Received: 11 October 2021 | Revised: 19 January 2022 | Accepted: 26 January 2022
DOI: 10.1111/sum.12790

Soil Use and Management
BRITISH SOCIETY OF SOIL SCIENCE
WILEY

RESEARCH ARTICLE

Grassland rehabilitation significantly increases soil carbon stocks by reducing net soil CO_2 emissions

Khatab Abdalla[1,2,3] | Macdex Mutema[4] | Pauline Chivenge[5,3] | Colin Everson[3] | Vincent Chaplot[6,3]

[1]Chair of Agroecology, Bayreuth Centre for Ecology and Environmental Research, University of Bayreuth, Bayreuth, Germany

[2]Environment, Natural Resources and Desertification Research Institute, National Centre for Research, Khartoum, Sudan

[3]School of Agricultural, Earth & Environmental Sciences, University of KwaZulu-Natal, Pietermaritzburg, South Africa

[4]Agricultural Research Council-Agricultural Engineering, Pretoria, South Africa

[5]African Plant Nutrition Institute UM6P Experimental Farm, Benguérir, Morocco

[6]Institut de Recherche pour le Développement (IRD), Laboratoire d'Océanographie et du Climat (LOCEAN), UMR 6159 CNRS/IRD/UPMC/MNHN, Paris Cedex 05, France

Correspondence
Khatab Abdalla, University of Bayreuth, Universitätsstraße 30, 95440, Bayreuth, Germany.
Emails: Khatab.Abdalla@Uni-Bayreuth.de; Khatab.hassan58@gmail.com

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 CO_2 emissions is still uncertain. Therefore, this study aimed at quantifying the impact of grassland rehabilitation on soil CO_2 emissions in a degraded grassland, South Africa. Commonly used rehabilitation practices were considered, that is rotational grazing (RG), livestock exclosure with fertilizer application (EF) and annual burning (AB), all being compared with traditional free grazing (FG). A total of 2880 in situ measurements of CO_2 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 CO_2 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 CO_2 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

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 SOCs depletion (Abdalla et al., 2018; Chaplot et al., 2016; Dlamini et al., 2014). However, SOCs 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 CO_2 by 30% compared with unburned grass in South Africa. Such an increase in CO_2 emissions from burned grassland soil was explained by soil aggregate instability (Abdalla, Chivenge, et al., 2021). Relatively, low differences in soil CO_2 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 CO_2 emission after compound fertilizer application in southern China.

In their latest global synthesis study, Conant et al. (2017) reported an increase in topsoil SOCs 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 forge production as a proxy of C inputs, promoting high SOCs (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 deleted SOCs.

The RG practice in this study was modified from Savory and Parsons (1980), which has rarely been investigated in terms of soil CO_2 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 SOCs 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 SOCs 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 SOCs by an average of 12.4 g C m^{-2} 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 SOCs induced by RG (Abdalla, Mutema, et al., 2021; Chaplot et al., 2016) would result in high gross soil CO_2 emissions (CO_2-C per m^2), but low net soil CO_2 emissions (CO_2-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°21′39.87″E; 28°48′40.48″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 1.86–5.86 $cmol_c$ 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).

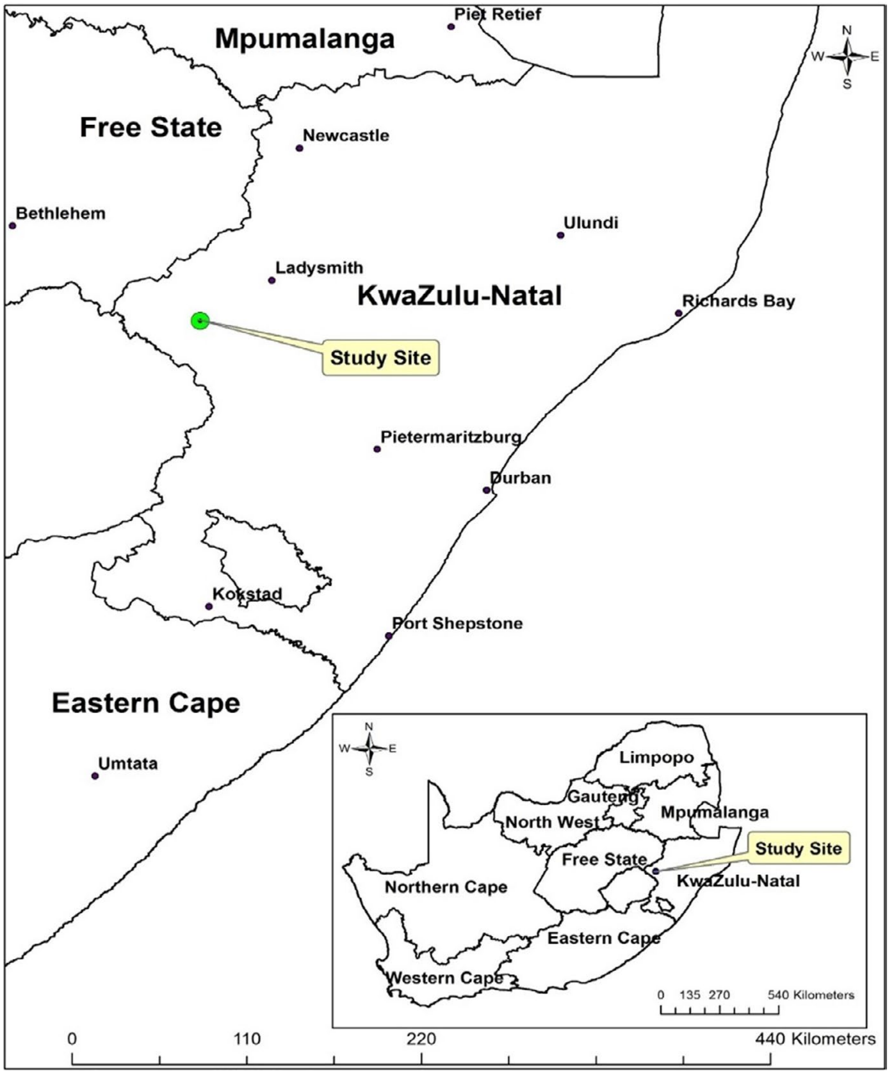
Grassland degradation was defined as the reduction in the capacity of grasslands to deliver their key ecosystem functions because of biodiversity and vegetation cover losses resulting from mismanagements such as overgrazing (FAO, 2011; Wick et al., 2016). Poor management of grasslands such as overgrazing 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 m × 50 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 m × 50 m), each showing all the degradation levels but with similar soil properties and subjected to different management practices to evaluate their effect on SOCs (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 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-m^2 (1 m × 1 m) areas were demarcated and used for soil CO_2 measurements, soil sampling and aboveground biomass to minimize possible variations resulting from different soil properties.

2.3 | Soil CO_2 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 tuffs within the demarcated 1-m^2 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 CO_2 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 CO_2 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 cm^3 and a surface area of 71.6 cm^2 was inserted 0.02 m into the PVC collar for each measurement.

The CO_2 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 CO_2 emission to the atmosphere, measured using the LI-COR 6400, was converted to gross CO_2 by dividing the measured CO_2 emission by the surface area of the PVC collar and expressed as g CO_2-C per unit of soil surface area. The net soil CO_2 emissions relative to SOCs, that is g CO_2-C per g of soil C, were an indirect measurement of SOC stabilization (Abdalla et al., 2018; Chaplot et al., 2015). The net CO_2 emissions were calculated as follows:

$$\text{Net soil } CO_2 \text{ emissions} = \frac{\text{Gross soil } CO_2 \text{ emissions}}{\text{SOCs}} \quad (1)$$

where SOCs 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 m^2. 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 ρb = 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 m^2. 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 (SOCs) were calculated following the equation by Batjes (2014):

$$\text{SOCs} = \text{SOCs} \times \text{Pb} \times \text{T}\left(1 - \frac{\text{Pf}}{100}\right)\text{b} \quad (3)$$

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); ρb = 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 (TNs) were computed following the same equation by replacing SOCc by TN content (TNc).

Soil bulk density and SOCs 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 SOCs 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 CO_2 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 m^2 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 (kg m^{-2} $year^{-1}$). 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 ± 0.21°C. The lowest average daily air temperature was recorded (13 ± 0.03°C) in June of both years, while the highest (38°C) occurred in September 2014. The average daily air temperature of 33 ± 0.27°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 AB > FG > RG > 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

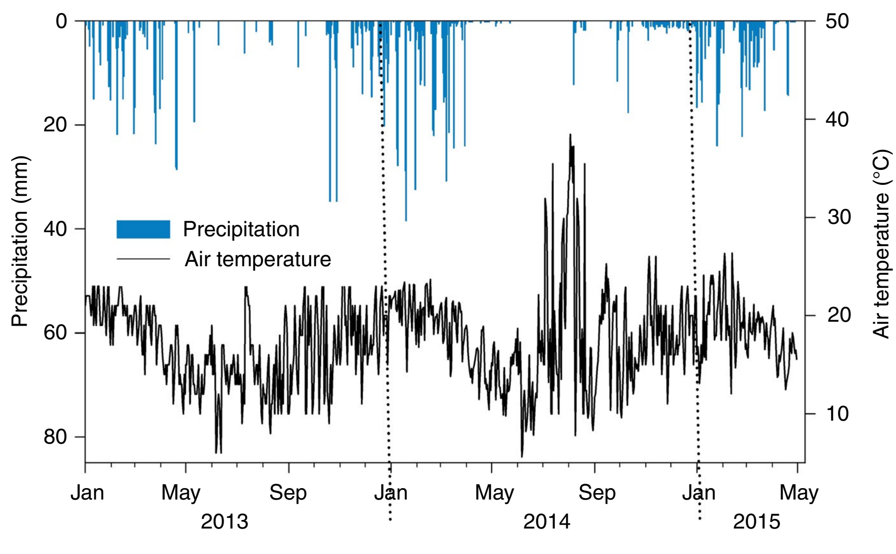


FIGURE 2 Daily precipitation and average air temperature during the study time

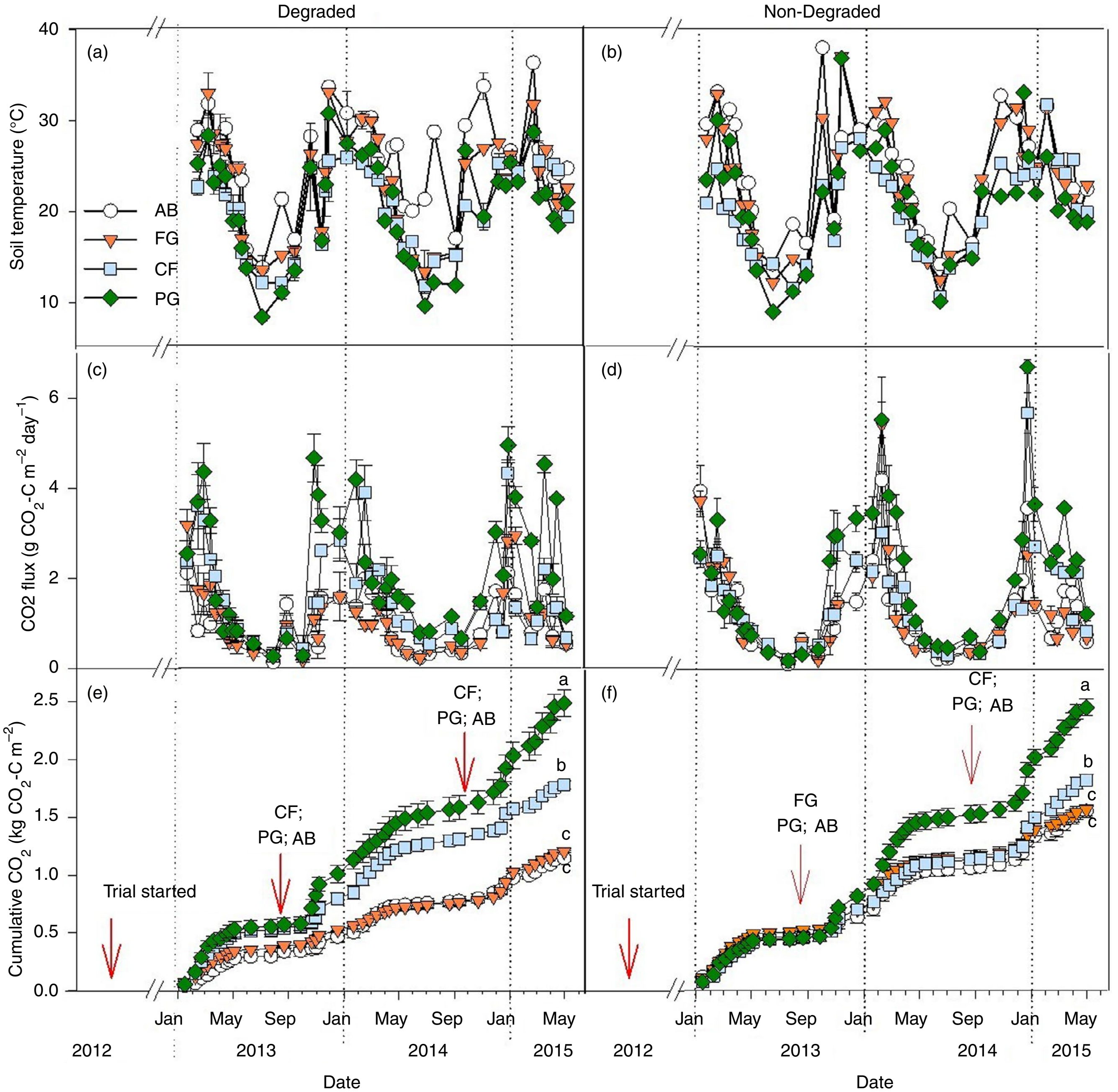


FIGURE 3 Soil temperature at 0–5 cm (a and b), daily (c and d) and cumulative (e and f) gross CO_2 emissions from rotational grazing (RG), livestock exclosure with NPK fertilizer application (EF), annual burn (AB) and traditional free grazing (FG) under degraded and non-degraded grasslands. Error bars 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 $p \le 0.05$

emissions than the non-degraded one in all treatments, except in RG. On the contrary, the overall average net CO_2 emissions (per g of C) were highest in EF and lowest in FG (Table 1). Under the degraded grasslands, RG had the lowest net CO_2 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 CO_2, 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 CO_2 sampling and their interactions significantly affect both gross and net CO_2 emissions from the study site (Table S1). The daily gross CO_2 emissions varied greatly with season and treatment under both degradation levels, mainly in summers, implying strong air temperature and precipitation influences on the soil CO_2 emissions (Figure 3c and d). The RG and EF treatments had higher cumulative gross CO_2 than FG and AB in the degraded grasslands, with a final cumulative value in RG

TABLE 1 Mean ± standard error (SE) of soil CO_2 emissions (gross and net CO_2) for the rehabilitation treatments (rotational grazing (RG), livestock exclosure 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 $p < 0.05$.

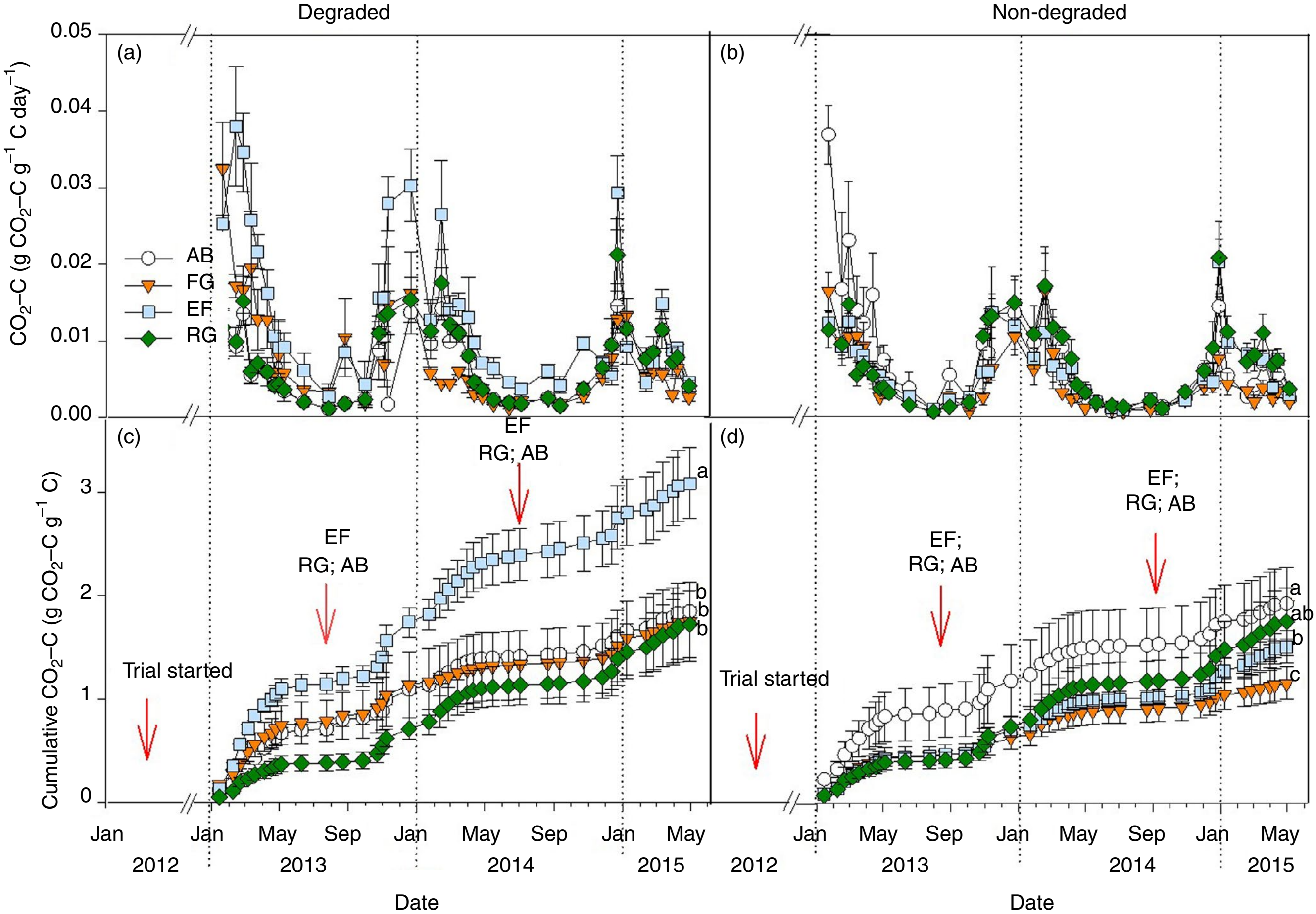


FIGURE 4 Daily and cumulative net CO_2 emissions from rotational grazing (RG), livestock exclosure with NPK fertilizer application (EF), annual burning (AB) and traditional free grazing (FG) in degraded and non-degraded grasslands. Error bars represent the standard error of the mean (N = 9). Different 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 CO_2–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 CO_2 value under the degraded grassland (Figure 3f).

The daily net CO_2 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 (ρb) in the topsoil (0–0.05 m) for the treatments (rotational grazing (RG), livestock exclosure 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 ($kg\ m^{-2}$)					ρb ($g\ cm^{-3}$)				
	2012–2013		2014–2015			2012–2013		2014–2015			2012–2013		2014–2015		
	Mean	SE	Mean	SE	% change	Mean	SE	Mean	SE	% change	Mean	SE	Mean	SE	% change
RG	0.15^a	0.03	0.29^a	0.04	93.33^a	0.50^a	0.04	0.75^a	0.10	50.00^b	1.60^a	0.02	1.48^b	0.30	-8.12^a
EF	0.14^a	0.01	0.23^b	0.06	64.29^b	0.46^a	0.03	0.74^a	0.04	60.86^a	1.55^a	0.03	1.50^b	0.02	-3.33^b
AB	0.13^a	0.02	0.14^c	0.03	7.69^d	0.44^a	0.03	0.43^c	0.02	-2.33^c	1.60^a	0.02	1.58^a	0.02	-1.26^c
FG	0.13^a	0.03	0.15^c	0.01	15.38^c	0.50^a	0.04	0.50^b	0.03	0.00^c	1.59^a	0.01	1.60^a	0.04	0.06^d
average	0.14	0.02	0.20	0.03		0.47	0.03	0.56	0.03		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 ± 0.04 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 ± 0.10 kg m^{-2}) and EF (0.74 ± 0.04 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 CO_2 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 CO_2 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 CO_2 correlated negatively with topsoil C and N but positively with aboveground biomass.

4 | DISCUSSION

4.1 | Link between aboveground biomass production and soil CO_2 emission

Despite the strong positive effects of soil temperature ($r = 0.6$) on the overall soil CO_2 emissions (CO_2 from degraded and non-degraded grasslands), the study results still supported the hypothesis that higher CO_2 emissions were driven mainly by higher aboveground biomass production and SOCs (Tables 2, 3 and Figure 5). Based on the overall average CO_2 emissions, RG emitted much higher gross soil CO_2 than other management practices, which was explained by a combination of higher aboveground biomass production and SOCs (Table 2). The strong positive correlation between gross soil CO_2 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 CO_2 and plant root activities in a semiarid mixed-grass prairie in North Dakota, USA. It is important to note that soil CO_2 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 CO_2 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 CO_2 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).

TABLE 3 Coefficients of determination (r) between gross soil CO_2 emissions and soil properties (SOC_C and SOCs, organic carbon content and stocks; TNc and TNs, total nitrogen content and stocks; C:N, carbon: nitrogen; ρb, bulk density; SWC, soil water content; and ST, soil temperature), weather conditions (AT, air temperature; P, precipitation) and aboveground biomass (AGB) (N = 28)

	Gross CO_2		
Parameters	Overall average	Degraded	Non-degraded
SOC_C	−0.85*	−0.87*	−0.78*
SOC_s	−0.85*	−0.86*	−0.82*
TNc	−0.88*	−0.92*	−0.76*
TNs	−0.82	−0.84*	−0.82*
C:N	0.53*	0.84*	0.01
ρb	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 $p < 0.05$.

4.2 | Impact of rotational grazing and fertilizer application on soil CO_2 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 CO_2 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 CO_2 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 SOCs (Table 2). However, three years after treatment implementation, RG and EF increased aboveground biomass by 93% and 64% and SOCs by 50% and 61%, respectively. In both cases, the nutrient input (from the chemical fertilizer and animal

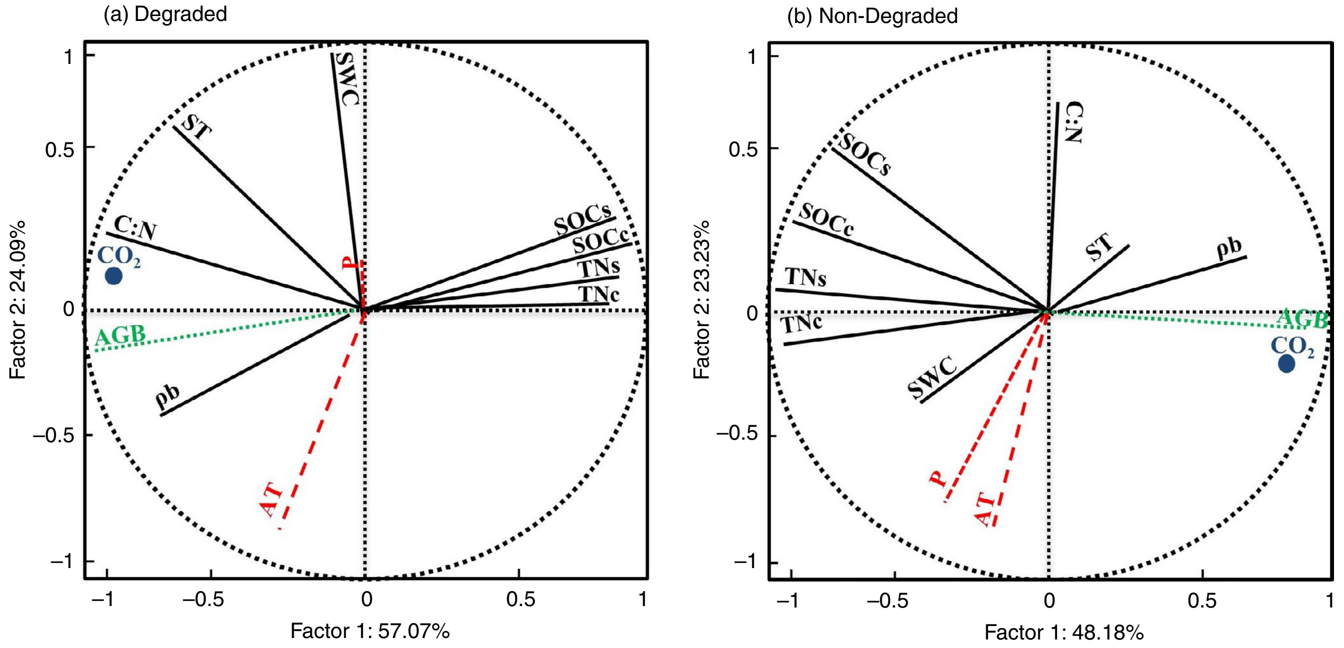


FIGURE 5 Principal component analysis (PCA) scatter diagrams for gross CO_2 emissions and soil properties (SOC_C and SOCs, organic carbon content and Stocks; TNc and TNs, total nitrogen content and stocks; C:N, carbon: nitrogen; ρb, bulk density; SWC, soil water content; and ST, soil temperature), weather 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 CO_2 emissions, probably because animal excreta (dung and urine) input provided soil microbes with readily available nutrients.

Oppositely, the low net CO_2 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 SOCs 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 CO_2 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 CO_2 emissions and soil C and N ($r = -0.85$ and -0.88 for N content and socks, respectively; Table 3). Similarly, higher net soil CO_2 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 CO_2 emissions (Bai et al., 2020; Chen et al., 2014).

4.3 | Impact of annual burning on soil CO_2 emissions

The lack of significant difference in soil CO_2 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 CO_2 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 CO_2 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 CO_2 emissions between the treatments. Another possible explanation for the lack of differences in soil CO_2 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 CO_2 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 CO_2 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 CO_2 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 CO_2 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 & Wyszkowska, 2016). The lack of soil water content effect on soil CO_2 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 CO_2 and C:N ratio under degraded grasslands (Table 2 and Figure 5a) implied strong relation between soil CO_2 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 CO_2 observed under degraded than the non-degraded grassland. On the contrary, the negative correlations between soil CO_2 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 CO_2 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 CO_2 emissions. On the contrary, annual burning showed low efficiency for grassland rehabilitation and it is associated with increased soil CO_2 emissions and other greenhouse gases emissions because of biomass burning (Abdalla, Chivenge, et al., 2021; Prosperi 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 CO_2 emitted from fossil fuel burning (Ramo et al., 2021).

However, being unable to separate the original source of the total soil CO_2 emission (root-derived CO_2 and SOM-derived CO_2) 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 CO_2 emissions than annual burning and free grazing of grass; (ii) rotational grazing enhanced aboveground biomass production and topsoil SOCs in degraded grasslands with significantly lower net soil CO_2 emission than other rehabilitation practices; and (iii) under non-degraded grasslands, soil CO_2 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 aboveground biomass production and SOCs with low net soil CO_2 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 CO_2, that is root-derived CO_2 and the CO_2 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.

ORCID

Khatab Abdalla https://orcid.org/0000-0003-2513-8826

REFERENCES

Abberton, M., Conant, R., & Batello, C. (2010). *Grassland carbon sequestration: management, policy and economics Proceedings of the Workshop on the role of grassland carbon sequestration in the mitigation of climate change*. Plant Production and Protection Division Food and Agriculture Organization of the United Nations (FAO). Integrated Crop Management Vol. 11–2010, 342.

Abdalla, K., Chivenge, P., Ciais, P., & Chaplot, V. (2021). Long-term (64 years) annual burning lessened soil organic carbon and nitrogen content in a humid subtropical grassland. *Global Change Biology*, *27*, 6436–6453. https://doi.org/10.1111/gcb.15918

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. https://doi.org/10.1016/j.geoderma.2016.07.009

Abdalla, K., Mutema, M., Chivenge, P., & Chaplot, V. (2021). Controlled grazing of maize residues increased carbon sequestration in no-tillage system: a case of a smallholder farm in South Africa. *Agronomy*, *11*, 1421. https://doi.org/10.3390/agronomy11071421

Abdalla, K., Mutema, M., Chivenge, P., Everson, C., & Chaplot, V. (2018). Grassland degradation significantly enhances soil CO_2 emission. *Catena*, *167*, 284–292. https://doi.org/10.1016/j.catena.2018.05.010

Acharya, B. S., Rasmussen, J., & Eriksen, J. (2012). Grassland carbon sequestration and emissions following cultivation in a mixed crop rotation. *Agriculture, Ecosystems & Environment*, *153*, 33–39. https://doi.org/10.1016/j.agee.2012.03.001

Adamczyk, B., Sietiö, O.-M., Straková, P., Prommer, J., Wild, B., Hagner, M., Pihlatie, M., Fritze, H., Richter, A., & Heinonsalo, J. (2019). Plant roots increase both decomposition and stable organic matter formation in boreal forest soil. *Nature Communications*, *10*, 3982. https://doi.org/10.1038/s41467-019-11993-1

Anderson, T. M., Ritchie, M. E., Mayemba, E., Eby, S., Grace, J. B., & McNaughton, S. J. (2007). Forage nutritive quality in the serengeti ecosystem: the roles of fire and herbivory. *American Naturalist*, *170*, 343–357. https://doi.org/10.1086/520120

Bahn, M., Rodeghiero, M., Anderson-Dunn, M., Dore, S., Gimeno, C., Drösler, M., Williams, M., Ammann, C., Berninger, F., Flechard, C., Jones, S., Balzarolo, M., Kumar, S., Newesely, C., Priwitzer, T., Raschi, A., Siegwolf, R., Susiluoto, S., Tenhunen, J., … Cernusca, A. (2008). Soil respiration in European grasslands in relation to climate and assimilate supply. *Ecosystems*, *11*, 1352–1367. https://doi.org/10.1007/s10021-008-9198-0

Bai, Y.-C., Chang, Y.-Y., Hussain, M., Lu, B., Zhang, J.-P., Song, X.-B., Lei, X.-S., & Pei, D. (2020). Soil chemical and microbiological properties are changed by long-term chemical fertilizers that limit ecosystem functioning. *Microorganisms*, *8*, 694. https://doi.org/10.3390/microorganisms8050694

Balogh, J., Biro, M., & Pintér, K. (2008). Root respiration in dry grassland. *Cereal Research Communications*, *36*, 355–358.

Batjes, N. H. (2014). Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, *65*, 10–21. https://doi.org/10.1111/ejss.12114_2

Baumert, V. L., Vasilyeva, N. A., Vladimirov, A. A., Meier, I. C., Kögel-Knabner, I., & Mueller, C. W. (2018). Root exudates induce soil macroaggregation facilitated by fungi in subsoil. *Frontiers in Environmental Science*, *6*, 140. https://doi.org/10.3389/fenvs.2018.00140

Borowik, A., & Wyszkowska, J. (2016). Impact of temperature on the biological properties of soil. *International Agrophysics*, *30*, 1–8. https://doi.org/10.1515/intag-2015-0070

Castaldi, S., de Grandcourt, A., Rasile, A., Skiba, U., & Valentini, R. (2010). CO_2, CH_4 and N_2O fluxes from soil of a burned grassland in Central Africa. *Biogeosciences*, *7*, 3459–3471. https://doi.org/10.5194/bg-7-3459-2010

Chaplot, V. (2021). Evidences of plants' impact on land degradation and climate change: An urgent call for new multidisciplinary research. *Geoderma*, *392*, 114984. https://doi.org/10.1016/j.geoderma.2021.114984

Chaplot, V., Abdalla, K., Alexis, M., Bourennane, H., 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. https://doi.org/10.1016/j.agee.2015.02.001

Chaplot, V., Dlamini, P., & Chivenge, P. (2016). Potential of grassland rehabilitation through high density-short duration grazing to sequester atmospheric carbon. *Geoderma*, *271*, 10–17. https://doi.org/10.1016/j.geoderma.2016.02.010

Chen, M., Chen, X., Hu, Z., Fan, T., Zhang, S., & Liu, Y. (2021). Contribution of root respiration to total soil respiration during non-growing season in mine reclaimed soil with different covering-soil thicknesses. *International Journal of Coal Science & Technology*, *8*, 1130–1137. https://doi.org/10.1007/s40789-020-00402-4

Chen, R., Senbayram, M., Blagodatsky, S., Myachina, O., Dittert, K., Lin, X., Blagodatskaya, E., & Kuzyakov, Y. (2014). Soil C and N availability determine the priming effect: microbial N mining and stoichiometric decomposition theories. *Global Change Biology*, *20*, 2356–2367. https://doi.org/10.1111/gcb.12475

Conant, R. T., Cerri, C. E. P., Osborne, B. B., & Paustian, K. (2017). Grassland management impacts on soil carbon stocks: a new synthesis. *Ecological Applications*, *27*, 662–668. https://doi.org/10.1002/eap.1473

Conant, R. T., & Paustian, K. (2002). Potential soil carbon sequestration in overgrazed grassland ecosystems. *Global Biogeochemical Cycles*, *16*, 90-1–90-9. https://doi.org/10.1029/2001GB001661

Conant, R. T., Paustian, K., & Elliott, E. T. (2001). Grassland management and conversion into grassland: effects on soil carbon. *Ecological Applications*, *11*, 13.

Curiel Yuste, J., Baldocchi, D. D., Gershenson, A., Goldstein, A., Misson, L., & Wong, S. (2007). Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture. *Global Change Biology*, *13*, 2018–2035. https://doi.org/10.1111/j.1365-2486.2007.01415.x

DeLacy, I. H., Basford, K. E., Cooper, M., Bull, J. K., & McLaren, C. G. (1996). Analysis of multi-environment trials-a historical perspective. In M. Cooper, & G. L. Hammer (Eds.), *Plant adaptation and crop improvement* (pp. 39–124). CAB Int.

Derner, J. D., & Schuman, G. E. (2007). Carbon sequestration and rangelands: A synthesis of land management and precipitation effects. *Journal of Soil and Water Conservation*, *62*, 77–85.

Díaz de Otálora, X., Epelde, L., Arranz, J., Garbisu, C., Ruiz, R., & Mandaluniz, N. (2021). Regenerative rotational grazing management of dairy sheep increases springtime grass production and topsoil carbon storage. *Ecological Indicators*, *125*, 107484. https://doi.org/10.1016/j.ecolind.2021.107484

Dignac, M.-F., Derrien, D., Barré, P., Barot, S., Cécillon, L., Chenu, C., Chevallier, T., Freschet, G. T., Garnier, P., Guenet, B., Hedde, M., Klumpp, K., Lashermes, G., Maron, P.-A., Nunan, N., Roumet, C., & Basile-Doelsch, I. (2017). Increasing soil carbon storage: mechanisms, effects of agricultural practices and proxies. A review. *Agronomy for Sustainable Development*, *37*, 14. https://doi.org/10.1007/s13593-017-0421-2

Dlamini, P., Chivenge, P., & Chaplot, V. (2016). Overgrazing decreases soil organic carbon stocks the most under dry climates and low soil pH: A meta-analysis shows. *Agriculture, Ecosystems & Environment*, *221*, 258–269. https://doi.org/10.1016/j.agee.2016.01.026

Dlamini, P., Chivenge, P., Manson, A., & Chaplot, V. (2014). Land degradation impact on soil organic carbon and nitrogen stocks of sub-tropical humid grasslands in South Africa. *Geoderma*, *235–236*, 372–381. https://doi.org/10.1016/j.geoderma.2014.07.016

Dlamini, P., Orchard, C., Jewitt, G., Lorentz, S., Titshall, L., & Chaplot, V. (2011). Controlling factors of sheet erosion under degraded grasslands in the sloping lands of KwaZulu-Natal, South Africa. *Agricultural Water Management*, Smallholder Systems Innovations for Integrated Watershed Management in Sub-Saharan Africa, *98*, 1711–1718. https://doi.org/10.1016/j.agwat.2010.07.016

Du, Z., Xie, Y., Hu, L., Hu, L., Xu, S., Li, D., Wang, G., & Fu, J. (2014). Effects of fertilization and clipping on carbon, nitrogen storage, and soil microbial activity in a natural grassland in Southern China. *PLoS One*, *9*, e99385. https://doi.org/10.1371/journal.pone.0099385

Dugas, W. A., Heuer, M. L., & Mayeux, H. S. (1999). Carbon dioxide fluxes over bermudagrass, native prairie, and sorghum. *Agricultural & Forest Meteorology*, *93*, 121–139. https://doi.org/10.1016/S0168-1923(98)00118-X

Elhottová, D., Koubová, A., Šimek, M., Cajthaml, T., Jirout, J., Esperschuetz, J., Schloter, M., & Gattinger, A. (2012). Changes in soil microbial communities as affected by intensive cattle husbandry. *Applied Soil Ecology*, *58*, 56–65. https://doi.org/10.1016/j.apsoil.2012.03.009

Everson, C. S., & Tainton, N. M. (1984). The effect of thirty years of burning on the Highland Sourveld of Natal. *Journal of the Grassland Society of Southern Africa*, *1*, 15–20. https://doi.org/10.1080/02566702.1984.9647976

FAO, (2011). *Land degradation assessment in drylands: Manual for local level assessment of land degradation and sustainable land management: Part 2, field methodology and tools*. Authors: Bunning, S., McDonagh, J., Rioux, J. Editor: Anne Woodfine. Food and Agriculture Organization of the United Nations.

Frank, A. B., & Dugas, W. A. (2001). Carbon dioxide fluxes over a northern, semiarid, mixed-grass prairie. *Agricultural & Forest*

Meteorology, *108*, 317–326. https://doi.org/10.1016/S0168-1923(01)00238-6

Friedlingstein, P., Jones, M. W., O'Sullivan, M., Andrew, R. M., Bakker, D. C. E., Hauck, J., Le Quéré, C., Peters, G. P., Peters, W., Pongratz, J., Sitch, S., Canadell, J. G., Ciais, P., Jackson, R. B., Alin, S. R., Anthoni, P., Bates, N. R., Becker, M., Bellouin, N., … Zeng, J. (2021). Global Carbon Budget 2021. *Earth System Science Data Discussions*. [preprint]. https://doi.org/10.5194/essd-2021-386

Fynn, R. (2008). Savory insights-is rangeland science due for a paradigm shift? *Grassroots, the Newsletter of the Grassland Society of Southern Africa*, *8*, 26–38.

Fynn, R. W. S., Morris, C. D., & Edwards, T. J. (2004). Effect of burning and mowing on grass and forb diversity in a long-term grassland experiment. *Applied Vegetation Science*, *7*, 1–10. https://doi.org/10.1111/j.1654-109X.2004.tb00589.x

Grossman, R. B., & Reinsch, T. G. (2002). Bulk density and linear extensibility. In J. H. Dane, & G. C. Topp (Eds.), *methods of soil analysis: Part 4* (pp. 201–225). Physical methods. Soil Science Society of America Inc.

Guntiñas, M. E., Leirós, M. C., Trasar-Cepeda, C., & Gil-Sotres, F. (2012). Effects of moisture and temperature on net soil nitrogen mineralization: A laboratory study. *European Journal of Soil Biology*, *48*, 73–80. https://doi.org/10.1016/j.ejsobi.2011.07.015

Hewins, D. B., Lyseng, M. P., Schoderbek, D. F., Alexander, M., Willms, W. D., Carlyle, C. N., Chang, S. X., & Bork, E. W. (2018). Grazing and climate effects on soil organic carbon concentration and particle-size association in northern grasslands. *Scientific Reports*, *8*, 1336. https://doi.org/10.1038/s41598-018-19785-1

Hillenbrand, M., Thompson, R., Wang, F., Apfelbaum, S., & Teague, R. (2019). Impacts of holistic planned grazing with bison compared to continuous grazing with cattle in South Dakota shortgrass prairie. *Agriculture, Ecosystems & Environment*, *279*, 156–168. https://doi.org/10.1016/j.agee.2019.02.005

IIUS-WRB (2014). *World reference base for soil resources 2014: international soil classification system for naming soils and creating legends for soil maps*. FAO.

Jacobo, E. J., Rodríguez, A. M., Bartoloni, N., & Deregibus, V. A. (2006). Rotational grazing effects on rangeland vegetation at a farm scale. *Rangeland Ecology & Management*, *59*, 249–257. https://doi.org/10.2111/05-129R1.1

Jia, X., Shao, M., & Wei, X. (2012). Responses of soil respiration to N addition, burning and clipping in temperate semiarid grassland in northern China. *Agricultural & Forest Meteorology*, *166–167*, 32–40. https://doi.org/10.1016/j.agrformet.2012.05.022

Keller, A. B., Borer, E. T., Collins, S. L., DeLancey, L. C., Fay, P. A., Hofmockel, K. S., Leakey, A. D. B., Mayes, M. A., Seabloom, E. W., Walter, C. A., Wang, Y., Zhao, Q., & Hobbie, S. E. (2021). Soil carbon stocks in temperate grasslands differ strongly across sites but are insensitive to decade-long fertilization. *Global Change Biology*, *28*(4), 1659–1677. https://doi.org/10.1111/gcb.15988

Kleber, M. (2010). What is recalcitrant soil organic matter? *Environmental Chemistry*, *7*, 320. https://doi.org/10.1071/EN10006

Kowalenko, C. G. (2001). Assessment of Leco CNS-2000 analyzer for simultaneously measuring total carbon, nitrogen, and sulphur in soil. *Communications in Soil Science and Plant Analysis*, *32*, 2065–2078. https://doi.org/10.1081/CSS-120000269

Lal, R. (2004a). Soil carbon sequestration to mitigate climate change. *Geoderma*, *123*, 1–22. https://doi.org/10.1016/j.geoderma.2004.01.032

Lal, R. (2004b). Soil carbon sequestration impacts on global climate change and food security. *Science*, *304*, 1623–1627. https://doi.org/10.1126/science.1097396

Lal, R., Suleimenov, M., Stewart, B. A., & Hansen, D. O. (2007). *Climate change and terrestrial carbon sequestration in Central Asia*, First Edition. ed. CRC Press, Taylor & Francis.

Lee, M., Nakane, K., Nakatsubo, T., & Koizumi, H. (2003). Seasonal changes in the contribution of root respiration to total soil respiration in a cool-temperate deciduous forest, in: J. Abe (Ed.), *Roots: The Dynamic Interface between Plants and the Earth: The 6th Symposium of the International Society of Root Research, 11–15 November 2001, Nagoya, Japan, Developments in Plant and Soil Sciences*. Springer Netherlands. pp. 311–318. https://doi.org/10.1007/978-94-017-2923-9_30

Lu, X., Kelsey, K. C., Yan, Y., Sun, J., Wang, X., Cheng, G., & Neff, J. C. (2017). Effects of grazing on ecosystem structure and function of alpine grasslands in Qinghai-Tibetan Plateau: a synthesis. *Ecosphere*, *8*, e01656. https://doi.org/10.1002/ecs2.1656

Mbaabu, P. R., Olago, D., Gichaba, M., Eckert, S., Eschen, R., Oriaso, S., Choge, S. K., Linders, T. E. W., & Schaffner, U. (2020). Restoration of degraded grasslands, but not invasion by Prosopis juliflora, avoids trade-offs between climate change mitigation and other ecosystem services. *Scientific Reports*, *10*, 20391. https://doi.org/10.1038/s41598-020-77126-7

Montané, F., Rovira, P., & Casals, P. (2007). Shrub encroachment into mesic mountain grasslands in the Iberian peninsula: Effects of plant quality and temperature on soil C and N stocks. *Global Biogeochemical Cycles*, *21*, GB4016. https://doi.org/10.1029/2006GB002853

Mucina, L., & Rutherford, M. C. (2006*). The vegetation of South Africa, Lesotho and Swaziland*. SNABI, Strelitzia, 19. South African National Biodiversity Institute, Pretoria, South Africa.

Norman, J. M., Garcia, R., & Verma, S. B. (1992). Soil surface CO_2 fluxes and the carbon budget of a grassland. *Journal of Geophysical Research: Atmospheres*, *97*, 18845–18853. https://doi.org/10.1029/92JD01348

Norman, J. M., Kucharik, C. J., Gower, S. T., Baldocchi, D. D., Crill, P. M., Rayment, M., Savage, K., & Striegl, R. G. (1997). A comparison of six methods for measuring soil-surface carbon dioxide fluxes. *Journal of Geophysical Research: Atmospheres*, *102*, 28771–28777. https://doi.org/10.1029/97JD01440

Oates, L. G., Undersander, D. J., Gratton, C., Bell, M. M., & Jackson, R. D. (2011). Management-intensive rotational grazing enhances forage production and quality of subhumid cool-season pastures. *Crop Science*, *51*, 892–901. https://doi.org/10.2135/cropsci2010.04.0216

Ojima, D. S., Schimel, D. S., Parton, W. J., & Owensby, C. E. (1994). Long- and short-term effects of fire on nitrogen cycling in tallgrass prairie. *Biogeochemistry*, *24*, 67–84. https://doi.org/10.1007/BF02390180

Peel, M., & Stalmans, M. (2018). The effect of holistic planned grazing^{TM} on African rangelands: a case study from Zimbabwe. *African Journal of Range & Forage Science*, *35*, 23–31. https://doi.org/10.2989/10220119.2018.1440630

Podwojewski, P., Janeau, J. L., Grellier, S., Valentin, C., Lorentz, S., & Chaplot, V. (2011). Influence of grass soil cover on water runoff and soil detachment under rainfall simulation in a sub-humid

South African degraded rangeland. *Earth Surface Processes and Landforms*, *36*, 911–922. https://doi.org/10.1002/esp.2121

Prosperi, P., Bloise, M., Tubiello, F. N., Conchedda, G., Rossi, S., Boschetti, L., Salvatore, M., & Bernoux, M. (2020). New estimates of greenhouse gas emissions from biomass burning and peat fires using MODIS Collection 6 burned areas. *Climatic Change*, *161*, 415–432. https://doi.org/10.1007/s10584-020-02654-0

Qiu, L., Zhang, Q., Zhu, H., Reich, P. B., Banerjee, S., van der Heijden, M. G. A., Sadowsky, M. J., Ishii, S., Jia, X., Shao, M., Liu, B., Jiao, H., Li, H., & Wei, X. (2021). Erosion reduces soil microbial diversity, network complexity and multifunctionality. *ISME Journal*, *15*, 2474–2489. https://doi.org/10.1038/s41396-021-00913-1

Ramo, R., Roteta, E., Bistinas, I., van Wees, D., Bastarrika, A., Chuvieco, E., & van der Werf, G. R. (2021). African burned area and fire carbon emissions are strongly impacted by small fires undetected by coarse resolution satellite data. *Proceedings of the National Academy of Sciences*, *118*, https://doi.org/10.1073/pnas.2011160118

Ritchie, M. E. (2020). Grazing management, forage production and soil carbon dynamics. *Resources*, *9*, 49. https://doi.org/10.3390/resources9040049

Savory, A., & Butterfield, J. (2016). *Holistic management, Third Edition: A commonsense revolution to restore our environment*, 3rd ed. Island Press.

Savory, A., & Parsons, S. D. (1980). The Savory grazing method. *Rangel Arched*, *2*, 234–237.

Schimel, D. S., Kittel, T. G. F., Knapp, A. K., Seastedt, T. R., Parton, W. J., & Brown, V. B. (1991). Physiological interactions along resource gradients in a tallgrass prairie. *Ecology*, *72*, 672–684. https://doi.org/10.2307/2937207

Schulze, R. E. (1997). *South African atlas of agrohydrology and -climatology*. WRC report (South Africa. Water Research Commission); TT82/96. Report No. 46, 1997.

Six, J., Elliott, E. T., & Paustian, K. (2000). Soil macroaggregate turnover and microaggregate formation: a mechanism for C sequestration under no-tillage agriculture. *Soil Biology & Biochemistry*, *32*, 2099–2103. https://doi.org/10.1016/S0038-0717(00)00179-6

Tao, X., Cui, J., Dai, Y., Wang, Z., & Xu, X. (2016). Soil respiration responses to soil physiochemical properties in urban different green-lands: A case study in Hefei, China. *International Soil and Water Conservation Research*, *4*, 224–229. https://doi.org/10.1016/j.iswcr.2016.08.001

Tessema, B., Sommer, R., Piikki, K., Söderström, M., Namirembe, S., Notenbaert, A., Tamene, L., Nyawira, S., & Paul, B. (2020). Potential for soil organic carbon sequestration in grasslands in East African countries: A review. *Grassland Science*, *66*, 135–144. https://doi.org/10.1111/grs.12267

Trollope, W. S. W. (1980). Controlling bush encroachment with fire in the savanna areas of South Africa. *Proceedings of the Annual Congresses of the Grassland Society of Southern Africa*, *15*, 173–177. https://doi.org/10.1080/00725560.1980.9648907

Valkó, O., Török, P., Deák, B., & Tóthmérész, B. (2014). Review: prospects and limitations of prescribed burning as a management tool in European grasslands. *Basic and Applied Ecology*, *15*, 26–33. https://doi.org/10.1016/j.baae.2013.11.002

van der Wal, R., Bardgett, R. D., Harrison, K. A., & Stien, A. (2004). Vertebrate herbivores and ecosystem control: cascading effects of faeces on tundra ecosystems. *Ecography*, *27*, 242–252. https://doi.org/10.1111/j.0906-7590.2004.03688.x

Virk, D. S., Pandit, D. B., Sufian, M. A., Ahmed, F., Siddique, M. A. B., Samad, M. A., Rahman, M. M., Islam, M. M., Ortiz-ferrara, G., Joshi, K. D., & Witcombe, J. R. (2009). REML is an effective analysis for mixed modelling of unbalanced on-farm varietal trials. *Experimental Agriculture*, *45*, 77–91. https://doi.org/10.1017/S0014479708007047

Wang, W., Guo, J., & Oikawa, T. (2007). Contribution of root to soil respiration and carbon balance in disturbed and undisturbed grassland communities, northeast China. *Journal of Biosciences*, *32*, 375–384. https://doi.org/10.1007/s12038-007-0036-x

Wang, W., Ohse, K., Liu, J., Mo, W., & Oikawab, T. (2005). Contribution of root respiration to soil respiration in a C3/C4 mixed grassland. *Journal of Biosciences*, *30*, 507–514. https://doi.org/10.1007/BF02703725

Wang, Z., Wan, X., Tian, M., Wang, X., Chen, J., Chen, X., Chang, S., & Hou, F. (2019). Response of soil respiration and soil microbial biomass carbon and nitrogen to grazing management in the Loess Plateau, China. *Biogeosciences Discussions*. [preprint]. https://doi.org/10.5194/bg-2018-531

Wang, Z., & Zheng, F. (2021). Impact of vegetation succession on leaf-litter-soil C:N: P stoichiometry and their intrinsic relationship in the Ziwuling Area of China's Loess Plateau. *Journal of Forest Research*, *32*, 697–711. https://doi.org/10.1007/s11676-020-01149-z

Wick, A. F., Geaumont, B. A., Sedivec, K. K., & Hendrickson, J. R. (2016). Chapter 11.2 - Grassland Degradation. In J. F. Shroder, & R. Sivanpillai (Eds.), *Biological and Environmental Hazards, Risks, and Disasters* (pp. 257–276). Academic Press. https://doi.org/10.1016/B978-0-12-394847-2.00016-4

Xu, W., & Wan, S. (2008). Water- and plant-mediated responses of soil respiration to topography, fire, and nitrogen fertilization in a semiarid grassland in northern China. *Soil Biology & Biochemistry*, *40*, 679–687. https://doi.org/10.1016/j.soilbio.2007.10.003

Zhang, H., Wang, R., Chen, S., Qi, G., He, Z., & Zhao, X. (2017). Microbial taxa and functional genes shift in degraded soil with bacterial wilt. *Scientific Reports*, *7*, 39911. https://doi.org/10.1038/srep39911

Zhao, X., Hu, S., Dong, J., Ren, M., Zhang, X., Dong, K., & Wang, C. (2019). Effects of spring fire and slope on the aboveground biomass, and organic C and N dynamics in a semi-arid grassland of northern China. *Journal of Arid Land*, *11*, 267–279. https://doi.org/10.1007/s40333-019-0052-3

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

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 CO_2 emissions. *Soil Use and Management*, 38, 1250–1265. https://doi.org/10.1111/sum.12790