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Excessive livestock grazing overrides the positive effects of trees on infiltration capacity and modifies preferential flow in dry *miombo* woodlands

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Abstract

The increase in livestock grazing in African drylands such as *miombo* woodlands threatens land productivity and ecosystem functioning. Trees have positive effects on soil hydraulic properties, but few studies have looked at grazing intensity and hydrological functioning in different land uses. Therefore, we conducted a biophysical survey in Morogoro Rural District, Tanzania, where we identified four main land uses and land cover types, that is, Forest reserve, open-access forest, cropland under fallow, and active cropland. We assessed grazing intensity, measured infiltration capacity, and conducted dye tracer experiments to assess the degree of preferential flow in 64 plots. We also tested the effect of grazing exclusion on infiltration capacity in 12-year-old fenced plots. Our results show that irrespective of land use or cover type, soil bulk density increased by 10% from low to high grazing intensity, whereas infiltration capacity and soil organic carbon decreased by 55% and 28%, respectively. We found a positive relationship between infiltration capacity and tree basal area in plots with lowest grazing intensities. However, at higher grazing, the infiltration capacity remained low independently of the basal area. Preferential flow in deeper soils was six-times higher in areas with no grazing, indicating higher deep soil and groundwater recharge potential at low grazing intensities. We conclude that the negative impacts on soil hydrological functioning of excessive livestock grazing override the positive effect of trees, but restricting grazing can reverse the impact.

KEYWORDS

grazing intensity, land use and land cover, *miombo* woodlands, preferential flow paths, ring infiltrometer, steady-state infiltration capacity

1 | INTRODUCTION

Drylands cover approximately 40% of the World's land area and support about two billion people, 90% of whom live in low and middle-income countries (UN, 2020). Water limitation is the key

factor governing dryland ecosystem functioning and community livelihood (Miller, 2005). Land use and (mis)management can further exacerbate the stress on ecosystems and livelihoods (Koch & Missimer, 2016). This pressure is expected to intensify in the future due to increased water demand as a result of

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population growth, infrastructure development, and increased demand for agricultural commodities (Jodha et al., 2012; Mittal, 2013; Ripple et al., 2017). In addition, drylands are extremely vulnerable to climatic variations and the impact of human disturbances such as deforestation, overgrazing, and unsustainable agricultural practices (Davies et al., 2012).

The influence of tree cover and land use on soil water dynamics in tropical drylands is poorly understood (FAO, 2016). Soil hydrological processes are complex, with high variability both spatially and temporally. Tree cover has been shown to have a strong influence on two of these processes in particular: infiltration capacity and preferential flow. Infiltration capacity is defined as the maximum rate at which water on the soil surface enters the soil (Ferré & Warrick, 2005; Kirkham, 2014), while the preferential flow is a rapid and uneven movement of water and solutes within the soil through regions of higher flux such as cracks and root channels (Guo & Lin, 2018; Jarvis et al., 2012). These two hydrological processes are affected by several factors, including both inherent and management-dependent soil properties such as soil texture and soil organic matter content, land use, and vegetation cover (Lozano Baez, 2019). At the plot level, studies show that trees positively influence soil structure, aggregate stability, and porosity through enhanced soil organic matter content and the activity of roots and tree-associated soil fauna, which, in turn, result in improved soil infiltration capacity and more preferential flow through macropores (Bargués-Tobella et al., 2014; Belsky et al., 1993; Benegas et al., 2014; Ekhuemelo, 2016; Eldridge & Freudenberger, 2005). Improvements in soil hydrological functioning caused by trees can ultimately enhance deep soil and groundwater recharge (Bargués-Tobella et al., 2014; Ilstedt et al., 2016). Improved preferential flow has been found to be positively correlated to infiltration capacity in several studies (Li et al., 2020; Zhang et al., 2018). However, understanding the main factors controlling soil hydraulic processes at the landscape scale requires measuring soil hydraulic properties over large areas beyond the plot level, and this is rather unusual due to the high cost and time associated with these measurements (Demand et al., 2019; Ilstedt et al., 2007; Zimmermann et al., 2006). This means there is a need for approaches that can combine plot-level measurements over several ecosystems or land uses at a scale of several kilometres.

Livestock keeping and farming are the major economic activities practiced by dryland communities (Powell et al., 2010; Scoones, 1991; Singh, 2018). Livestock supports the livelihoods of about 70% of the rural dryland population of West and East Africa. Twenty percent of these livestock keepers depend exclusively on livestock (pastoralists), while the rest derive a portion of their income from cropping (agropastoralists) (Cornelis de, 2016). Because of this high dependency and population increase, livestock grazing is exhibiting an increasing trend in dryland ecosystems (Gumbo et al., 2018). While sustainable intensification of the animal population can have a positive influence on natural ecosystems (Blache et al., 2016; Harry et al., 2014; Kairis et al., 2015; Saleem, 1998), poor management and lack of technical know-how is common and has led to severe overgrazing in many drylands (Busso & Pérez, 2019; Cortina et al., 2011; Yirdaw et al., 2017).

Overgrazing is considered a serious threat to ecosystem health due to its negative impacts on land productivity and soil stability, particularly on slopes, causing severe erosion and reducing the soil water holding capacity (Czeglédi & Radácsi, 2005; Wang, 2014), as well as soil organic carbon (Dlamini et al., 2016). High livestock grazing intensities also reduce the regeneration of young woody plants (Kikoti et al., 2015; Lohbeck et al., 2020) and increase soil compaction as a result of trampling (Sharrow, 2007). The frequent and continuous movement of large herds of livestock disrupts soil aggregates and can create an impervious compaction layer within the topsoil (Russell & Bisinger, 2015), which, in turn, can result in decreased soil infiltration capacity (Hiernaux et al., 1999; Savadogo et al., 2007) and less preferential flow paths for deep soil water percolation (Dreccer & Lavado, 1993). In tropical pasturelands, it has been shown that interactions between trees and livestock lead to spatial variations in soil hydraulic properties, with soil infiltration capacity and preferential flow through macropores being greater in the vicinity of trees than in adjacent open areas (Benegas, 2018). However, when anthropogenic disturbances are high, the positive effects of trees may be diluted or even suppressed. For example, results from Ghimire et al. (2014, 2013) show that reforestation of severely degraded land was not effective in restoring soil hydraulic properties due to the heavy usage of such land - including litter collection, livestock grazing, and harvesting of fuelwood. However, the specific effects of varying tree cover and livestock grazing intensities in dryland forests and woodlands have yet to be examined.

Miombo is a commonly used term for the seasonally dry deciduous woodlands dominated by the genera *Brachystegia*, *Julbernardia*, and/or *Isoberlinia* (Leguminosae, subfamily Caesalpinioideae) which are widespread across Africa (Williams et al., 2008). *Miombo* constitutes the most extensive tropical seasonal woodland and dry forest type in Africa, covering an area between 2.7 and 3.6 million km² across the Central African Plateau and its escarpment (CIFOR, 1996). *Miombo* extends from Tanzania and southern DRC in the north to Zimbabwe in the south, and across the continent from Angola, through Zambia, to Malawi and Mozambique (Walker & Desanker, 2004). In Tanzania, *miombo* woodland accounts for the largest dryland vegetation land cover, amounting to as much as 90% of all forested land (MNRT, 2015). However, it faces intense pressure from rapid deforestation and degradation through socioeconomic activities, with a mean rate of decline of about 1.13% per year since the 1990s (Abdallah & Monela, 2007; Sawe et al., 2014). Such deforestation is mainly due to increased demand for firewood, charcoal production, shifting cultivation, illegal lumber production for building materials, a high frequency of wildfires, and livestock grazing, all coupled with rapid population growth and urbanization (Manyanda et al., 2020; Sangeda & Maleko, 2018). Since livestock grazing has been and still is a growing practice in *miombo* woodlands (Abdallah & Monela, 2007; Cauldwell et al., 1999; Sangeda & Maleko, 2018), understanding its ecological implications is essential, in particular those related to water security.

In this study, we determined how varying livestock grazing intensity, forest protection, and land use influence soil hydraulic properties

in *miombo* woodlands. We selected a $10 \times 10 \text{ km}^2$ study area, which included a protected forest reserve and surrounding communities practicing agriculture and livestock keeping. Across this landscape, we measured a range of soil properties related to soil hydrological functioning: bulk density, soil texture, and soil organic carbon. We examined 160 plots randomly distributed, but following a nested hierarchical sampling design (Vågen et al., 2018; Vågen & Winowiecki, 2020). In 64 of these 160 plots, we also measured two additional key soil hydraulic properties – soil infiltration capacity and degree of preferential flow. We classified the plots into four primary land use and land cover types: forest reserve, open-access forest, cropland under fallow, and cropland under cultivation. Within the forest reserve, we also established a separate study to measure soil properties inside and outside two exclosures, from which livestock had been excluded for 12 years. In all plots, we assessed relative livestock grazing intensity and hypothesized that (i) Infiltration capacity and preferential flow increase with increased tree cover, (ii) Infiltration capacity and preferential flow decrease with increased intensity of grazing.

2 | MATERIALS AND METHODS

2.1 | Study site

We conducted this study within a $10 \times 10 \text{ km}^2$ site covering the northeastern part of the Kitulangalo Forest Reserve (KFR) and

surrounding landscape, some 35 km northeast of Morogoro Municipality in Morogoro Rural District, along the Morogoro – Dar es Salaam Highway and 150 km inland from the city of Dar es Salaam, Tanzania (central coordinates $6^\circ 38' 1'' \text{ S}$, $37^\circ 58' 46'' \text{ E}$, Figure 1). KFR covers the ridge between the main road and the Sangasanga River from an altitude of 350–774 m above mean sea level (Mwandosya et al., 1998). The climate of the area is a tropical dry subhumid, with mean annual rainfall and temperature of 850 mm and 24.3°C , respectively (Holmes, 1995). The rainfall is unimodal, with a rainy season spread over 5–6 months (November to May) and a dry season extending from June to October.

The KFR was officially established in 1955 and declared in the Government Gazette GN 198 of 3rd June 1955 as being designated for conservation and water catchment protection purposes (SUA, 2018). KFR was first classified by the government as a 'productive reserve', meaning that wood harvesting is allowed by those who obtain a license. Later, in 1985, harvesting was forbidden, even though illegal encroachment for wood harvesting and livestock

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

Site/depth (cm)	Sand (%)	Clay (%)	Silt (%)	Number of samples (n)
0 to 20	67 (11)	22 (11)	11 (4)	160

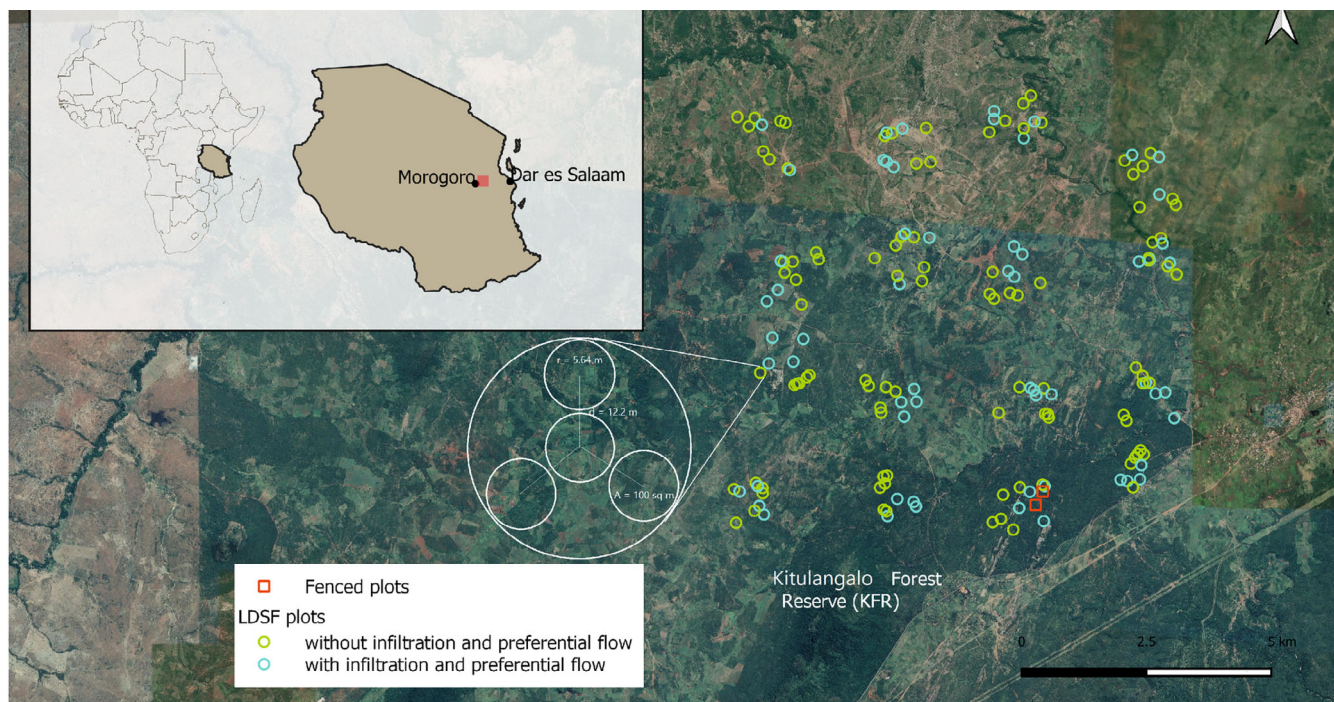


FIGURE 1 Map showing the location of the $10 \times 10 \text{ km}^2$ study site in Morogoro, Tanzania. The site covers the northeastern part of the Kitulangalo Forest reserve. We used a nested hierarchical sampling design, following the land degradation surveillance framework (LDSF) (Vågen et al., 2018; Vågen & Winowiecki, 2020). The map shows the location of the LDSF plots, 160 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 [Colour figure can be viewed at wileyonlinelibrary.com]

grazing still occurs (Hammarstrand & Särnberger, 2013; Njoghomi et al., 2020). Vegetation cover in the KFR and surrounding areas is typical open dry *miombo* woodland dominated by *Julbernardia globiflora*, *Brachystegia boehmii*, and *Pterocarpus rotundifolius*, with a canopy height of up to 20 m (Nduwamungu et al., 2009). Soil texture at our study site (Table 1) is relatively uniform and is classified as sandy clay-loam. The KFR is surrounded by seven villages (Gwata, Mazizi, Maseyu, Geza ulole, Lubondo, Mavulu, and Lukwambe) with farming, animal herding, and charcoal production as their main economic activities. The populations of these villages depend greatly on the woodlands in and outside the reserve for their livelihoods.

2.2 | Sampling design

In this study, we adopted the sampling design from the Land Degradation Surveillance Framework (LDSF) (Vågen & Winowiecki, 2020). The LDSF is a hierarchical field survey and sampling protocol consisting of sites 100 km² in size (10 × 10 km), clusters within sites, and plots within clusters. Each LDSF site is divided into 16 tiles 2.5 × 2.5 km² in size, and random centroid locations for clusters within each tile are generated. Each cluster, in turn, consists of 10 plots with randomized center-point locations. Each plot is 1000 m² in size and consists of four subplots, 100 m² in size (Figure 1).

To test the effects of total livestock exclusion, we designed a separate study using two 12-year-old fenced 30 × 90 m² plots within the forest reserve (Figure 1) that were established by the Tanzania Forest Research Institute (TAFORI) in 2005. These enclosures were set out to test and quantify the effects of anthropogenic activities within the forest. At the time these plots were established, the two areas we compared (inside and outside) were both affected by grazing and had a similar disturbance level (Njoghomi et al., 2020).

2.3 | Land use and vegetation assessment

By combining interviews on the history of land use and land cover changes with the communities in villages surrounding the KFR and physical observation, each LDSF plot was classified into one of the following classes:

1. Forest reserve (FR): These are areas classified and managed by the government as forest reserves that have not been cultivated for at least the last 30 years.
2. Open-access forest outside the reserve (OAF): These are areas outside the reserve that have not been cultivated for the last 30 years, mostly covered by natural vegetation and not under any official governance.
3. Cropland under fallow (CUF): Croplands that have not been cultivated for at least the past 5 years.
4. Cropland under cultivation (CUC): Areas that have been cultivated at least during the last growing season.

TABLE 2 Mean basal area (standard error, SE) of trees with diameter at breast height (DBH) > 5 cm in the Kitulungalo 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)	Number of plots (n)
Forest reserve	5.3 (0.6)	904 (22)	36
Open-access forest	1.7 (0.1)	590 (13)	39
Cropland under fallow	0.6 (0.1)	285 (8)	46
Cropland under cultivation	0.2 (0.1)	81 (5)	38

Vegetation assessments were conducted at the subplot level, where we measured and counted all trees (woody vegetation taller than 3 m and with a DBH greater than 5 cm). These data were then used to calculate the basal area for each of the four land use and land cover types (Table 2).

2.4 | Soil sampling and analysis

At the center of each of the four subplots within an LDSF plot, we dug a 50 cm deep soil pit from which to collect soil samples; these were taken from the pit wall at 0–20 cm depth. We mixed the samples from all four subplots within a plot to obtain one composite sample. In the enclosures, we also collected one soil sample from each sampling point (Figure 2). Additionally, we collected soil samples for topsoil bulk density assessment. Bulk density samples were collected using a stainless steel cylinder of volume 98.17 cm³ (5 cm height and 5 cm inner diameter) at the middle of the 0–20 cm depth interval on one of the pit walls. One bulk density sample was collected at the center of each of the four subplots within an LDSF plot and the center of each sampling point in the enclosures. We choose to focus on just the topsoil because of the nature of the parameters we are studying (grazing effect and land use). Soil compression caused by grazing, which we measured as an increased bulk density, occurs within the upper 20 cm of the topsoil. Land use, especially farming, in these areas does not involve heavy machinery; instead, hand hoes are mostly used, and these do not penetrate down to the subsoil. Using the samples, we conducted laboratory analyses of soil organic carbon by the Walkley-Black chromic acid wet oxidation method (Bremner & Jenkinson, 1960), soil texture by the hydrometer method, and bulk density.

2.5 | Soil infiltration capacity measurements

We measured soil infiltration capacity (also known as soil infiltrability; Hillel, 2003) in 64 LDSF plots, one measurement per plot in four randomly selected plots per cluster (Figure 1), and 16 paired samples, with points inside and outside each of the enclosure (Figure 2). However, we removed four plots from the 64 LDSF plots

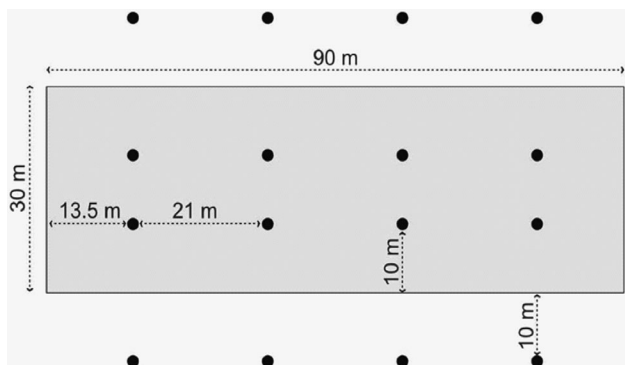


FIGURE 2 Layout of sampling points inside and outside the exclosures/fenced plots to test the effects of livestock exclusion on soil infiltration capacity in Kitulungalo Forest Reserve, Morogoro, Tanzania. Thick black box line = fence around the $30 \times 90 \text{ m}^2$ plot, dots = measuring points we established

during the data cleaning phase because of errors in field measurements, retaining 60 infiltration measurements that we used in our analysis. We measured soil infiltration capacity at the center of each selected plot using a single ring infiltrometer (Di Prima et al., 2018) with an inner diameter and height of 30 and 27 cm, respectively. In each of the plots, we inserted the ring 5 cm into the soil. We then conducted prewetting by carefully pouring two liters of water into the ring and allowing it to completely infiltrate before we started recording infiltration rates. During the infiltration measurements, the ring was carefully filled with water up to the 20 cm level, as stated in the LDSF field guide (Vågen & Winowiecki, 2020). The water level within the ring was recorded after 5 min, and the ring was immediately refilled to the initial start level (20 cm). This procedure was repeated every 5 min during the first half-hour of the infiltration experiment and every 10 min during the second half-hour for a minimum period of 1 hr, depending on whether a steady infiltration rate had been reached or not. During the 10-min interval period, we stopped taking measurements once we obtained similar readings in three consecutive measurements; sometimes, this took up to 70 min in total. For each time interval, infiltration capacity rates were calculated by subtracting the final water level from the initial one and dividing it by the time interval. Steady-state infiltration capacity was estimated using the *SSphilip* function from the package 'HydroMe' in R, which is a self-starting function for estimating infiltration parameters in the Philips model (Omuto, 2013).

2.6 | Preferential flow

Following the completion of each infiltration measurement, we conducted a dye experiment to study the water infiltration patterns. We could only do this in the 64 LDSF plots, as we were not allowed to disturb the soil further in the exclosures. After we finished taking infiltration measurements, 200 mm of a brilliant blue FCF (C.I.42090) dye solution of concentration 4 g L^{-1} equivalent to 14.1 L was added into

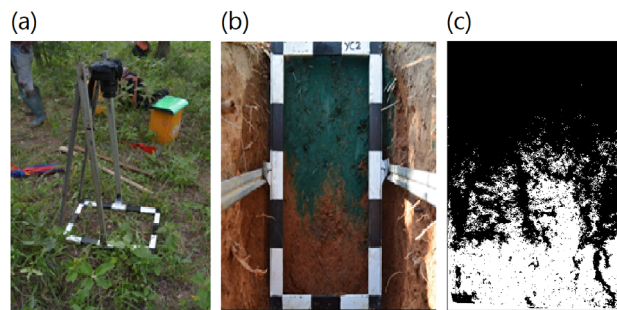


FIGURE 3 Pictures covering the process from acquiring a picture of a stained soil profile to obtaining the classified image (stained vs. nonstained classes); (a) a camera and a graded frame for soil profile photography, (b) photo of a stained soil profile, (c) classified soil profile image showing the dye stained and nonstained area of the profile [Colour figure can be viewed at wileyonlinelibrary.com]

the infiltration ring and allowed to soak completely. Thirty minutes after complete infiltration of the dye solution, after the removal of the infiltration ring, we carefully dug a 0.45 m wide by 2 m long and 0.6 m deep pit cutting across the dye stained surface to expose a vertical stained soil profile. The exposed face was then leveled carefully to avoid smearing before taking photos. A Nikon D5200 camera with a 35 mm focal length and a graded frame with inner dimensions of $0.3 \times 0.5 \text{ m}^2$ (width and height, respectively) (Figure 3a) were used to take the pictures of the stained soil profiles. The camera was placed 1.5 m from the centre of the photo frame. Photos (Figure 3b) were taken in daylight under an umbrella to avoid direct radiation causing too much reflection. Photos were then analyzed using ERDAS IMAGINE-version 9.2 (ERDAS Inc., 2008) and ARC MAP-version 10.2 software (ESRI Inc., 2013). First, photos were preprocessed to correct for geometric distortion, and then individual pixels were classified into dye-stained and nonstained classes using supervised image classification in ERDAS Imagine (Figure 3c).

After completing the classification, we created a shapefile in ArcMap comprising 100 rectangular polygons of 15 cm^2 (30 cm wide and 0.5 cm high) that divided our images into grids. We then calculated the area within each of these rectangular polygons covered by stained and nonstain pixels. From this, we calculated dye stained area for each profile where; uniform dye stained area is 80% and more while nonuniform stained areas are all below 80%. These figures were then used to calculate the different indices of preferential flow. From the classified images and corresponding dye coverage curves, the following preferential flow indices were calculated;

1. *Total dye coverage* (DC, %) (Flury et al., 1994); is the percentage ratio of the dye-stained area to the total profile area (dye stained and nondye stained). Soils with a higher degree of preferential flow will have a low value of this parameter.

$$DC = 100 \cdot \left(\frac{D}{D + ND} \right), \quad (1)$$

Where: DC (%) is percentage dye coverage, D is the dye coverage area (cm^2), and ND is the nonstained area

$$\text{PF} - \text{fr} = 100 \cdot \left(1 - \frac{\text{UniFr} \cdot 30}{\text{TotStAr}} \right) \quad (2)$$

2. *Uniform infiltration depth* (UniFr, cm) (Van Schaik, 2009): the depth at which the dye coverage decreases below 80%; this represents the depth of the uniform infiltration front where the infiltration process is dominated by the uniform flow. Below this depth, it is assumed that the flow is preferential. Soils showing high preferential patterns will therefore have low values of this parameter.
3. *Preferential flow fraction* (Van Schaik, 2009): the fraction of the total infiltration that flows through preferential flow paths.

Where: PF-fr is the preferential flow fraction (%), UniFr is the uniform infiltration depth (cm), TotStAr is the total stained area (cm^2), 30 is the width in cm of our graded photo frame.

High values of this parameter are indicative of unevenness of pore space distribution in a soil column. Thus, soils with a high degree of preferential flow will have high values of this parameter

4. *Preferential flow at 45–50 cm* (PF₄₅₋₅₀, %): this is the preferential flow in deeper soils, it refers to the dye coverage percentage in the 45–50 cm depth range if this is below the uniform infiltration depth.

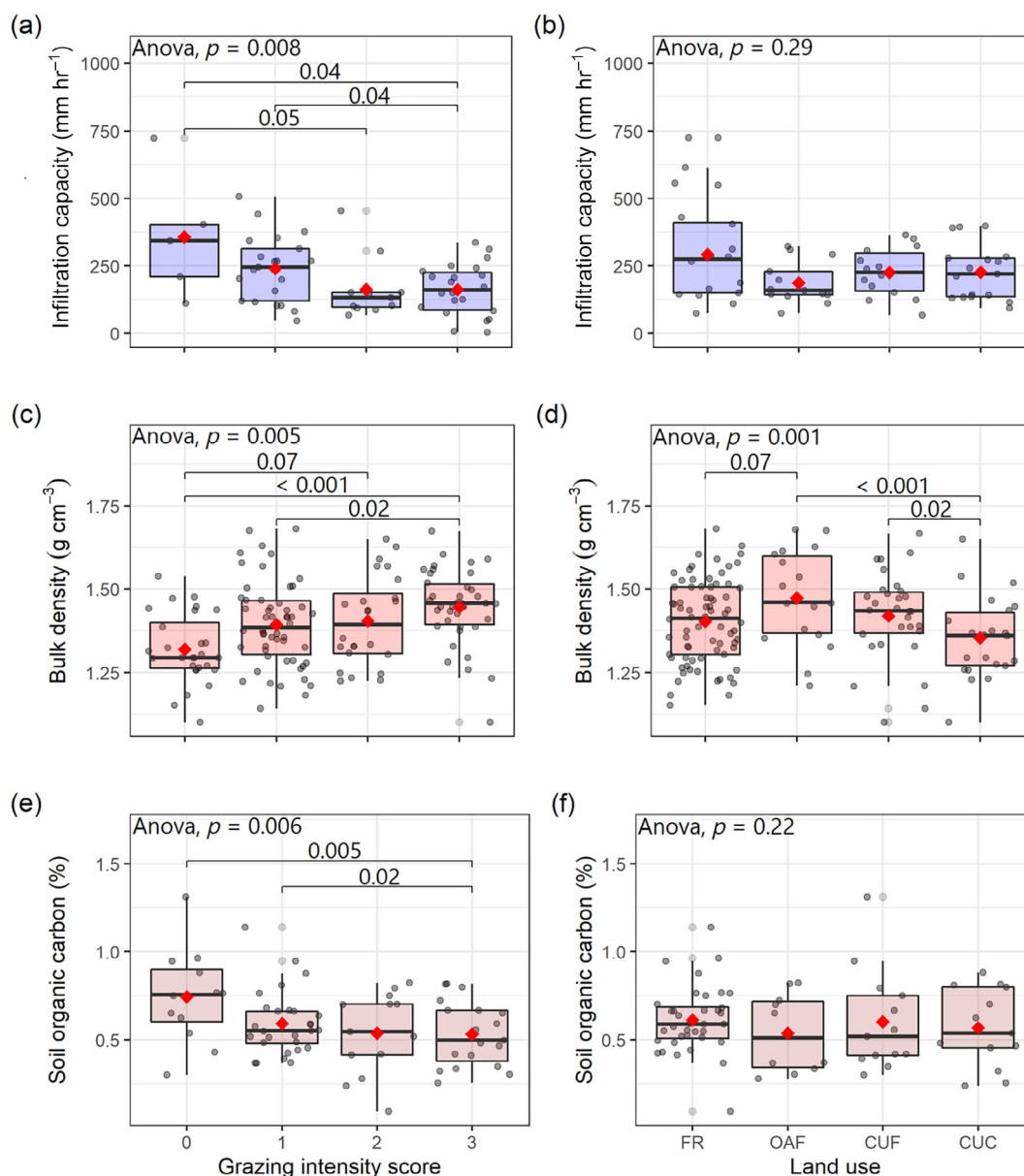


FIGURE 4 Boxplot (median, first and third quartile) of (a, b) steady-state infiltration capacity (mm hr^{-1}), (c, d) bulk density (g cm^{-3}), and (e, f) soil organic carbon (%) for the different classes of grazing intensity (left column) and land use/land cover (right column) within a $10 \times 10 \text{ km}^2$ area in Kitulungalo, Morogoro, Tanzania; significance values (p) are given. Red dots indicate the mean value. FR = forest reserve, OAF = open-access forest, CUF = cropland under fallow, CUC = cropland under cultivation [Colour figure can be viewed at wileyonlinelibrary.com]

This measure indicates the presence of preferential flow at this depth when the uniform infiltration depth is above 45 cm, which was the case in all our plots. The selection of this depth interval was based on the dimensions of the frame we used (30 × 50 cm) but can change depending on the height of the photo frame.

2.7 | Grazing intensity

We established a grazing intensity score to allow us to study the effects of different livestock grazing intensities. In this study, the grazing intensity score is related to the visible impacts of livestock grazing. We based the scoring on individual observations 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. We assigned a value between 0 and 3 for each parameter separately according to its severity (where 0 = no sign observed and 3 = most severe condition observed); we then summed them to obtain the overall plot score (0–9), which we then used to reclassify grazing intensity into four distinct classes: 0 = no observations of the parameters considered, 1 = 1–3, 2 = 4–6, 3 = 7–9.

2.8 | Statistical analyses

All statistical analyses were performed in R version 3.6.1 (R Core Team, 2019). Before starting the analyses, we checked for data normality by plotting q-q plots. Given that the sampling design employed in this study was hierarchical or nested, we first constructed linear mixed-effects models using the lme() function from the package 'nlme' by Pinheiro, Bates, DebRoy, Sarkar, & R Core Team (2020), to estimate the effects of different soil parameters, land use and land cover types, and tree cover on steady-state infiltration capacity and preferential flow indices. We used the hypothesis testing method suggested by Zuur et al. (2009), with $\sigma^2 = 0$, where σ^2 is the variance of the random intercept (clusters). In this case, we could not reject the null hypothesis. We also compared the Akaike information criterion (AIC)

between models with different random effects structures (with and without clusters as a random effect). The model without the random effect was better. This suggested that there was no advantage in incorporating clustering as a random effect in the model. At the same time, it revealed the presence of an extremely low correlation between observations within the same cluster, confirming the absence of autocorrelation and meaning that it was appropriate to use a regular linear regression (fixed effects only). We ran regression analysis for infiltration capacity and preferential flow using tree cover (basal area) and grazing intensity as covariates. We used an ANOVA test (the aov() function in R) to identify significant differences in infiltration capacity (mm hr^{-1}), bulk density (g cm^{-3}), soil organic carbon (%), and all other preferential flow indices (TotStAr, UniFr, PFfr, and PF_{45-50}) between land use/land cover types and different grazing intensities. We conducted the ANOVAs after checking for equality of variance among groups by using Levene's test (the LeveneTest() function in R from the package 'car'), confirming the absence of heteroscedasticity. A paired t-test (the t.test() function in R) was used to compare infiltration capacity (mm hr^{-1}), bulk density (g cm^{-3}), and soil organic carbon (%) between sampling points located inside and outside the enclosures.

3 | RESULTS

3.1 | Infiltration capacity, soil organic carbon, and bulk density

We observed no clear relationship between steady-state infiltration capacity and land use/land cover type ($p = 0.29$; Figure 4b). Instead, across all land use and land cover classes, steady-state infiltration capacity decreased with increasing livestock grazing intensity ($p = 0.008$; Figure 4a); Mean steady-state infiltration capacity for plots with low grazing intensity (score 0) was 357 mm hr^{-1} ($\text{SE} \pm 104$), double that in plots with high grazing intensity ($160 \pm 20 \text{ mm hr}^{-1}$). Regression analysis showed that there was a clear positive relationship between steady-state infiltration capacity and tree basal area in locations with a grazing intensity score of 0 ($p = 0.02$) (Figure 5, Table 3). However, this relationship

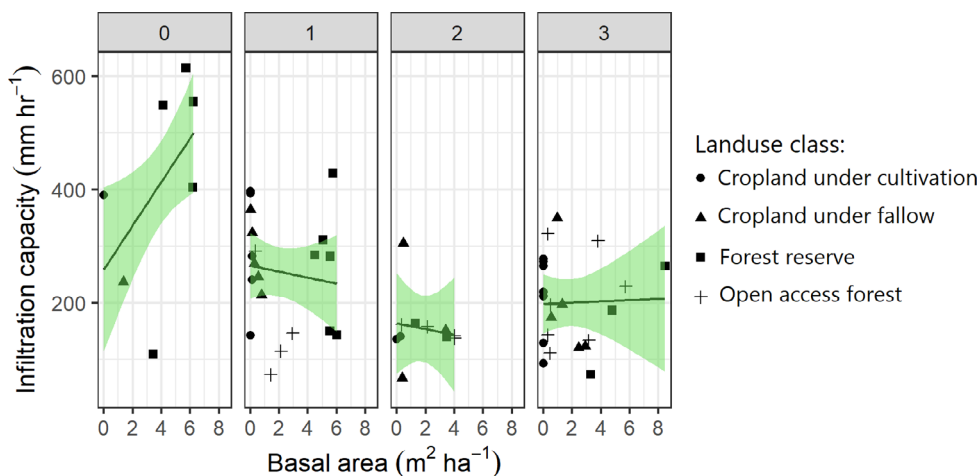


FIGURE 5 Scatter plots showing the relationship between steady-state infiltration capacity (mm hr^{-1}) and basal area ($\text{m}^2 \text{ ha}^{-1}$) in relation to different grazing intensity scores within the LDSF site in Kitulangalo, Morogoro, Tanzania. Numbers at the top of the plot (0, 1, 2, and 3) represent grazing intensity scores. Regression lines are shown [Colour figure can be viewed at wileyonlinelibrary.com]

TABLE 3 Regression coefficients and p -values for the linear model showing the relationship between infiltration capacity (mm hr^{-1}) and basal area (g cm^{-3}) associated with the different grazing intensity scores (gr1, gr2, and gr3) as treatments from the LDSF site in Kitulungalo, Morogoro, Tanzania

Parameter	BA	gr1	gr2	gr3	BA*gr1	BA*gr2	BA*gr3
Coefficients	38.65	6.55	-95.56	-60.97	-43.94	-43.55	-37.56
p -values	(0.02)	(0.93)	(0.27)	(0.44)	(0.02)	(0.08)	(0.05)

seemed to disappear in the presence of grazing (grazing intensity score 1, 2, 3; Figure 5). Mean bulk density increased from 1.32 ± 0.03 to $1.45 \pm 0.02 \text{ g cm}^{-3}$ from grazing score 0–3 ($p = 0.005$; Figure 4c). However, for bulk density, land use/land cover also had a significant effect ($p = 0.001$; Figure 4d), with the highest bulk density ($1.46 \pm 0.02 \text{ g cm}^{-3}$) in open-access forest and the lowest in farms under cultivation ($1.34 \pm 0.02 \text{ g cm}^{-3}$). Mean soil organic carbon decreased to 1/3 with increasing grazing intensity ($p = 0.006$; Figure 4e) from 0.72 ± 0.06 to $0.52 \pm 0.02\%$ (grazing score 0 to 3), but no clear relationship was observed in relation to land use/land cover ($p = 0.22$; Figure 4f).

Soil properties generally improved with the exclusion of livestock grazing. Mean steady-state infiltration capacity in paired plots inside and outside grazing enclosures was near twice the level inside compared to outside ($p = 0.03$; Figure 6a), that is, 442 ± 53 and $279 \pm 49 \text{ mm hr}^{-1}$, respectively. Mean steady-state infiltration capacity for the paired plots outside the enclosures was similar to that for the LDSF plots within the forest reserve (Figure 4b) ($284 \pm 51 \text{ mm hr}^{-1}$) where enclosures were located. Mean bulk density was 1.64 ± 0.01 and $1.45 \pm 0.04 \text{ g cm}^{-3}$ outside and inside enclosures, respectively ($p < 0.001$; Figure 6b). Mean soil organic carbon was about double ($p < 0.001$; Figure 6c) inside the enclosures ($1.46 \pm 0.03\%$) compared to outside ($0.72 \pm 0.03\%$).

3.2 | Infiltration patterns and preferential flow

The degree of preferential flow was affected by livestock grazing intensity but not by land use/land cover type (Figures 7 and 8). Both Total stained area (Figure 7a) and Uniform infiltration depth (Figure 7b) decreased with increasing grazing intensities (1065 ± 59 to $679 \pm 29 \text{ cm}^2$; $p < 0.001$ and 30 ± 3 to $19 \pm 1 \text{ cm}$; $p = 0.004$ for grazing score 0–3 respectively). The preferential flow fraction increased with increasing grazing intensity ($p = 0.012$; Figure 7c) from $18 \pm 5\%$ at grazing score 0 to $29 \pm 3\%$ at grazing score 3. Preferential flow in the bottom 5 cm of the profile (45–50 cm depth) was six-times higher in areas where no grazing was observed ($55 \pm 5\%$) ($p < 0.001$; Figure 7d) than in areas with grazing intensity score 3 ($9 \pm 1\%$), but did not show any clear relationship with land use/land cover type ($p > 0.05$; Figure 7h). Regression analysis between preferential flow and basal area gave a very low r^2 value of 0.009, which suggests no correlation.

4 | DISCUSSION

We hypothesized that in a *miombo* dryland landscape, tree cover would decrease soil bulk density and have a positive effect on steady-

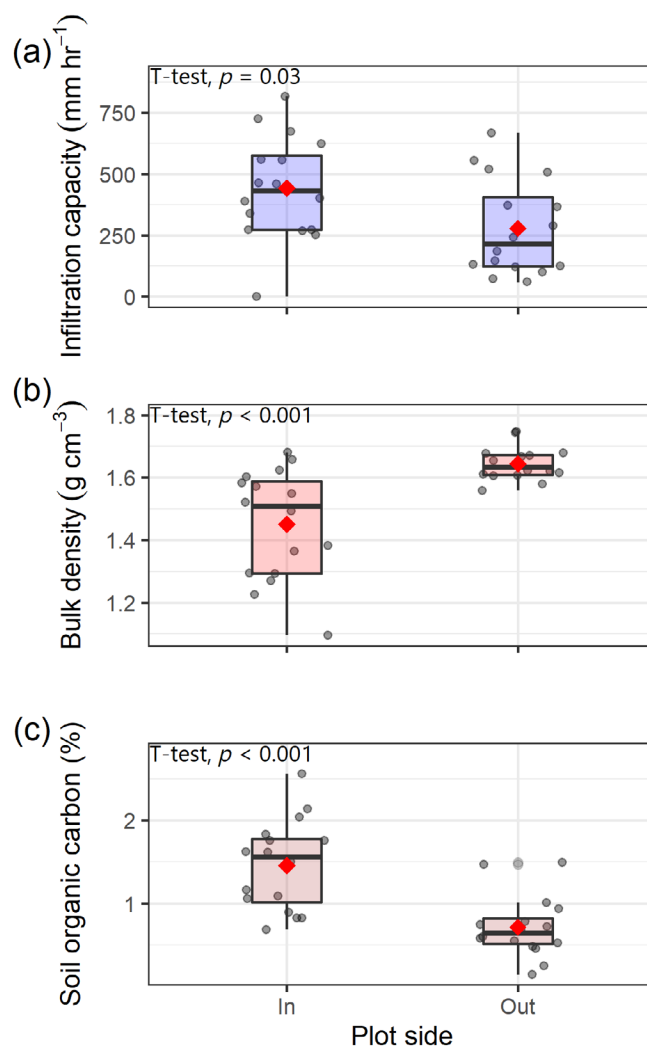


FIGURE 6 Boxplot (median, first and third quartile) of infiltration capacity (mm hr^{-1}) (a), bulk density (g cm^{-3}) (b), and soil organic carbon (%) (c), inside and outside grazing enclosures in Kitulungalo Forest Reserve, Morogoro, Tanzania. Significance values (p) are given. Red dots indicate the mean value [Colour figure can be viewed at wileyonlinelibrary.com]

state infiltration capacity, degree of preferential flow, and soil organic carbon, while livestock grazing intensity would have the opposite effects. As hypothesized, increasing grazing intensity led to higher bulk density and lower steady-state infiltration capacity and soil organic carbon, regardless of land use and land cover type. However, it was observed that, in the absence of grazing (0 grazing intensity score), there was a clear positive relationship between steady-state

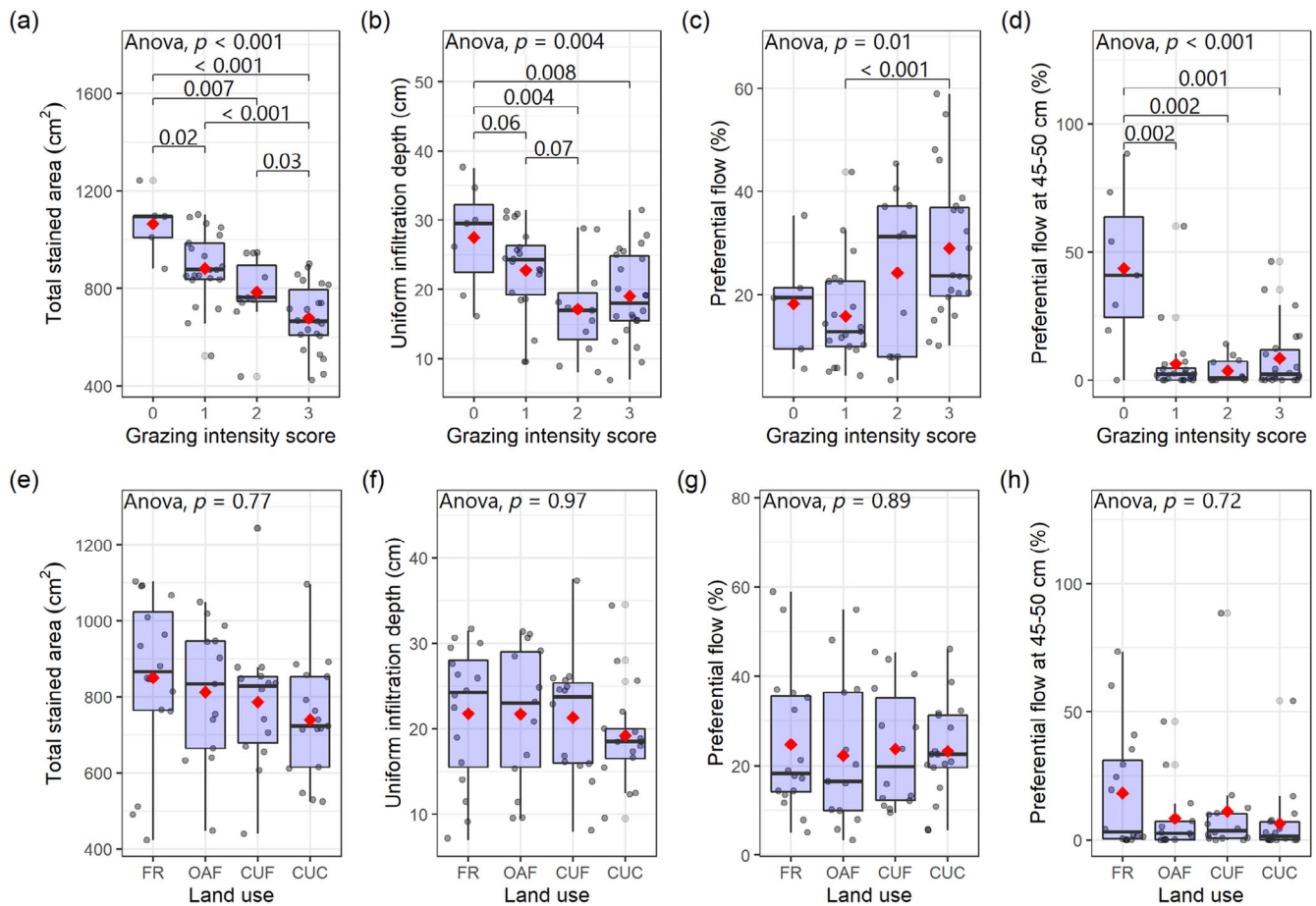


FIGURE 7 Boxplot (median, first and third quartile), of the different preferential flow indices for the different classes of grazing intensity (upper row) and land use/land cover (lower row) within a $10 \times 10 \text{ km}^2$ area in Kitulangalo, Morogoro, Tanzania; significance values (p) are given. Red dots indicate the mean value of each index for the respective grazing intensity and land use/land cover class. (a, e) Total stained area (cm^2), (b, f) uniform infiltration depth (cm), (c, g) preferential flow fraction (%), and (d-h) preferential flow at 45–50 cm (%). FR = forest reserve, OAF = open-access forest, CUF = cropland under fallow, CUC = cropland under cultivation [Colour figure can be viewed at wileyonlinelibrary.com]

infiltration capacity and basal area, which declined with grazing activities regardless of land-use class. This is the reason why all four land use and land cover types, from the forest reserve to cropland under cultivation, had similar steady-state infiltration capacity levels and degree of preferential flow. The preferential flow indices, which considered the entire soil profile, indicated higher preferential flow at high grazing intensities and little influence of land use and the land cover type, the opposite situation to the one we hypothesized. However, preferential flow at 45–50 cm depth, which indicates deep profile drainage, was six-times higher at the lowest grazing intensity compared to areas with high grazing intensities.

Similar to our study, most studies have shown a positive effect of trees on soil hydrological functioning; this has been attributed to their well-established root systems, that improve porosity and soil aggregation and, consequently, increase infiltration and preferential flow (Bargués-Tobella et al., 2014; Benegas et al., 2014; Cardwell, ; Cui et al., 2019; Ekhuemelo, 2016; Kan et al., 2019; Liu et al., 2020; Lozano Baez, 2019; Wu et al., 2021). Considering this, and recurring soil

disturbance that disrupts vertical pore continuity in agricultural lands, forests have been reported to have higher soil infiltration capacity than cultivated land (Fan et al., 2013; He et al., 2009; Ilstedt et al., 2007; Nyberg et al., 2012; Yimer et al., 2008). This was the case in our study area in the absence of grazing. We attribute the absence of a clear effect of trees on soil hydraulic properties in the presence of intensive grazing to the severe soil disturbance caused by livestock. Livestock trampling has been reported to cause soil compaction, decrease soil hydrological functioning (Donkor et al., 2002; Dreccer & Lavado, 1993; Dudley et al., 2002), and reduce soil organic carbon (Dlamini et al., 2016). Similarly, results from our study also indicate an increase in soil bulk density and decreasing soil organic carbon with increasing grazing intensity. In our study area and many other tropical drylands, livestock grazing is mostly undertaken based on the convenience of pasture availability regardless of the primary land use or land cover (Boerma & Koohafkan, 2007). This, coupled with the low biomass production capacity typical of dryland ecosystems, results in an overall decrease in soil organic carbon across landscapes (De Deyn

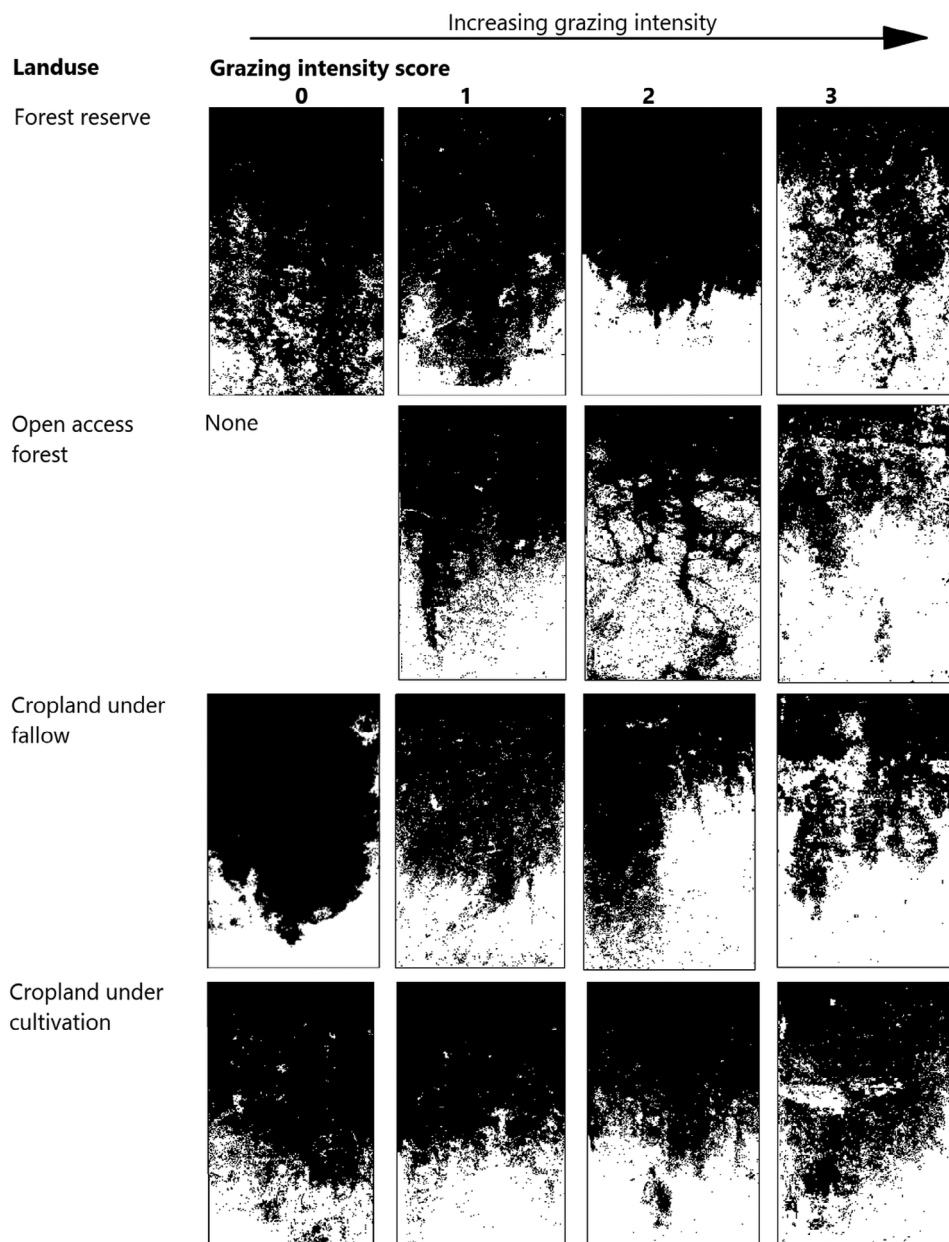


FIGURE 8 Examples of classified stained profiles (black: Dye stained soil, white: nonstained soil) for different classes of grazing intensity and land uses/land cover from a $10 \times 10 \text{ km}^2$ area in Kitulungalo, Morogoro, Tanzania

et al., 2008), which, in turn, negatively impacts soil hydraulic properties. At the same time, high wild forest fire incidence, mostly in woodlands and forested land, reduces the amount of soil carbon, moving it towards the levels similar to those of other less vegetated areas (Mganga et al., 2015; Ryan et al., 2011). Frequent movement of grazing animals over time causes the collapse of the soil structure, particularly in the topsoil, creating a compaction layer, which leads to reduced and uneven distribution of pore space down the soil column (Russell & Bisinger, 2015). Since the rate of infiltration and flow through the soil profile depends on soil porosity as a function of pore size and pore continuity (Osanyinpeju & Dada, 2018), infiltration becomes slower with increasing soil compaction (Zhang et al., 2006).

Unexpectedly, three of four preferential flow indices showed an increasing degree of preferential flow with increasing grazing intensity. Most likely, this is an effect of soil compaction. Many soils have

infiltration patterns characterized by uniform flow close to the soil surface and a higher degree of preferential flow at depth (Zhang et al., 2019). When the topsoil is compacted or eroded, the area of uniform flow is reduced, and in several preferential flow indices, this would appear as an increase in the degree of preferential flow. Another possible explanation for this observation is that uniform flow is higher when there is an even distribution of pore space and water can pass evenly through the soil column, whereas the opposite is the case for preferential flow (Kan et al., 2019). Sandy soils, under normal conditions, typically exhibit a uniform infiltration front due to their coarse texture (Duley & Kelly, 1939). However, livestock trampling may create nonuniform compression patterns in soils that we then see as increasing preferential flow at the same time that infiltration capacity decreases. Because livestock grazing is prevalent across various land uses and land cover types, this could potentially also explain

the absence of land use and land cover effect in our observations. Our findings, however, indicate that preferential flow at 45–50 cm depth was six-times higher for the areas with a zero-grazing intensity score than for the highest grazing intensity, showing the importance of including indices of preferential flow that are independent of measures of dye cover in the topsoil.

Decreased soil infiltration capacity can result in increased surface runoff and ponding of water on the soil surface (Haghnazari et al., 2015) and, consequently, more erosion. Reduced infiltration capacity and preferential flow may also lead to an increased residence time of water in the soil surface and topsoil layer, with additional exposure to evaporation (Bargués-Tobella et al., 2014). This translates to reduced deep soil and groundwater recharge potential (Stako et al., 2012). Our study indicates that there is significantly higher deep drainage (preferential flow at 45–50 cm depth) in areas with zero grazing intensity compared to those areas that are more affected by livestock, and this can be explained by less compaction of the topsoil and presence of vertical continuity of macro-pores at depth. Since deep soil and groundwater recharge depend greatly on deepwater percolation (David et al., 2016), these findings emphasize the need to consider grazing as one of the key factors when managing drylands for local and downstream water resources. While trees play a pivotal role in enhancing soil hydraulic properties, they also use water through evapotranspiration. If increases in tree cover do not lead to enhanced soil hydraulic properties, the net impact of more trees on groundwater recharge will always be negative. Because of this, maintaining or restoring tree cover alone may be ineffective to improve water availability if livestock grazing and other anthropogenic activities that impact soils are not well managed (Ghimire et al., 2013, 2014). Reduced infiltration from high livestock grazing may be a more serious problem in forest land than in other land uses since more water is lost through evapotranspiration from trees. Thus, if tree-based restoration activities in these areas disregard the need to reduce livestock grazing intensity beyond the tree establishment phase, the net impact of trees on local water availability may be negative.

The effect of grazing exclosures was an increase in soil infiltration capacity and soil organic carbon, while bulk density decreased. We attribute the differences to the exclusion of livestock grazing, considering that the two areas (inside and outside the exclosures) had similar properties when the exclosures were installed 12 years ago. Increased infiltration capacity inside the exclosures resulted in increased ground vegetation cover, which, together with tree roots and soil animals, can restore the soil structure after removing the compression agent (livestock). Higher vegetation cover reduces surface runoff and adds plant litter, which, in turn, increases soil carbon, improving soil water holding capacity and, eventually, soil and groundwater recharge (Descheemaeker et al., 2006). Trees and other plants produce root network systems that increase soil aggregation and stability and create macropores that act as pathways for rapid water flow (Guo et al., 2019; Johnson & Lehmann, 2006). This suggests that vegetation might be most effective in improving soil hydrological functioning when livestock grazing intensity is reduced. However, these fences

were simply used to test what happens when there is complete exclusion of livestock grazing activities in a particular location. Because of their limited spatial scale, distribution across land cover classes, and number, results from these fenced areas cannot be extrapolated to the whole study area.

5 | CONCLUSIONS

Unsustainable land-use practices in drylands may accelerate land degradation and render drylands uninhabitable (Oba et al., 2000). We show here that livestock grazing intensity along with tree density is crucial in the sustainable management of water resources in *miombo* drylands. Moreover, overgrazing could override the positive influence of trees on infiltration capacity and eventually on drainage at deeper soil depth. To maintain and enhance soil infiltration capacity and water security, we recommend that: (i) Tree-based restoration efforts in drylands involve the control of livestock grazing intensity beyond the tree establishment phase; (ii) strong policies are put in place to protect dryland forest reserves and other forested areas from excessive livestock grazing; (iii) rangelands measures that restrict grazing pressure and allow the soil to recover are implemented through rotational grazing, enclosures, and so forth. Future research is needed to understand and establish the appropriate grazing intensities management that would benefit both dryland dwellers and ecosystem sustainability.

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DATA AVAILABILITY STATEMENT

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

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