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Land use change, tree cover and livestock in miombo woodlands

Interacting effects on soil carbon and hydrological properties

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Cover: Livestock grazing in miombo woodlands in Kilosa district, Tanzania.
(photo: Lufunyo Lulandala)

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Land use change, tree cover, and livestock in miombo woodlands: Effects on soil hydrological properties

Abstract

Miombo woodlands stretch across eastern and southern Africa and occupy an area of around 2.7 million km^2. These forests provide a wide range of ecosystem services that are essential to the livelihoods of communities around them and play a crucial role in the carbon and hydrological cycles at regional and global scales. However, miombo woodlands are affected by deforestation and forest degradation, mainly due to agricultural expansion, charcoal production, timber and firewood harvesting, and livestock grazing. This thesis aims to assess the impacts of these land uses and tree cover on key soil properties, particularly soil organic carbon and soil hydrological properties such as infiltration capacity, which are important indicators of ecosystem health. The central hypothesis was that tree cover positively influences soil hydrological properties and soil organic carbon, while land uses that involve a decrease in tree cover or disturb the soil have a negative impact. I conducted the studies in two different sites in Tanzania; Kitulangalo forest reserve and the surrounding areas in Morogoro Rural district, and Ulaya mbuyuni village in Kilosa district. I measured infiltration capacity, preferential flow, tree basal area, livestock grazing intensity, and different soil properties, including soil organic carbon, bulk density, and texture. Results show that soil hydrological properties and soil organic carbon increased with increasing tree cover. Hence, croplands had relatively lower infiltration capacity and soil organic carbon than forest land. Both soil organic carbon and soil hydrological properties decreased with increasing livestock grazing intensity across land uses. Findings also indicate that the positive effect of trees on soil hydrological properties and soil organic carbon was significantly reduced when livestock grazing intensity was high. In addition, the combination of croplands and high livestock grazing intensity resulted in lower infiltration capacity and organic carbon than the combination of forest and high grazing intensity. When comparing small and large clearings for charcoal production, large clearings had lower values of infiltration capacity and soil organic carbon. I concluded that forest conversion to croplands reduces soil hydrological functioning and soil organic carbon. Trees could be an important tool to restore soils and their hydrological function in degraded landscapes, but the presence of high livestock grazing intensities reduces their effectiveness.

Keywords: Soil hydrological properties, land use, livestock grazing intensity, miombo woodlands, charcoal production, silvopastoralism, drylands

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Mabadiliko ya matumizi ya ardhi, mfuniko wa miti, na mifugo kwenye misitu ya miombo: Athali kwenye kaboni na sifa za kihaidrolojia za udongo

Muhtasali

Misitu ya miombo imeenea kote mashariki na kusini mwa Afrika na kuchukua eneo la karibu kilomita za mraba milioni 2.7. Misitu hii hutoa huduma mbali mbali za mfumo ikolojia ambazo ni muhimu kwa maisha ya jamii zinazoizunguka na huchukua jukumu muhimu katika mzunguko wa kaboni na kihaidrolojia katika mizani ya kikanda na kimataifa. Hata hivyo, misitu ya miombo huathiriwa na ukataji miti na uharibifu wa misitu, hasa kutokana na upanuzi wa kilimo, uzalishaji wa mkaa, uvunaji wa mbao na kuni, na malisho ya mifugo. Tasnifu hii inalenga kutathmini athari za matumizi haya ya ardhi na mfuniko wa miti kwenye sifa kuu za udongo, hasa kaboni hai ya udongo na sifa za kihaidrolojia za udongo kama vile uwezo wa kupenyeza, ambazo ni viashirio muhimu vya afya ya mfumo ikolojia. Dhana kuu ilikuwa kwamba mfuniko wa miti huathiri vyema tabia ya kihaidrolojia ya udongo na kaboni hai ya udongo, wakati matumizi ya ardhi ambayo yanahusisha kupungua kwa kifuniko cha miti au kuvuruga udongo yana athari mbaya. Nilifanya tafiti hizo katika maeneo mawili tofauti nchini Tanzania; Hifadhi ya msitu wa Kitulangalo na maeneo jirani katika Wilaya ya Morogoro Vijijini, na kijiji cha Ulaya mbuyuni wilayani Kilosa. Nilipima uwezo wa kupenyeza, mtiririko wa upendeleo, eneo la msingi wa miti, ukubwa wa malisho ya mifugo, na sifa tofauti za udongo, ikiwa ni pamoja na kaboni hai ya udongo, msongamano mkubwa na umbile. Matokeo yanaonyesha kuwa tabia ya kihaidrolojia ya udongo na kaboni hai ya udongo iliongezeka kutokana na kuongezeka kwa mifuniko ya miti. Kwa hivyo, ardhi ya kilimo ilikuwa na uwezo wa chini wa kupenyeza na kaboni hai ya udongo kuliko ardhi ya misitu. Tabia zote mbili za kaboni hai ya udongo na kihaidrolojia ya udongo zilipungua kwa kuongezeka kwa malisho ya mifugo katika matumizi ya ardhi. Matokeo ya utafiti pia yanaonyesha kuwa athari chanya ya miti kwenye tabia ya kihaidrolojia ya udongo na kaboni hai ya udongo ilipunguzwa kwa kiasi kikubwa wakati kiwango cha malisho ya mifugo kilikuwa kikubwa. Aidha, mchanganyiko wa mashamba ya mazao na wingi wa malisho ya mifugo ulisababisha kupungua kwa uwezo wa kupenyeza na kaboni hai kuliko mchanganyiko wa misitu na malisho mengi. Wakati wa kulinganisha maeneo madogo na makubwa kwa ajili ya uzalishaji wa mkaa, usafishaji mkubwa ulikuwa na viwango vya chini vya uwezo wa kupenyeza na kaboni ya kikaboni ya udongo. Nilihitimisha kuwa ubadilishaji wa ardhi ya misitu kuwa mashamba ya mazao hupunguza utendaji kazi wa kihaidrolojia wa udongo na kaboni hai ya udongo. Miti inaweza kuwa chombo muhimu cha kurejesha udongo na sifa zake za kihaidrolojia katika mandhari iliyoharibiwa, lakini kuwepo kwa kiwango ch juu cha ulishaji wa mifugo hupunguza ufanisi wao.

Maneno Muhimu: Sifa za kihaidrolojia za udongo, matumizi ya ardhi, ukubwa wa malisho ya mifugo, misitu ya miombo, uzalishaji wa mkaa, silvopastoralism, maeneo kavu

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Dedication

To my precious wife Abela and my amazing son Laurence

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. Lulandala, L., Bargués-Tobella, A., Masao, C. A., Nyberg, G., & Ilstedt, U. (2021). Excessive livestock grazing overrides the positive effects of trees on infiltration capacity and modifies preferential flow in dry Miombo woodlands. *Land Degradation & Development, 33*(4), 581 -595. doi:https://doi.org/10.1002/ldr.4149

II. Lulandala, L., Bargués-Tobella, A., Masao, C. A., Nyberg, G., & Ilstedt, U. (20XX). Soil organic carbon in dry miombo landscape decreases with higher grazing intensity, but trees can counteract the effect. *Manuscript.*

III. Lulandala, L., Bargués-Tobella, A., Masao, C. A., Nyberg, G., & Ilstedt, U. (20XX). The size of clearings for charcoal production in miombo woodlands affects soil hydrological properties and soil organic carbon. *Manuscript submitted to Forest Ecology and Management Journal. Submitted in September 2022.*

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*Corresponding author

The contribution of Lufunyo Lulandala to the papers included in this thesis was as follows:

I. Participated in the planning of the work. Conducted field experiments and sampling. Conducted data analysis. Wrote the manuscript with inputs from all other authors. Acted as corresponding author.

II. Participated in the planning of the work. Conducted field experiments and sampling. Conducted data analysis. Wrote the manuscript with inputs from all other authors. Acted as corresponding author.

III. Participated in the planning of the work. Conducted field experiments and sampling. Conducted data analysis. Wrote the manuscript with inputs from all other authors. Acted as corresponding author.

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Abbreviations

CUC	Cropland under cultivation
CUF	Cropland under fallow
FMU	Forest management units
FR	Forest reserve
KFR	Kitulangalo forest reserve
LDSF	Land degradation surveillance framework
OAF	Open access forest
SCP	Sustainable charcoal project
SOC	Soil organic carbon
SSA	Sub-Saharan Africa
TAFORI	Tanzania forest research institute
TTCS	Transforming Tanzania's charcoal sector

1. Introduction

1.1 Land use, tree cover, and groundwater

Land use - the human use of land - encompasses both the economic and cultural activities taking place in an area, including agricultural, residential, mining, and recreational uses (EPA, 2017). Land use change is a global concern as it is one of the main reasons for environmental change (Sharma et al., 2019). In efforts to provide food, fiber, water, and shelter to the growing global population, forests, farmlands, waterways, and air are being modified globally (Foley et al., 2005). In the course of two centuries, global population has increased by 7-folds to the current approximate of 7.8 billion people (PRB, 2020). Such increase in population comes with an increased demand for basic human necessities, including energy and food, which drives land use change and negatively impacts natural ecosystems through loss of biodiversity, increased greenhouse gas emissions, and the degradation of soil and water resources (Foley et al., 2005).

Soil Organic Carbon (SOC) improves the water-holding capacity of the soil, improves soil structure and stability, and increases microbial activities. (Milne et al., 2015). Considering that soil is a major carbon reservoir - with two times the amount of carbon in the atmosphere and three times that in vegetation (Powlson et al., 2011)- conserving soils is essential both for climatic regulation and agricultural productivity.

Agricultural expansion makes up about 90 % of global deforestation, most of which occurs within the tropics (FAO, 2021; Pendrill et al., 2022; Ramesh et al., 2019). Converting forest land to annual cropland is frequently associated with carbon loss from both the soil and vegetation. The impact of this change depends on several climatic and environmental factors, including temperature, moisture availability, soil characteristics, and frequency of disturbance (Malhi et al., 1999). In the temperate region, the conversion of a natural forest to agricultural land leads to a 60 % loss in soil organic carbon, up to 75 % in the tropics, while sub-Saharan Africa (SSA) has lower carbon loss than in the overall tropics with 63 % (Devi, 2021; Lal, 2004; Vågen et al., 2005). This makes tropical ecosystems, particularly vast and densely populated tropical drylands, more vulnerable to land use changes (Gaur & Squires, 2018).

Trees offer a wide range of services (Bargués-Tobella et al., 2014; Benegas et al., 2014; Ekhuemelo, 2016). Apart from carbon storage and climate regulation, trees maintain biodiversity, prevent soil erosion, improve soil fertility and soil aggregation, regulate the hydrologic cycle, and provide different products like fruits, timber, and fiber (Barrios et al., 2018; Cavender-Bares et al., 2022; Cunningham et al., 2015; Salmond et al., 2016). However, the influence of trees and how different land uses in a landscape, particularly in drylands, interact and collectively affect soil water dynamics is still poorly understood. Through the improvement of soil quality, trees positively influence soil hydrological properties. Soil hydrological properties are complex and are influenced by several factors both inherent and management-dependent, including vegetation cover, soil texture, soil organic matter content, and land use (Lozano-Baez et al., 2021). In the face of climatic changes, drylands are confronted with even bigger risks of increased drought spells and temperatures (Easterling et al., 2000; Hughes, 2003). To avoid degradation, it is important to understand how different co-occurring land uses in dryland landscape influence hydrological processes and how to optimize the land use-soil hydrology relationship for conservation. However, information on how different landscape components, whether separately or in interaction, affect soil hydrological properties in drylands is still scarce but highly needed.

Groundwater, sometimes referred to as subsurface water, is that water found beneath the Earth's surface, in rock and soil pore spaces, and in fractures of rock formations (Holmes, 2000). Groundwater makes up approximately 30.1 % of all freshwater worldwide, while 68.7 % is fixed in ice caps and the other 1.2 % is surface freshwater in the form of lakes, rivers, and dams (Gleick, 1996). Groundwater comes from precipitation and the process by which water drains deep into the ground is called recharge (Ajami, 2021). Groundwater is a vital source of drinking water. Globally, about 50 % of all drinking water comes from groundwater (Beckie, 2013) and it also accounts for 38 % of all water used for irrigation (Siebert et al., 2010). In addition, groundwater supports rivers, lakes, and wetlands, particularly during the drier month when there is little or no rain input; hence, it supports the biodiversity of plants and animals (Aylward, 2005). All these make groundwater a precious item that needs conservation, particularly in drylands with limited rainfall. However, both groundwater quality and quantity are being threatened by over-exploitation from uncontrolled wells drilling (Chesnaux, 2012), contamination from both industrial wastes and chemical agricultural fertilizers (Hansen et al., 2012), reduced recharge quantities due to the disruption of groundwater recharge systems as a result of land use changes, as well as climatic changes (Green et al., 2011).

Soil hydrological processes, including water infiltration and preferential flow, runoff generation, soil-water storage, redistribution, drainage, evaporation, and transportation, have important consequences on the overall soil water budget (Zhang et al., 2016). Water flows from above ground into the subsurface through infiltration, and the maximum rate at which soil can absorb water in a given condition is termed infiltration capacity (Ferré & Warrick, 2005) or infiltrability (Hillel, 2003). Once on the ground, water movement through the soil is influenced by two major forces: capillary and adsorptive forces. When soils are not yet saturated, water moves downward by the adhesion force, however, when the soil is near a point of saturation matric potential decreases and large pores in the soil are filled and water moves rapidly through them by the gravitational pull downward (Voroney, 2019). The relatively slow and even movement of water and solutes through the soil in pores that are small enough to retain water against the force of gravity is called matrix flow (Zhang et al., 2017), while the rapid, uneven movement of water and solutes through regions of higher flux in larger soil pores such as from root and animal channels and cracks is known as preferential flow (Lei Guo et al., 2019). Hydrological processes are complex, with spatial and temporal variabilities, and are influenced by many natural and management-dependent factors (Easterling et al., 2000; Eger et al., 2017; Gwak & Kim, 2017; Hughes, 2003; Yair & Raz-Yassif, 2004). Destruction of macro-pores will decrease infiltration capacity, and hence that risks reducing the flow of water needed to recharge groundwater reserves during intense rain events in tropical areas (Bargues-Tobella et al., 2020). In order to avoid degradation, it is essential to understand how different land uses in dryland landscape influences hydrological processes and how to optimize land use-soil hydrology relationships for soil and water conservation.

1.2 Drylands

Drylands cover about 40 - 45 % of the world's land area (Huang et al., 2016; Schimel, 2010), and support around 2.1 billion people worldwide of whom 90 percent live in low and middle-income countries (UN, 2020). Africa is the continent with the largest share of the total global dryland area (32%), followed by Asia, North America, South America, and Europe (Maestre et al., 2021). Drylands can be broadly defined by using an aridity index, which is a quantitative indicator of the degree of water deficiency calculated as the ratio of mean annual precipitation to mean annual potential evapotranspiration (UNEP, 1992). UNEP classifies drylands as areas with an aridity index of less than 0.65, that are characterized by high evapotranspiration rates exceeding the available rainfall or snowfall (UNEP, 1992). Based on the aridity index, drylands can further be classified into hyper-arid (< 0.5), arid (0.05 – 0.20), semi-arid (0.20 – 0.50), and dry sub-humid (0.50 – 0.65) making a major ecosystem in both tropical and temperate regions throughout all continents (Prăvălie, 2016; UN, 2011). Despite the level of aridity, drylands support a diverse mosaic of contrasting landscapes and biodiversity of both plants and animals found in these ecosystems, as well as human communities (Chakrabarti, 2016). Drylands are extremely vulnerable to climatic variations and anthropogenic activities that include unsustainable agriculture practices, deforestation, and overgrazing (Davies et al., 2012).

1.2.1 Land use in drylands

Land use in dryland regions is highly dynamic and not fully understood (Fu et al., 2021). Land use dynamics are influenced by natural factors like climate, soil, and topography, as well as socioeconomic factors, including population, economic status, and culture (Gaur & Squires, 2018). Globally, of the 6.1 billion hectares of drylands, 28 % of it is considered barren/unproductive land too dry to support life, 25 % is classified as grasslands, 18 % as forest land, and 14 % as croplands (Figure 1). It is estimated that drylands support up to 44 % of the world's cultivated systems and are the source of 50 % of the world's livestock production (Chakrabarti, 2016). Traditionally, drylands have primarily been used for livestock production, particularly in arid and semi-arid climates, but reports show an ever-increasing rate of rangeland conversion into cropland (UN, 2018). Since global livestock production also keeps growing (Pandey & Upadhyay, 2022), this could imply an increasing overlap of the two land uses (livestock keeping and crop cultivation), as it has also been reported in the drylands of Africa (Mortimore, 1991). There is a need for thorough studies on how these conversions of rangelands to croplands and the overlapping of livestock grazing and cropping land uses affect different aspects of dryland ecosystems.

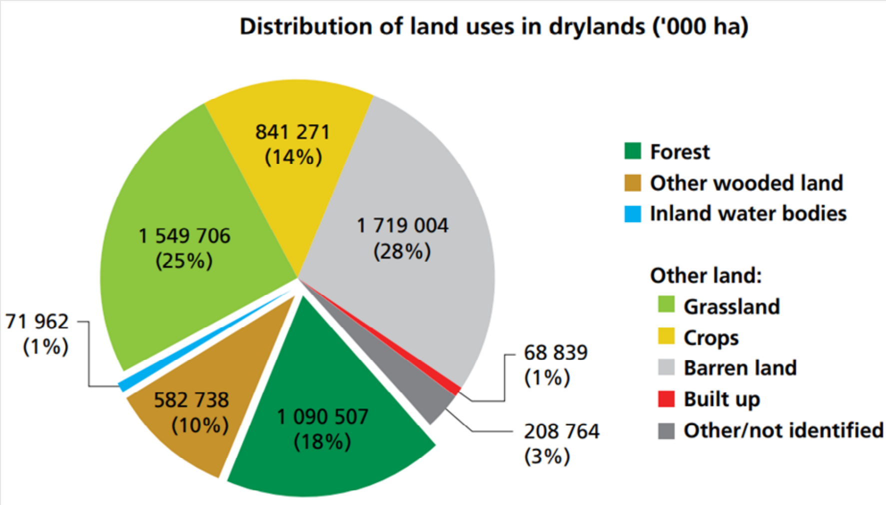

Figure 1. Major land use classification, land area coverage, and percentages in the global drylands (FAO, 2019).

Compared to other ecosystems, drylands display a uniquely close interdependence between nature and dryland dwellers that is mainly attributed to limited available resources (Sietz, 2011). Approximately 1/3 of the population living in drylands depends on agriculture for their livelihood (UN, 2021). This close interdependence has been and still is one of the most substantial influencing factors on spatial-temporal land modification in drylands (Silva et al., 2016). The population density in drylands increases with decreasing aridity, from 10 people km^{-2} in hyper-arid areas to 71 people km^{-2} in dry sub-humid areas (Gaur & Squires, 2018). At the same time, drylands are home to the poorest and most marginalized communities in the world, and 16 % of their population live in chronic poverty and hence are greatly dependent on ecosystem services for their survival (UNCCD & DIE, 2019). Population growth in drylands is not, in principle, a direct driver of environmental degradation per se, but it increases the pressure on natural resources (Milas, 1985). Population growth is expected to increase further the demand for, e.g., settlement land and agricultural and animal-based products, which in turn may lead to shorter fallow periods (Pelzer, 1964) and more encroachment on rangelands, woodlands, and forests, resulting in even more land and ecosystem degradation (Spinoni et al., 2021). Evidence on the implications of these modifications on soil and water resources in drylands is scarce but highly relevant and urgently needed.

Although dryland communities also depend on crop cultivation for their livelihood, dryland climate does not offer favorable conditions to support crop cultivation (Gimenez et al., 1997). Apart from water limitations due to low and unreliable rainfall coupled with high evapotranspiration rates (Miller, 2005), dryland agriculture is also faced with other severe biotic and abiotic challenges, including declining soil quality, pests and diseases, and climate change (Holmgren et al., 2006; Venkateswarlu & Shanker, 2012). Dryland soils are characterized by low fertility, low organic matter content, and are easily eroded by wind and water (Venkateswarlu & Shanker, 2012). Considering these poor characteristics and the fact that 80 % of the world's agricultural land area is rain-fed (Lamptey, 2022), drylands are far disadvantaged and often characterized by low productivity and land degradation (Parr et al., 1990). To efficiently manage dryland farming systems, considerable quantities of inputs in the form of nutrients, irrigation systems, and appropriate management strategies are needed. However, these options are economically inaccessible to the majority of dryland farmers, and most of them opt for shifting cultivation (Gordon et al., 2013). Shifting cultivation, which is a form of cultivation where farm plots are temporarily abandoned for vegetation to grow while the farmer moves to another location (Hillel, 2008), has been one of the prominent factors for vegetation manipulation and land modification in drylands.

As a result of low organic matter content, dryland soils have low aggregate stability and hence are highly vulnerable to degradation (Chen et al., 2022). In addition, drylands are characterized by limited water availability, which limits primary productivity. As a result, soil organic carbon (SOC) (Ramesh et al., 2019) is low compared to more humid systems, and vegetation is often scattered or sparsely distributed, leaving the soil exposed to direct agents of degradation like rain, overland flow, sunlight, and wind (Reynolds, 2001). Poor qualities of soils in the drylands make them highly vulnerable to common unsustainable land uses like overgrazing and shifting cultivation.

1.3 Miombo woodlands

Miombo is a common term used to identify the most extensive tropical seasonal woodland and dry forest formation in Africa, covering an estimated area of 2.7 - 3.6 million km^2 across the central African plateau and its escarpment (Campbell et al., 1996; Frost, 1996). Miombo woodlands cover about 10 % of the African landmass (Figure 2) (Malmer, 2007) and extend from Tanzania and southern parts of the Democratic Republic of Congo (DRC) in the north to Zimbabwe in the south, and across the continent from Angola, through Zambia, to Malawi and Mozambique (Walker & Desanker, 2004). The overstory in Miombo woodlands is mainly dominated by the genera *Braychystegia, Julbernadia,* and/or *Isoberlinia* (Leguminosae, sub-family Caesalpinioideae) (Williams et al., 2008). The miombo region has an estimated 8500 species of higher plants, over 54% of which are endemic (Rodgers et al., 1996). Miombo woodlands have been widely classified into two types based on the amount of rainfall they receive, dry miombo (< 1000 mm/year), and wet miombo (> 1000 mm/year) (Munishi et al., 2011; White, 1983), which also vary slightly in their vegetation structure and composition. Dry miombo is usually characterized by a lower canopy height (< 15 m) and canopy cover (30 - 60 % of the ground), and a lower floristic composition compared to wet miombo (Ribeiro et al., 2020). Miombo woodlands are characterized by the presence of trees with an umbrella-shaped canopy, scattered sub-canopy trees, a discontinuous layer of understory saplings and shrubs, and patchy grasses and forbs (Frost, 1996).

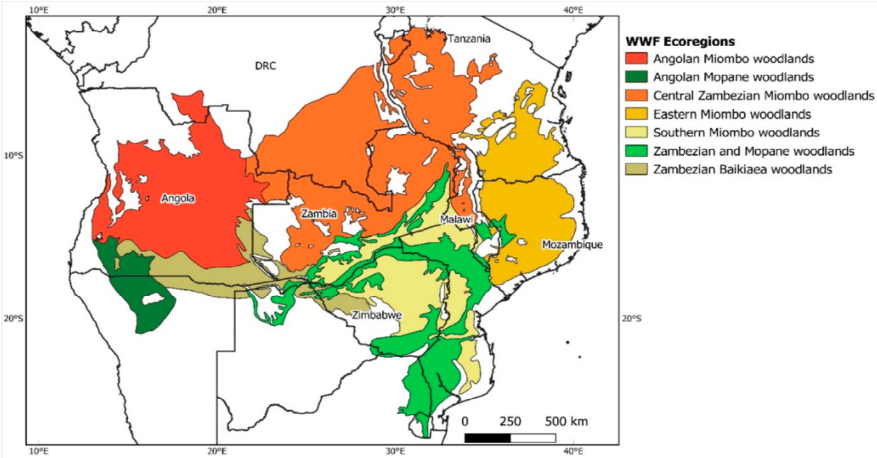

Figure 2. Map showing the distribution of miombo woodlands in eastern, central, and southern African countries (Source: Maquia et al., 2019).

1.3.1 Land use/ land cover dynamics in miombo woodlands

Over 100 million rural and urban communities directly or indirectly depend on miombo woodlands for their livelihood, through the provision of goods and services like timber and non-timber products, including fruits, mushrooms, traditional medicines, and bee products (Chirwa et al., 2008). About 80 % of the communities living in miombo woodlands are agro-pastoralists, practicing livestock keeping and farming as their main economic activities together with some charcoal production (Njana et al., 2013). Poor management and increased utilization pressure have led to extensive miombo degradation.

For a long time, shifting cultivation has been the standard agricultural practice in the miombo region (Grogan et al., 2013; Kilawe et al., 2018; Luoga et al., 2000; Stromgaard, 1988). It is mainly practiced on a subsistence basis by small-scale rural farmers, often with small farm sizes of < 2 ha (Ribeiro et al., 2013). This mode of farming has widely been used in many parts of the world, including Southeast Asia and South America, and it is still widely practiced in both dry and humid Sub-Saharan Africa (Hillel, 2008). It involves the rotation of farming fields with variation in fallow times ranging from 1 to 20 years depending on the rate of recovery and demand for arable land (Kilawe et al., 2018). In this form of cultivation, soil fertility is restored by long periods of fallowing through the recycling of nutrients between vegetation rather than by off-farm inputs of fertilizers (Lal, 2005). Shifting cultivation systems are ecologically viable even in harsh and fragile ecosystems like the tropical drylands as long as there is enough land for long restorative fallow (Lal, 2005). However, soaring population growth in Sub-Saharan Africa and increased demand for settlement areas and arable lands has led to high pressure on natural ecosystems like miombo woodlands and, thereby, the shortening of fallow periods (Dalle & De Blois, 2006). Consequently, this form of cultivation has generally been identified as the major cause of land cover changes and deforestation in the tropics (O'Brien, 2002). Under extreme reduction of fallow periods, productivity is drastically reduced, leading to system breakdown and severe soil degradation (Alajangi et al., 2021).

Livestock grazing in miombo woodlands

Livestock keeping is one of the main economic activities of dryland communities like those living in miombo woodlands (Njana et al., 2013; Powell et al., 2010). Miombo woodlands have been used as grazing lands by indigenous communities for a long time (Ruvuga et al., 2020), however, studies of how livestock grazing influence miombo ecology are still scarce. Livestock supports an estimated 70 % of the rural dryland population of West and East Africa, where about 20 % of these livestock keepers depend exclusively on livestock (pastoralists), while the rest derive a portion of their income from cropping (agro-pastoralists) (Cornelis de, 2016). The increasing trend of the global demand for animal-based products is the main reason for the current increasing pattern of livestock keeping in drylands (Abdallah & Monela, 2007; Gumbo et al., 2018).

Studies of how livestock grazing affects the miombo ecosystem are few, and most of that available focus on vegetation stratum. Such studies show that overgrazing reduces regeneration and species diversity in the long run (Mtimbanjayo & Sangeda, 2018; Nduwamungu et al., 2009; Sangeda & Maleko, 2018), but how livestock grazing affects soil properties, particularly hydrological functions, is still not well understood.

Charcoal production in miombo woodlands

About 65 % of the world's charcoal is produced in Sub-Saharan Africa (SSA) (Mensah et al., 2020), with Nigeria, Ethiopia, the Democratic Republic of

Congo, Mozambique, Tanzania, Ghana, and Egypt being the top seven producers (Mensah, 2021). Global sustainable development goal number 7, directing toward access to affordable, clean, and sustainable energy in low- and middle-income regions by 2030 (UN, 2015). However, charcoal is still the primary domestic fuel in most SSA countries (Doggart et al., 2020), where more than 80 % of households use firewood and charcoal for cooking and heating (Meng et al., 2021). Charcoal production is one of the main sources of disturbance in miombo woodlands (Sedano et al., 2016), and it is closely associated with many environmental problems that have both regional and global scale consequences, including deforestation, forest degradation, pollution and contributes to climate change (Doggart et al., 2020; Ramanathan & Carmichael, 2008). Wood fuels alone are estimated to generate around 1.9 to 2.3 % of the global greenhouse gas emission (Bailis et al., 2015), and with a rapidly growing population in SSA, which is predicted to double by 2050 (Shepard, 2019), this also means the increase in demand and consumption of wood fuels if no appropriate interventions are implemented.

Charcoal production is not considered a main economic activity for most SSA communities, but rather is practiced to obtain financial support during dry seasons when farming is not an option or when clearing vegetation during farm preparations (Mabele, 2019). These activities are often done without any form of licensing or permits and hence are not well documented or regulated (Sedano et al., 2016). Around this region, charcoal is produced mainly using traditional earth kilns with low efficiency of around 10 to 20 % depending on different parameters like humidity, the wood size used, and the overall control of the carbonization process (Schure et al., 2019), while improved kilns range between 30 to 42 % efficiency (Adam, 2009). Lack of technical know-how and poor technologies are the two main reasons for low productivity among traditional charcoal producers and low charcoal quality output (Adam, 2009). Because of this, much more wood is cleared to cover kiln inefficiencies and low charcoal selling prices due to poor charcoal quality, increasing deforestation and forest degradation rates.

The harvesting of wood for charcoal production is usually done by two methods: i. Selective cutting, where specific tree species capable of producing high-quality charcoal are identified based on their dimensions and wood density (Chidumayo & Gumbo, 2013). Selective cutting may eventually lead to the degradation and disappearance of some species mostly preferred for charcoal production (Silva et al., 2019), ii. Clear felling, where all vegetation is cut down irrespective of species. When clear-felling for farm preparation, the land is then used for crop cultivation; otherwise, it is left to regenerate over time, and open to other land uses like livestock grazing (Jew et al., 2016). Studies on how these two contrasting wood harvesting methods for charcoal production in miombo woodlands affect an ecosystem are still lacking, and the few available are mainly focused on the vegetation (FAO, 2010; Kutsch et al., 2011; Zulu, 2010). However, studies from other ecosystems show that different soil characteristics, like physical and chemical properties, are reactive to variations in tree cover (Cui et al., 2005) and clearing sizes for charcoal wood harvest even without the presence of agricultural activities. There is still the need for studies in miombo woodlands aiming to understand the implication of charcoal production activities especially on the under-explored and yet important ecosystem component, the soil.

1.4 The study hypotheses

The general hypothesis was that in the miombo landscape, tree cover positively influences soil infiltration capacity, preferential flow, and soil organic carbon. I hypothesized that land uses within the miombo landscape affect soil hydrological properties and soil organic carbon and that the land use effect increases when more than one land use occurs simultaneously within the same area (figure 3). From these main hypotheses, I developed the following more specific hypotheses;

Hypothesis 1: High tree cover improves soil hydrological properties across the studied land uses in miombo woodlands, and hence the more the tree cover, the more the infiltration capacity and preferential flow (Paper I & III). However, livestock grazing counteracts this effect (Paper I).

Hypothesis 2: Soil organic carbon and hydrological properties in miombo woodlands decrease with increasing livestock grazing intensity, but more tree cover can counteract the negative effects of livestock (Paper I and II).

Hypothesis 3: Livestock grazing affects soil hydrological properties and soil organic carbon negatively in miombo forests, but these properties recover after excluding livestock and allow vegetation to recover (Paper I & II).

Hypothesis 4: The studied land uses with less tree cover (agricultural land and forest clearings for charcoal production) have less soil organic carbon content than those with more tree cover (forests and fallows), but the level of grazing intensity influences this effect (Paper II and III).

Hypothesis 5: Large forest clearings for charcoal production are more detrimental to soil hydrological properties and soil organic carbon than small clearings.

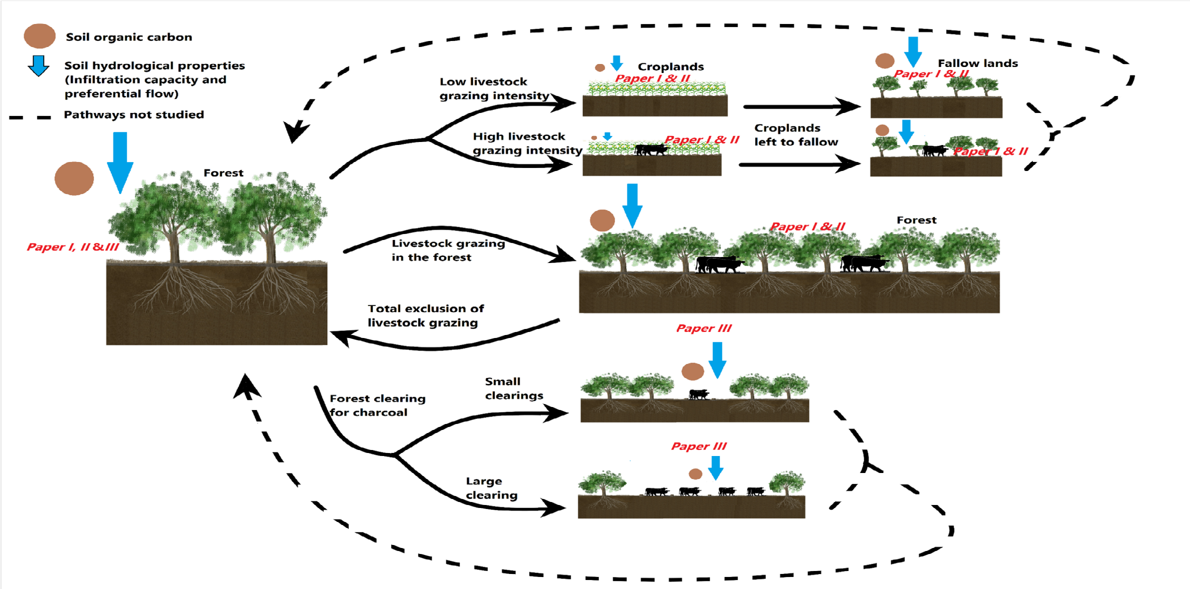

Figure 3. Conceptual framework of common land use change dynamics within miombo landscapes and their influence on soil hydrological properties and soil organic carbon. In the forest, there is high biomass input, deep root activity, good soil structure and stability, and microorganism activity, and hence better soil hydrological properties, less surface runoff, and higher soil organic carbon. In croplands, there is shallow root activity and exposed and disturbed soil with low addition of biomass which leads to poor hydrological properties, poor soil structure, increased surface runoff, and less resilient soil to grazing disturbance. In fallows, the soil is recovering from the cultivation disturbance, There is less soil protection compared to the forest, low biomass accumulation, and relatively weak soil structure, and hence vulnerable to disturbance by livestock grazing. Forest clearings for charcoal production leave the land exposed to degradation, large clearing means more land area is exposed and vice versa is true. Across these different land uses, increasing livestock grazing intensity leads to reduced soil organic carbon and hydrological functioning.

2. Objectives

The overall aim of this thesis is to attain better understanding on how the major land uses and their collective effect influence soil hydrological properties and soil carbon in miombo woodlands.

Specific objectives of the studies described in the papers were;

- To analyze the effect of grazing intensity and tree cover on soil hydrological properties in protected and cultivated miombo woodlands (Paper I & III).

- To assess and understand the effect of grazing intensity and tree cover on soil organic carbon in miombo woodlands (Paper I & II).

- To assess how charcoal harvesting clearing size affects soil hydrological properties and soil organic carbon (Paper III).

3. Material and Methods

3.1 Study sites

Fieldwork was conducted in two different study sites (Figure 4). Paper I and II took place in Kitulangalo Forest Reserve (KFR), which is located in Morogoro Rural District, 35 km northeast of Morogoro Municipality along the Morogoro - Dar es Salaam highway. The study for paper III was conducted in Ulaya Mbuyuni village, Kilosa district, approximately 300 km inland from Dar es Salaam. I chose KFR and the surrounding area because it gave me not only the possibility to study different land uses within the miombo landscape but also the land cover management effect because of the presence of the forest reserve. Furthermore, I chose Kilosa because of the presence of the *Transforming Tanzania's Charcoal Sector* project, which allowed me to study charcoal production with controlled clearing sizes of known time since harvesting. The soils of both study areas can generally be classified as ferralsols with sandy clay-loam texture (Msanya et al., 1995). These are weathered soils characterized by low inherent fertility (Kögel-Knabner & Amelung, 2014).

The vegetation of both study areas is typical of miombo woodlands and dominated by trees of the *Fabaceae* family (subfamily Caesalpinioedae within genera *Brachystegia, Julbernardia,* and *Isoberlinia*) (Kajembe et al., 2013; Nduwamungu et al., 2009).

3.1.1 Kitulangalo forest reserve

Kitulangalo forest reserve covers the ridge between the Morogoro-Dar es Salaam highway and the Sangasanga River between an altitude of 350 – 774 m above the mean sea level (Mwandosya et al., 1998). The climate of KFR

is tropical dry sub-humid with mean annual rainfall and temperature of 850 mm and 24.3° C, respectively (Holmes, 1995).
Kitulangalo forest reserve was established in 1955 for water catchment protection and conservation (SUA, 2018). The government first classified it as a "productive reserve", denoting that wood harvesting is allowed under license. However, the status of KFR was changed in 1985, prohibiting any harvesting, although illegal encroachment for wood harvesting and livestock grazing still occurs (Hammarstrand & Särnberger, 2013).

3.1.2 Ulaya Mbuyuni

Ulaya Mbuyuni village is one of the 30 villages where the Sustainable Charcoal Project (SCP) is being undertaken as a pilot project within the TTCS. According to the village land use plan classification, there were two land use classes within my study site; village land forest reserve set aside for protection purposes and forest management units (FMU) used for charcoal harvesting. Within the FMUs, I identified several small clearings (50 × 50 m) of varying ages (harvested from 2016 onwards) in a checkerboard pattern and a large clearing (300 × 300 m) harvested in 2013. Both village land forest reserve and large clearing had small roads passing beneath them.

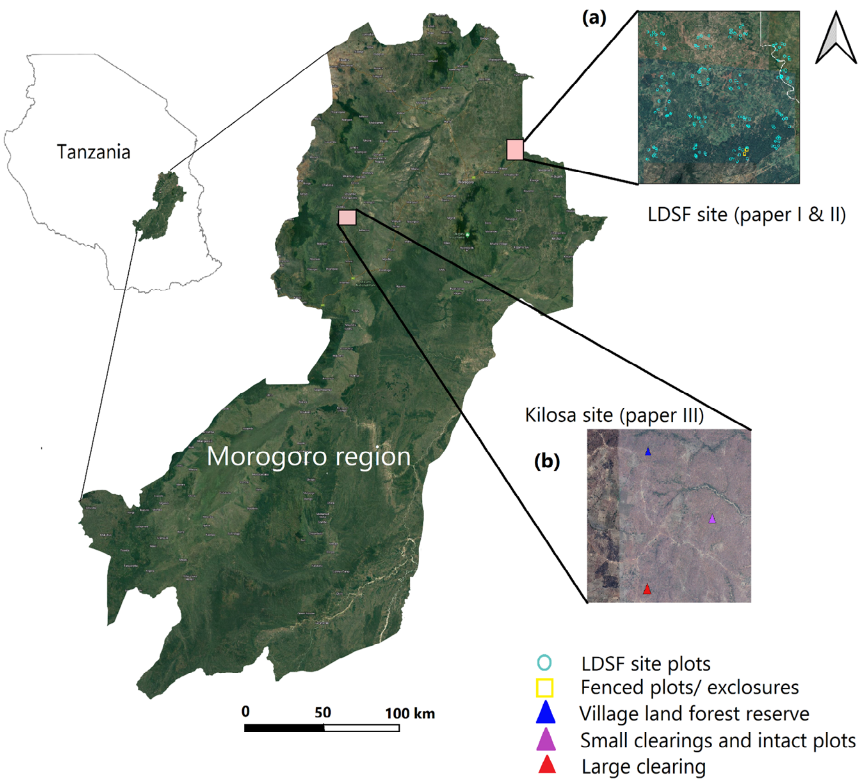

Figure 4. Map showing the location of the two study sites in Tanzania: a) LDSF site in Kitulangalo forest reserve and surrounding areas in Morogoro Rural District, b) Kilosa site in the Ulaya mbuyuni village in Kilosa District.

3.2 Experimental design

3.2.1 Papers I and II

I designed the sampling protocol for papers I and II to assess the effect of livestock grazing, land uses, and tree basal area on steady-state infiltration capacity (Hillel, 2003), preferential flow, and SOC at a landscape level. I sampled at a landscape level to capture the effects of different land uses and land covers. I adopted the Land Degradation Surveillance Framework (LDSF), which is a hierarchical sampling protocol (Vågen & Winowiecki, 2020), that involved the establishment of a 100 km^2 (10 ×10 km) size site with a central coordinate 6° 38´1´´S, 37° 58´46´´E, covering the northeastern KFR and the surrounding landscape. The site is divided into 16 (4 × 4) tiles of 2.5 × 2.5 km; within each tile, random centroid locations for clusters were generated. Each cluster consisted of 10 plots of 1000 m^2 size randomly stablished, making a total of 160 plots for the whole site. Each plot had four sub-plots, each with an area of 100 m^2.

I also designed a separate study to test the effect of total livestock exclusion by using two 12-years old fenced 30 × 90 m^2 plots within the forest reserve. The two fenced plots were established by the Tanzania Forest Research Institute (TAFORI) in 2005, to investigate and quantify the effect of anthropogenic activities within the forest.

I combined interviews with the communities in villages surrounding the KFR on the land use history and land cover changes with visual assessments. This enabled me to classify each LDSF plot into four primary land use and land cover types, namely: forest reserve, open-access forest, cropland under fallow, and cropland under cultivation. The classifications were based on the following definitions;

- 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.
- 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.
- Cropland under fallow (CUF): Croplands that have not been cultivated for at least the past 5 years.
- Cropland under cultivation (CUC): Areas that have been cultivated at least during the last growing season.

For vegetation assessment, I measured and counted all trees (woody vegetation taller than 3 m and with DBH greater than 5 cm). I then used this data to calculate the basal area for each of the four land use and land cover types (Table 1).

Table 1. 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 of papers I and II.

Land use/ land cover	Basal area (m^2ha^{-1})	Stem density (stems ha^{-1})	Number of plots (paper I) Topsoil	Number of plots paper II	
				Topsoil	Subsoil
Forest reserve	5.3 (0.6)	904 (22)	36	36	15
Open access forest	1.7 (0.1)	590 (13)	39	40	17
Cropland under fallow	0.6 (0.1)	285 (8)	46	35	14
Cropland under cultivation	0.2 (0.1)	81 (5)	38	38	13

In each cluster of 10 plots, I took one soil infiltration capacity measurement and performed a blue dye tracer experiment in four randomly selected plots, totaling 64 plots for the entire site (Paper I). By the time I identified the 64 plots, I did not have plots with a fallow time below five years; hence, croplands under fallow did not include shorter fallows. For paper I, I collected only the topsoil samples in all plots. For paper II, I collected topsoil soil samples in 149 plots after removing 11 plots that fell in a group with fallows below five years. Since I did not have the resources to collect subsoil samples for all 149 plots, I randomly selected 59 plots to collect subsoil samples. The 59 plots were selected randomly within land use based on the fractional representation of different land uses on all 149 plots to avoid oversampling of some land uses over others (Table 1). I also established 16 pairs of sampling points inside and outside each of the exclosures for infiltration measurements and soil sample collection (papers I and II).

3.2.2 Paper III

I designed the sampling for paper III to test the effect of clearing sizes for charcoal production on soil infiltration capacity and SOC. I established four study treatments in the study area: Village land forest reserve, large clearing (300 × 300 m), small intact plots (50 × 50 m), and small clearings (50 × 50 m). The small intact plots and small clearings were all within the checkerboard pattern harvested plots. For the village land forest reserve and large clearing, I established 6 parallel 90 m long transects, 50 m apart, starting from a road. Along each transect, I located sampling plots at a distance of 10, 50, and 90 m from the road resulting in 18 sampling plots per site. For the checkerboard pattern harvested clearings, I established a total of 44 sampling plots with a radius of 6 m at the center of both small clearings (22 plots) and small intact plots (22 plots).

3.3 Field measurements and sampling

3.3.1 Soil infiltration capacity and preferential flow

In each sampling position, I measured infiltration capacity using a single-ring infiltrometer with a height and diameter of 27 cm and 30 cm, respectively. I filled the ring and recorded the water level at an interval of 5 minutes in the first half an hour, and then at an interval of 10 minutes until steady-state infiltration was reached (Hillel, 2003). After each data reading, I filled the ring to the original water level. I obtained an infiltration rate by subtracting the final reading of an interval from the original reading of the water level (200 mm) (Figure 5).

After the infiltration capacity measurement, I used a dye solution to study the water flow pattern. I poured fourteen liters of Brilliant Blue FCF (C.I.42090) solution at a concentration of 4 g L^{-1} into the ring to a water level of 200 mm and let it infiltrate completely. Half an hour later, I dug a 0.45 m wide and 2 m long, and 0.6 m deep pit cut across the dye-stained surface to expose a vertical stained soil profile for photographing. Photos were taken using a Nikon D5200 camera with a 35 mm focal length mounted 1.5 m from the center of a graded frame with dimensions of 0.3×0.5 m^2.

3.3.2 Soil samples collection

I collected soil samples by digging a 50 cm deep pit. Samples were collected from one of the pit walls at two consecutive depth intervals, 0 - 20 cm for topsoil and 20 – 50 cm for subsoil. I collected bulk density soil with a stainless cylinder of volume 98.17 cm^3 (5 cm height and inner diameter) at the middle of the depth intervals for topsoil and subsoil. For the LDSF plots, I collected soil samples at the center of each of the four subplots within a plot, then mixed them to obtain a composite sample for the plot.

3.3.3 Grazing intensity

I studied grazing intensity by establishing an index called grazing intensity score. The score rated the visible impacts of livestock grazing through observation of several indicators, which were; (i) signs of livestock presence (droppings, sounds, etc.); (ii) animal paths and hoof prints on the soil surface; and (iii) grazed vegetation. I assigned a value between 0 and 3 for each parameter separately according to its severity (where 0 = no sign observed, and 3 = severe condition observed).

Figure 5. Land uses and experimental procedures for studying infiltration capacity and preferential flow: (a) Croplands (U. Ilstedt), (b) Wood stalking for charcoal production (U. Ilstedt), (c) Fenced plot/ exclosure (U. Ilstedt), (d) and (e) Livestock grazing in miombo woodlands, (f) Single ring infiltrometer, (g) Preparation of a stained soil profile for photo taking, (h) A graduated frame for photo taking, (i) Dye-stained soil profile ready for photographing, and (j) Classified dye-stained image.

3.4 Laboratory analyses

3.4.1 Soil organic carbon, texture, and bulk density determination

All soil samples for organic carbon determination were analyzed using the Walkey-Black wet oxidation method (Bremner & Jenkinson, 1960). Soil texture was determined with the hydrometer method that calculates the physical proportions of soil particles based on their setting rates in an aqueous solution (Bouyoucos, 1936). Bulk density samples were analyzed using the oven-dry method to constant weight at 105° C (Blake & Hartge, 1986).

3.5 Data analysis

I performed all the analyses by using R-studio version 3.6.1 (R Core Team, 2019). For papers I and II, because of the hierarchical nature of the sampling design, I started by checking for autocorrelation based on clusters as a random effect. I found an extremely low auto-correlation which allowed me to use regular regression models when analyzing steady steady-state infiltration capacity (paper I). However for paper II when studying soil organic carbon, I observed a correlation between observations within similar land uses, and hence I used a mixed effect model with land use as a random variable.

3.5.1 Estimation of steady-state infiltration capacity and degree of preferential flow (Paper I, III, and IV)

I estimated steady-state infiltration capacity from the measured infiltration rates using a self-starting *SSphilip* function from the "HydroMe" package in R (Omuto, 2013). I fitted and analyzed all measurements from the single-ring infiltrometer to get the final steady-state infiltration capacity values for each plot.

I classified all photographs from dye-stained soil profiles into stained and non-stained areas using supervised classification in ERDAS Imagine-version 9.2 (Erdas Inc, 2008). Then, I used the classified image to calculate different preferential flow parameters, including total stained area (Flury et al., 1994), uniform infiltration depth (Van Schaik, 2009), preferential flow fraction (Van Schaik, 2009), and preferential flow at 45-50 cm (Lulandala et al., 2021).

3.5.2 Statistical analysis of the livestock grazing, land use and tree cover effect on hydrological properties and soil organic carbon (paper I – III)

I used different tools to analyze variations in soil hydrological properties and SOC with changes in land use, depending on the study's experimental setting. I used ANOVA to compare steady-state infiltration capacity, preferential flow indices, and SOC across different land use and land cover classes and different grazing intensities (papers I & II). To test the effect of tree cover on soil hydrological properties in the presence of livestock grazing, I used regression analysis, having tree cover and grazing intensity as interacting covariates (paper I). I also used a mixed-effects model having land use as a random effect to test the effect of interaction between basal area, varying grazing intensity, and land use on SOC (paper II). I used paired t-test to compare steady-state infiltration capacity and SOC between inside and outside of the fenced plots (papers I & II).

In paper III, I used ANOVA to compare steady-state infiltration capacity and SOC across different charcoal production clearing sizes (large clearings, small clearings, small intact plots, and village land forest reserve). I also used ANOVA to test the variation in steady-state infiltration capacity, SOC, and basal area at different distances from the road into the forest and large clearing. Finally, I used the Mann-Whitney test to perform a pairwise comparison between distances from the road (paper III).

4. Results and Discussions

The three papers appended to this thesis investigated the influence of land use & land cover and livestock grazing intensity on soil hydrological properties and SOC in dry miombo woodlands and analyzed the underlying mechanisms. The following is a collective discussion of the main findings of the three papers. Further detailed information is found in the appended papers (I – III).

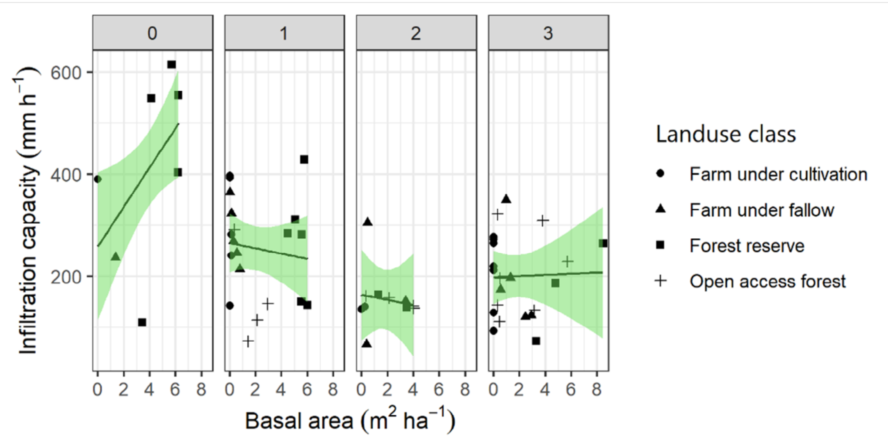

Figure 6. Scatter plots showing the relationship between steady-state infiltration capacity (mm h^{-1}) and basal area (m^2 ha^{-1}) in relation to different grazing intensity scores within the LDSF site in Kitulangalo, Morogoro, Tanzania. Numbers on top of the plot (0, 1, 2, and 3) represent grazing intensity scores.

4.1 Effect of tree cover on soil hydrological properties

Findings from my studies indicate a positive relationship between tree cover and soil hydrological properties (papers I & III). Data from paper I comparing different land use classes showed that steady-state infiltration capacity was highest in the forest reserve (284 ± 51 mm h^{-1}; paper I), and village land forest reserve (400 ± 5 mm h^{-1}; paper III) which also had the highest basal area (Table 1). Furthermore, regression analysis between steady-state infiltration capacity and tree basal revealed a positive association between the two (papers I and II), which was sometimes hard to see when mixed land uses are involved like the presence of livestock grazing in different land uses. Results from paper III indicate that in the village land forest reserve, where forest disturbance from wood harvesting was minimal, steady-state infiltration capacity was significantly higher than in both large and small clearings.

Concurrently, I showed in paper I that different preferential flow indices like total stained area and preferential flow at 45-50, had more or less the same pattern as steady-state infiltration capacity being relatively higher in the forest reserve. Forest reserve had a higher total stained area than any other land use class as well as preferential flow at the bottom 5 cm of the profile, which was higher than any other land use/land cover.

These findings agree with several other studies from different parts of the world showing that trees positively influence soil hydrology. This influence is a result of increased SOC from the addition of biomass from litter-fall, roots, and dead branches (Barros & Fearnside, 2016; Devi, 2021; Stockmann et al., 2015), increased soil porosity and aggregation through woody vegetation well-established root systems, and enhanced tree-associated soil fauna (Guo & Lin, 2018). Together, these mechanisms lead to increased soil infiltration capacity and preferential flow in the vicinity of trees (Bargués-Tobella et al., 2020; Bargués-Tobella et al., 2014; Belsky et al., 1993; L. Benegas et al., 2014). This can also be seen in our study in paper II, which shows a positive relationship between SOC and tree basal area, as well as in paper I, where land uses with high SOC also have high infiltration capacity.

My study revealed that the relationship between trees and soil hydrological properties may sometime not be observed, not because it is not there, but can be obscured by other land uses within the area since the relationship differs for different land uses (figure 6, paper I). This is important because, in drylands where water availability is scarce, it is also where land use dynamics are highly pronounced.

4.2 Effect of different land uses on soil hydrological properties and soil organic carbon

My studies showed that land use has a significant influence on soil hydrological properties and SOC. Here I explain the effect of land uses that we observed in our study area and are most common in the miombo woodlands landscape setting.

4.2.1 Livestock grazing

Data from paper I show that steady-state infiltration capacity in areas with no visible signs of livestock grazing (357 ± 104 mm h^{-1}) was more than double that in areas with the highest grazing intensity (160 ± 20 mm h^{-1}). The same was observed in our study on exclosures (paper I), which showed that steady-state infiltration capacity inside the exclosures (442 ± 53 mm h^{-1}) was almost double that of the outside (279 ± 49 mm h^{-1}). I also showed that high grazing intensity overrides the positive effects of trees on soil infiltration capacity regardless of land use (Figure 6).

All our preferential flow indices were affected by livestock grazing (paper I). Both total stained area and uniform infiltration depth decreased by 36 % and 37 % with increasing grazing intensity from 0 to 3 grazing intensity scores, respectively, while preferential flow fraction increased by 61 %. Preferential flow at 45-50 cm depth in areas where no visible signs of livestock grazing (score 0) was 6-times higher than in any other grazing intensity score (1-3). This suggests that although trees enhance the preferential flow and deep water drainage through the root activities, if the topsoil layer is disturbed and compressed, then the positive effect of trees is minimized.

In our study, higher livestock grazing intensities were also associated with increasing soil disturbance and compaction, as indicated by increasing soil bulk density. For example, we observed a 10 % increase in bulk density with increasing grazing intensity from 0 to 3 intensity score (paper I), while up to 40 % (18 tonnes ha^{-1}) decrease in SOC was observed for the same change in grazing intensity across land uses (paper II). This change in soil properties that led to declining soil hydrological functioning can be explained by the collapsing of soil structure as a result of excessive recurring trampling by grazing animals (Donkor et al., 2002; Dreccer & Lavado, 1993; Dudley et al., 2002), as well as reduced SOC as a result of overgrazed vegetation that is the source of SOC (Boerma & Koohafkan, 2007). Drylands have low primary productivity and relatively high decomposition rates during parts of the year, which makes them sensitive to livestock grazing. Our exclosures experiment further confirmed this, as shown by improved soil hydrological properties and SOC after the total exclusion of livestock grazing (paper 1).

Changes in SOC as a result of livestock grazing may also be influenced by other factors, including climate, soil texture, and frequency and duration of grazing. According to a meta-analysis on the impact of livestock grazing, there is a persistent decline in SOC with heavy grazing intensity, particularly in more sandy than clayey soils, warmer than cold climates, and drier than wetter climates (Lai & Kumar, 2020). This suggests that tropical dryland soils are particularly vulnerable to heavy livestock grazing. Nevertheless, studies from wetter climates also show that low to moderate livestock grazing can enhance nutrient cycling and promote vigorous vegetation growth through animal dropping, which promotes healthy soils (Blache et al., 2016).

4.2.2 Agricultural activities

Findings from papers I and II indicate a decrease in soil hydrological functioning and SOC with changes in land use from forest land to croplands, which is in line with findings from (Fan et al., 2013; J. He et al., 2009; Ilstedt et al., 2007; Nyberg et al., 2012). Croplands under cultivation and fallows were the land uses with the lowest steady-state infiltration capacity (245 ± 31 mm h^{-1} and 188 ± 22 mm h^{-1}) and bulk density (1.34 ± 0.02 g cm^{-3} and 1.4 ± 0.03 g cm^{-3}, respectively) values (paper I). The study in paper II showed up to a 55 % decrease in SOC in the forest as compared to croplands. Of all preferential flow indices, only the total stained area showed a clear declining trend from forest lands to croplands.

A decline in soil hydrological properties and SOC in croplands can be explained by; i. reduced biomass inputs from vegetation that is cleared when opening up land for agricultural purposes (Aweto, 1981), this also leads to decreased soil fauna as well as root activities, ii. burning of the crop residues after harvest instead of incorporating them into the soil as biomass(Grogan et al., 2013), iii. soil cultivation involving tillage, and leaching of dissolved carbon into water percolating downward (Nakhavali et al., 2021). Soil tillage causes soil disturbance and disrupts the capillary continuity that allows for the effective vertical movement of water (Gómez et al., 1999). This, together with the loss of vegetation cover that promotes soil aggregation, porosity enhancement, and litter addition, explains SOC's effective loss.

Land use change from forest land to agricultural lands is associated with a 60 % loss in SOC in temperate regions and up to 75 % or more in the tropics (Devi, 2021; Lal, 2004; Vågen et al., 2005). As explained earlier, the high percentage loss in SOC within the tropics is due to high temperatures that speed up decomposition and the low primary productivity particularly in tropical drylands compared to wet tropics. Considering this, and given that shifting cultivation is the predominant land use system in tropical drylands, agricultural practices within these areas significantly impact soil hydrology and the global carbon cycle in general.

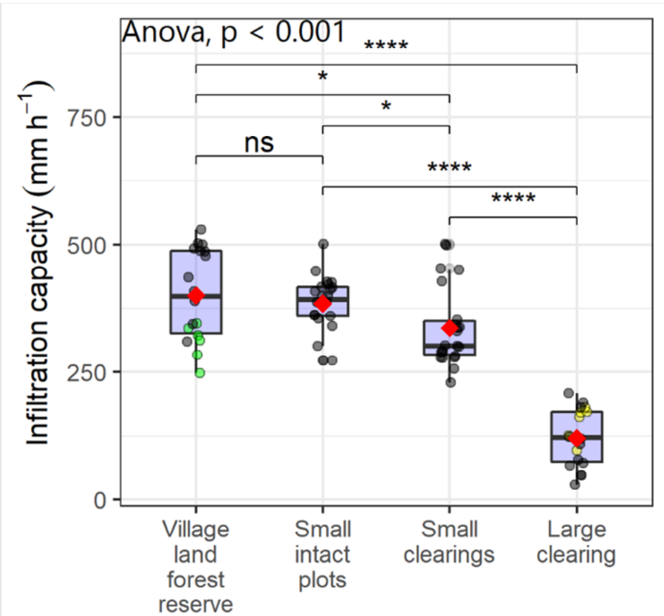

Figure 7. Means (red diamond) and boxplot (median, first and third quartile) of soil steady-state infiltration capacity (mm h^{-1}) for the four different study treatments within the study area in Ulaya Mbuyuni village, Kilosa, Morogoro region, Tanzania. Green dots show the steady-state infiltration capacity (mm h^{-1}) of plots at 10m from the road within the village forest reserve. Yellow dots show the steady-state infiltration capacity (mm h^{-1}) of plots at 10m from the road within the large clearing. Gray dots show steady-state

infiltration capacity (mm h^{-1}) in different treatments within the study area. Asterisks denote significance levels ('***'= <0.001, '**' = 0.001-0.01, '*' = 0.01-0.05, 'ns' >0.05).

4.2.3 Forest clearing for charcoal production

The study on the impacts of harvesting for charcoal production and clearing sizes on soil hydrological properties and SOC (paper III), showed that it is not just the cutting of trees that matters but also how this is done. Findings from this paper indicated that large clearings (300 × 300 m) had the lowest mean steady-state infiltration capacity (121 ± 3 mm h^{-1}) and SOC (12 ± 0.2 tonnes ha^{-1}) values, representing only 30% and 75 %, respectively, of those in the village land forest reserve. On the other hand, small clearings had significantly higher mean steady-state infiltration capacity (337 ± 4 mm h^{-1}) and SOC (15 ± 0.2 tonnes ha^{-1}) than large clearings, although still below mean values in the village land forest reserve and small intact plots (Figure 7).

Although the village land use plan dictates that, once harvested, these clearings should be protected from other land uses, especially livestock grazing, there are reports of grazing activities within these clearings, particularly during dry seasons (Mabele, 2019), that could influence the patterns we are observing.

Small clearings in our study area had nearly three times higher steady-state infiltration capacity than large clearing, which is largely due to the ability of trees to influence their neighboring open areas through their root system, which extend beyond the canopy edge (Bargués-Tobella et al., 2014; Benegas, 2013). Studies show that the ratio of the canopy to root system radius in trees and shrubs can be as small as 1/10 (Lejeune et al., 2004). Small clearings benefit from the additive effect of tree roots and canopies from all sides of the clearing, enhancing the soil properties more efficiently than in large clearings. This has also been reported in other studies like studies in agroforestry parklands, which also reported higher preferential flow and deep water drainage in smaller open areas than in larger ones (Bargués-Tobella et al., 2014 & 2020), and improvement in soil properties like SOC, bulk density, and porosity from the center to the edge of the forest gaps (He et al., 2015).

Large clearings tend to leave a significant piece of land exposed to agents of degradation like direct rainfall, sun, rainfall, and wind. These can lead to the deterioration of soil qualities through increased surface runoff and soil erosion (Haghnazari et al., 2015), which in turn might lead to reduced soil and underground water recharge (Bargués-Tobella et al., 2014). On the other hand, small clearings through relatively high infiltration capacity and soil protection show a promising potential for soil and groundwater recharge. Furthermore, in small clearings, there is a reduction in water loss through transpiration because of reduced tree density, while the infiltration capacity is maintained, which could result in improved groundwater recharge compared to large clearings or more dense forests (Ilstedt et al., 2016).

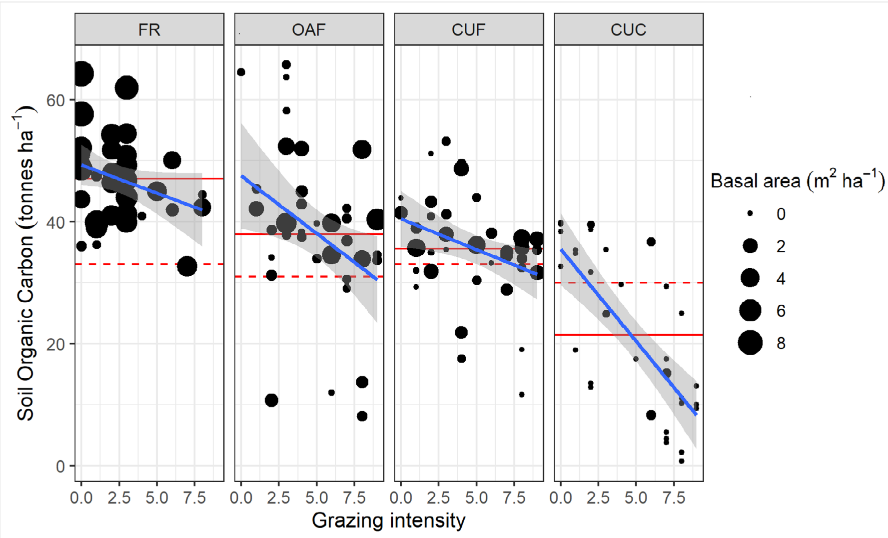

Figure 8. The relationship between topsoil organic carbon (tonnes ha^{-1}) and livestock grazing intensity (score 0-9) for each land use and land cover class (FR = forest reserve, OAF = open access forest, CUF = cropland under fallow, CUC = cropland under cultivation) in 149 plots across the 10 x 10 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 shows mean topsoil and subsoil soil organic carbon (tonnes ha^{-1}), respectively, in each land use class.

4.3 The interactive effect of livestock grazing intensity and different land uses on soil hydrological properties and soil organic carbon

My studies partly revealed that different land uses influence soil hydrological properties and soil organic carbon differently, and interactions between land uses within the landscape tend to reduce or amplify their effects. Paper I showed a positive relationship between steady-state infiltration capacity and tree basal area in plots with no visible signs of livestock grazing (score 0); however, this relationship disappears with increasing grazing intensity (Figure 6). Although different studies have looked at the effect of land use on soil infiltration capacity (Fu et al., 2000; Shukla et al., 2003; Sun et al., 2018; Yimer et al., 2008), and found a consistent decline in soil infiltration capacity with changes in land use from forested land to croplands. However, in the presence of moderate to high grazing intensity (score = 2 and 3) in our study area, these relationships could not be observed, and instead, all land uses/ land covers appeared to show more or less the same values of infiltration capacity (paper 1). When I compared steady-state infiltration capacities based on grazing intensity while disregarding land uses, plots with 0-grazing intensity had the highest value of infiltration capacity. Preferential flow indices in paper I showed the same pattern with land uses as infiltration capacity, suggesting that livestock grazing intensity has a strong influence on soil hydrological properties that can override even the effects of other land uses.

Findings from paper II showed that SOC varied greatly across land uses/ land covers, and that increasing livestock grazing intensity led to lower SOC in all land uses (Figure 8). Changes in SOC following the conversion of forests to cropland are well documented (Allen, 1985; Aweto, 1981; Grogan et al., 2013; Szott et al., 2004; Walker & Desanker, 2004). Here I showed that, because of the interactions of these changes in land use and the presence of higher livestock grazing intensities, the outcome can become more detrimental, particularly in croplands with less tree cover. Within the forest reserve, areas with the highest grazing intensity (score = 3) had 82 % of the SOC that was found in areas where no livestock grazing signs were observed. In croplands under active cultivation, on the other hand, areas with no visible signs of livestock grazing had more than double the SOC found in areas with the highest grazing intensity. Interestingly, the mean SOC in areas with the highest grazing intensity within the forest reserve was 1.3 times higher than that in areas within croplands under cultivation with no signs of livestock grazing observed.

Findings from paper III indicate that the negative effect of livestock grazing on SOC and soil hydrological properties is more severe in land uses with less tree cover. Concurrent changes from forest land to cropland and livestock grazing have a combined effect that might lead to higher soil degradation and loss of soil hydrological function than conversions from forest to cropland alone. Since livestock grazing is an important component of the miombo ecosystem, it is important to consider implementing more adaptive solutions like silvopastoral systems that would encompass even the grazing needs and reduce pressure and degradation.

4.4 Potential of trees to counteract the effects of livestock grazing

Trees' positive influence on soil properties might explain the observed variations in the effect of livestock grazing in different land uses. Livestock grazing reduces vegetation cover when feeding on grass, herbs, and seedlings (Mtimbanjayo & Sangeda, 2018; Wang, 2014), and disrupts the soil structure by trampling (Li et al., 2015). However, trees can counteract this through increased inputs of litter-fall from above biomass and roots (Cui et al., 2005), the creation of niches with a favorable microclimate that enhance SOC (Thomas et al., 2018), and root activity, thereby enhancing soil aggregation and porosity, which in turn improves soil hydrological function (Benegas, 2013). However, results from papers I show that trees are ineffective in improving soil hydrological properties under high livestock grazing intensity.

5. Conclusions

In this thesis, I studied how land uses in miombo landscape affect soil hydrological properties (steady-state infiltration capacity and preferential flow) and soil organic carbon. First, I identified the most common land uses within the miombo landscape in our study area: crop cultivation (active cultivation and fallow land), livestock grazing, charcoal production, open access forest, and forest reserve. Then, I investigated how these land uses and tree cover affect the focus soil properties, individually or in combination.

The main conclusions are:

- High tree cover influenced soil hydrological properties positively (infiltration capacity and preferential flow) when livestock grazing intensity is low to moderate. However, at the highest grazing intensities, the positive influence of trees is removed.

- Excessive grazing overrides the positive effects of trees on soil organic carbon and hydrological properties, but trees improve soil hydrological properties and soil organic carbon at low to moderate grazing intensities

- Soil hydrological properties and soil organic carbon can be restored by the exclusion of livestock grazing in miombo forests.

- Clearings of forests to croplands or for charcoal production leads to the loss of soil organic carbon, and the effect is more severe with increasing livestock grazing intensity (Paper II). Hence, the impact of livestock on soil properties is more negative in croplands than in forested lands.

- Large forest clearings for charcoal production are more detrimental to soil hydrological properties and soil organic carbon than small clearings, which retain more favorable hydrological properties for preventing surface runoff of water, and erosion, and promoting groundwater recharge.

6. Future research directions

Understanding the consequences when different land uses in a landscape are combined is important for the accurate prediction of management outcomes and planning. This in turn can enable policymakers to make founded decisions and appropriate policy measures that could enhance the sustainability of natural and agricultural ecosystems.

This thesis contributes to a deeper understanding of how major and common land uses within the miombo landscape influence their ecosystem separately and when they co-exist, an occurrence that is most of the time overlooked. However, more work is still needed.

Results from this thesis contribute to the understanding of how overgrazing can negatively influence soil hydrology and soil organic carbon, and override even the positive restorative effect of trees. Considering the vast area of miombo woodlands and the increasing trend of livestock grazing in the miombo ecosystem, there is a need for further studies of appropriate measures to take, which may include; i. Determining the appropriate livestock stocking rate that would be environmentally friendly, and would work with either free or zero grazing, ii. because the complete control of livestock grazing may not be a possible option, there is a need for further studies of other agroforestry systems apart from agropastoral systems that would emphasize livestock pasture availability like silvopastoral system.

This thesis established an important link between livestock grazing and soil hydrology. However, there is still a need for further understanding of the actual effects on the underground water recharge by measuring the quantities of recharging water associated with different grazing intensities as well as variation in tree cover within a landscape.

This thesis explains the general effect of livestock as a whole, but most of the time there is a diversity of livestock, both browsers and grazers moving together. Since different livestock species have different feeding requirement and weights per animal, they would have a variation of impacts on both soil hydrology and organic carbon. Heavier and bigger animals would have more impact on the soil structure than small and lighter ones, while small animals like goats are browsers and hence they also feed on tree leaves and hence may have a significant impact on biodiversity as well in the long run.

Forest clearing sizes proved to have an influence on soil hydrology and soil organic carbon. Since the study in this thesis had only two sizes to compare, it is not possible to establish an optimum clearing size and harvesting styles/patterns that would have an optimum effect on soil hydrology as well as soil organic carbon. There is a need for further study that would measure not only infiltration and preferential flow, but also the actual quantities of recharge of underground water in relation to a spectrum of clearing sizes and effects of the presence of livestock grazing in the clearings.

7. References

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Popular science summary

Miombo woodlands cover about 10 % of the African landmass and 100 million rural and urban communities directly or indirectly depend on them for their livelihoods. These areas are drylands, where the main physical limitation for people and ecosystems is the availability of water. Around 80 % of the population are agro-pastoralists, practicing livestock keeping and farming together with some wood harvesting for producing charcoal. Due to high levels of population growth, there has been a large expansion of arable and pastoral land. This together with poor management, has led to widespread degradation of both vegetation and soil quality, especially soil organic carbon and hydrological properties, which are important factors in ecosystem health and productivity. There is therefore increasing interest in the effects of different land uses and practices, including the restoration of degraded lands. Although these types of land uses rarely occur independently, their impacts have generally been studied separately. This is a problem since the effects on climate mitigation and water resources of different land uses when acting together could potentially be greater than the sum of their individual effects separately. Conversely, the positive effects of one land use could be canceled by another land use. In my thesis, I, therefore, studied the collective effects of co-occurring land use on soil carbon and soil hydrological properties in dryland miombo woodland ecosystems in Tanzania. I showed that changes from the forest to croplands as well as excessive livestock grazing substantially reduce soil organic carbon. Higher livestock grazing intensities are more detrimental to soil organic carbon in croplands that do not have trees compared to forest lands. This shows the potential of trees to counteract the negative effects of livestock grazing. However, my results also show that for soil hydrological properties, the positive effects of trees disappeared where the grazing intensity was highest.

Restoration projects that use trees to restore soil and water resources may therefore be wasted if not controlling livestock. These results provide new insights into how to plan and manage dryland ecosystems, taking into account the increasing trend of grazing due to increased global demands for livestock products.

Populärvetenskaplig sammanfattning

Miombo-skogarna täcker cirka 10 % av Afrikas landmassa, och 100 miljoner människor på landsbygden och i städerna är direkt eller indirekt beroende av dem för sin försörjning. Dessa områden är torra och den största fysiska begränsning för människorna och ekosystemen är vattentillgången. Omkring 80 % av befolkningen är jordbrukspastoralister som bedriver boskapsskötsel och jordbruk samt viss avverkning för framställning av träkol. På grund av den kraftiga befolkningsökningen har det skett en stor expansion av odlings- och betesmarkerna. Detta har tillsammans med dålig förvaltning lett till en omfattande försämring av både vegetation och markkvalitet, särskilt organiskt kol i marken och hydrologiska egenskaper, som är viktiga faktorer för ekosystemens hälsa och produktivitet. Det finns därför ett ökat intresse för effekterna av olika markanvändning och metoder, inklusive återställande av skadade marker. Trots att dessa typer av markanvändning sällan förekommer oberoende av varandra har deras effekter i allmänhet studerats separat. Detta är ett problem, eftersom effekterna på klimatet eller på vattenresurser av olika markanvändningar som samverkar kan vara större än summan av deras enskilda effekter separat. Omvänt kan positiva effekter av en markanvändning upphävas av en annan markanvändning. I min avhandling har jag därför studerat de kollektiva effekterna av samverkande markanvändning på markens kol- och hydrologiska egenskaper i miombo-ekosystem Tanzania. Mina resultat visade att förändringar från skog till åkermark samt överdrivet betande av boskap minskar det organiska kolet i marken drastiskt. Högre betesintensitet är mer skadligt för det organiska kolet i marken på åkermark utan träd jämfört med skogsmark. Detta visar på trädens potential att motverka de negativa effekterna av boskapsbete. Mina resultat visar dock också att när det gäller markens hydrologiska egenskaper försvann de positiva effekterna av träd där betesdriften var störst.

Restaureringsprojekt där träd används för att återställa mark- och vattenresurser kan därför vara bortkastade om man inte kontrollerar betet. Resultaten ger nya insikter om hur man kan planera och förvalta ekosystem i torra områden, med hänsyn till den ökande trenden betande boskap och globala efterfrågan på animaliska produkter.

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RESEARCH ARTICLE

WILEY

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 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 (agro-pastoralists) (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 *Braychystegia*, *Julbernadia*, and/or *Isoberlinia* (Leguminosae, subfamily Caeasalpinioideae) 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 km^2 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 × 10 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 hierarchal 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 × 10 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° 38′ 1″ S, 37° 58′46″ 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

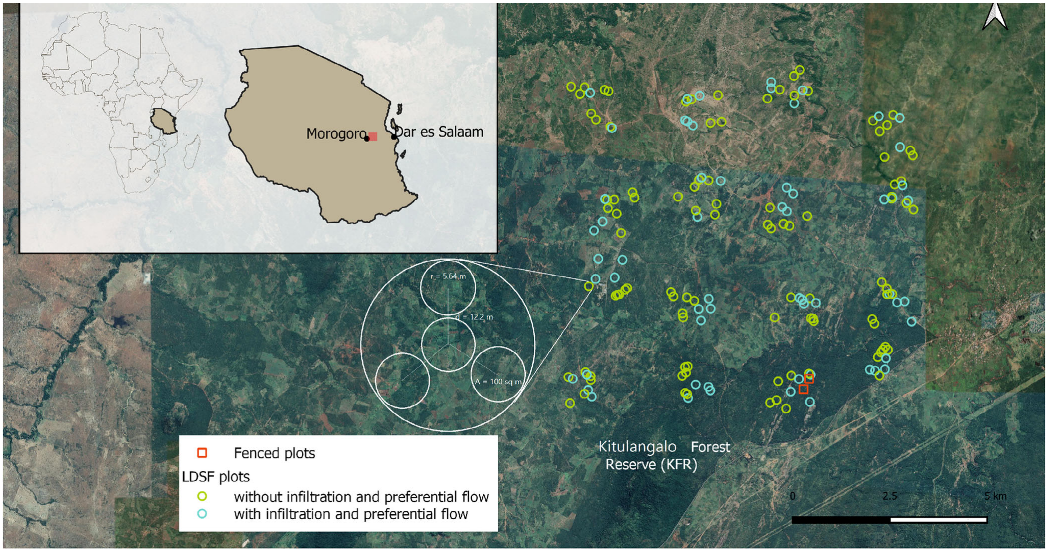

FIGURE 1 Map showing the location of the 10 × 10 km^2 study site in Morogoro, Tanzania. The site covers the northeastern part of the Kitulanghalo Forest reserve. We used a nested hierarchical sampling design, following the land degradation surveillance framework (LDSF) (Vågen et al., 2018; Vågen & Winowiecki, 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 km^2 in size (10 $\times$ 10 km), clusters within sites, and plots within clusters. Each LDSF site is divided into 16 tiles 2.5 $\times$ 2.5 km^2 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 m^2 in size and consists of four subplots, 100 m^2 in size (Figure 1).

To test the effects of total livestock exclusion, we designed a separate study using two 12-year-old fenced 30 $\times$ 90 m^2 plots within the forest reserve (Figure 1) that were established by the Tanzania Forest Research Institute (TAFORI) in 2005. These exclosures 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 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})	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 exclosures, 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 cm^3 (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 exclosures. 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 exclosure (Figure 2). However, we removed four plots from the 64 LDSF plots

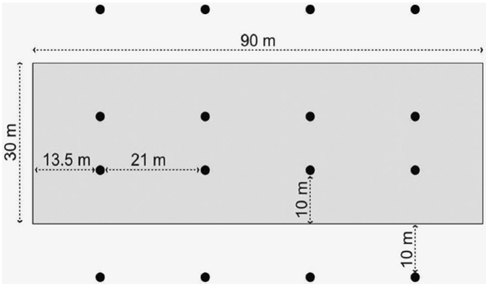

FIGURE 2 Layout of sampling points inside and outside the exclosures/fenced plots to test the effects of livestock exclusion on soil infiltration capacity in Kitulangalo Forest Reserve, Morogoro, Tanzania. Thick black box line = fence around the 30 × 90 m^2 plot, dots = measuring points we established

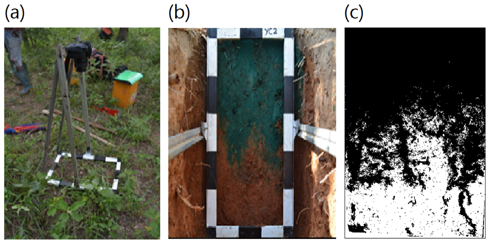

FIGURE 3 Pictures covering the process from acquiring a picture of a stained soil profile to obtaining the classified image (stained vs. nonstained classes); (a) a camera and a graded frame for soil profile photography, (b) photo of a stained soil profile, (c) classified soil profile image showing the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

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 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 × 0.5 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.

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

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

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.

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

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* ($PF_{45\text{-}50}$, %): 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.

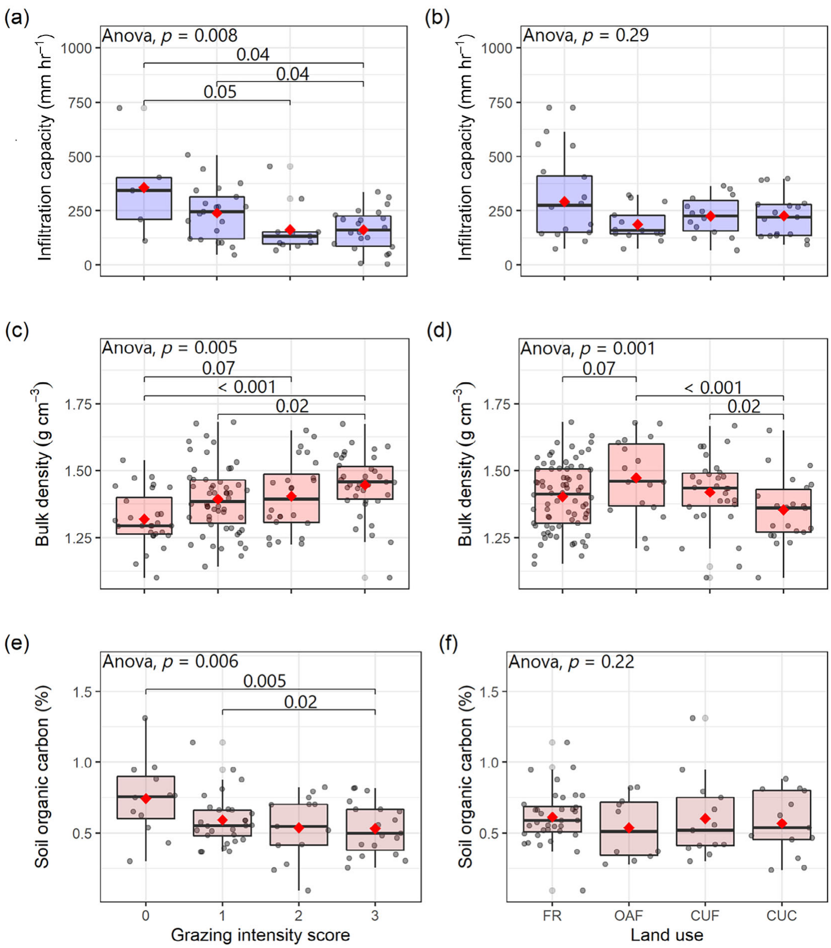

FIGURE 4 Boxplot (median, first and third quartile) of (a, b) steady-state infiltration capacity (mm hr^{-1}), (c, d) bulk density (g cm^{-3}), and (e, f) soil organic carbon (%) for the different classes of grazing intensity (left column) and land use/land cover (right column) within a 10 × 10 km^2 area in Kitulangalo, 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 sigma^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 $PF_{45\text{-}50}$) 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 exclosures.

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} (SE ± 104), double that in plots with high grazing intensity (160 ± 20 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

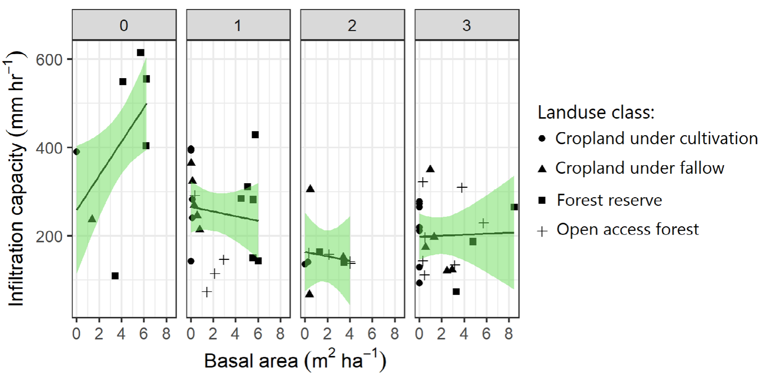

FIGURE 5 Scatter plots showing the relationship between steady-state infiltration capacity (mm hr^{-1}) and basal area (m^2 ha^{-1}) in relation to different grazing intensity scores within the LDSF site in Kitulangalo, Morogoro, Tanzania. Numbers at the top of the plot (0, 1, 2, and 3) represent grazing 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 Kitulangalo, 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 ± 0.02 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 ± 0.02 g cm^{-3}) in open-access forest and the lowest in farms under cultivation (1.34 ± 0.02 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 ± 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 exclosures was near twice the level inside compared to outside ($p = 0.03$; Figure 6a), that is, 442 ± 53 and 279 ± 49 mm hr^{-1}, respectively. Mean steady-state infiltration capacity for the paired plots outside the exclosures was similar to that for the LDSF plots within the forest reserve (Figure 4b) (284 ± 51 mm hr^{-1}) where exclosures were located. Mean bulk density was 1.64 ± 0.01 and 1.45 ± 0.04 g cm^{-3} outside and inside exclosures, respectively ($p < 0.001$; Figure 6b). Mean soil organic carbon was about double ($p < 0.001$; Figure 6c) inside the exclosures (1.46 ± 0.03%) compared to outside (0.72 ± 0.03%).

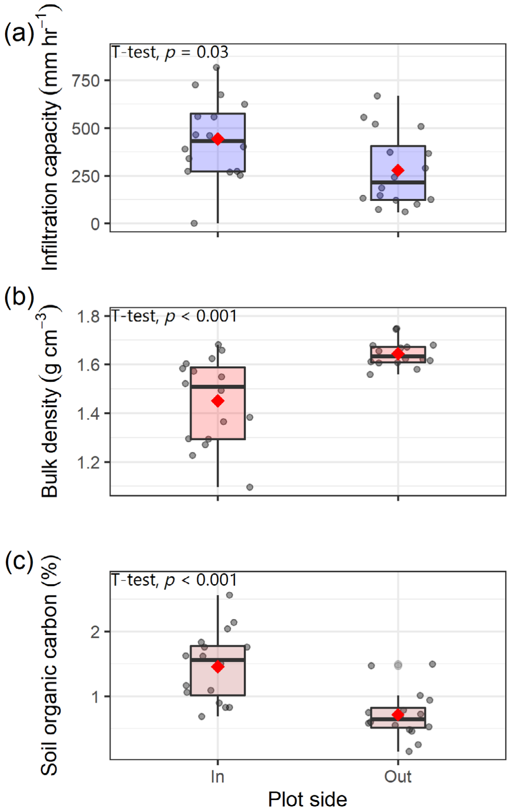

FIGURE 6 Boxplot (median, first and third quartile) of infiltration capacity (mm hr^{-1}) (a), bulk density (g cm^{-3}) (b), and soil organic carbon (%) (c), inside and outside grazing exclosures in Kitulangalo Forest Reserve, Morogoro, Tanzania. Significance values (*p*) are given. Red dots indicate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

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 ± 29 cm^2; $p < 0.001$ and 30 ± 3 to 19 ± 1 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 ± 5% at grazing score 0 to 29 ± 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 ± 5%) ($p < 0.001$; Figure 7d) than in areas with grazing intensity score 3 (9 ± 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-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

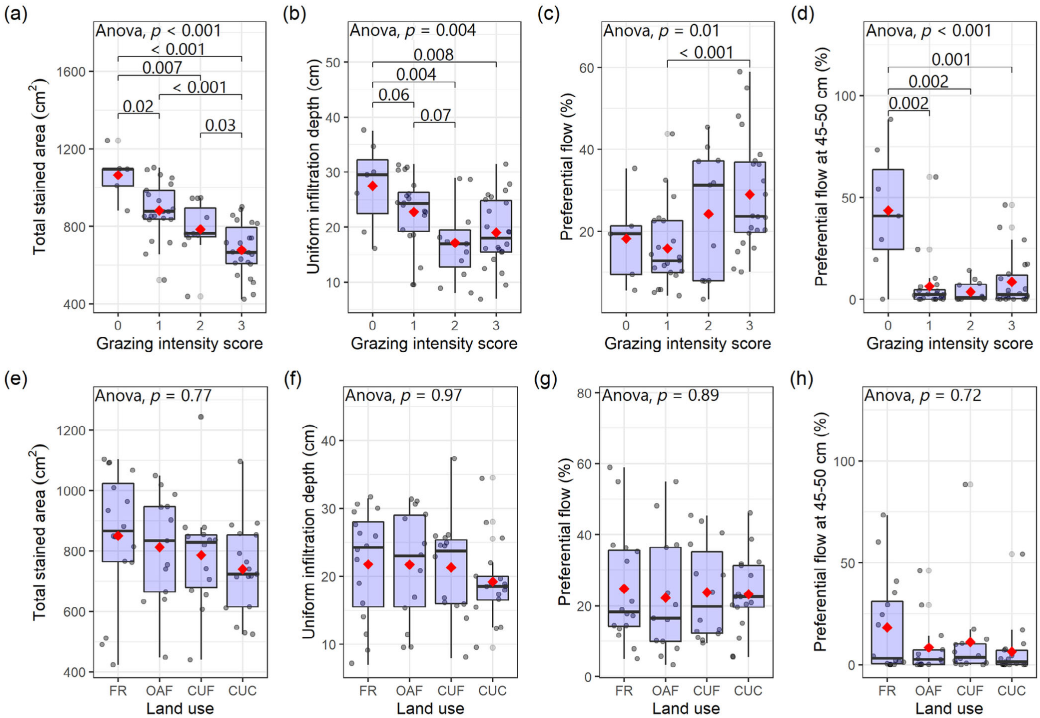

FIGURE 7 Boxplot (median, first and third quartile), of the different preferential flow indices for the different classes of grazing intensity (upper row) and land use/land cover (lower row) within a 10 × 10 km^2 area in Kitulangalo, Morogoro, Tanzania; significance values (p) are given. Red dots indicate 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

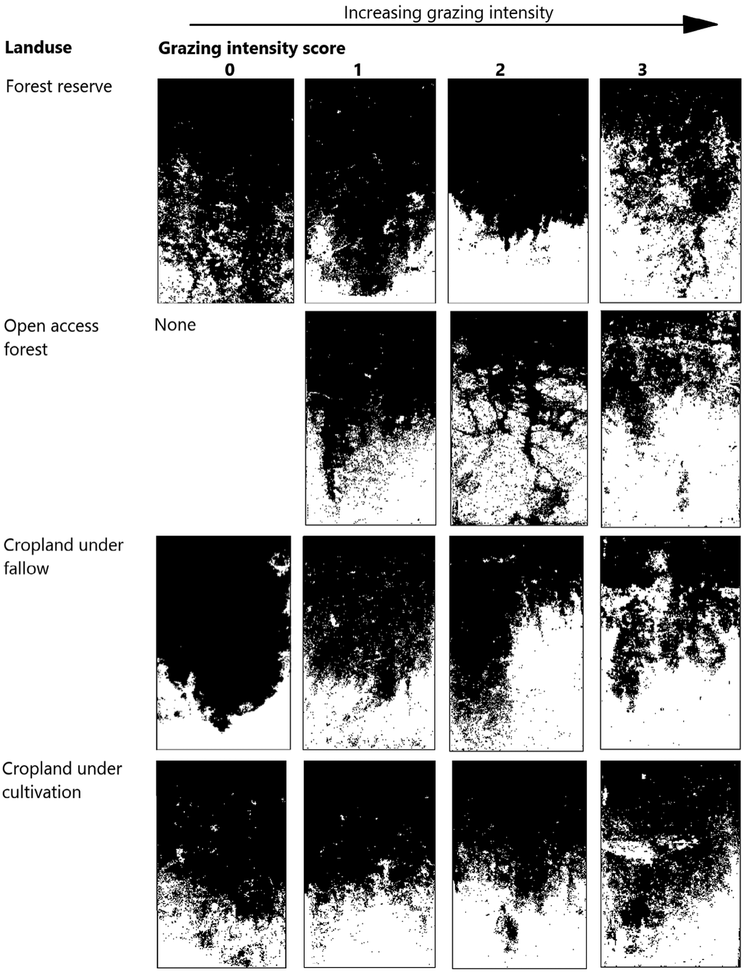

FIGURE 8 Examples of classified stained profiles (black: Dye stained soil, white: nonstained soil) for different classes of grazing intensity and land uses/land cover from a 10×10 km^2 area in Kitulangalo, Morogoro, Tanzania

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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This thesis analysed the interactive effect of different land uses, tree cover, and livestock grazing on soil organic carbon and hydrological properties in miombo woodlands. Results show that higher livestock grazing intensities are more detrimental in croplands than in forest lands. It also shows that trees can counteract the negative effects of grazing and agriculture, but for soil hydrological properties only at low to moderate grazing intensities and not at high intensities. This provides more insight on how to better manage and restore miombo landscapes.

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