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RESEARCH

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Soil organic carbon in dry miombo landscapes decreases with higher grazing intensity, but trees can counteract the effect

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Abstract

Soil organic carbon is one of the key determinants of soil quality and productivity, contributing to food production and mitigation of climate change. Vegetation, particularly trees, is essential in maintaining and enhancing soil organic carbon. However, there is a lack of knowledge on the ability of trees to enhance soil organic carbon in the presence of high-intensity livestock grazing. Here, we conducted a study in Morogoro Rural District, Tanzania, where we established a 10 × 10 km study site covering part of the Kitulangalo forest reserve and surrounding areas. We identified four main land uses and land cover types: forest reserve, open access forest, cropland under fallow, and cropland under cultivation. We assessed soil organic carbon stocks, livestock grazing intensity, and tree basal area in 149 plots. We also tested the effect of total grazing exclusion in two 12-years-old fenced plots located within the forest reserve. Topsoil organic carbon stocks were higher in land use classes with higher tree cover; At low grazing intensity, the forest reserve had the highest mean topsoil organic carbon (51 ± 8 tones ha^{-1}) and that this decreased with decreasing tree cover across land uses, with croplands under cultivation having the lowest value (32 ± 12 tones ha^{-1}). We found that soil organic carbon declined with increased grazing intensity, but this decrease was higher in croplands, with a 64% decrease to 12 ± 8 tones ha^{-1} when comparing the lowest and highest grazing intensities. In contrast, the decrease in forested lands was 18% to 42 ± 17 tones ha^{-1}. We conclude that increasing livestock grazing intensity has a negative impact on soil organic carbon, particularly in land uses with a low tree cover, and that more trees in the landscape have the potential to counteract the adverse effects of livestock grazing.

Keywords: Soil organic carbon, livestock grazing intensity, land use, land cover, and miombo landscape

Introduction

Soil makes the largest carbon pool on Earth (Jackson *et al.*, 2017) and is critical for stabilizing atmospheric CO_2 concentration and greenhouse gases (Jackson *et al.*, 2017). An estimated 1500 Gt of organic carbon is contained within the top meter of the soil profile (Scharlemann *et al.*, 2014), accounting for 62% of the total soil carbon (Lal, 2004). Soil organic carbon (SOC) is also a key indicator of soil health and governs a wide range of soil functions (FAO, 2017; Wiesmeier *et al.*, 2019; Tully and McAskill, 2020), including soil water holding capacity, soil structure and stability, nutrient retention and availability, soil aeration and drainage, and biological activity (Gaiser and Stahr, 2013; Gan *et al.*, 2013; Schjønning *et al.*, 2018). Drylands cover around 40% of the global land area (Cherlet *et al.*, 2018) and hold around 27 to 36% of the global SOC (Bernoux and Chevallier, 2014). Hence, changes in SOC over such vast areas can significantly impact the atmospheric carbon concentration (Stockmann *et al.*, 2013). Tropical drylands are of particular concern as they are faced with high rates of deforestation, unsustainable agricultural practices, and excessive livestock grazing (Lambin *et al.*, 2003; Fu *et al.*, 2021), with negative impacts on SOC (Solomon *et al.*, 2000, 2002; Vågen *et al.*, 2005). While these changes do not occur in isolation, past field studies on their effects on SOC have primarily addressed them separately.

Miombo woodlands are Africa's most extensive dryland forest formation, covering an estimated 2.7 million km^2 across ten countries in eastern, central, and southern Africa (Ryan *et al.*, 2016). Due to the high proportion of endemic and near-threatened species and their extensive area coverage, miombo woodlands are considered one of the five globally prioritized wilderness areas (Mittermeier *et al.*, 2003). However, miombo woodlands face

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intense degradation pressures, mostly of anthropogenic origin and linked to unsustainable agricultural practices, charcoal extraction, and overgrazing (Jew *et al*., 2016; Manyanda *et al*., 2020; Manyanda *et al*., 2021). The population of sub-Saharan Africa is expected to double by 2050 (Eastwood and Lipton, 2011), which will likely exacerbate the pressure on these ecosystems (Dewees *et al*., 2010).

Miombo woodlands are inhabited mainly by pastoral and agro-pastoral communities practicing livestock keeping and crop cultivation as their primary livelihood strategy (Manyanda *et al*., 2021). However, while ecological effects of crop cultivation in miombo woodlands are relatively well studied (Stromgaard, 1988, 1990; Luoga *et al*., 2000; Grogan *et al*., 2013; Jew *et al*., 2017), studies on the impacts of livestock grazing, particularly on SOC, are still scarce. Considering that communities surrounding miombo woodlands are mostly agro-pastoral (Isango *et al*., 2007; Giliba *et al*., 2011), the shortage of information on the impacts of livestock keeping in miombo woodlands is alarming. Overgrazing results in bare land being exposed to wind and running water for extended periods without sufficient time to recover (Catto, 2013). While low to moderate grazing intensity can improve soil health through enhanced SOC and nitrogen inputs from animal droppings (Hui and Jackson, 2005; Li *et al*., 2011; Ahmed *et al*., 2020), overgrazing has detrimental effects on soils, including reduced soil organic matter, decreased soil water infiltration and retention, and decreased quality and quantity of the vegetation cover (Tomer, 2014; Panakoulia *et al*., 2017). But little is known about how different livestock grazing intensities interact with co-occurring factors in miombo woodlands, for instance, land use and degree of tree cover, or the magnitude of these interactions and their corresponding collective effect on SOC.

Subsistence farming, mainly in the form of shifting cultivation, has been a source of livelihood for communities in the miombo ecosystem for a long time (Grogan *et al*., 2013). This practice involves rotating farming fields in varying fallow periods ranging from 1 to 20 years, depending on the recovery rate and demand for agricultural land (Kilawe *et al*., 2018). Because of this, miombo landscapes are characterized by patches of land with active farming and fallows of varying ages surrounding forests (Kalaba *et al*., 2013). If fallow periods are too short, this form of cultivation can be detrimental (Pelzer, 1964; Christanty, 1986; Ryan *et al*., 2016), causing soil erosion and the loss of SOC after subsequent slash and burn cycles in farm preparation processes (Osman *et al*., 2013). It is estimated that converting natural forests to agricultural lands results in a 60% loss of SOC in temperate regions and up to 75% or more in the tropics (Lal, 2004; Vågen *et al*., 2005; Devi, 2021). Yet, the combination of these changes and different livestock grazing intensities within the miombo landscape are not well studied.

Trees provide a wide range of ecosystem services, including regulatory services like erosion and flood control, carbon storage, and climate and hydrological regulation (Mengist *et al*., 2020). The benefits of trees, particularly their positive effect on SOC, have been reported in a wide range of systems, including croplands and forests (Gaiser and Stahr, 2013; Lorenz and Lal, 2014; Cardinael *et al*., 2015; Pardon *et al*., 2017; Hou *et al*., 2020). Trees provide organic matter from dead plant materials like leaves, bark, branches, and tree trunks that become part of the soil's organic carbon pool (Li *et al*., 2019). Through this, trees help maintain the quality of soil by enhancing its water-holding capacity and reducing overland flow and soil erosion (Cui *et al*., 2005), increasing soil aggregation and structure through the activity of their roots, reducing soil compaction (Ramesh *et al*., 2019), and promoting soil microbial activity (Thomas *et al*., 2018). However, there is still a lack of evidence on the effectiveness of trees in enhancing or maintaining SOC in the presence of high-intensity livestock grazing. Overgrazing can override the positive effects of trees on soil hydrological functioning in miombo landscapes (Lulandala *et al*., 2021), but whether the same is the case for SOC density is still unclear.

Most miombo studies on land use and its impact explore and explain single factors in isolation e.g., how land cover changes (Lupala *et al*., 2015), livestock grazing, or management of these woodlands (Chidumayo, 2019) affect different aspects of the ecosystem. However, at the landscape scale, these factors do not act in isolation but interact and have a collective impact that is often overlooked. SOC is influenced by natural factors – including climate, soil type, soil moisture, topography, vegetation cover, and soil organisms – and anthropogenic factors like land use and land management (Hu *et al*., 2018). Because of this, plot-level measurements alone cannot sufficiently explain landscape interactions, hence the need to perform these measurements at the landscape level. Fenced plots (exclosures) have been used in several studies to isolate the effects of livestock grazing on different ecosystem properties like vegetation and soil properties (Descheemaeker *et al*., 2006; Njoghomi *et al*., 2020). However, exclosure studies have some limitations: (i) they often exclude interactions that are part of the natural ecosystem in a landscape, and (ii) they are of limited area, and so, can only be used to explain local variations but not factors with extensive area coverage like land uses. These reasons make it difficult to infer the outcomes from these exclosures to the landscape level. Combining measurements from exclosures and at the landscape level provides a powerful way of understanding ecosystem interactions that would otherwise be masked.

In this study, we determined how varying livestock grazing intensity, forest protection, and land use influence SOC in a miombo landscape in Tanzania. We established a study site of 10 × 10 km covering a forest reserve and some public areas surrounding it under different land uses, and established 149 plots using a nested hierarchical random sampling design (Vågen and Winowiecki, 2020). We classified these 149 plots into four main land use classes: forest reserve, open-access forest, cropland under fallow, and cropland under cultivation. Across this landscape, we assessed livestock grazing intensity, measured tree basal area, and a range of soil properties, including bulk density, soil texture, and SOC. Within the forest reserve, we also took advantage of two 12 years old exclosures. We collected soil samples inside and outside of two exclosures to test the effect of complete livestock grazing exclusion on soil properties in the forest. We hypothesized that (i) soils within the forest reserve have higher SOC stocks than those in croplands under cultivation, croplands under fallow, or open access forests; (ii) lower livestock grazing intensity results in higher SOC; and (iii) higher tree density can counteract the adverse effects of livestock grazing on SOC.

Methods

STUDY AREA

We established a 10 × 10 km study site located in the Kitulangalo forest reserve (KFR) and surrounding area in Morogoro district, Tanzania (site center coordinates: 6° 38′ 1″ S, 37° 58′46″ E). The KFR is located approximately 50 km north-east of Morogoro municipality and 150 km from Dar es Salaam city. The climate is tropical dry sub-humid, with a mean annual temperature and rainfall of 24.3°C and 850 mm, respectively (Holmes, 1995). The vegetation is characterized by an open dry miombo woodland, with the overstorey dominated by *Brachystegia boehmii*Taub, *Julbernardia globiflora* (Benth.) Troupin, and *Pterocarpus rotundifolius*(Sond.) Druce(Prance, 1984). Dominant woody species in the understory include *Combretum* molle R.Br. ex G.Don, *Diplorhynchus condylocarpon* (Müll.Arg.) Pichon and *Dichrostachys cinerea* (L.) Wight & Arnand, while the most common grass genus is *Hyparrenia* (Nduwamungu *et al*., 2009). The dominant soil texture class in and around KFR is sandy clay loam (USDA, 1987) (Table 1).

Communities in surrounding villages are primarily low-income agro-pastoralists practicing farming, livestock keeping, and charcoal production as their main livelihood strategies (Nduwamungu *et al*., 2009), and are highly dependent on miombo woodlands for their livelihoods. As a result, illegal tree-cutting and livestock grazing are common (Hammarstrand and Särnberger, 2013).

SAMPLING DESIGN

We employed a hierarchical sampling protocol following the Land Degradation Surveillance Framework (LDSF) (Vågen and Winowiecki, 2020). First, a 100 km^2 (10 × 10 km) site was established and divided into 16 (4 × 4) tiles 2.5 × 2.5 km in size. Within each tile, random centroid locations for clusters were generated. Within each cluster, 10 plots 1000 m^2 in size were randomly established. Each plot contained four sub-plots, each with an area of 100 m^2. We also tested the effect of total exclusion of livestock grazing by using two 12 years old plots 90 × 30 m in size enclosed with fence wire (i.e., exclosures) that were established in the KFR by the Tanzania Forest Research Institute (TAFORI) for research purposes. These plots were initially established to quantify the effects of anthropogenic activities within the forest reserve, and by that time, both inside and outside locations had similar disturbance levels (Njoghomi *et al.*, 2020).

LAND USE AND LAND COVER ASSESSMENT

We combined field surveys and interviews with the local communities surrounding the KFR to understand the land-use history of the study area. From this, we classified our plots into four land use and land cover classes; (i) Forest reserve (FR): all areas that are currently under official institutional management, being regarded as a reserve, and which have not been under cultivation during at least the past 30 years, (ii) Open-access Forest (OAF): all areas within our study site that are outside the forest reserve or under any kind of official protection and have not been cultivated for at least 30 years, (iii) Cropland under fallow (CUF): cropland that has not been cultivated for at least the past 5 years, and (iv) Cropland under cultivation (CUC): all areas in our study site that are actively being cultivated or were cultivated during the previous growing season. We measured the DBH of all woody vegetation with height and DBH above or equal to 3 m and 5 cm, respectively, in all subplots (Table 1). All data collection was done during the dry season (Fig. 1).

Table 1. Mean (standard error, SE) for sand, clay, and silt content (%) of the topsoil (0–20 cm) and subsoil (20–50) samples collected in the Kitulangalo Forest Reserve and surrounding villages, Tanzania.

Soil depth (cm)	Sand (%)	Clay (%)	Silt (%)	Number of samples (n)
0 to 20	66.81 ± 11.2	22.39 ± 10.7	10.87 ± 4.4	149
20 to 50	67.67 ± 0.16	21.27 ± 0.15	11.06 ± 0.07	59

SOIL SAMPLING AND ANALYSIS

From the 160 sampling plots in the LDSF site, we removed 11 plots that had been under fallow for less than 5 years and remained with 149 plots. We dug four 50 cm deep soil pits, one at the center of each subplot, totaling 596 soil sampling locations across the site. We collected topsoil (0–20 cm) samples in each soil pit for SOC and bulk density analysis. In addition, subsoil (20–50 cm) samples were collected in 40% of the plots (59 in total). The 59 plots to take subsoil samples were selected based on fractional representations of different land uses on all 149 plots to avoid oversampling some land uses over others. The distribution and number of plots (topsoil, subsoil) relative to land uses were as follows; forest reserve (36, 15), open-access forest (40, 17), cropland under fallow (35, 14), and cropland under cultivation (38, 13). For each plot, we mixed the soil samples from each depth interval into two composite soil samples, one for the topsoil and one for the subsoil. Bulk density samples were collected using a stainless-steel cylinder with a known volume of 98.21 cm^3 (5 cm height and 5 cm inner diameter).

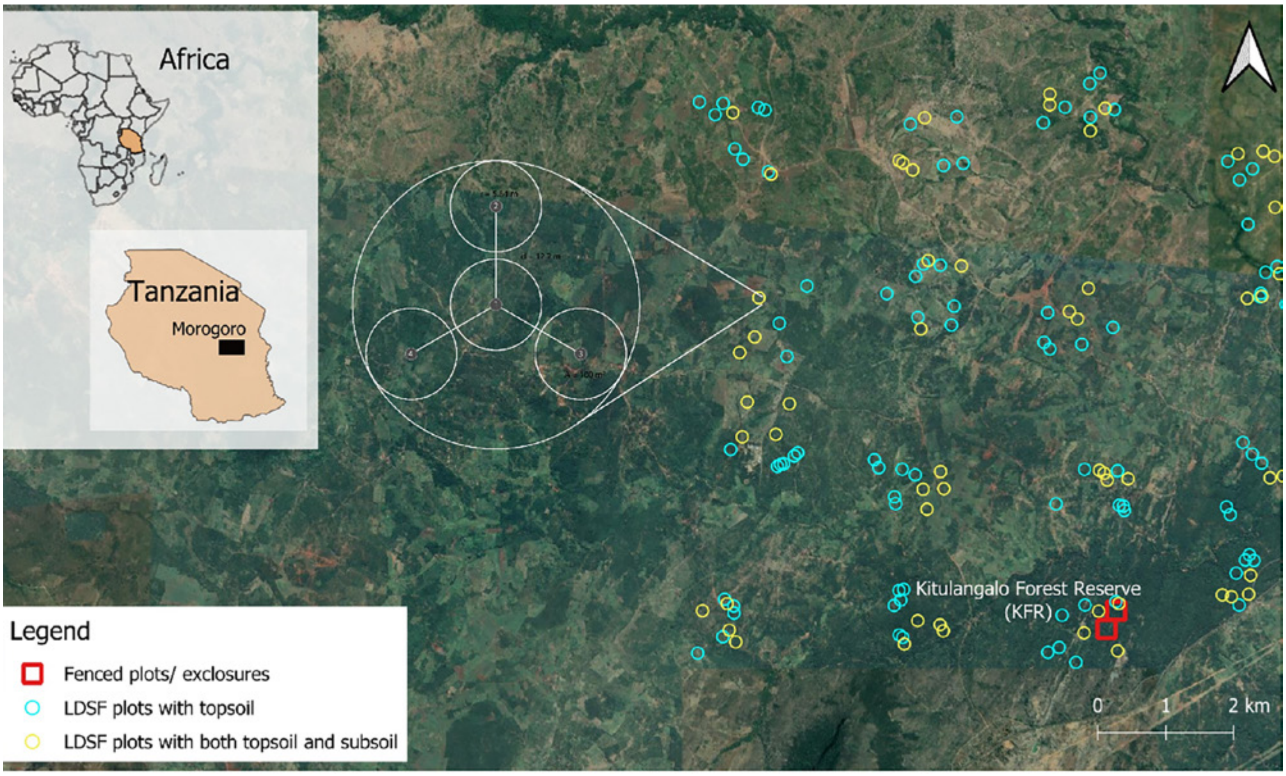

Fig. 1. Map showing the location of the 10 × 10 km^2 study site in Morogoro, Tanzania. The site covers the northeastern part of the Kitulanghalo Forest reserve. We used a nested hierarchical sampling design, following the land degradation surveillance framework (LDSF) (Vågen and Winowiecki, 2020). The map shows the location of the LDSF plots, 149 in total, and that of the two fenced plots where livestock was excluded. Each LDSF plot is 1000 m^2 in size and contains four subplots 100 m^2 in size, as shown in the plot layout.

In the two exclosures, we established 8 sampling points both inside and outside, totaling 16 sampling points per exclosure and 32 in total (Fig. 2). We also collected SOC and bulk density samples at each sampling point established inside and outside the exclosures in both soil depth intervals. All samples were measured for fresh weight, labeled, and packed in the field. Laboratory analysis of SOC was done by the Walkley-Black chromic acid wet oxidation method (Bremner and Jenkinson, 1960), soil texture by hydrometer method (Ashworth *et al.*, 2001), and bulk density by oven-drying to constant weight at 105°C (Al-Shammary *et al.*, 2018) (Table 2).

GRAZING INTENSITY

To study the effect of grazing on SOC across our study site, we established a grazing intensity variable based on the visible impacts of the following parameters: (i) signs of livestock presence (droppings, sounds, etc.); (ii) Animal paths and hoof prints on the soil surface; and (iii) Grazed vegetation. For each parameter, we assigned a score between 0 and 3, where 0 means no sign was observed on that particular parameter and 3 means extreme observation. In each plot, we added the values from each parameter to get the overall plot score, which ranged between 0 and 9, where 0 corresponds to no visible signs of livestock grazing and 9 represents the highest grazing intensity. We also created a factor version of this variable called "grazing intensity score" by reclassifying grazing intensity variable into distinct classes: 0 = no observations, 1 = 1–3, 2 = 4–6, 3 = 7–9 for further analysis and comparison.

DATA ANALYSIS

We calculated the soil carbon stock (tones ha^{-1}) for the two consecutive soil depths by using Eqn 1 (GRDC, 2014);

$$SOC\left(tonnes\,ha^{-1}\right) = \frac{SOC\%}{100} \times BD\left(g\,cm^{-3}\right) \times Depth\left(m\right) \times 10{,}000 \tag{1}$$

Where: SOC% = Percent soil organic carbon (g of OC per 100 g of dry soil), BD = Soil bulk density (g cm^{-3}), and Depth = Soil depth (m).

We performed all our statistical analyses in R studio version 3.6.1 (R Core Team, 2019). We first checked our data normality by using q–q plots. Considering the hierarchical sampling design of the study and cluster setup of our sample plots, we had two possible random variables, cluster and land use/ land cover class. We started by fitting a linear mixed-effects model using the lme() function from the package "nlme" (Pinheiro *et al.*, 2020), to test whether to include clusters in our model as a random effect or not. The mixed-effects model had livestock grazing intensity and basal area as fixed effects against SOC, and cluster as a random effect. We then tested the null hypothesis that sigma^2 = 0 (Zuur *et al.*, 2009), where sigma^2 is the variance of the random intercept (clusters), but we could not reject it, meaning that cluster variations were too small to influence the data and would not add value to our model and hence we dropped it. We then tested using land use/ land cover as a random intercept where basal area and grazing intensity were set as covariates against SOC, and we managed to reject the hypothesis, meaning that variations related to land use/ land cover are large enough to influence our data. We also ran a regression analysis of SOC against basal area in plots with the highest grazing intensity scores (3) from different land use classes. This was done to test the effect of trees in the presence of heavy grazing. We performed an Analysis of Variance (ANOVA) using the aov() function in R to test for significant differences in SOC (tones ha^{-1}) between different grazing intensities and land uses for both topsoil (0–20 cm) and subsoil (20–50 cm) within the LDSF site. We used paired t-tests to compare SOC between plots inside and outside the exclosures.

Results

Mean topsoil organic carbon was significantly higher in the forest reserve (47 ± 7 tones ha^{-1}; mean, std dev; Fig. 3) than in the open-access forest (37 ± 13 tones ha^{-1}), cropland under fallow (36 ± 8 tones ha^{-1}) and cropland under cultivation (21 ± 13 tones ha^{-1}) (all with p-values of <0.001). No significant difference was observed in mean topsoil SOC between open-access forest and cropland under fallow (p = 0.61), however, topsoil SOC in both open-access forest and cropland under fallow was significantly higher than in cropland under cultivation (p = 0.02 and p = 0.02, respectively). Within the forest reserve, areas with the lowest grazing intensity (grazing intensity score = 0) had 1.2 times higher topsoil SOC (9 tones ha^{-1} higher) than areas with the highest grazing intensity (grazing intensity score = 3; p = 0.05). In croplands under active cultivation, areas with the lowest grazing intensity had more than double topsoil SOC (20 tones ha^{-1} higher; p < 0.001) than areas with the highest grazing intensity. In the grazing exclosures, mean topsoil SOC (43 ± 10 tones ha^{-1}) was significantly higher than outside the exclosures (34 ± 8 tones ha^{-1}; p = 0.007), and within the same range as plots located in the forest reserve (Fig. 4). Soil

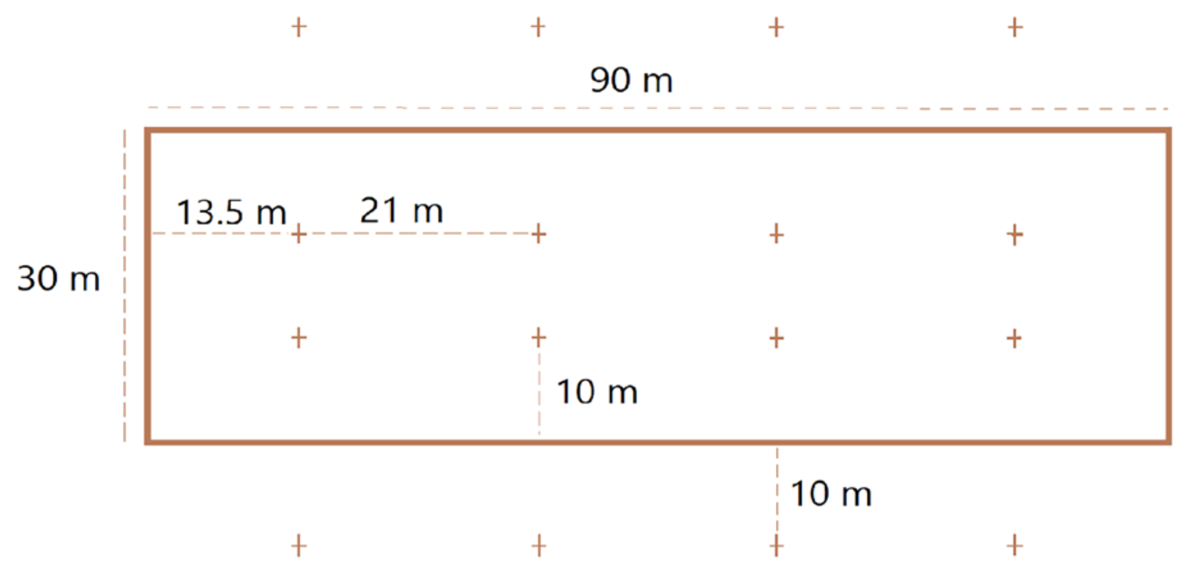

Fig. 2. Layout of sampling points inside and outside the exclosures/ fenced plots to test the effects of livestock exclusion on soil organic carbon in Kitulangalo Forest Reserve, Morogoro, Tanzania. Thick brown box line = fence around the 30 × 90 m plot, Cross signs = measuring points we established.

organic carbon in the subsoil (20–50 cm) did not show any clear trend across land use and land cover classes nor between inside (30 ± 7 tones ha^{-1}) and outside exclosures (32 ± 9 tones ha^{-1}, p = 0.74).

The regression analysis of SOC versus the interaction between land use and land cover class and grazing intensity showed a negative relationship between topsoil SOC (tones ha^{-1}) and grazing intensity that was steeper for those land use and land cover classes with lower tree cover, like cropland under cultivation, compared to the forest reserve (Fig. 3 and Table 3).

Regression analysis of SOC against tree basal area in areas with the highest grazing intensity (score 3) showed an increasing trend in SOC with increasing basal area (Fig. 5). Croplands (cropland under fallow and cropland under cultivation) showed a steeper association compared to forested land covers like forest reserve and open access forest.

Table 2. Mean basal area ± standard error, SE of trees with diameter at breast height (DBH) > 5 cm in the Kitulangalo Forest Reserve and surrounding areas (Tanzania), for the four land use and land cover types considered in the study.

Land use	Basal area (m^2 ha^{-1})	Stem density (stems ha^{-1})
Forest reserve	5.3 ± 0.6a	904 ± 22a
Open access forest	1.7 ± 0.1b	590 ± 13b
Cropland under fallow	0.6 ± 0.1c	285 ± 8c
Cropland under cultivation	0.2 ± 0.1d	81 ± 5d

Letters a, b, c, d represent significance, same letters means no difference within the group and vice-versa.

Discussion

As we had hypothesized, topsoil SOC decreased with increasing livestock grazing intensity but increased with increasing tree cover. The decrease with grazing intensity was larger in croplands than in forest lands. SOC in the forest reserve decreased by approximately 18% between areas with the lowest grazing intensity (51 ± 8 tones ha^{-1}) and those with the highest grazing intensity (42 ± 17 tones ha^{-1}), while we observed a 64% decrease in mean SOC from 32 ± 12 tones ha^{-1} to 12 ± 8 tones ha^{-1} with the same change in grazing intensity within croplands.

Effects of livestock grazing on SOC are highly variable in different studies depending on various environmental factors, including soil properties, climate, altitude, the intensity of grazing, and time (Dlamini *et al*., 2016). Some studies show a decrease in SOC due to heavy grazing intensities (Steffens *et al*., 2008; Martinsen *et al*., 2011). Others report an increase in topsoil SOC, particularly with low to moderate grazing intensities (Derner *et al*., 1997; Bauer *et al*., 1987), or no change at all (Johnston *et al*., 1971). Further, higher tree basal area was related to higher topsoil carbon stocks across grazing intensities and land use classes. These studies come from a wide range of climatic conditions, from temperate to tropical climates, and have different ranges of grazing intensity. However, most of them are from grasslands and do not consider the impact of livestock on forests or other wooded lands, or croplands. Our landscape study highlights the importance of such frequently overlooked factors and their influence on ecosystems.

We observed a 40% (18 tones ha^{-1}) decrease in topsoil SOC with increasing grazing intensity between areas with no grazing and areas with the highest grazing intensity averaged across all land uses. Results from the experimental 12-year-old exclosures support those from the landscape survey since the average SOC inside the exclosures was similar to that for the plots in the forest reserve with

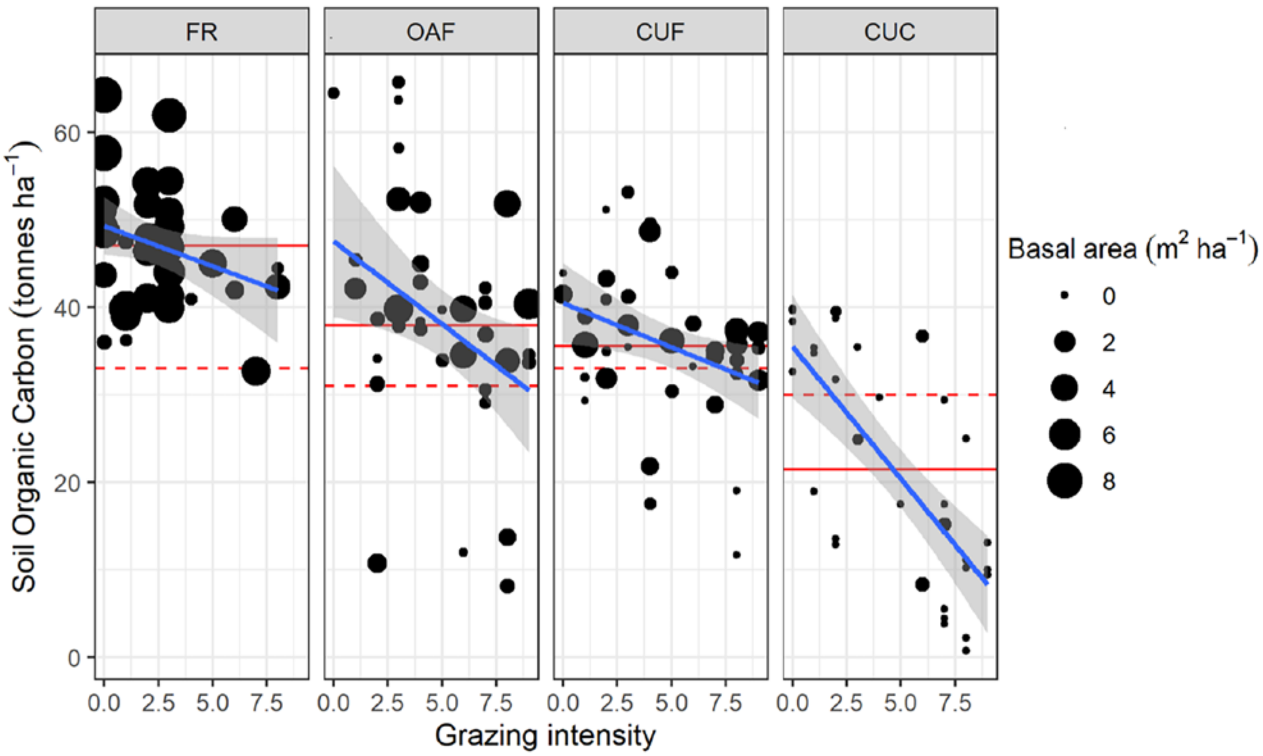

Fig. 3. The relationship between topsoil organic carbon (tones ha^{-1}) and livestock grazing intensity (score 0–9) for each land use and land cover class (FR = forest reserve, OAF = open access forest, CUF = cropland under fallow, CUC = cropland under cultivation) in 149 plots across the 10 × 10 km study site in Kitulangalo, Morogoro, Tanzania. Circle sizes are relative to the plot tree basal area (m^2 ha^{-1}). Solid blue lines represent regression lines within each land use. Solid and dashed red lines indicate the mean topsoil and subsoil soil organic carbon (tones ha^{-1}), respectively, gray shading shows 95% confidence interval in each land use/ land cover class.

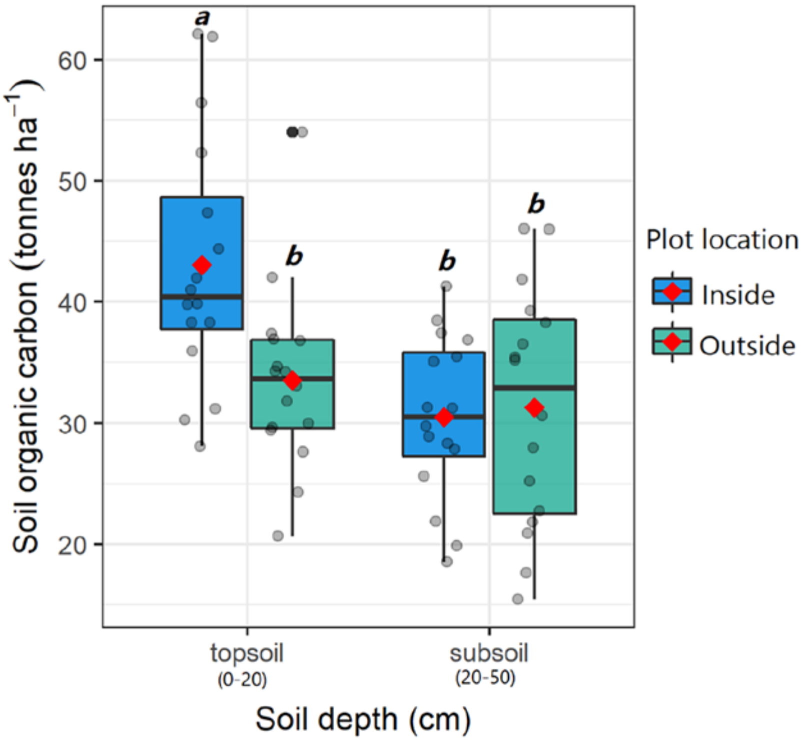

Fig. 4. Boxplot (median, first and third quartile) of soil organic carbon (tones ha^{-1}) in two consecutive soil depth intervals inside and outside grazing exclosures in Kitulangalo Forest Reserve, Morogoro, Tanzania. Red diamond symbols show the mean. Black dots show soil organic carbon (tones ha^{-1}) individual observations. Letters on top of the boxplots are significant levels, same letters = no significant difference, different letters = significant difference present.

Table 3. Regression coefficients and associated p-values showing the relationship between soil organic carbon (tones ha^{-1}) against land uses and different grazing intensities and interactions between grazing and land use with forest reserve as a reference class within the study site in Kitulangalo, Morogoro, Tanzania. Parameters: Gr = Grazing intensity (score: 0–9), OAF = open access forest, CUF = cropland under fallow, CUC = cropland under cultivation.

Parameter	Coefficient	p-value
Gr	−0.92	0.18
OAF	−1.77	0.65
CUF	−8.81	0.01
CUC	−13.78	<0.001
Gr: OAF	−0.97	0.27
Gr: CUF	−0.09	0.91
Gr: CUC	−2.10	0.01

low grazing intensity score (Fig. 3), while our measurements just outside the 12-year exclosures were at a similar level as plots with high grazing intensity. Subsoil carbon stock (20–50 cm) did not show clear trends with either land use or grazing intensity.

While low to moderate grazing may increase SOC (Li *et al.*, 2011; Blache *et al.*, 2016), findings from a global meta-analysis on the impacts of livestock grazing on soil properties show a persistent decline in SOC with heavy grazing intensity, particularly in the case of coarse-textured soils and more arid climates (Lai and Kumar, 2020). Tropical drylands like miombo woodlands exhibit low net primary productivity while having relatively high decomposition rates during parts of the year (Govers *et al.*, 2013; Johnston *et al.*, 2009). This, combined with the prevalence of coarse-textured soils, which on average lose up to 31% more SOC than fine-textured soils (Jobbágy and Jackson, 2000; Potter *et al.*, 2001; Vågen and Winowiecki, 2013; Winowiecki *et al.*, 2016), make miombo soils particularly vulnerable to degradation by livestock grazing.

In our study, the forest reserve had the highest mean topsoil (0–20 cm) organic carbon stock (47 ± 7 tones ha^{-1}) followed by open access forest (37 ± 13 tones ha^{-1}), cropland under fallow (36 ± 8 tones ha^{-1}), and cropland under cultivation (21 ± 13 tones ha^{-1}; Fig. 3). Topsoil carbon stocks in our study are within the range of reported stocks in other miombo studies, with forests having a range between 21 and 54 tones ha^{-1} and fallows from 10 to 52 tones ha^{-1} (fallow length 1–30 years) (Rossi *et al.*, 2009; Mapanda *et al.*, 2010; Kutsch *et al.*, 2011). We found a consistent decrease in SOC with changes from forest to agricultural lands, in agreement with previous studies in miombo woodlands (Bulusu *et al.*, 2021; Walker and Desanker, 2004). We observed a 36% decrease in soil carbon stock from forest to cropland at the lowest grazing intensities. This is within the range of 20 to 50% reported in semi-arid areas within the tropics when converting from forest to agricultural lands (Davidson and Ackerman, 1993; Solomon *et al.*, 2000). The observed significant difference in SOC between cropland under fallow and cropland under cultivation in our study area shows that if sufficient fallow time is allowed, it is possible to recover SOC in croplands (Szott *et al.*, 2004). The decrease in SOC after converting forests to cropland can be explained in three ways; (i) reduced biomass inputs from vegetation cover following deforestation (Aweto, 1981), (ii) the cropping system involving the burning of crop residues (Grogan *et al.*, 2013) and soil cultivation (e.g., tillage) (West and Post, 2002), and (iii) leaching of dissolved organic carbon (Nakhavali *et al.*, 2021).

High livestock grazing intensity had a more severe negative impact on SOC in land uses with low basal area/tree cover than in forest land. We observed an 18% decrease in SOC with an increase in grazing intensity from 0 to 3 intensity scores within the forest reserve, while in the croplands with active cultivation, the observed decrease in SOC was three times higher (64%) with the

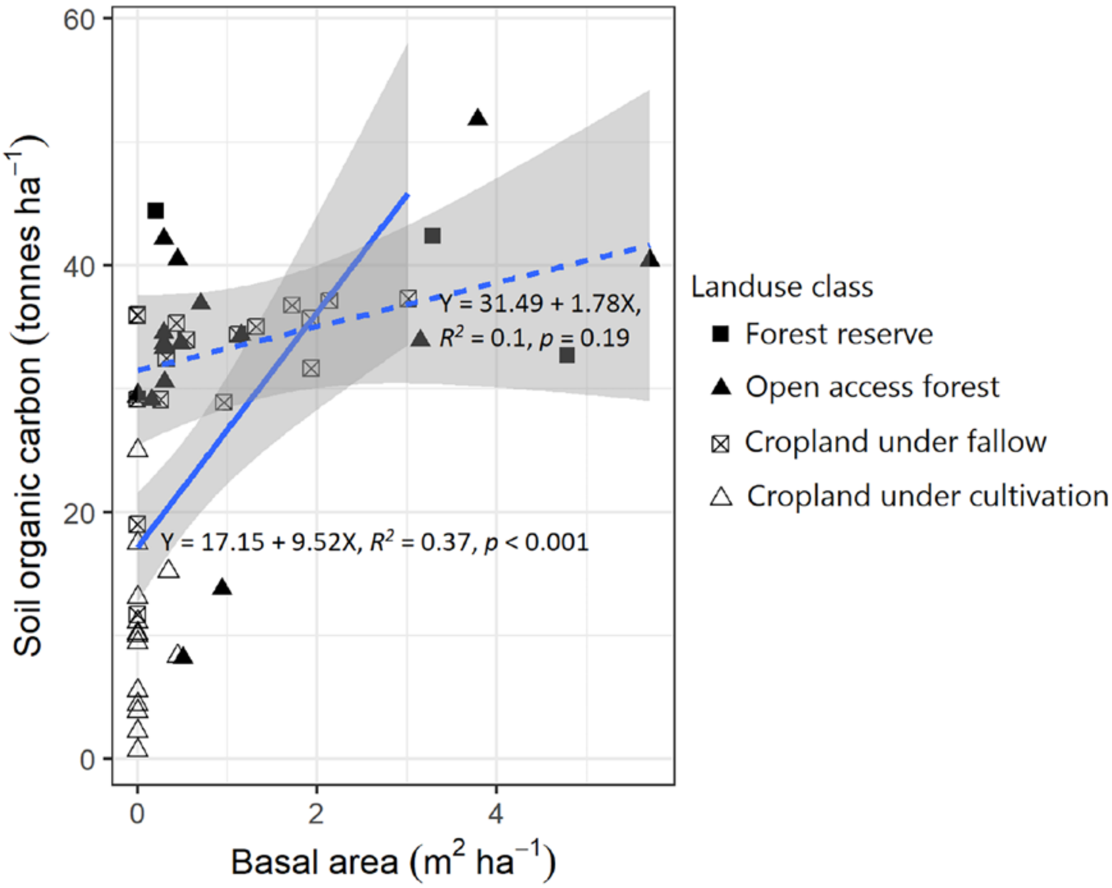

Fig. 5. Scatter plot of the topsoil soil organic carbon (tones ha^{-1}) against basal area (m^2 ha^{-1}) in plots with the highest grazing intensity score (3) for different land use classes (FR = forest reserve, OAF = open access forest, CUF = cropland under fallow, CUC = cropland under cultivation) within the study site in Kitulangalo, Morogoro, Tanzania. The dashed blue line shows the linear relationship found between the two within forested land covers (forest reserve and open access forest). The solid blue line shows the linear relationship between the two in croplands (cropland under fallow and cropland under cultivation).

same change in grazing intensity (Fig. 3). Overgrazing causes a significant reduction of vegetation cover (Wang, 2014; Mtimbanjayo and Sangeda, 2018), thus reducing carbon inputs into the soil (Li *et al*., 2015). The loss of vegetation cover and frequent trampling by grazing animals can increase the breakdown of soil structural aggregates, soil erosion by both wind and water, and loss of SOC (Li *et al*., 2019). This, coupled with soil disturbance by tillage, makes agricultural lands even more susceptible to degradation (Blevins and Frye, 1993; Haddaway *et al*., 2017). Trees counteract this by adding carbon back to the soil through above- and below-ground litter inputs. In addition, trees may help maintain soil carbon through their positive effect on soil infiltration, which, in turn, reduces the risk of overland flow and water erosion (Cui *et al*., 2005), and the creation of niches with favorable microclimates for soil microbes that enhance SOC (Thomas *et al*., 2018). This might help explain why mean SOC stocks in the forest reserve with the highest grazing intensity were still 30% higher than that of cropland under cultivation with no visible signs of grazing. Our study shows that, by complete exclosure of only grazing in croplands, it is possible to triple carbon stocks, which has approximately the same effect as leaving it to fallow. This could probably further be increased through enhanced tree cover, however, because higher tree densities were less common in cropped land in our study area, it was not possible to ascertain the effect of trees in the interaction within croplands.

Topsoil SOC was significantly higher within exclosures than outside. The inside and outside of the exclosures were intensively grazed and similar in characteristics when the exclosures were installed 12 years ago (Lulandala, 2008; Njoghomi *et al*., 2020). Hence, our results suggest an increase of SOC in areas inside the exclosures of 10 tones ha^{-1} (Fig. 4) with complete exclusion of livestock due to increased biomass, reduced soil compaction, and improved soil aeration, leading to enhanced soil biological activity and enhanced water infiltration (Descheemaeker *et al*., 2006; Johnson and Lehmann, 2006; Lulandala *et al*., 2021).

CONCLUSION

Our study fills a critical knowledge gap in the management of tropical woodlands which may have consequences for increasing and maintaining soil carbon and enhancing climate change mitigation. We show that converting miombo woodlands to agricultural lands reduces SOC. If forest conversion is combined with high grazing intensity, this leads to accelerated loss of SOC. Reciprocally, leaving cropland into fallow can increase SOC, but heavy grazing delays recovery. Our study also shows that trees play an important role in maintaining SOC; hence, maintaining and enhancing tree cover in forests and agricultural fields could reduce the adverse effects of grazing on soil carbon stocks. Instead of completely clearing forests for agricultural uses, trees may be deliberately left on farms to help maintain soil quality and enhance recovery once in fallow. However, the most significant improvements in SOC result from controlling livestock grazing intensity. These findings stress the importance of land-use management practices and policies across the 2.7 million km^2 of miombo woodlands, especially given the increasing trends of livestock grazing (Manyanda *et al*., 2021), and high rates of land use changes (Ribeiro, 2016) in the region. By exposing a 1 million km^2 of cropland to light grazing intensity instead of heavy grazing, it would be possible to gain up to

1.8 gigatones of soil carbon, and by adding low density trees cover on these farms, up to 0.9 gigatones of additional soil carbon could be gained. Conversely, if heavy grazing is introduced to 1 million km^2 of forested land, 0.9 gigatones of soil carbon can be lost. This can further increase to 3.9 gigatones if the same land is converted to cropland, substantially impacting future atmospheric CO_2 levels. With an increasing trend of livestock grazing in drylands, management practices such as improved silvopastoralism and zero grazing are needed to minimize its adverse effects on soil carbon stocks and climate change.

CONFLICT OF 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 article. All relevant institutional and ethical guidelines were followed during the research process, including obtaining necessary approvals from ethical review boards and securing informed consent where applicable.

The research was conducted independently, without any influence from sponsors, and all data have been presented transparently to ensure accuracy and integrity in the publication of this manuscript.

ETHICS STATEMENT

This study was conducted in accordance with all applicable ethical guidelines and regulations. The research adhered to the ethical principles outlined by National Health Research Ethics Committee (NatHREC) under the Tanzania Commission for Science and Technology (COSTECH) as well as Swedish Ethical Review Authority (Etikprövningsmyndigheten), ensuring compliance with the highest standards of academic integrity and ethical research practices. In cases where human participants or sensitive environmental data were involved, appropriate consent was obtained, and ethical clearance was granted by the relevant bodies.

All authors confirm that the research was performed in accordance with relevant guidelines, regulations, and best practices. Any data collected and used during this study was handled with the utmost care to ensure confidentiality, integrity, and security. The authors declare no conflict of interest related to the publication of this manuscript.

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AUTHOR CONTRIBUTIONS

Following the authorship criteria outlined by McNutt *et al.* (PNAS, 2018), each author made a substantial contribution to the work presented in this paper. The specific contributions are as follows:

LL contributed in conceptualization of the study, development of the methodology, data collection and analysis, validation of results, and primary responsibility for writing the original draft of the manuscript; UIU contributed in conceptualization of the study, development of the methodology, securing funding, and providing critical revisions to the manuscript; GN performed supervision of the research project, securing funding, and providing critical revisions to the manuscript; CAM performed visualization and preparation of figures, data management, and substantial contribution to the final manuscript revision; and ABT contributed in visualization and preparation of figures, data management, and critical contribution to the final manuscript revision. All authors contributed to the interpretation of data and approved the final version of the manuscript. They agree to be accountable for all aspects of the work to ensure the accuracy and integrity of any part of the study.

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DATA AVAILABILITY

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

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