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No changes in soil organic carbon and nitrogen following long-term prescribed burning and livestock exclusion in the Sudan-savanna woodlands of Burkina Faso

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Abstract

Fire and overgrazing reduce aboveground biomass, leading to land degradation and potential impacts on soil organic carbon (SOC) and total nitrogen (TN) dynamics. However, empirical data are lacking on how prescribed burning and livestock exclusion impact SOC in the long-term. Here we analyse the effects of 19 years of prescribed annual burning and livestock exclusion on tree density, SOC and TN concentrations in the Sudanian savanna ecoregion at two sites (Tiogo and Laba) in Burkina Faso. Results revealed that neither livestock exclusion nor prescribed burning had significant impact on SOC and TN concentrations. The results at both sites indicate that 19 years of livestock and fire exclusion did not result in a significant increase in tree density compared to grazing and annual prescribed burning. The overall mean (± SEM) of SOC stocks in the 0−50 cm depth increment in the unburnt (53.5 ± 4.7 Mg C ha^{-1}) and annually burnt (56.4 ± 4.3 Mg C ha^{-1}) plots at Tiogo were not statistically different. Similarly, at Laba there was no significant difference between the corresponding figures in the unburnt (37.9 ± 2.6 Mg ha^{-1}) and in the annually burnt plots (38.6 ± 1.9 Mg ha^{-1}). Increases in belowground inputs from root turnover may have countered changes in aboveground biomass, resulting in no net change in SOC and TN. We conclude that, contrary to our expectation and current policy recommendations, restricting burning or grazing did not result in increase in SOC stocks in this dry savanna ecosystem.

Keywords: Carbon sequestration; Climate change mitigation; Ecosystem services; Land restoration; Soil health; Soil organic matter

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Introduction

Soil organic carbon (SOC) is key to ecosystem functioning (Milne et al., 2015) but it varies in pattern and process between ecological zones. When properly managed, soils have the potential to not only maintain the productivity and health of the land, but also to mitigate climate change impacts (Lal, 2015). Land management practices that maintain or increase soil organic carbon not only protect soil from erosion, but they are important for water holding capacity, soil nutrient cycling and aboveground productivity. On the other hand, practices such as frequent burning or overgrazing can deplete SOC stocks by altering the structure of soil biotic communities (Sileshi & Mafongoya, 2006), reducing organic matter inputs (Higgins et al., 2007) and increasing vulnerability to erosion (Ansley, Boutton & Skjemstad, 2006; Piñeiro, Paruelo, Oesterheld & Jobbágy, 2010) in savannas. The effect of both fire and grazing on the structure and function of savanna ecosystems deserves to be better understood (Aynekulu et al., 2017; Milne et al., 2016). Prescribed burning and exclusion of grazing in rangelands are considered important management practices for SOC sequestration (Conant, 2010; Derner & Schuman, 2007), and continue to receive support as part of (inter)national policies and schemes for climate change mitigation (e.g., World Bank, 2019; Steiner, Franzluebbers, Neely, Ellis & Aynekulu, 2014). While savanna ecosystems are vulnerable to land degradation and climate change, they are also a potential target for land-based actions to mitigate climate change (Bastin et al., 2019).

Grazing land occupies about 36% of the global terrestrial land which is not covered by ice providing important ecosystem services (Erb, Gaube, Krausmann, Plutzar, Bondeau & Haberl, 2007). Savannas make up a large proportion of tropical and subtropical grazing lands. Fire, herbivory, soil properties and rainfall are known to be the primary factors for the co-dominance of grass and woody species in savannas (Sankaran et al., 2005). Due to the large area they occupy, savannas (Gherardi & Sala, 2020) are important for global soil *C* stocks, even though their net primary productivity (above and belowground) and soil *C* stocks are lower, per unit area, than those in various types of forest.

Fire plays a key role in regulating carbon balance in African savannas, which cover roughly 50% of the continents' land area (Ciais et al., 2011). Fire also changes abundance of different species and biomass allocation patterns (Beringer, Hutley, Tapper & Cernusak, 2007; Knicker, 2007; McLauchlan et al., 2020). The functioning and structure of ecosystems can be modified by fire and grazing which in turn affects soil organic carbon (SOC) and nutrient stocks (Pathak, Nath, Sileshi, Lal & Das, 2017; Pineiro, Paruelo, Oesterheld & Jobbagy, 2010). For example, the effects of repeated fires on SOC and total nitrogen (TN) in savannas can be both severe and compounding (Bird, Veenendaal, Moyo, Lloyd & Frost, 2000; Chidumayo & Kwibisa, 2003; Pathak et al., 2017; Yan et al., 2012).

African ecosystems are known to contribute about 40% to global fire emissions, mostly from savanna burning (Ciais et al., 2011). In Burkina Faso, approximately 2.2 million hectares of savanna are burnt annually (FAOSTAT, 2020). Burkina Faso generated about 0.57 $MtCO_2$-e yr^{-1}or 81% of its greenhouse gas emission from fire, in early dry season (October−December) burning and 0.14 $MtCO_2$-e yr^{-1} or 19% from late dry season (February−April) savanna burning (Pellegrini et al., 2018). The 1981 forest policy of Burkina banned bush fires (Mäkelä & Hermunen, 2007). This was revised in the 2006 National Strategy for Fire Management in Rural Areas with the aim of minimizing damages caused by forest fires by using prescribed early fire as a tool in sustainable forest and agricultural management (Mäkelä & Hermunen, 2007). In 2010 the government introduced measures that discourage bush fire frequencies in its Reducing Emissions from Deforestation and Forest Degradation (REDD) program (MEDD, 2012).

In African savannas, prescribed burning is used regularly by wildlife managers and livestock herders to prevent dominance of trees and shrubs, and to remove any low quality forage (Fynn, Haynes & O'Connor, 2003; Sankaran et al., 2005). However, fire does not consume already grown trees due to niche differentiation (Sankaran et al., 2005) and often has the use thick and insulating bark as a fire resistant trait (Bond & Keeley, 2005). Prescribed fire inhibits the survival and growth of tree seedling and samplings to maintain low tree density in the savanna (Bond & Keeley, 2005). Increase in litter deposition due to fire that affects above ground biomass can also lead to increases in SOC stocks (Pathak et al., 2017; Yan et al., 2012).

Although several studies have shown that grazing limits aboveground *C* stocks in grazing lands, its influences on SOC stocks remain unclear (Wigley, Augustine, Coetsee, Ratnam & Sankaran, 2020). Some studies predicted that reduced grazing pressure enhances SOC stocks (Conant, Paustian & Elliott, 2001). Several experiments (Bikila, Tessema & Abule, 2016; Mekuria & Aynekulu, 2011; Viglizzo, Ricard, Taboada & Vázquez-Amábile, 2019; Wigley et al., 2020) that focused on traditional rangeland exclosures reported significant increases in SOC. However, different land management practices lead to spatial and temporal variation in SOC improvements under exclosures making it difficult to generalise about the effectiveness of grazing exclusion in increasing SOC stocks. Decreases were also found in other studies (Reid, Thornton, Crabb, Kruska, Atieno & Jones, 2004; Schuman, Reeder, Manley, Hart & Manley, 1999) and increases in yet more (Derner, Briske & Boutton, 1997; Yong-Zhong, Yu-Lin, Jian-Yuan & Wen-Zhi, 2005) SOC stocks following reduced grazing intensity. Therefore, gaining a clear understanding of the long-term impacts of reduced grazing intensity on SOC stocks is crucial for improved management of grazing lands.

As it is difficult to make generalizations about the impact of fire and grazing in savanna ecosystems on SOC and TN stocks from surveys alone, controlled experiments with replicated and randomized treatments are needed to establish cause-effect relations in the impact of prescribed burning and grazing intensity on SOC and TN dynamics. Therefore, our objective was to quantify changes in SOC stocks following 19 years of a prescribed burning and livestock exclusion treatment in a researcher-designed experiment in Burkina Faso in the dry savanna ecosystem. We hypothesized that prescribed burning and livestock exclusion treatments and their combinations would lead to increases in SOC and TN compared to the control.

Materials and methods

Study area

The study was conducted in Tiogo (12°13′ N, 2°42′ W) and Laba (11°40′ N, 2° 50′ W) sites located at an altitude of 300 m above sea level in Burkina Faso (Fig. 1). The sites fall in the Sudanian savanna ecosystem where bush fire occurs annually (Dayamba, Savadogo, Sawadogo, Zida, Tiveau & Oden, 2011; Musyimi et al., 2017).

The mean (Mean ± SD) annual rainfall was 916 ± 158 mm at Laba and 837 ± 158 mm at Tiogo. Mean daily minimum and maximum temperatures range between 16 and 26 °C and 32 and 40 °C, respectively. The typical vegetation consists of bushland and woodland with an understory dominated by annual grasses *Andropogon pseudapricus* and *Loudetia togoensis* and the perennial grass *Andropogon gayanus*. The dominant tree species in both sites were *Detarium microcarpum, Combretum nigricans, Acacia macrostachya, Entada africana, Lannea acida, Anogeissus leiocarpus* and *Vitellaria paradoxa* (Sawadogo, Tiveau & Nygård, 2005). According to results of soil analysis in the current study, clay content varied between 31−34% on the Laba site and 57−58% on the Tiogo site. The Laba site has shallow soils (< 50 cm depth) while the Tiogo site has deep (> 50 cm) soils.

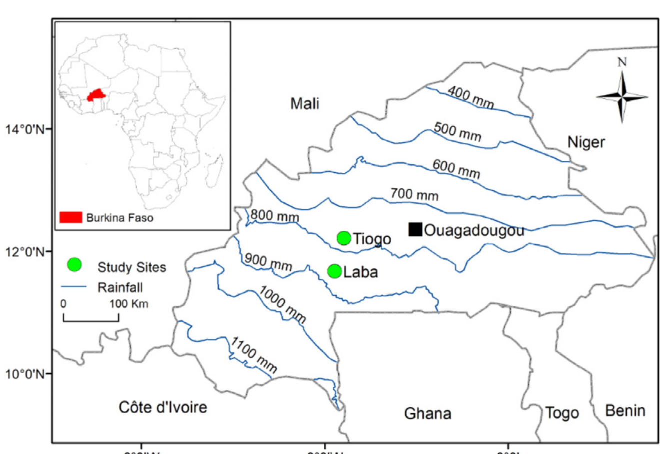

Fig. 1. Location of Laba and Tiogo study sites in Sudanian savanna-woodlands in Burkina Faso

Soil sampling and analysis

This study is part of a long-term split-plot experiment established in 1992 to explore the ecological impacts of prescribed fire and grazing intensity. The experiment had four replicates with a total area of 4.5 ha each. The experimental site measuring 18 ha was split into two contiguous main plots of which 9 ha was fenced off with barbed wire to restrict cattle from entering exclosures (i.e., closed) at the beginning of the dry season in December 1992 and the other left open for livestock grazing. Each main plot was further divided into 4 blocks of 2.25 ha, each containing 9 subplots of 0.25 ha (50 × 50 m). The plots and blocks were separated from each other by 30 m firebreaks. The treatments are fire protection and annual prescribed burning. The prescribed early fire was applied at the end of the rainy season (October−November) each year beginning 1992, when the grass layer humidity was approximately 40%. This mirrors the common practice found in the study region. The grass from the exclosure plots were not cut for livestock or any other use. Detailed description of the experimental design can be found in Sawadogo et al. (2005). Soil samples were collected from 72 plots in Laba and 72 plots in Tiogo sites. As treatments were randomly allocated, we assumed that all treatments had the same initial SOC and TN prior to the establishment of the experiments. The grazing pressure at Tiogo and Laba sites were about 0.7 and 0.5 tropical livestock unit ha^{-1}, respectively (Sawadogo et al., 2005). However, we presume some inter-annual variation existed in the intensity of these factors. Qualitatively, fire intensity was generally low because burning early in the dry season tend to produce low intensity fire as the fuel load is dominated by the herbaceous layer still holding moisture from the wet season (Savadogo, Sawadogo & Tiveau, 2007). Both grazed plots had a similar history of moderate and discontinuous grazing intensity throughout the year (Savadogo et al., 2007).

Data on tree density was estimated by counting trees and shrubs with height of greater than 1.5 m using four 100 m^2 subplots within each 50 × 50 m plots. Soil samples were collected from five sampling points (from the four corners and one from the middle) in each plot at 0−20 and 20−50 cm depths using a 7.6 cm diameter soil auger. Soil samples from the five points were combined, mixed thoroughly and about 1kg composite sample was collected using a coning and quartering method for each plot and depth range. We collected 144 soil samples from the two depths per study site. Samples were air dried and sieved through 2 mm sieve for analysis of SOC and TN using a Thermal Scientific Flash 2000 CN Analyzer (Skjemstad & Baldock, 2008). To determine bulk density, soil samples were collected from the center of each plot for the two depths (Aynekulu, Vågen, Shepherd & Winowiecki, 2011). We calculated SOC stocks per soil layer as: $SOC\ stock = C/100 \times \rho \times D \times 10{,}000$, where SOC stock is given in Mg C

ha^{-1}; C is SOC concentration (%); ρ is the dry soil bulk density fine soil fraction (Mg m^{-3}); D is the depth of the sampled soil layer (m); and 10,000 was used to convert the unit to Mg C ha^{-1}.

Data analysis

The first analysis focused on comparing tree density and SOC stocks in the 0−50 cm depth on the two sites. This was followed by a more detailed analysis of SOC, TN concentrations, C:N ratios, and SOC stocks. The study was a hierarchical (nested) design, where exclosure treatment were nested in blocks, burning treatments were nested in exclosure treatments and plots were nested in burning treatments. Therefore, we used a linear mixed-effects model to account for the hierarchical nature of the design implemented via the nlme package (Pinheiro & Bates, 2000) of *R* (R Core Team, 2018). The fixed effects were grazing management (exclosure vs. open) and prescribed burning (burned vs. unburned). The block interaction effect was entered in the model as a random effect. Data from the two sites were analysed individually as the sites significantly differed in soil texture and soil depth.

Our linear mixed effects model was expressed in the following form:

$$y_{ijk} = \mu + B_j + E_{ij} + F_{jk} + (E \times F)_{ijk} + \varepsilon_{ijk}$$

where B_j is the random effect of block j; E_{ij} is the fixed effect of exclosure; F_{jk} is the fixed effect of fire regime k; $(E \times F)_{ijk}$ is the interaction effect of exclosure ij with fire jk in the j^{th} block, and ε_{ijk} is a random error term.

As variation in nutrient stocks under the different treatments may be influenced by variation in bulk density, we used SOC and TN concentration to evaluate the impact of burning and grazing on SOC and TN.

Results

Variation in tree density with treatments

We did not find statistically significant differences in tree densities between open and exclosure treatments at both Laba and Tiogo sites (Table 1). Mean tree densities in unburnt plots were not significantly different from burnt plots on the Tiogo site (Table 1). On the other hand, on the Laba site mean tree density under prescribed burning was significantly ($P = 0.009$) lower than mean density in control plots.

Variation in soil organic carbon stocks with treatments

In both sites, bulk density did not significantly vary with the main effects of exclosure and burning or the interaction effects of exclosure and burning. As expected, the average bulk density in the 0−20 cm depth (1.2 g cm^{-3}) was smaller than in the 20−50 cm depth (1.5 g cm^{-3}). Across treatments, SOC stocks were higher on the Tiogo site than on the Laba site (Fig. 2). However, we found no difference in SOC stocks between exclosure and open grazing management systems in the 0-50 cm soil depth at both the Laba and Tiogo sites. On the other hand, plots under prescribed burning had slightly higher SOC stocks over the control by 5.4% and 1.8% at the Laba and Tiogo sites, respectively. The overall mean (± SEM) of SOC stocks in the 0−50 cm depth were 53.5 ± 4.7 Mg C ha^{-1} in the unburnt plots and 56.4 ± 4.3 Mg C ha^{-1} in annually burnt plots on the Tiogo site, while the corresponding figures on the Laba site were 37.9 ± 2.6 Mg C ha^{-1} in the unburnt and 38.6 ± 1.9 Mg C ha^{-1} in annually burnt plots. Fig. 2 summarizes the interaction effect of exclosure and prescribed burning on SOC stocks in the 0−20 cm and 20−50 cm soil depths.

Table 1. Variations in mean (±SEM) tree density (stem ha^{-1}) with exclosure and prescribed burning on the Laba and Tiogo sites in Burkina Faso.

Effects	Treatment combinations	Laba	Tiogo
Grazing	Exclosure	603 ± 44 [a]	508 ± 50 [a]
	Open	640 ± 44 [a]	586 ± 50 [a]
Burning	Unburnt	706 ± 51[a]	586 ± 52 [a]
	Burnt	537 ± 36 [b]	508 ± 39 [a]
Interaction	Exclosure * Unburnt	692 ± 72 [a]	542 ± 73 [a]
	Exclosure *Burnt	514 ± 51 [a]	475 ± 55 [a]
	Open * Unburnt	721 ± 72 [a]	631 ± 73 [a]
	Open * Burnt	559 ± 51 [a]	541 ± 55 [a]

Figures followed by the same letters in the same column are not significant at $p < 0.05$

Variation in soil organic carbon and nitrogen concentrations with treatment and soil depth

We found higher variation in SOC, TN concentrations, and C:N ratios at Laba than Tiogo sites (Fig. 3). However, we found no significant differences ($P > 0.05$) between the exclosure and the open areas in SOC, TN concentrations, and C:N ratio for both soil depths (0−20 and 20−50 cm) (Fig. 3). Burning also did not have a statistically significant effect on SOC and TN concentrations, and C:N ratio for both depths at both sites (Fig. 3).

The interaction effect of burning and livestock exclusion also did not have a statistically significant effect on SOC, TN concentrations, and C:N ratio for both depths at both Laba and Tiogo sites (Table 2). SOC, TN concentrations in the topsoil (0−20 cm) were higher than in the 20−50 cm soil depth in all treatments at both the Laba and Tiogo sites (Table 2).

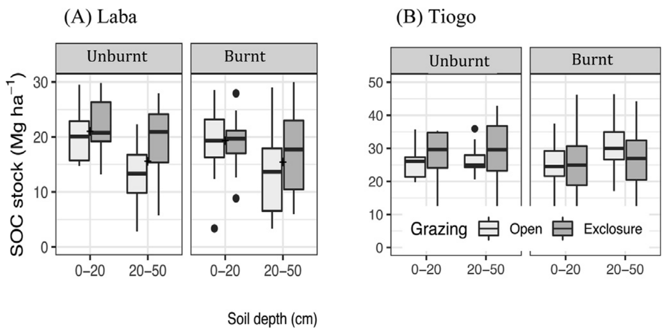

Fig. 2. Variation in soil organic carbon stocks (Mg C ha^{-1}) in 0−20 and 20−50 cm depths with exclosure and prescribed burning on Laba and Tiogo sites in Sudanian savanna-woodlands in Burkina Faso. Lower and upper box boundaries 25th and 75th percentiles, respectively, line inside box median, lower and upper error lines 10th and 90th percentiles, respectively, filled circles data falling outside 10th and 90th percentiles.

Discussion

Our results did not reveal significant changes in mean tree densities due to prescribed burning on the Tiogo site. On the other hand, mean tree density was lower in burnt plots than in the control plots. The impact of excluding grazing and fire treatments on vegetation composition and biomass production have been carefully studied and reported in Sawadogo et al. (2005). That study documented large inter-annual variation in above ground biomass. It also showed reduction of total biomass by the presence of livestock but no effect fromprescribed burning (Sawadogo et al., 2005). During the early dry season, the moisture content of the fuel biomass is high and thus the fires tend to be less expansive and the amount of energy emitted is low and thus the effect of the fires on the vegetation is minimised (Savadogo et al., 2007). Therefore, the indirect effect of fire on SOC and TN following the fire on woody species was expected to be insignificant.

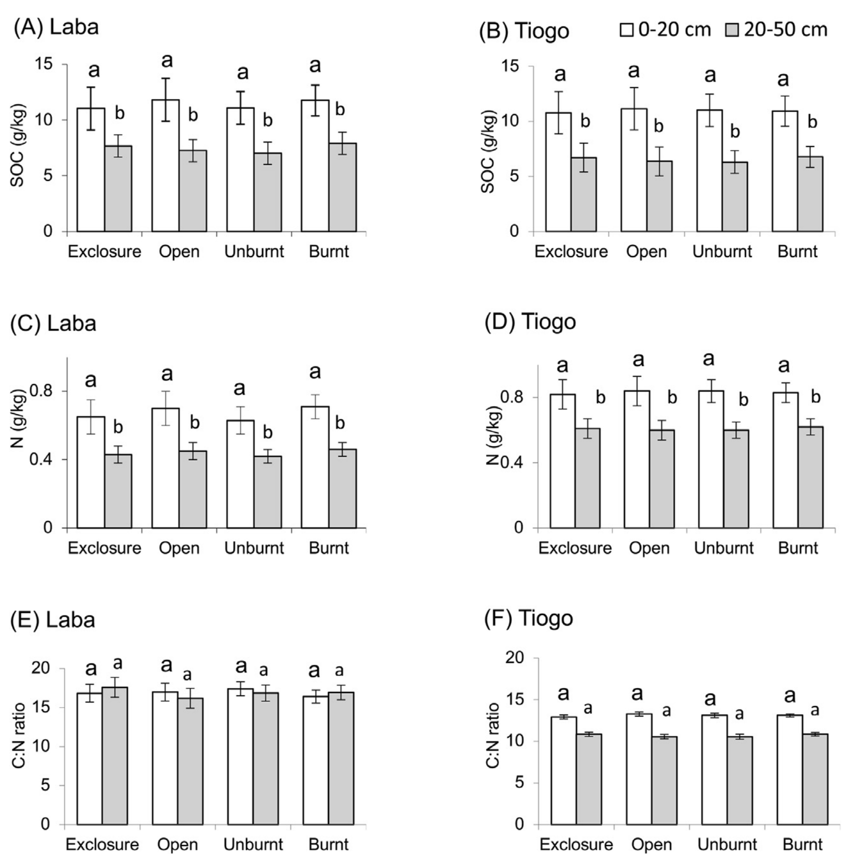

Fig. 3. Effects of long-term prescribed fire and livestock exclusion on soil organic carbon and nitrogen concentrations, and C:N ratio (mean ± SEM) on Laba and Tiogo sites in Sudanian savanna-woodlands in Burkina Faso. Different letters show significant difference at 0.05

Table 2. Variations in mean (±SEM) soil carbon concentration, nitrogen concentration, and C/N ratio with exclosure and prescribed burning on the Laba and Tiogo sites in Burkina Faso.

Variable	Effects	Treatment combinations	Laba 0-20 cm	20-50 cm	Tiogo 0-20 cm	20-50 cm
Carbon ($g\ kg^{-1}$)	Grazing	Exclosure	10.89 ± 1.92^{a}	7.74 ± 1.31^{b}	10.96 ± 1.13^{a}	6.90 ± 0.72^{b}
		Open	12.20 ± 1.92^{a}	7.50 ± 1.31^{b}	10.95 ± 1.13^{a}	6.35 ± 0.72^{b}
	Burning	Unburnt	11.10 ± 1.47^{a}	7.02 ± 1.03^{b}	11.00 ± 0.88^{a}	6.34 ± 0.57^{b}
		Burnt	11.77 ± 1.38^{a}	7.92 ± 0.95^{b}	10.93 ± 0.82^{a}	6.77 ± 0.52^{b}
	Interaction	Exclosure*Unburnt	11.50 ± 2.09^{a}	7.48 ± 1.45^{b}	10.55 ± 1.24^{a}	6.36 ± 0.81^{b}
		Exclosure*Burnt	10.59 ± 1.95^{a}	7.88 ± 1.34^{b}	11.02 ± 1.15^{a}	7.08 ± 0.74^{b}
		Open*Unburnt	10.69 ± 2.09^{a}	6.56 ± 1.45^{b}	11.45 ± 1.24^{a}	6.26 ± 0.81^{b}
		Open*Burnt	12.95 ± 1.95^{a}	7.97 ± 1.34^{b}	10.85 ± 1.15^{a}	6.45 ± 0.74^{b}
Nitrogen ($g\ kg^{-1}$)	Grazing	Exclosure	0.65 ± 0.10^{a}	0.43 ± 0.05^{a}	0.82 ± 0.09^{a}	0.61 ± 0.06^{b}
		Open	0.72 ± 0.10^{a}	0.46 ± 0.05^{b}	0.84 ± 0.09^{a}	0.60 ± 0.06^{a}
	Burning	Unburnt	0.63 ± 0.08^{a}	0.42 ± 0.04^{b}	0.84 ± 0.07^{a}	0.63 ± 0.05^{b}
		Burnt	0.71 ± 0.07^{a}	0.46 ± 0.04^{b}	0.83 ± 0.06^{a}	$0.60 \pm 0.05^{\mathbf{b}}$
	Interaction	Exclosure*Unburnt	0.65 ± 0.11^{a}	0.41 ± 0.06^{b}	0.80 ± 0.09^{a}	0.59 ± 0.07^{b}
		Exclosure*Burnt	0.65 ± 0.10^{a}	0.45 ± 0.05^{b}	0.85 ± 0.09^{a}	0.64 ± 0.07^{b}
		Open*Unburnt	0.62 ± 0.11^{a}	0.43 ± 0.06^{b}	0.88 ± 0.09^{a}	0.60 ± 0.07^{b}
		Open*Burnt	0.78 ± 0.10^{a}	0.48 ± 0.05^{b}	0.81 ± 0.09^{a}	0.60 ± 0.07^{b}
C:N ratio	Grazing	Exclosure	$16.61.14^{a}$	17.50 ± 1.27^{a}	12.90 ± 0.24^{a}	10.85 ± 0.26^{b}
		Open	16.89 ± 1.14^{a}	16.32 ± 1.27^{a}	13.30 ± 0.24^{a}	10.64 ± 0.26^{b}
	Burning	Unburnt	17.41 ± 0.89^{a}	16.86 ± 1.04^{a}	13.10 ± 0.27^{a}	10.55 ± 0.30^{b}
		Burnt	16.42 ± 0.83^{a}	16.94 ± 0.93^{a}	13.11 ± 0.19^{a}	10.85 ± 0.21^{b}
	Interaction	Exclosure*Unburnt	17.55 ± 1.27^{a}	17.89 ± 1.47^{b}	13.03 ± 0.38^{a}	10.80 ± 0.42^{b}
		Exclosure*Burnt	16.14 ± 1.17^{a}	17.32 ± 1.32^{b}	12.84 ± 0.27^{a}	10.88 ± 0.30^{b}
		Open*Unburnt	17.26 ± 1.27^{a}	15.83 ± 1.47^{b}	13.17 ± 0.38^{a}	10.30 ± 0.42^{b}
		Open*Burnt	16.70 ± 1.17^{a}	16.57 ± 1.32^{b}	13.37 ± 0.27^{a}	10.81 ± 0.30^{b}

Under each variable, figures followed by the same letters in the same column are not significant at $p < 0.05$.

Contrary to our expectations, 19 years of exclusion of grazing and prescribed annual burning treatments did not significantly increase SOC to 50 cm soil depth in this study. Therefore, our hypothesis that long-term livestock exclusion and banning of savanna burning significantly increases SOC stocks compared to the open grazing prescribed burning was rejected. This finding is relevant at a time when climate change policies have reignited interest in fire as a widely used traditional management tool to balance grass and woody cover in tropical savannas (Milne et al., 2016). Our conclusion is in agreement with Richards, Cook & Lynch (2011) who found no impact of fire on SOC, and Coetsee, Bond and February (2010) who failed to detect a significant impact of 50 years of periodic burning on SOC. Similarly, a 58 year burning study in a dry savanna ecosystem in South Africa reported no significant difference in SOC stocks (Pellegrini, Hedin, Staver & Govender, 2015). On the other hand, Oluwole, Sambo & Sikhalazo (2008) found increase in SOC after 25 years of annual burning in semiarid rangelands in South Africa. While livestock exclusion did not result in a significant increase in SOC in our study, Wigley et al. (2020) found 54% increase in SOC in the 30 cm depth increment after, herbivore exclusion for 20 years on a semiarid savanna in Kenya. These results indicate that the effects of prescribed burning and livestock exclusion are context-specific.

A carbon isotope analysis over 58 years showed that grass biomass was the major contributor to SOC (Pellegrini et al., 2015). Therefore, frequent burning may lead to decline in SOC and TN by reducing organic inputs to soils (Pellegrini et al., 2018). In some savanna ecosystems, annual burning can also increase nutrient cycling, biomass production and SOC content (Pathak et al., 2017). For example, in *Imperata cylindrica* grasslands in India, fire increased SOC and TN contents by 20−35% and supported 25% higher biomass production and 21% higher SOC stocks in the burnt site compared to an unburnt site (Pathak et al., 2017). Our results which did not detect change in SOC following fire exclusion or grazing may likely be attributed to the fact that the sites were covered by perennial grasses (*Andropogon gayanus* and *Diheteropogon amplectens*), which have slow root biomass turnover due to high lignin contents in the stems (Rossiter-Rachor, Setterfield, Douglas, Hutley, Cook & Schmidt, 2009).

Although regular burning could lead to a decline in SOC in grasslands (Ansley et al., 2006; Piñeiro et al., 2010), its short-term effect is not clear. For example, Bird et al. (2000) found higher SOC in unburned plots in a sub-humid savanna in Zimbabwe in contrast to Oluwole et al. (2008) who found increased SOC in plots which were frequently burned compared to unburned plots in dry savanna in South Africa. Discordance across locations may be the result of frequent fires resulting in the disruption of the cycling of carbon and various nutrients (Pellegrini et al., 2015). It should also be noted that any change in SOC occurs slowly (Ansley et al., 2006), and the duration of this study (19 years) may not be sufficient for significant changes in SOC and TN concentrations to be detected.

SOC in Tiogo (fine-textured soil) in the 0−20 cm depth was higher than Laba (coarse-texture soil). Bird et al. (2000), found high SOC stocks in fine-textured soils in savanna ecosystems when compared to coarse-textured soils in the same ecosystems. Bird et al. (2000) attributed this to higher nutrient and moisture holding capacities found in fine-textured soils, and the enhanced ability of fine mineral material to stabilize and protect any SOC added to the soil (Bird et al., 2000).

(Inter)national climate policy requires reliable data on the existing soil *C* stocks, coupled to understanding of how land management by pastoralists and farmers, including grazing and fire management, interacts with the ecological determinants of the grass-tree-soil interactions of *C*, water and nutrient cycling (Fig. 4). At a process level, however, the assumed effectiveness of controlling grazing and fire in increasing SOC has not been confirmed (Pathak et al., 2017) and further studies and more nuanced analyses are warranted. We present a conceptual framework (Fig. 4) to fully understand the three nested scales that need to be jointly understood.

The need for a deeper, process-oriented understanding of belowground effects of vegetation on SOC storage also emerged in a recent analysis of the effects of elevated atmospheric CO_2 concentrations. Contrary to expectations of proportionality of aboveground biomass and soil *C* storage, a recent metareview found that experimental effects of elevated CO_2 concentrations on soil carbon storage are inversely proportional to their effects on aboveground biomass (Terrer et al., 2021). Positive effects on soil *C* were found in nutrient-limited situations with small aboveground biomass responses, and neutral (or even negative) soil *C* changes where aboveground biomass response was strong. These findings are aligned with early predictions for elevated CO_2 effects from a 'functional equilibrium' perspective on the root to shoot ratio in plants in nutrient-limited, but not in water-limited, situations (van Noordwijk, Martikainen, Bottner, Cuevas, Rouland & Dhillion, 1998). In the relationship between modified root biomass and soil *C* storage, root (and mycorrhizal hyphae) turnover will increase soil *C* inputs, but root-mediated mineralization (for example by phosphatase release) may work in the opposite direction. The conclusion of Terrer et al. (2021) that current models miss presentation of such processes (and that this misrepresents future changes) is relevant, but conceptual building blocks for including the feedbacks observed have existed for some time.

While absolute belowground biomass inputs decrease with mean annual rainfall, the relative contribution (compared to aboveground biomass formation) increases. Belowground inputs are understood to be 61% (± 4.8) of total annual productivity in shrublands and savanna, versus 46% (± 7.4) as global average (Grandy & Robertson, 2007) and 41% (± 4.8) in broadleaf forest. Consequences of organic inputs (Mg ha^{-1} y^{-1}) for time-averaged *C* stocks (Mg ha^{-1}) depend on the mean residence time (y), which may be higher for below- than for aboveground inputs. Belowground net primary productivity (BNPP) strongly affects the global carbon cycle, as root turnover (Schmidt et al., 2011; van Noordwijk & Brouwer, 1997) and rhizodeposition (Sokol, Kuebbing, Karlsen-Ayala & Bradford, 2019) are the main sources of soil organic carbon.

A recent global estimate for the shrublands and savanna biome of 210 ± 53 g m^{-2} y^{-1} (Grandy & Robertson, 2007) translates to soil *C* stocks of 20−40 Mg C ha^{-1} for mean residence times of 10−20 years, with details depending on the organic-quality related rapid turnover fraction, the soil texture-dependent slow fraction and the carbonisation-dependent inert fraction (van Noordwijk, Cerri, Woomer, Nugroho & Bernoux, 1997). It suggests that management controls of soil *C* stocks are the Net Primary Production

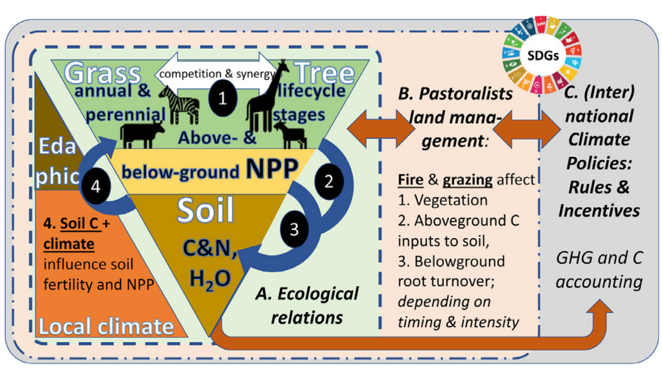

Fig. 4. Three nested scales that need to be jointly understood: A. the ecological interactions between grass, trees, herbivores and soil as influenced by B. grazing and fire management as farmer/pastoralist activities, and to be reflected in C. in national SOC accounting and subject to rules and incentives that are part of (inter)national climate policy

(related to the effective Leaf Area Index throughout the year), its partitioning to BNPP, its allocation over rapid, slow and inert fractions, and the mean residence times for each fraction. Whereas grazing will affect standing biomass, it may have relatively small effects on NPP as long as patch-level green cover is re-established rapidly after grazing; its effects on the relative partitioning to BNPP may be based on a larger structural root turnover contribution and a smaller rhizodeposition, as fine roots can die during the plants recovery of a disturbed shoot to root ratio due to grazing (va Noordwijk, Lawson, Hairiah & Wilson, 2015). Aboveground emissions due to fire ae evident, concentrating in a short period of time what would otherwise decompose more slowly. It is, however, not immediately clear how much impact fire has on belowground *C* storage as SOC. Fire, after the main growing season, may primarily reduce aboveground necromass, which may have a relatively small contribution to soil C.

Referring to Fig. 4, we don't have a satisfactory account of the underlying processes in this experiment but interpret the data in terms of a prominence of belowground organic inputs as dominating the soil *C* balance. Existing estimates of belowground net primary production together with reasonable turnover time estimates can indeed account for the measured soil *C* stocks of around 40 Mg C ha^{-1}, with likely only small fractions below 50 cm depth. Treatments may, in the short term, have increased root turnover and compensated any possible decline in NPP with an increased belowground allocation. It is likely, however, that such compensation would not apply in more extreme cases of overgrazing where large vegetation changes are induced.

Fire and grazing influence the TN cycle by influencing denitrification, nitrification, mineralization (Coetsee, Jacobs & Govender, 2012). However, TN (measured here) is overwhelmingly dominated by organic N material (formed from the breakdown of organic matter inputs). It might be important to make this distinction: small changes in available mineral N are unlikely to be detected by the analysis of TN concentrations. The N that is mineralized per annum in West African savanna was estimated to be less than 5 kg ha^{-1} $year^{-1}$ while most of nitrogen comes from the root materials (Abbadie, Mariotti & Menaut, 1992). Coetsee, February & Bond (2008) reported no significant differences in daily nitrogen mineralization rates after 50 years of different burning treatments from annual burning to fire exclusion in South African savanna. This is partly because savannas have inherent ecosystem characteristics which minimize nitrogen loss with fire. These include physical adaptations such as thick, fire-resistant bark in savanna trees, greater allocation to root biomass which confers greater resprouting ability and nitrogen resorption by trees and grass roots in the dry season when fires usually take place (Abbadie et al., 1992; Hoffmann, Orthen, Vargas & Nascimento, 2003; Singh, 1994). As a result, a shift in greater grass to tree ratios with frequent fire will not cause a decrease in available N through the altered quality of litter inputs (Coetsee et al. 2008).

Herbivores decrease fire intensity by removing biomass thereby decreasing N loss though volatilization. Grazers may also conserve up to 50% of nitrogen by moving TN from aboveground to belowground pools via urinary and fecal excretion (Hobbs, Schimel, Owenby & Ojima, 1991). Widespread woody encroachment has probably been, in part, a consequence of this fire suppression. The results of this study lend credence to the idea that fire can be used much more vigorously in African savannas to manipulate tree-grass ratios without negative effects on TN cycling (Coetsee et al., 2008).

Conclusions

In conclusion, 19 years of exclusion of fire and livestock did not result in increase in SOC stocks in this dry savanna ecosystem. However, we remain cautious that our conclusion might not hold under higher grazing and fire intensities than those reported here. Therefore, climate change mitigation projects of dry savanna ecosystems should consider the pros and cons of traditional practice of prescribed burning as a management tool. Since soil carbon change may take several decades, we recommend monitoring accompanied by modelling of SOC change over longer time periods than this study. We also suggest studies consider net GHG emissions from burning practices in addition to carbon sequestration.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Abbadie, L., Mariotti, A., & Menaut, J. C. (1992). Independence of savanna grasses from soil organic matter for their nitrogen supply. *Ecology*, *73*, 608–613.

Ansley, R. J., Boutton, T. W., & Skjemstad, J. O. (2006). Soil organic carbon and black carbon storage and dynamics under different fire regimes in temperate mixed-grass savanna. *Global Biogeochemical Cycles*, *20*(3), 1–11. doi:10.1029/2005GB002670.

Aynekulu, E., Vågen, T. G., Shepherd, K., & Winowiecki, L. A. (2011). A protocol for measurement and monitoring soil carbon stocks in agricultural landscapes. *XXX.*

Aynekulu, Ermias, Mekuria, W., Tsegaye, D., Feyissa, K., Angassa, A., de Leeuw, J., & Shepherd, K. (2017). Long-term livestock exclosure did not affect soil carbon in southern Ethiopian rangelands. *Geoderma*, *307*(July), 1–7. doi:10.1016/j.geoderma.2017.07.030.

Bastin, J. F., Finegold, Y., Garcia, C., Gellie, N., Lowe, A., Mollicone, D., & Crowther, T. W. (2019). Response to comments on "the global tree restoration potential. *Science*, *366* (6463), 1–7. doi:10.1126/science.aay8108 (New York, N.Y.).

Beringer, J., Hutley, L. B., Tapper, N. J., & Cernusak, L. A. (2007). Savanna fires and their impact on net ecosystem productivity in North Australia. *Global Change Biology*, *13*(5), 990–1004. doi:10.1111/j.1365-2486.2007.01334.x.

Bikila, N. G., Tessema, Z. K., & Abule, E. G. (2016). Carbon sequestration potentials of semi-arid rangelands under traditional management practices in Borana, Southern Ethiopia. *Agriculture, Ecosystems & Environment*, *223*, 108–114. doi:10.1016/j.agee.2016.02.028.

Bird, M. I., Veenendaal, E. M., Moyo, C., Lloyd, J., & Frost, P. (2000). Effect of fire and soil texture on soil carbon in a sub-humid savanna (Matopos, Zimbabwe). *Geoderma*, *94*, 71–90.

Bond, W. J., & Keeley, J. E. (2005). Fire as a global "herbivore": The ecology and evolution of flammable ecosystems. *Trends in Ecology & Evolution*, *20*(7), 387–394. doi:10.1016/j.tree.2005.04.025.

Chidumayo, E. N., & Kwibisa, L. (2003). Effects of deforestation on grass biomass and soil nutrient status in miombo woodland, Zambia. *Agriculture, Ecosystems and Environment*, *96*, 97–105. doi:10.1016/S0167-8809(02)00229-3.

Ciais, P., Bombelli, A., Williams, M., Piao, S. L., Chave, J., Ryan, C. M., & Valentini, R. (2011). The carbon balance of Africa: Synthesis of recent research studies. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, *369*(1943), 2038–2057. doi:10.1098/rsta.2010.0328.

Coetsee, C., Bond, W. J., & February, E. C. (2010). Frequent fire affects soil nitrogen and carbon in an African savanna by changing woody cover. *Oecologia*, *162*(4), 1027–1034. doi:10.1007/s00442-009-1490-y.

Coetsee, C., February, E. C., & Bond, W. J. (2008). Nitrogen availability is not affected by frequent fire in a South African savanna. *Journal of Tropical Ecology*, *24*(6), 647–654. doi:10.1017/S026646740800549X.

Coetsee, C., Jacobs, S., & Govender, N. (2012). An overview of nitrogen cycling in a semiarid savanna: Some implications for management and conservation in a large African park. *Environmental Management*, *49*(2), 387–402. doi:10.1007/s00267-011-9779-0.

Conant, R. T. (2010). In R. Conant (Eds.), *Challenges and opportunities for carbon sequestration in grassland systems: 9. Challenges and opportunities for carbon sequestration in grassland systems. Challenges and opportunities for carbon sequestration in grassland systems*. Rome: Food and Agriculture Organization of the United Nations (FAO).

Conant, R. T., Paustian, K., & Elliott, E. T. (2001). Grassland management and conversion into grassland: Effects on soil carbon. *Ecological Applications*, *11*(2), 343–355.

Dayamba, S. D., Savadogo, P., Sawadogo, L., Zida, D., Tiveau, D., & Oden, P. C. (2011). Dominant species' resprout biomass dynamics after cutting in the Sudanian savanna-woodlands of West Africa: Long term effects of annual early fire and grazing. *Annals of Forest Science*, *68*, 555–564. doi:10.1007/s13595-011-0055-5.

Derner, J. D., Briske, D. D., & Boutton, T. W. (1997). Does grazing mediate soil carbon and nitrogen accumulation beneath C 4, perennial grasses along an environmental gradient? *Plant and Soil*, *191*, 147–156.

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*(2), 77–85.

Erb, K. H., Gaube, V., Krausmann, F., Plutzar, C., Bondeau, A., & Haberl, H. (2007). A comprehensive global 5 min resolution land-use data set for the year 2000 consistent with national census data. *Journal of Land Use Science*, *2*(3), 191–224. doi:10.1080/17474230701622981.

FAOSTAT. (2020). Emissions shares. Retrieved December 30, 2020, from http://www.fao.org/faostat/en/#data/GH

Fynn, R. W. S., Haynes, R. J., & O'Connor, T. G. (2003). Burning causes long-term changes in soil organic matter content of a South African grassland. *Soil Biology and Biochemistry*, *35*(5), 677–687. doi:10.1016/S0038-0717(03)00054-3.

Gherardi, L. A., & Sala, O. E. (2020). Global patterns and climatic controls of belowground net carbon fixation. *Proceedings of the National Academy of Sciences*, *117*(33), 20038–20043. doi:10.1073/pnas.2006715117.

Grandy, A. S., & Robertson, G. P. (2007). Land-use intensity effects on soil organic carbon accumulation rates and mechanisms. *Ecosystems*, *10*, 58–73.

Higgins, S. I., Bond, W. J., February, E. C., Bronn, A., Euston-Brown, D. I. W., Enslin, B., & Trollope, W. S. W. (2007). Effects of four decades of fire manipulation on woody vegetation structure in Savanna. *Ecology*, *88*(5), 1119–1125.

Hobbs, N. T., Schimel, D. S., Owenby, C. E., & Ojima, D. S. (1991). Fire and grazing in the tallgrass prairie: Contingent effects on nitrogen budgets. *Ecology*, *72*, 1374–1382.

Hoffmann, W. A., Orthen, B., Vargas, D., & Nascimento, P. K. (2003). Comparative fire ecology of tropical savanna and forest trees. *Functional Ecology*, *17*, 720–726.

Knicker, H. (2007). How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. *Biogeochemistry*, *85*(1), 91–118. doi:10.1007/s10533-007-9104-4.

Lal, R. (2015). Restoring soil quality to mitigate soil degradation. *Sustainability*, *7*(5), 5875–5895. doi:10.3390/su7055875 (Switzerland).

Mäkelä, M., & Hermunen, T. (2007). Fire management on rural lands in Burkina Faso: A community-based approach. Helsinki. Retrieved from http://www.fao.org/sustainable-forest-management/toolbox/cases/case-detail/en/c/236904/

McLauchlan, K. K., Higuera, P. E., Miesel, J., Rogers, B. M., Schweitzer, J., Shuman, J. K., & Watts, A. C. (2020). Fire as a fundamental ecological process: Research advances and frontiers. *Journal of Ecology*. doi:10.1111/1365-2745.13403.

MEDD. (2012). Readiness preparation plan for REDD. *XXX.*

Mekuria, W., & Aynekulu, E. (2011). Exclosure land management for restoration of the soils in degraded communal grazing lands in Northern Ethiopia. *Land Degradation & Development*. doi:10.1002/ldr.1146 n/a−n/a.

Milne, E., Aynekulu, E., Bationo, A., Batjes, N. H., Boone, R., Conant, R., & Williams, S. (2016). Grazing lands in Sub-Saharan Africa and their potential role in climate change mitigation: What we do and don't know. *Environmental Development*, *19*, 70–74. doi:10.1016/j.envdev.2016.06.001.

Milne, E., Banwart, S. A., Noellemeyer, E., Abson, D. J., Ballabio, C., Bampa, F., & Zheng, J. (2015). Soil carbon, multiple benefits. *Environmental Development*, *13*(January), 33–38. doi:10.1016/j.envdev.2014.11.005.

Musyimi, Z., Said, M. Y., Zida, D., Rosenstock, T. S., Udelhoven, T., Savadogo, P., & Aynekulu, E. (2017). Evaluating fire severity in Sudanian ecosystems of Burkina Faso using Landsat 8 satellite images. *Journal of Arid Environments*, *139*, 95–109. doi:10.1016/j.jaridenv.2016.11.005.

Oluwole, F. A., Sambo, J. M., & Sikhalazo, D. (2008). Long-term effects of different burning frequencies on the dry savannah grassland in South Africa. *African Journal of Agricultural Research*, *3*(2), 147–153.

Pathak, K., Nath, A. J., Sileshi, G. W., Lal, R., & Das, A. K. (2017). Annual burning enhances biomass production and nutrient cycling in degraded Imperata grasslands. *Land Degradation & Development*, *28*, 1763–1771.

Pellegrini, A. F. A., Ahlström, A., Hobbie, S. E., Reich, P. B., Nieradzik, L. P., Staver, A. C., & Jackson, R. B. (2018). Fire frequency drives decadal changes in soil carbon and nitrogen and ecosystem productivity. *Nature*, *553*(7687), 194–198. doi:10.1038/nature24668.

Pellegrini, A. F. A., Hedin, A. L. O., Staver, A. C., & Govender, N. (2015). Fire alters ecosystem carbon and nutrients but not plant nutrient stoichiometry or composition in tropical savanna. *Ecology*, *96*, 1275–1285.

Piñeiro, G., Paruelo, J. M., Oesterheld, M., & Jobbágy, E. G. (2010). Pathways of grazing effects on soil organic carbon and nitrogen. *Rangeland Ecology & Management*, *63*(1), 109–119. doi:10.2111/08-255.1.

Pinheiro, J. C., & Bates, D. M. (2000). *Mixed-effects models in S and S-PLUS*. New York: Springer-Verlag.

R Core Team. (2018). *R: a language and environment for statistical computing*. Vienna: R Foundation for Statistical Computing.

Reid, R. S., Thornton, P. K., Crabb, G. J. M. C., Kruska, R. L., Atieno, F., & Jones, P. G. (2004). Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics? *Environment, Development and Sustainability*, *6*, 91–109.

Richards, A. E., Cook, G. D., & Lynch, B. T. (2011). Optimal fire regimes for soil carbon storage in tropical Savannas of Northern Australia. *Ecosystems*, *14*(3), 503–518. doi:10.1007/s10021-011-9428-8.

Rossiter-Rachor, N. A., Setterfield, S. A., Douglas, M. M., Hutley, L. B., Cook, G. D., & Schmidt, S. (2009). Invasive Andropogon gayanus (gamba grass) is an ecosystem transformer of nitrogen relations in Australian savanna. *Ecological Applications*, *19*(6), 1546–1560. doi:10.1890/08-0265.1 doi.

Sankaran, M., Hanan, N. P., Scholes, R. J., Ratnam, J., Augustine, D. J., Cade, B. S., & Zambatis, N. (2005). Determinants of woody cover in African Savannas. *Nature*, *438*(7069), 846–849. doi:10.1038/nature04070.

Savadogo, P., Sawadogo, L., & Tiveau, D. (2007). Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso. *Agriculture, Ecosystems and Environment*, *118*, 80–92. doi:10.1016/j.agee.2006.05.002.

Sawadogo, L., Tiveau, D., & Nygård, R. (2005). Influence of selective tree cutting, livestock and prescribed fire on herbaceous biomass in the savannah woodlands of Burkina Faso, West Africa. *Agriculture, Ecosystems & Environment*, *105*(1−2), 335–345. doi:10.1016/j.agee.2004.02.004.

Schmidt, M. W. I., Torn, M. S., Abiven, S., Dittmar, T., Guggenberger, G., Janssens, I. A., & Trumbore, S. E. (2011). Persistence of soil organic matter as an ecosystem property. *Nature*, *478*(7367), 49–56. doi:10.1038/nature10386.

Schuman, G. E., Reeder, J. D., Manley, J. T., Hart, R. H., & Manley, W. A. (1999). Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland. *Ecological Applications*, *9*(1), 65–71.

Sileshi, G., & Mafongoya, P. L. (2006). The short-term impact of forest fire on soil invertebrates in the miombo. *Biodiversity and Conservation*, *15*, 3153–3160.

Singh, R. S. (1994). Changes in soil nutrients following burning of dry tropical savanna. *International Journal of Wildland Fire*, *4*, 187–194.

Skjemstad, J., & Baldock, J. A. (2008). Total and organic carbon. In M. R. Carter, & E. G. Gregorich (Eds.), *Soil sampling and methods of analysis* (pp. 225−238). Soil Science Society of Canada.

Sokol, N. W., Kuebbing, S. E., Karlsen-Ayala, E., & Bradford, M. A. (2019). Evidence for the primacy of living root inputs, not root or shoot litter, in forming soil organic carbon. *New Phytologist*, *221*(1), 233–246. doi:10.1111/nph.15361.

Steiner, J. L., Franzluebbers, A. J., Neely, C., Ellis, T., & Aynekulu, E. (2014). In R. Lal, & B. A. Stewart (Eds.), *Enhancing soil and landscape quality in smallholder grazing systems* (pp. 63−112). CRC Press. doi:10.1201/b17747-5.

Terrer, C., Phillips, R. P., Hungate, B. A., Rosende, J., Pett-Ridge, J., Craig, M. E., & Jackson, R. B. (2021). A trade-off between plant and soil carbon storage under elevated CO_2. *Nature*, *591*, 599–603.

va Noordwijk, M., Lawson, G., Hairiah, K., & Wilson, J. (2015). Root distribution of trees and crops: Competition and/or complementarity. *Tree-crop interactions: agroforestry in a changing climate* (pp. 221−257). Wallingford, UK: CABI. doi:10.1079/9781780645117.0221.

van Noordwijk, M., & Brouwer, G. (1997). Roots as sinks and sources of nutrients and carbon in agricultural systems. *Soil ecology in sustainable agricultural systems* (pp. 71−89). Boca Raton: Adv Agroeco CRC-Press.

van Noordwijk, M., Cerri, C., Woomer, P. L., Nugroho, K., & Bernoux, M. (1997). Soil carbon dynamics in the humid tropical forest zone. *Geoderma*, *79*(1−4), 187–225.

van Noordwijk, M., Martikainen, P., Bottner, P., Cuevas, E., Rouland, C., & Dhillion, S. S. (1998). Global change and root function. *Global Change Biology*, *4*(7), 759–772.

Viglizzo, E. F., Ricard, M. F., Taboada, M. A., & Vázquez-Amábile, G. (2019). Reassessing the role of grazing lands in carbon-balance estimations: Meta-analysis and review. *Science of the Total Environment*, *661*, 531–542.

Wigley, B. J., Augustine, D. J., Coetsee, C., Ratnam, J., & Sankaran, M. (2020). Grasses continue to trump trees at soil carbon sequestration following herbivore exclusion in a semiarid African savanna. *Ecology*. doi:10.1002/ecy.3008.

World Bank. (2019). *Stories from the field - a look at world bank carbon finance projects in Africa*. World Bank.

Yan, Y., Tian, J., Fan, M., Zhang, F., Li, X., Christie, P., & Six, J. (2012). Soil organic carbon and total nitrogen in intensively managed arable soils. *Agriculture, Ecosystems & Environment*, *150*, 102–110. doi:10.1016/j.agee.2012.01.024.

Yong-Zhong, S., Yu-Lin, L., Jian-Yuan, C., & Wen-Zhi, Z. (2005). Influences of continuous grazing and livestock exclusion on soil properties in a degraded sandy grassland, Inner Mongolia, Northern China. *Catena*, *59*(3), 267–278. doi:10.1016/j.catena.2004.09.001.