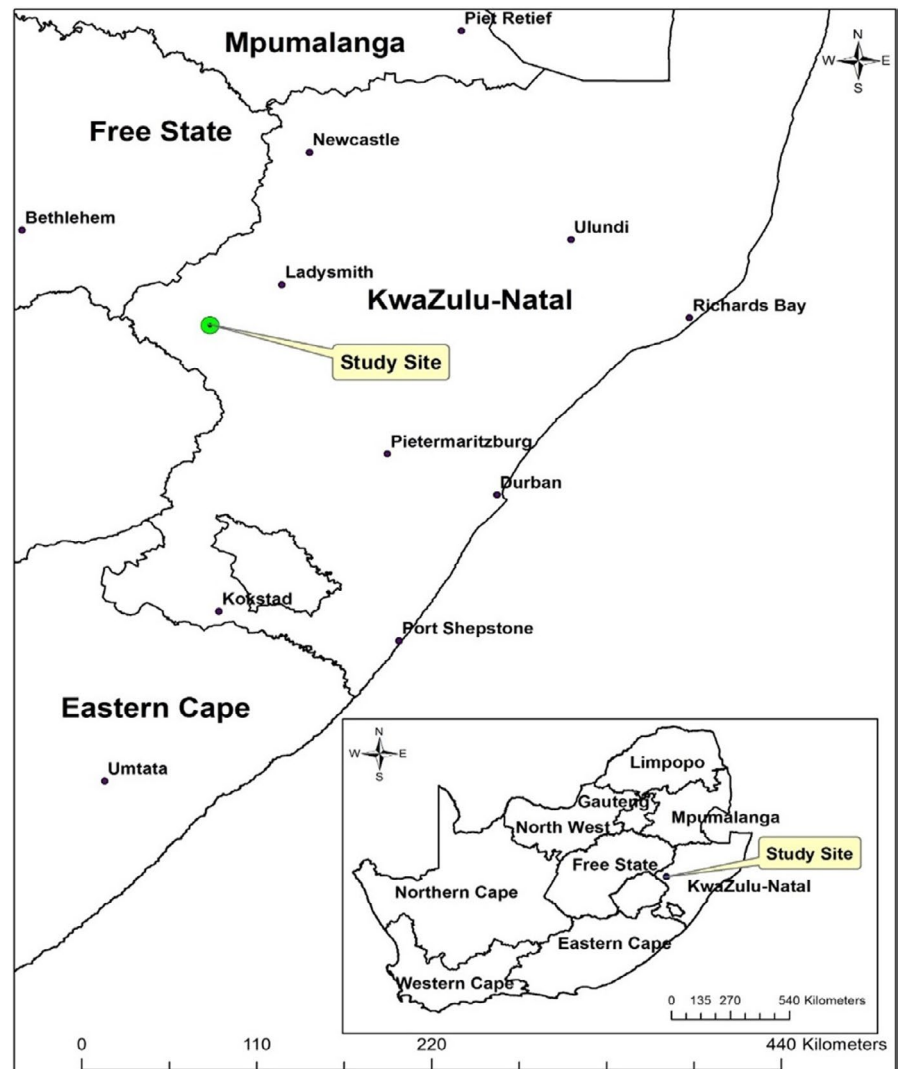
functions because of biodiversity and vegetation cover losses resulting from mismanagements such as overgrazing (FAO, 2011; Wick et al., 2016). Poor management of grasslands such as overgrazing associated with high anthropogenic activities because of population growth in the area has led to severe grass and land degradation with subsequent soil C losses because of increased runoff and associated soil erosion (Dlamini et al., 2011, 2014; Podwojewski et al., 2011). Besides the traditional free grazing, grassland burning for increasing fodder production, species composition and controlling bush encroachment is also a common practice in the area (Everson & Tainton, 1984; Fynn et al., 2004).

2.2 | Experimental design

Dlamini et al. (2011) established the experimental site in June 2011 and characterized the soil by high short-range properties variations. In 2012, Dlamini et al. (2014) demarcated a 1500-m^2 ($30 \text{ m} \times 50 \text{ m}$) space with homogeneous bedrock and soil properties but showing grassland degradation gradient from heavily degraded, that is aerial grass cover (Cov) of <5% on the upslope, to non-degraded, that is 100% cover on the downslope position. The area was subdivided into six subplot ($5 \text{ m} \times 50 \text{ m}$), each showing all the degradation levels but with similar soil properties and subjected to different management practices to evaluate their effect on SOC (Chaplot et al., 2016). This site was also utilized by Abdalla et al. (2018) to investigate the impact of grassland degradation on soil CO_2 emissions.

The present study considers two degradation levels (non-degraded, >75% Cov >100%; and degraded, >25% Cov >50%) and four grassland management practices, namely rotational grazing (RG), livestock exclusion with NPK fertilizer applied (EF), annual burning (AB) and traditional free grazing (FG). The RG was achieved through adopting a high stocking rate of 38 Nguni cattle (equivalent of $1200 \text{ heads ha}^{-1}$) supplied by the local community for three consecutive days in June, from 2011 to 2014, followed by livestock exclusion for the remaining 362 days each year. The EF treatment involved complete livestock exclusion by wire fence and application of a single dose of nitrogen (N), phosphorus (P) and potassium (K) combined +NPK fertilizer (2:3:3, 22) at 0.2 t ha^{-1} once a year in September before the rainy season. The fertilizer dosage was decided upon the results of pre-soil fertility analysis for the respective nutrients. Aboveground grass biomass was removed in the fenced EF treatment plots. For AB treatment, fenced plots were burned once a year in June. Grassland burning is a common management practice widely used to remove unpalatable vegetation and control bush encroachment in the African savannah.

FIGURE 1 Study site location in KwaZulu-Natal province, South Africa



The FG corresponded to the common practice where livestock grazed freely, and the grazing area was open for cattle grazing throughout the year. Therefore, the stocking rate was not known and possibly changed throughout the years. Within each subplot, three 1-m² (1 m × 1 m) areas were demarcated and used for soil CO₂ measurements, soil sampling and aboveground biomass to minimize possible variations resulting from different soil properties.

2.3 | Soil CO₂ emission measurements

Three polyvinyl chloride (PVC) plastic collars (12 cm diameter * 5 cm height) were inserted 0.03 m into the soil leaving 0.02 m above the soil surface. The three collars were inserted randomly between the grass tufts within the demarcated 1-m² plots (9 collars per treatment in each degradation level). Live grass shoots inside the PVC collars were removed regularly by hand. The collars were inserted into their positions two weeks before the first CO₂ measurements to eliminate the effect of soil disturbance (e.g.

Bahn et al., 2008; Norman et al., 1997). They were kept in place throughout the experiment, except for RG and AB where it was necessary to remove them to allow grazing and burning, respectively.

The measurements were performed in situ twice to thrice a month in the wet summer and once a month in the dry winter. The frequency of measurement was higher in summer because CO₂ emission was more variable due to higher rainfalls, soil moisture and temperature change. The measurements were done between 10:00 and 13:00 h to limit the impact of diurnal variations (Castaldi et al., 2010). A LI-COR 6400XT gas exchange system (LI-COR, Lincoln, NE, USA) fitted with a LI-COR 6400-09 soil respiration chamber was used to perform the measurements (Norman et al., 1992, 1997). The chamber with an internal volume of 991 cm³ and a surface area of 71.6 cm² was inserted 0.02 m into the PVC collar for each measurement.

The CO₂ emissions were measured one time per collar, which was computed from three measurement cycles resulting in nine values per treatment in each degradation level. A total of 40 measurements over two years and a half

resulted in 2880 (9 per treatment * 4 treatments * 2 degradation levels * 40 events) in situ measurements over the experiment duration. The total soil CO₂ emission to the atmosphere, measured using the LI-COR 6400, was converted to gross CO₂ by dividing the measured CO₂ emission by the surface area of the PVC collar and expressed as g CO₂-C per unit of soil surface area. The net soil CO₂ emissions relative to SOC_s, that is g CO₂-C per g of soil C, were an indirect measurement of SOC stabilization (Abdalla et al., 2018; Chaplot et al., 2015). The net CO₂ emissions were calculated as follows:

$$\text{Net soil CO}_2 \text{ emissions} = \frac{\text{Gross soil CO}_2 \text{ emissions}}{\text{SOC}_s} \quad (1)$$

where SOC_s represent soil organic C stocks (kg C m⁻²) at the 0.05 m depth.

2.4 | Soil sampling and analysis

Sampling for soil bulk density determination was performed using metallic cylinders of 0.075 m diameter and 0.05 m height. The cylinders were used to collect undisturbed samples from 0 to 0.05 m soil depths in all the demarcated 1 m². The samples were placed in airtight plastic bags and later oven-dried at 105°C until constant weight is reached. The soil bulk density was calculated using the below equation (Grossman & Reinsch, 2002):

$$\rho_b = \frac{\text{odw} - \text{rf} - \text{cw}}{\text{cv} - \left(\frac{\text{rf}}{\text{dr}}\right)} \quad (2)$$

where ρ_b = soil bulk density of <2 mm material (g cm⁻³); odw = oven dry weight of sample (g); rf = weight of rock fragments in sample (g); cw = weight of empty sampling core (g); cv = volume of sampling core (cm⁻³); and dr = density of the rock fragments (g cm⁻³).

Another soil sample for soil carbon (C) and total nitrogen (TN) content analysis was collected at the same time with the bulk density sampling and from the exact soil depth at three randomly selected positions within the demarcated 1 m². Therefore, nine topsoil samples (N = 9) were collected per treatment in each degradation level. The samples were air-dried for 48 h and sieved through 2-mm sieves and stored for the C and N analysis. The sieved samples were analysed for total C and N using a LECO CNS-2000 Dumas dry matter combustion analyser (LECO Corp., St. Joseph, MI). The LECO CNS-2000 provides excellent total C and N results quickly and simply (Kowalenko, 2001). The total soil C content was considered equivalent to soil organic C (SOC) content because

no significant reactions were observed after adding a few drops of 1 M HCl. SOC stocks (SOC_s) were calculated following the equation by Batjes (2014):

$$\text{SOC}_s = \text{SOC}_c \times \rho_b \times T \left(1 - \frac{\text{Pf}}{100}\right) b \quad (3)$$

where SOC_s = soil organic C stock (kg C m⁻²); SOC_c = soil organic C content in the ≤2 mm soil material (g C kg⁻¹ soil); ρ_b = soil bulk density (kg m⁻³); T = thickness of the soil layer (m); PF = proportion of fragments of >2 mm in per cent; and b = constant equal to 0.001. Nitrogen stocks (TN_s) were computed following the same equation by replacing SOC_c by TN content (TN_c).

Soil bulk density and SOC_s data obtained in 2011–2012 before implementing the treatments were considered baseline data, which were compared with the data obtained in 2014–2015 after three years to evaluate the performance of the treatments on soil compactions and SOC_s replenishment under degraded grasslands.

2.5 | Soil temperature, water content and weather data

Topsoil (0.05 m) temperature and water content were measured in conjunction with CO₂ emission measurements (N = 9), using a thermocouple connected to the LI-COR chamber and a portable hydrosense soil moisture meter (Campbell Scientific, Inc., USA), respectively. The soil temperature and water content measurements were performed as close to PVC collars as possible. While the soil temperature was measured over the study period, the soil moisture measurements only started in December 2014 because of shipping delays and the probe malfunctioning at the beginning. Precipitation and air temperature data were obtained from an automatic weather station (Campbell Scientific Africa (Pty) Ltd) located (29°21'39.34"E; 28°48'28.56"S, 1344 m.a.s.l.) approximately 100 m from the experimental plots. The weather station recorded data every 10 min for the weather parameters.

2.6 | Aboveground biomass

Aboveground grass biomass was assessed by harvesting all the grass materials within each demarcated one m² once a year in June, immediately before burning and grazing. The grass materials were clipped at the soil surface level and all aboveground material pocketed. The materials were later oven-dried at 65°C for 48 h and weighed to estimate dry weights, which were subsequently used to

compute the aboveground biomass ($\text{kg m}^{-2} \text{ year}^{-1}$). The initial biomass data in 2011–2012 were compared against the one obtained in 2014–2015 to evaluate the treatment's effect on biomass productions.

2.7 | Data analysis

The experimental data were tested for normality using the Shapiro–Wilk normality test, which showed that the data are normally distributed ($p > 0.05$). Hence, the data are unbalanced (unequal time interval between CO_2 , aboveground biomass and soil parameters), the data analysis was conducted using the residual (or restricted) maximum-likelihood (REML) model in GenStat 14 (VSN International, Hemel Hempstead, UK, 2011), which is an effective analysis method of unbalanced on-farm data (Virk et al., 2009). The REML algorithm estimates the variance components using residual maximum likelihood and uses these variances to generate generalized least square estimates of the treatment effects and the best linear unbiased predictors of the random effects (DeLacy et al., 1996).

In this study, the treatments and degradation levels were considered fixed effects and the demarcated subplot was the random effects. Since soil CO_2 emissions were measured repeatedly at the same PVC collar positions, a mixed repeated-measures model was used to quantify the treatment and grassland degradation effects on the CO_2 emissions. A significant threshold of $p \leq 0.05$ was used for mean treatment comparisons, with the mean variations documented using standard error. Univariate correlations between soil CO_2 emissions and soil properties were performed separately for degraded and non-degraded grassland conditions. Principal component analysis (PCA) based on a correlation matrices was used to further investigate the main factors that drive CO_2 fluxes. The

figures were produced using SigmaPlot (version 12; Systat Software Inc., Richmond, California, USA, 2013).

3 | RESULTS

3.1 | Precipitation, air and soil temperatures

Annual precipitation for 2013 and 2014 was 718 and 562 mm, respectively. Most of the precipitation (about 90%) occurred in summer, that is between November and April each year (Figure 2). The mean annual air temperature in both years was $17 \pm 0.21^\circ\text{C}$. The lowest average daily air temperature was recorded ($13 \pm 0.03^\circ\text{C}$) in June of both years, while the highest (38°C) occurred in September 2014. The average daily air temperature of $33 \pm 0.27^\circ\text{C}$ for July–September 2014 was much higher than in 2013 for the same period (Figure 3). Soil temperature changed significantly in response to changes in the air temperature (Figure 2a,b). Soil temperatures were, in most cases, lower in RG and EF than in FG and AB in both the degraded and non-degraded grassland conditions. However, the treatment means were all significantly different from each other and decreased in the order $\text{AB} > \text{FG} > \text{RG} > \text{EF}$.

3.2 | Soil CO_2 emissions from the different management strategies

Considering the overall average gross soil CO_2 emissions (per m^2) of the degraded and non-degraded grasslands, RG and EF emitted 78% and 30% higher gross soil CO_2 than FG, respectively (Table 1). However, the gross CO_2 emission from AB did not differ significantly from FG. The degraded grassland tended to have lower gross CO_2

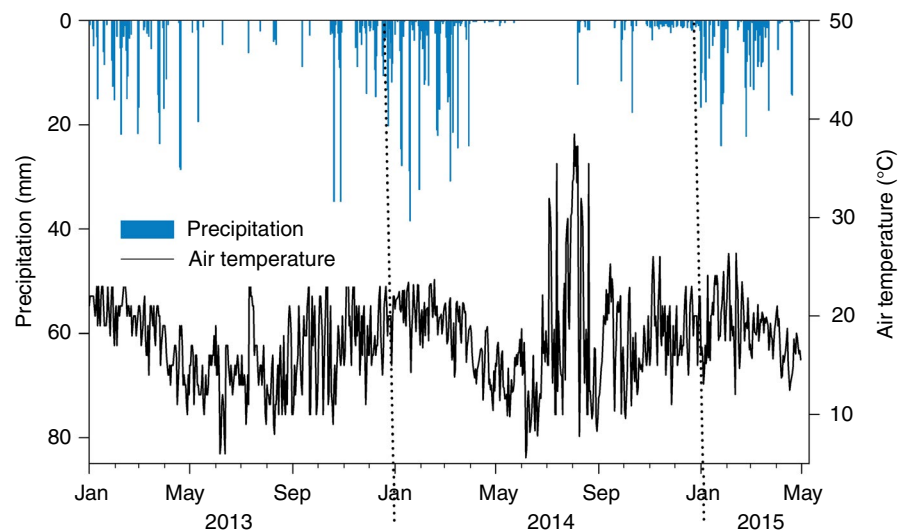


FIGURE 2 Daily precipitation and average air temperature during the study time

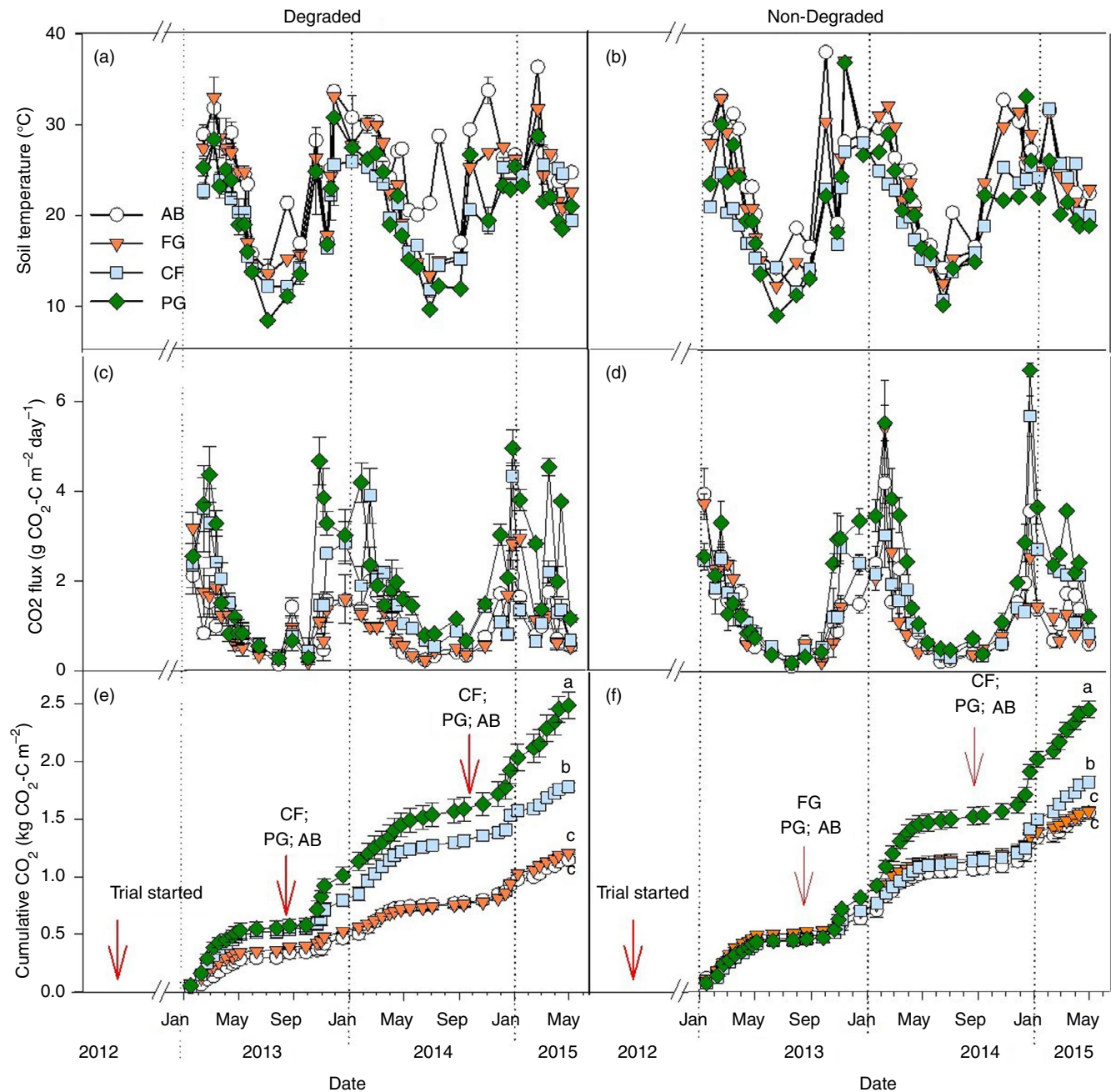


FIGURE 3 Soil temperature at 0–5 cm (a and b), daily (c and d) and cumulative (e and f) gross CO₂ emissions from rotational grazing (RG), livestock enclosure with NPK fertilizer application (EF), annual burn (AB) and traditional free grazing (FG) under degraded and non-degraded grasslands. Error bars represent the standard error of the mean (N = 9). Different lower-case letters to the right of the final cumulative flux values indicate significant differences between treatments at $p \leq 0.05$

emissions than the non-degraded one in all treatments, except in RG. On the contrary, the overall average net CO₂ emissions (per g of C) were highest in EF and lowest in FG (Table 1). Under the degraded grasslands, RG had the lowest net CO₂ emissions, which was 17% lower than FG and as much as 106% than EF. The AB and FG treatments emitted the highest and lowest net CO₂, respectively, under the non-degraded grassland condition; however, only FG was significantly lower than other treatments.

The mixed repeated measures showed that treatments, degradation levels, date of CO₂ sampling and their interactions significantly affect both gross and net CO₂ emissions from the study site (Table S1). The daily gross CO₂ emissions varied greatly with season and treatment under both degradation levels, mainly in summers, implying strong air temperature and precipitation influences on the soil CO₂ emissions (Figure 3c and d). The RG and EF treatments had higher cumulative gross CO₂ than FG and AB in the degraded grasslands, with a final cumulative value in RG

TABLE 1 Mean ± standard error (SE) of soil CO₂ emissions (gross and net CO₂) for the rehabilitation treatments (rotational grazing (RG), livestock enclosure with NPK fertilizer application (EF), annual burning (AB) and traditional free grazing (FG) (N = 120)

	Soil CO ₂ emissions							
	Gross CO ₂ (g CO ₂ -C m ⁻² day ⁻¹)				Net CO ₂ (g CO ₂ -C g ⁻¹ C day ⁻¹)			
	RG	EF	AB	FG	RG	EF	AB	FG
Overall average								
Mean	2.07 ^a	1.51 ^b	1.13 ^c	1.16 ^c	1.36 ^c	1.93 ^a	1.58 ^b	1.22 ^c
SE	0.16	0.11	0.09	0.10	0.1	0.18	0.14	0.16
Degraded (D)								
Mean	2.07 ^a	1.50 ^b	0.97 ^c	1.02 ^c	1.26 ^c	2.60 ^a	1.56 ^b	1.48 ^b
SE	0.21	0.15	0.09	0.11	0.13	0.30	0.17	0.20
Non-degraded (ND)								
Mean	1.29 ^c	1.51 ^b	2.04 ^a	1.30 ^c	1.46 ^a	1.25 ^a	1.60 ^a	0.96 ^b
SE	0.98	1.03	1.47	1.05	1.00	0.83	1.43	0.83

*Means in each row with the same letter are not significant at $p < 0.05$.

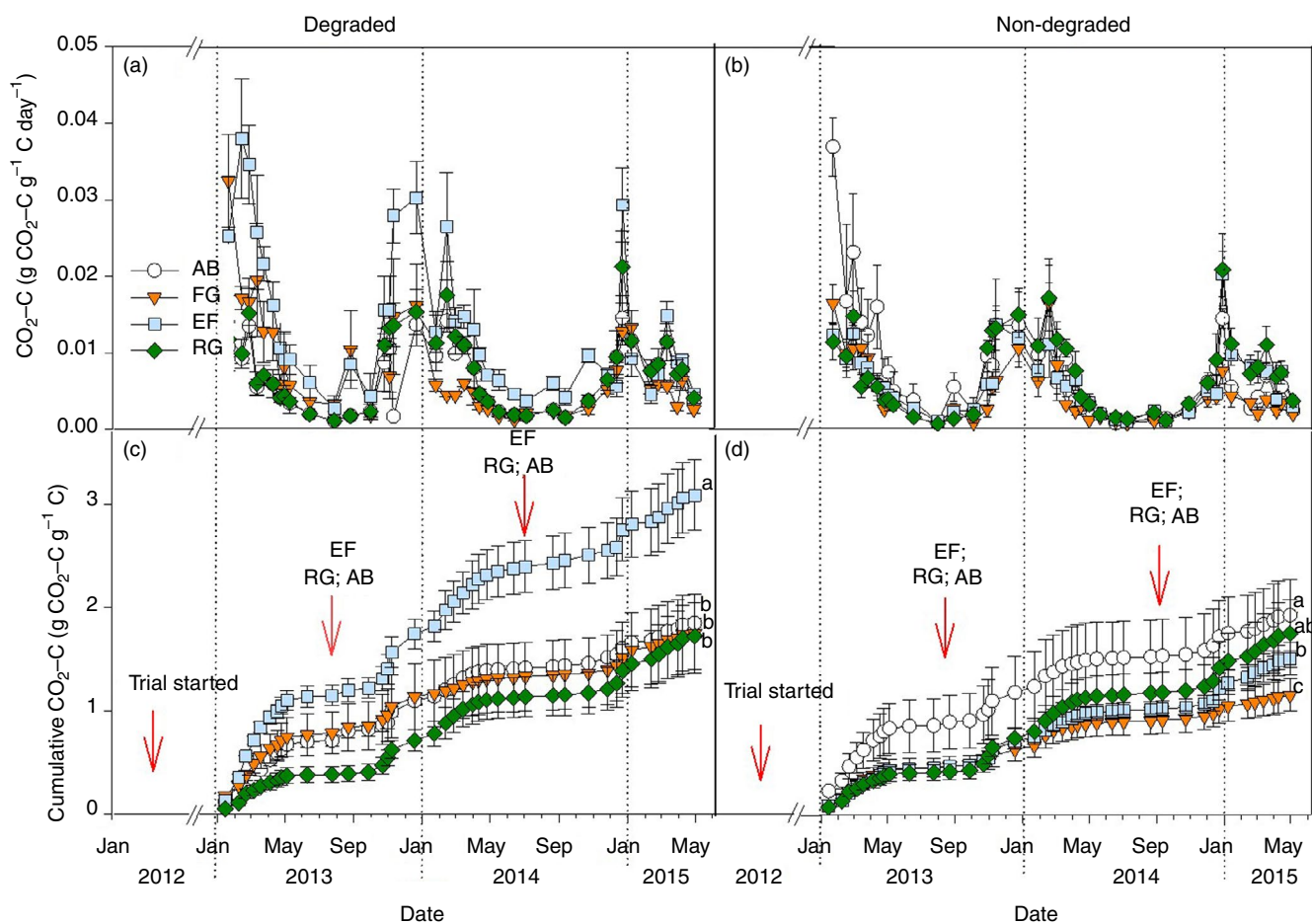


FIGURE 4 Daily and cumulative net CO₂ emissions from rotational grazing (RG), livestock enclosure with NPK fertilizer application (EF), annual burning (AB) and traditional free grazing (FG) in degraded and non-degraded grasslands. Error bars represent the standard error of the mean (N = 9). Different lower-case letters to the right of the final cumulative flux values indicate significant differences between treatments at $p \leq 0.05$

of 2.51 ± 0.28 kg CO₂-C m⁻², which was 36% higher than the average of AB and FG, but only 14% higher than EF (Figure 3e). Likewise, RG recorded the highest cumulative gross CO₂ value under the degraded grassland (Figure 3f).

The daily net CO₂ emissions also varied significantly over time, with more variation between the treatments in summer with less differences under non-degraded compared with the degraded grassland (Figure 4a and b).

TABLE 2 Changes in aboveground biomass (AGB), soil organic carbon stocks (SOCs) and soil bulk density (ρ_b) in the topsoil (0–0.05 m) for the treatments (rotational grazing (RG), livestock enclosure with NPK fertilizer application (EF), annual burning (AB) and traditional free grazing (FG) under degraded grasslands) from season 2012–2013 to 2014–2015 ($N = 9$)

Treatments	AGB (kg m^{-2})				SOCs (kg m^{-2})				ρ_b (g cm^{-3})				
	2012–2013		2014–2015		2012–2013		2014–2015		2012–2013		2014–2015		
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	
RG	0.15 ^a	0.03	0.29 ^a	0.04	0.50 ^a	0.04	0.75 ^a	0.10	50.00 ^b	1.60 ^a	0.02	1.48 ^b	0.30
EF	0.14 ^a	0.01	0.23 ^b	0.06	0.46 ^a	0.03	0.74 ^a	0.04	60.86 ^a	1.55 ^a	0.03	1.50 ^b	0.02
AB	0.13 ^a	0.02	0.14 ^c	0.03	0.44 ^a	0.03	0.43 ^c	0.02	–2.33 ^c	1.60 ^a	0.02	1.58 ^a	0.02
FG	0.13 ^a	0.03	0.15 ^c	0.01	0.50 ^a	0.04	0.50 ^b	0.03	0.00 ^c	1.59 ^a	0.01	1.60 ^a	0.04
average	0.14	0.02	0.20	0.03	0.47	0.03	0.56	0.03	0.00 ^c	1.58	0.02	1.56	0.03

*Mean values in each column accompanied by the same letter and/or letter combination were not significantly different at $p < 0.05$.

Under degraded grasslands, net CO_2 emissions from the EF were the highest than the other treatments, and RG was the lowest in most cases (Figure 4a). However, AB showed significantly greater net CO_2 in 2013 under non-degraded grasslands (Fig. b). Cumulatively, the net CO_2 under degraded grasslands descended in the following order; EF > AB > FG > RG (Figure 4c). In contrast, the order under non-degraded grassland was AB > RG > EF > FG (Figure 4d).

3.3 | Aboveground biomass production, SOCs and soil bulk density

The aboveground biomass production among the treatments was not significantly different at the beginning of the experiment in the growing season of 2012–2013 (Table 2). However, the biomass production values descended in the order RG > EF > AB > FG. The same trend prevailed in the final season (2014–2015), but the values were significantly different except AB and FG treatments. For instance, RG had the most significant value ($0.29 \pm 0.04 \text{ kg m}^{-2}$), while AB and FG had the least significant values. On average, aboveground biomass production increased over the study period by 93%, 64%, 15% and 8% for RG, EF, FG and AB, respectively.

Similarly, SOCs also showed no significant differences in the 2012–2013 season, but significant differences were observed in 2014–2015 with RG ($0.75 \pm 0.10 \text{ kg m}^{-2}$) and EF ($0.74 \pm 0.04 \text{ kg m}^{-2}$) being significantly higher than FG and AB. While EF and RG induced a significant increase in SOCs by 60.86% and 50%, respectively, AB caused a reduction of –2.33% and FG showed no effect after three years of implementation (Table 2). The initial topsoil bulk densities also showed no significant differences among the treatments in the 2012–2013 season. While the soil bulk densities generally declined over the study period, FG showed an increase of 0.06% in the 2014–2015 season. The greatest decrease (–8.12%) was recorded in RG, followed by EF (–3.33%) and lastly AB (–1.26%).

3.4 | Factors controlling soil CO_2 emissions

Overall average gross soil CO_2 emissions increased significantly with aboveground biomass soil temperature and C:N and decreased with the increase in soil C and N (Table 3). Similarly, soil CO_2 emissions decreased with the increase in soil C and N (both content and stocks) and increased mostly with aboveground biomass ($r = 0.94$), C:N and soil temperature under degraded grasslands. However, under non-degraded conditions, CO_2 emissions

were not affected by the C:N but rather by aboveground biomass and soil temperature in a positive direction and soil C and N in a negative direction. The principal component analysis (PCA) shows the multiple correlations between the soil CO₂ emissions from the two grassland degradation levels and the aboveground biomass, soil and weather drivers (Figure 5a and b). Factors 1 and 2 of the first PCA (Figure 5a) accounted for 81% of the total dataset variance with 57% of the total variance represented by factor 1. In this PCA, soil CO₂ showed a strong positive correlation with aboveground biomass and C:N ratio. However, soil C and N showed a negative correlation under degraded grassland. The major factors (1 and 2) of the second PCA (Figure 5b) accounted for 71% of the data set total variance, with 48% of the variation correlated to factor 1. In this PCA, gross CO₂ correlated negatively with topsoil C and N but positively with aboveground biomass.

4 | DISCUSSION

4.1 | Link between aboveground biomass production and soil CO₂ emission

Despite the strong positive effects of soil temperature ($r = 0.6$) on the overall soil CO₂ emissions (CO₂ from degraded and non-degraded grasslands), the study results still supported the hypothesis that higher CO₂ emissions were driven mainly by higher aboveground biomass production

TABLE 3 Coefficients of determination (r) between gross soil CO₂ emissions and soil properties (SOC_C and SOC_S, organic carbon content and stocks; TN_C and TN_S, total nitrogen content and stocks; C:N, carbon: nitrogen; ρ_b , bulk density; SWC, soil water content; and ST, soil temperature), weather conditions (AT, air temperature; P, precipitation) and aboveground biomass (AGB) (N = 28)

Parameters	Gross CO ₂		
	Overall average	Degraded	Non-degraded
SOC _C	-0.85*	-0.87*	-0.78*
SOC _S	-0.85*	-0.86*	-0.82*
TN _C	-0.88*	-0.92*	-0.76*
TN _S	-0.82	-0.84*	-0.82*
C:N	0.53*	0.84*	0.01
ρ_b	0.44	0.68*	0.27
SWC	0.14	0.31	-0.31
ST	0.60*	0.68*	0.64*
AT	0.10	0.12	0.12
P	0.05	0.16	-0.06
AGB	0.91*	0.94*	0.92*

*Statistically significant determinants at $p < 0.05$.

and SOC_S (Tables 2, 3 and Figure 5). Based on the overall average CO₂ emissions, RG emitted much higher gross soil CO₂ than other management practices, which was explained by a combination of higher aboveground biomass production and SOC_S (Table 2). The strong positive correlation between gross soil CO₂ emission and aboveground biomass production ($r = 0.91$; Table 3) agreed with results from other studies (e.g. Bahn et al., 2008; Frank & Dugas, 2001; Wang et al., 2019). Wang et al. (2019) found that aboveground biomass, which is related to belowground biomass, was a critical biotic factor controlling soil respiration in the Loess Plateau, China. In support, Frank and Dugas (2001) reported direct links between gross CO₂ and plant root activities in a semiarid mixed-grass prairie in North Dakota, USA. It is important to note that soil CO₂ emissions emanate from living plant roots and soil fauna (microbial respiration), which decompose the soil organic matter (Adamczyk et al., 2019). Even though the current study did not separate the relative contributions of roots and soil fauna respirations to the total CO₂ emissions, the dominant source was found to be root respiration in the dry grasslands (Balogh et al., 2008; Dugas et al., 1999). However, the contribution of root respiration to the total soil CO₂ varied from 15% to 91%, depending on the climate, crop and soil types, and sampling time during the growing season (Chen et al., 2021; Lee et al., 2003; Wang et al., 2005, 2007).

4.2 | Impact of rotational grazing and fertilizer application on soil CO₂ emissions

The greater gain rate of aboveground biomass under RG and EF (Table 2) than FG could be attributed to nutrient inputs from cattle dung and urine (in the case of RG) and chemical fertilizer application (EF). While the present study reported higher gross soil CO₂ emissions from plots with higher aboveground grass biomass (Tables 1 and 2), some studies reported no significant differences (e.g. Acharya et al., 2012), which they attributed to the effects of relatively higher root biomass in old grasslands. Unlike aboveground parts, roots are not harvested; therefore, old grasslands tend to have relatively higher belowground biomass than new grasslands. Hence, soil CO₂ emissions could be higher in the old grasslands despite their lower aboveground biomass production levels.

At the beginning of the experiment (season 2012–2013), no significant difference was found between the treatments in the aboveground biomass and SOC_S (Table 2). However, three years after treatment implementation, RG and EF increased aboveground biomass by 93% and 64% and SOC_S by 50% and 61%, respectively. In both cases, the nutrient input (from the chemical fertilizer and animal

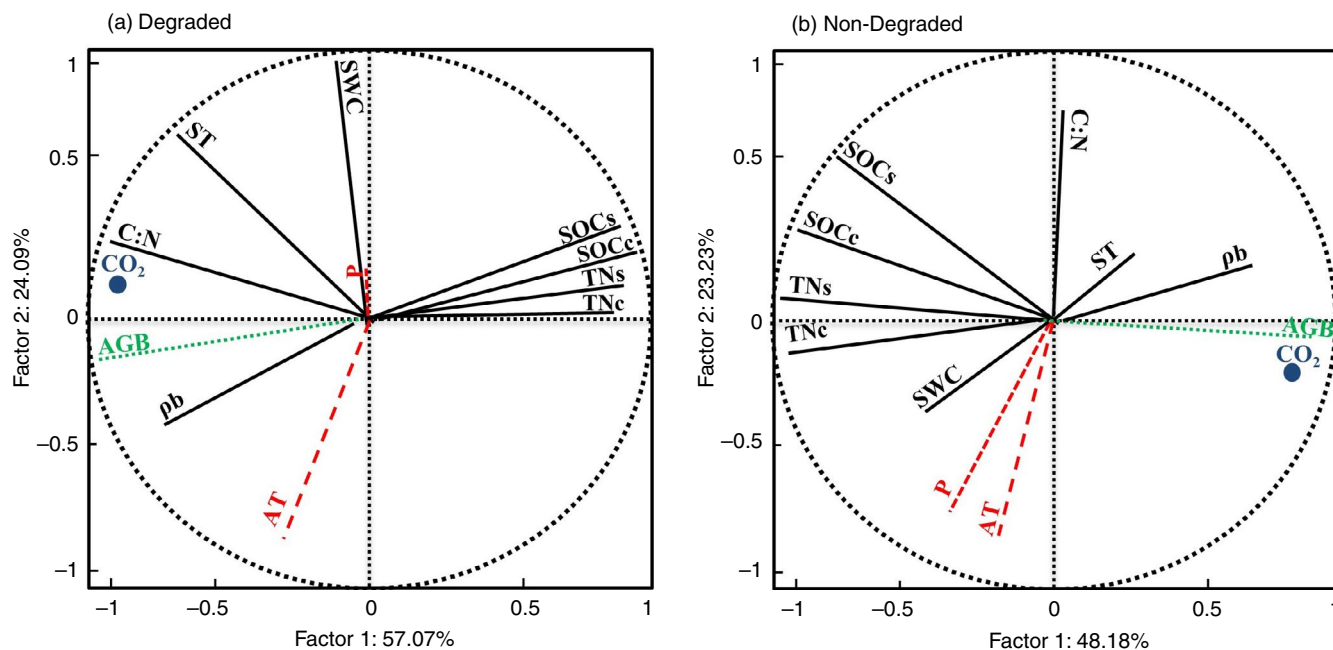


FIGURE 5 Principal component analysis (PCA) scatter diagrams for gross CO₂ emissions and soil properties (SOC_C and SOC_T, organic carbon content and Stocks; TN_C and TN_S, total nitrogen content and stocks; C:N, carbon: nitrogen; ρ_b , bulk density; SWC, soil water content; and ST, soil temperature), weather conditions (AT, air temperature; P, precipitation) and aboveground biomass (AGB) as active variables under degraded (a) and non-degraded (b) grasslands. $N = 28$

excreta) explains such an increase in biomass production. In addition, the long resting period from herbivores in the RG approach allows the recovery of the grasses. Free grazing also slightly increased grass biomass (i.e. 15%) and induced low net CO₂ emissions, probably because animal excreta (dung and urine) input provided soil microbes with readily available nutrients.

Oppositely, the low net CO₂ emissions from RG treatment (Table 1; Figure 4) under degraded grasslands point to a possible persistence of C in soil aggregates under this treatment, suggesting better soil aggregation (Kleber, 2010). As mentioned above, RG is associated with higher belowground biomass production, which promotes better soil aggregation and build-up of SOC_C because of increased C input from the litter and root turnover and possibly root exudates (Baumert et al., 2018; Dignac et al., 2017). Under such conditions, the net positive impact is better soil C protection leading to high recalcitrant C in the soils (Kleber, 2010; Six et al., 2000).

Rotational grazing induced relatively high net soil CO₂ under the non-degraded grassland conditions (Table 1), possibly because of the high input of cattle excreta resulting from the high stocking rate. Animal excreta gave soil microbes readily available nutrients from dung and urine, thus increasing soil microbial activities and microbial respiration without affecting the native soil organic matter (Elhottová et al., 2012; van der Wal et al., 2004). The previous explanation is supported by the negative correlation observed between CO₂ emissions and soil C

and N ($r = -0.85$ and -0.88 for N content and socks, respectively; Table 3). Similarly, higher net soil CO₂ emission from plots treated with mineral fertilizers (EF) than the FG treatment resonates with the notion that chemical fertilizer application increases soil microbial activities, thereby stimulating higher soil CO₂ emissions (Bai et al., 2020; Chen et al., 2014).

4.3 | Impact of annual burning on soil CO₂ emissions

The lack of significant difference in soil CO₂ emission between AB and FG (Table 1), both associated with low aboveground biomass production under degraded grassland (Table 2), suggests that AB might have a low capacity to rehabilitate degraded grasslands. Both grassland management strategies are widely practised in Southern Africa, with AB more popular among commercial livestock farmers, while communal farmers practise FG. In addition to the low capacity to rehabilitate degraded grasslands, grassland burning is also well known to increase soil erosion and nutrient losses (Valkó et al., 2014; Zhao et al., 2019) because of bare patches created and hydrophobicity of soils subjected to intense fire temperatures. The higher net soil CO₂ emission induced by AB compared with FG under non-degraded grassland (Table 1) could be attributed to the increased C:N ratio of grasses, which are the primary nutritional source to soil microbes

(Anderson et al., 2007; Ojima et al., 1994). Thus, soil microbial activities require more energy to decompose low-quality substrate (i.e. litter with high C:N), which might induce more soil respiration under annually burned grassland. However, the similarities in CO₂ emissions between AB and FG under degraded grassland could be explained by the lack of variations in aboveground biomass (Table 2) and reduced grass cover. This might induce equal effects on soil temperature and soil microbial activity resulting in equivalent CO₂ emissions between the treatments. Another possible explanation for the lack of differences in soil CO₂ from AB and FG under degraded grassland could be that degraded soils generally have poor microbial populations because of, for example, lower substrate input and poor soil quality (Qiu et al., 2021; Zhang et al., 2017).

4.4 | Relation between soil properties and soil CO₂ emissions

The higher aboveground biomass production is associated with a higher soil C sequestration rate (Table 3), implying that high soil cover by grass reduces soil water losses via evaporation and topsoil temperature fluctuations (e.g. Abdalla et al., 2016; Bahn et al., 2008; Guntiñas et al., 2012). Soil temperature, which correlated positively to CO₂ emission (Table 3, Figure 5), is an accurate proxy for estimating soil respiration in the absence of water stress (Bahn et al., 2008). In general, soil respiration is driven not only by individual factors but also by their interactions. For example, soil CO₂ fluxes in the topsoil layer highly depend on soil temperature, which is regulated by aboveground biomass, grass cover, season, daytime, substrate inputs and quality (Curiel Yuste et al., 2007; Wang et al., 2019). In support, Abdalla et al. (2016) reported positive correlations between soil CO₂ emission on the one hand, and topsoil moisture and temperature on the other hand in another burnt grassland trial. Similarly, Tao et al. (2016) reported significant correlations between soil respiration and soil temperature (within 2–31°C range), and also with soil moisture (18%–25% range), in urban green lands of China. However, the results are only valid within optimal moisture and temperature conditions for soil microbial activities (Borowik & Wyszowska, 2016). The lack of soil water content effect on soil CO₂ emissions observed in the present study could be because of the fact that soil water content was measured over one growing season.

The significantly positive correlation ($r = 0.84$) between soil CO₂ and C:N ratio under degraded grasslands (Table 2 and Figure 5a) implied strong relation between soil CO₂ and the mature perennial grass leaf-litter quality observed in the degraded plots. Generally, mature grasses and their leaf litter have high C:N ratios, making them

less digestible to microbes (Schimel et al., 1991; Wang & Zheng, 2021). At the same time, mature grasses are more likely to self-shade (Schimel et al., 1991), leading to reductions in below-canopy temperatures and possibly higher soil emission rates. This trend explains the greater net CO₂ observed under degraded than the non-degraded grassland. On the contrary, the negative correlations between soil CO₂ emissions and SOC and N content under degraded grassland (Table 3) point to the limited role of soil C and N content on soil C outputs from soils under degraded grassland conditions. As a final note, rotational grazing has great potential in increasing soil C stocks, reducing soil CO₂ emissions and enhancing land productivity (Abdalla, Mutema, et al., 2021; Chaplot et al., 2016). In addition to its high potential on rehabilitated degraded grass in a short time (e.g. three years) compared with commonly used practices, that is, annual burning and free grazing reduce net CO₂ emissions. On the contrary, annual burning showed low efficiency for grassland rehabilitation and it is associated with increased soil CO₂ emissions and other greenhouse gases emissions because of biomass burning (Abdalla, Chivenge, et al., 2021; Prospero et al., 2020; Ramo et al., 2021). This makes rotational grazing an ideal option for increasing smallholder farmers' resilience to climate change by increasing grassland sustainability and food security. Rotational grazing could be a beneficial practice to mitigate climate change not only in the study site but rather in the whole of sub-Saharan Africa, where the total burned area represents 70% of the global burned area and accounts for 14% of the global CO₂ emitted from fossil fuel burning (Ramo et al., 2021).

However, being unable to separate the original source of the total soil CO₂ emission (root-derived CO₂ and SOM-derived CO₂) in the present study is considered a limitation of the current study. Another limiting factor is the experimental design because of the limited space and the associated cost. Nevertheless, the current study provided insights on the potential importance of using small-scale trials (with homogeneous soil) to detect initial differences in soil C sequestration rates using cost-effective grassland rehabilitation practices that the local community can easily adopt. Hewins et al. (2018) demonstrated that greater sampling intensities on small-scale trials located on homogeneous soils (i.e. 15*30 m) were important to detect (positive or negative) changes in soil C sequestration in Alberta, Canada. They justified the use of small-scale SOC studies under grazing practices because of practical (e.g. logistical and cost) limitations to maximizing sample sizes. However, adequate sample size and number of spatial replications are crucial, given that most of these results are used in estimating global C budgets. The spatial variation of soil properties is an essential factor with a significant influence on the size of trial plots. The observed

inconsistent effects of grazing in many studies are because of varying soil types and properties (Derner & Schuman, 2007; Hewins et al., 2018; Hillenbrand et al., 2019).

5 | CONCLUSIONS

Three main conclusions were drawn from the study results: (i) degraded grassland soils submitted to rotational grazing and chemical fertilizer application emitted higher gross soil CO₂ emissions than annual burning and free grazing of grass; (ii) rotational grazing enhanced above-ground biomass production and topsoil SOC_s in degraded grasslands with significantly lower net soil CO₂ emission than other rehabilitation practices; and (iii) under non-degraded grasslands, soil CO₂ emissions were mainly driven by weather conditions, that is precipitation, air temperature and associated soil parameters such as soil temperature and water content. The study results suggested that rotational grazing could be a good option for rehabilitating degraded grasslands because it enhanced above-ground biomass production and SOC_s with low net soil CO₂ emissions, indicating greater soil C protection. The positive effect of rotational grazing results from the interaction between the C inputs associated with the high stocking rate and the long resting periods. The more extended the resting periods, the better for grass and soil recovery; however, farmers incur opportunity costs by resting plots for a long time as that land will be idle. However, further research using true replications to confirm these results through long-term trials under different resting periods, stocking rates and environmental conditions is still required. Research is also needed to discriminate the sources of soil CO₂, that is root-derived CO₂ and the CO₂ resulting from soil organic matter decomposition using advanced isotope labelling techniques.

ACKNOWLEDGEMENTS

The authors are grateful to the Water Research Commission (WRC) for funding this research (project number K5/2266). We also would like to thank Alexander von Humboldt Foundation for financially supporting the first author during the data analysis and manuscript writing. The authors also acknowledged the Department of Soil Science, University of KwaZulu-Natal, Agricultural Research Council-Agricultural Engineering; the Institut de Recherche pour le Développement (IRD); and the chair of Agroecology, BayCEER, University of Bayreuth, for providing facilities and support. Special thanks to the Potshini community for allowing us to experiment on their land. Open access funding enabled and organized by ProjektDEAL.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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How to cite this article: Abdalla, K., Mutema, M., Chivenge, P., Everson, C., & Chaplot, V. (2022). Grassland rehabilitation significantly increases soil carbon stocks by reducing net soil CO₂ emissions. *Soil Use and Management*, 38, 1250–1265. <https://doi.org/10.1111/sum.12790>