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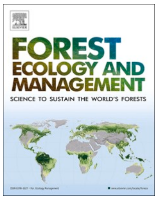

The size of clearings for charcoal production in miombo woodlands affects soil hydrological properties and soil organic carbon

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ABSTRACT

Charcoal production is a major driver of forest degradation in miombo woodlands. Forests play a crucial role in regulating the hydrological cycle, so it is critical to understand how forest degradation and management practices impact water availability, particularly in drylands. Few studies have examined the effect of forest clearing size on the hydrological functioning of soil, particularly under real-world conditions where, following clearing, forests are subject to multiple and prolonged anthropogenic disturbances, as occurs in miombo woodlands which are cleared for charcoal production and commonly used for livestock grazing. The pilot project *Transforming Tanzania's Charcoal Sector* was established in 2012 with the aim of establishing a sustainable wood harvesting system for charcoal production based on rotational harvesting cycles that allow for natural forest regeneration. Two clearing sizes were established: large clearings (300 × 300 m) harvested by clear-felling, and small clearings (50 × 50 m) harvested in a checkerboard pattern. We examined the effect of these two clearing sizes on soil hydrological properties and soil organic carbon (SOC) in Kilosa district, Morogoro, Tanzania. Our analysis included four treatments: large clearings, small clearings, small intact plots (unharvested plots within the checkboard pattern), and village land forest reserve. For each treatment we assessed the tree cover and measured soil infiltration capacity, soil bulk density, SOC stock, and texture. We also examined the relationship between these variables and the distance to the closest road to better understand the impact of livestock and human disturbance. Our results show that large clearings had the lowest mean infiltration capacity (121 ± 3 mm h^{-1}) and SOC stock content (12 ± 0.2 tonnes ha^{-1}), and the highest bulk density (1.6 ± 0.005 g cm^{-3}) of all the treatments. We found a positive relationship between infiltration capacity and basal area ($R^2 = 0.71$) across all treatments. We also found that infiltration capacity, SOC stock and tree basal area increased with increasing distance from the closest road, while bulk density decreased. We conclude that, in terms of their impact on soil hydrological functioning and SOC stock, small clearings, while not completely unaffected, are better than larger ones. In small clearings, concurrent reductions in tree cover and a relatively low impact on soil hydrological properties could result in increased soil and groundwater recharge compared to unharvested forest areas. Controlling livestock grazing can further minimize soil degradation, producing additional gains.

1. Introduction

Deforestation and forest degradation threaten forests worldwide (Olander et al., 2008) and have an impact on water dynamics and supply from local to global scales (Ellison et al., 2017). Globally, an estimated 10.6 million hectares of forests are being lost through deforestation each year, and another 100 million hectares are affected by degradation (FAO, 2020). About 95 % of this forest loss occurs within the tropics (Ritchie & Roser, 2021), and is highest in Africa (FAO, 2020). The primary drivers of deforestation are the expansion of cities and agriculture (Houghton, 2012; Pendrill et al., 2022), while forest degradation is caused mainly by timber and fuelwood harvesting, and forest fires (Pearson et al., 2017), often in combination with other human impacts such as livestock grazing and agriculture (de Andrade et al., 2017).

Although there is an established understanding of the impacts of forest cover on catchment water budgets in general (FAO, 2016), little is known about the influence of tree cover on soil water dynamics in human-impacted landscapes in tropical drylands (Ellison et al., 2017; Ilstedt et al., 2016). Catchment studies around the world have shown that reductions in tree cover reduce evapotranspiration and increase

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total yearly streamflow (Zhang et al., 2001). However, soil water movement is influenced by both natural and anthropogenic factors (Baez, 2019). Trees enhance soil aggregation and macroporosity through above- and below-ground litter additions and the activity of their roots and associated soil fauna (Bargués-Tobella et al., 2014). Improved soil structure and increased macroporosity lead to increased soil water holding capacity, infiltration capacity and degree of preferential flow (Bargués-Tobella et al., 2020 & 2014; Ellison et al., 2017; Ilstedt et al., 2007). This can, in turn, result in increased water percolation into deeper soil layers and, eventually, improved groundwater recharge (Ilstedt 2016). In addition to trees, land use and management practices that cause significant vegetation and soil disturbance also affect the rate and quantity of water infiltration and overall groundwater recharge (Woltemade, 2010). Land use and management practices that damage the soil structure – including soil compaction by livestock or agricultural practices such as tilling- are particularly important (Pagliai et al., 2004).

Miombo woodlands are the most extensive tropical dryland forests in central and southern Africa, occupying about 2.7 million km^2 (Ribeiro et al., 2015). Miombo woodlands have a high level of species endemism and constitute one of the top five high-biodiversity wilderness areas globally (Mittermeier et al., 2003). Furthermore, miombo woodlands provide a wide range of ecosystem services that are essential to supporting the livelihoods of>65 million people in the region (Monela & Abdallah, 2007; Ribeiro et al., 2015), including provisioning services such as supply of fuelwood, and others such as water flow regulation, carbon sequestration and storage, and habitat provision (Gumbo et al., 2018). However, miombo woodlands are under tremendous degradation and deforestation pressure, mainly due to charcoal production, firewood harvest, and shifting cultivation (Chidumayo, 1987; Kutsch et al., 2011; Luoga et al., 2000; Manyanda et al., 2021) jeopardizing the provision of these critical ecosystem services.

In sub-Saharan Africa, charcoal production for urban consumption is one of the main drivers of forest degradation (Sedano et al., 2016).. Charcoal is primarily produced by either clear-felling or selective cutting, in forests and woodlands in both rural areas and around urban centers (Sedano et al., 2016). Clear-felling is typically associated with land preparation for agricultural purposes, where charcoal production is just a by-product (Jew et al., 2016). In contrast, selective cutting involves selecting tree species with high calorific value and dimensions suitable for producing high-quality charcoal (Chidumayo & Gumbo, 2013). Selective cutting for charcoal production often leads to forest degradation and threatens to reduce preferred species to the verge of extinction (Silva et al., 2019). There have been several efforts to reduce pressure on miombo woodlands by increasing the efficiency of charcoal production, trade, and consumption (Branch et al., 2022). The pilot project *Transforming Tanzania's Charcoal Sector* was established in 2012 with the aim of creating a sustainable wood harvesting system for charcoal production, based on rotational harvesting cycles that allowed for natural forest regeneration. The project started by establishing large clearings of 300 × 300 m^2, but managers observed that these attracted livestock which they considered to have a negative effect on soils and regeneration (Titima. M, personal communication, June, 17, 2018). Because of this, the project approach changed, establishing smaller clearings of 50 × 50 m^2 distributed in a checkerboard pattern, and leaving trees of diameter at breast height (dbh) < 15 cm standing on these clearings. Although it is known that forest clearing size affects natural regeneration (Clarke, 2004; Coates & Burton, 1997; Myers et al., 2000), its effects on soil properties, including hydrological functioning, remain poorly understood.

Most of the available studies on the effects of forest openings or gaps on soil water dynamics are based on artificially created clearings and those resulting from natural tree-fall (Arunachalam & Arunachalam, 2000; Denslow et al., 1998; Muscolo et al., 2007; Yu et al., 2018; Zulkiflee & Blackburn, 2010), which may not necessarily reflect real-world conditions involving multiple and prolonged anthropogenic disturbances. Forest gaps are primarily a result of forest disturbances (McCarthy, 2001), and the effects of these gaps depend on several factors, including the source and intensity of disturbance, size and age of the gap, forest type, climate, and land use following the opening of the gap (Arunachalam & Arunachalam, 2000; Lu et al., 2018). The most common cause of forest gaps in miombo woodlands is subsistence agriculture (Jew et al., 2016). The impacts of forest conversion to cropland are well documented and include the reduction of biomass and soil organic carbon (SOC), increased soil erosion (Murty et al., 2002), and reduced infiltration capacity and preferential flow (Nyberg et al., 2012; Yimer et al., 2008). However, there is less understanding of the implications of forest clearings that do not involve crop cultivation, such as those for charcoal production, or are not located within protected areas and are hence open to other uses such as livestock grazing. Heavy livestock grazing in miombo woodlands has a negative influence on the physical and hydrological properties of soils, including bulk density, infiltration capacity, and preferential flow (Lulandala et al., 2021). This influence can vary spatially depending on forest accessibility, which itself depends on factors such as the presence of forest roads that serve as pathways for the livestock *en route* to different grazing lands. Studies of small forest gaps (85–300 m^2) in drylands outside the tropics show that larger clearing size may be associated with increased bulk density, reduced pore-space, and SOC stock (Amolikondori et al., 2020). However, these are much smaller clearings than those generally employed for wood harvesting in tropical dry forests, including in our study site (2,500 – 90,000 m^2). Studies in tropical agroforestry parklands, which are used for both cropping and livestock grazing, also show that infiltration capacity and preferential flow decrease with increasing gap size (Bargués-Tobella et al., 2014). However, the effects of charcoal clearings on soil hydrological properties in unprotected dry forests, including miombo, are still not well understood.

In this study, we aimed to explore the influence of clearing size on SOC stocks and soil hydrological properties in rotational charcoal harvesting systems in miombo woodlands. We did this by studying the two contrasting clearing sizes implemented by the pilot project *Transforming Tanzania's Charcoal Sector* (TTCS). The TTCS pilot project started in 2011 with the aim of contributing to a "Pro-poor and climate-resilient transformation of the economy and the governance of the forest product value chains, including charcoal and biomass energy". Based on previous studies of other systems (Arunachalam & Arunachalam, 2000; Bargués-Tobella et al, 2014 & 2020; He et al., 2015), we expected smaller clearings containing some retained trees to have better soil hydrological properties than larger clearings containing only regrowth. More specifically, we hypothesized that: a) large clearings will have higher soil bulk density and lower SOC stock and infiltration capacity than small clearings; b) in both large and small clearings tree basal area will have a negative influence on soil bulk density and a positive influence on SOC stock and infiltration capacity; and c) the distance to the nearest road will influence SOC stock and infiltration capacity positively and bulk density negatively due to increased access and disturbance by people and livestock.

2. Methodology

2.1. Study area

We conducted our study in Ulaya Mbuyuni village in Kilosa district (Fig. 1), which lies approximately 300 km inland from Dar es Salaam, between 6°S and 8°S, and 36°30′E and 38°E. Kilosa district borders the Tanga Region to the north and Morogoro District to the east (Wassena et al., 2013). Mean annual rainfall and temperature in Kilosa district are 900 mm and 25 °C, respectively, with the driest period occurring between June and September (Kajembe et al., 2013). The dominant vegetation locally is miombo woodland with an overstorey dominated by trees of the *Fabaceae* family (subfamily *Caesalpinioedae*, mainly within the genera *Brachystegia, Julbernardia*, and *Isoberlinia)* and an

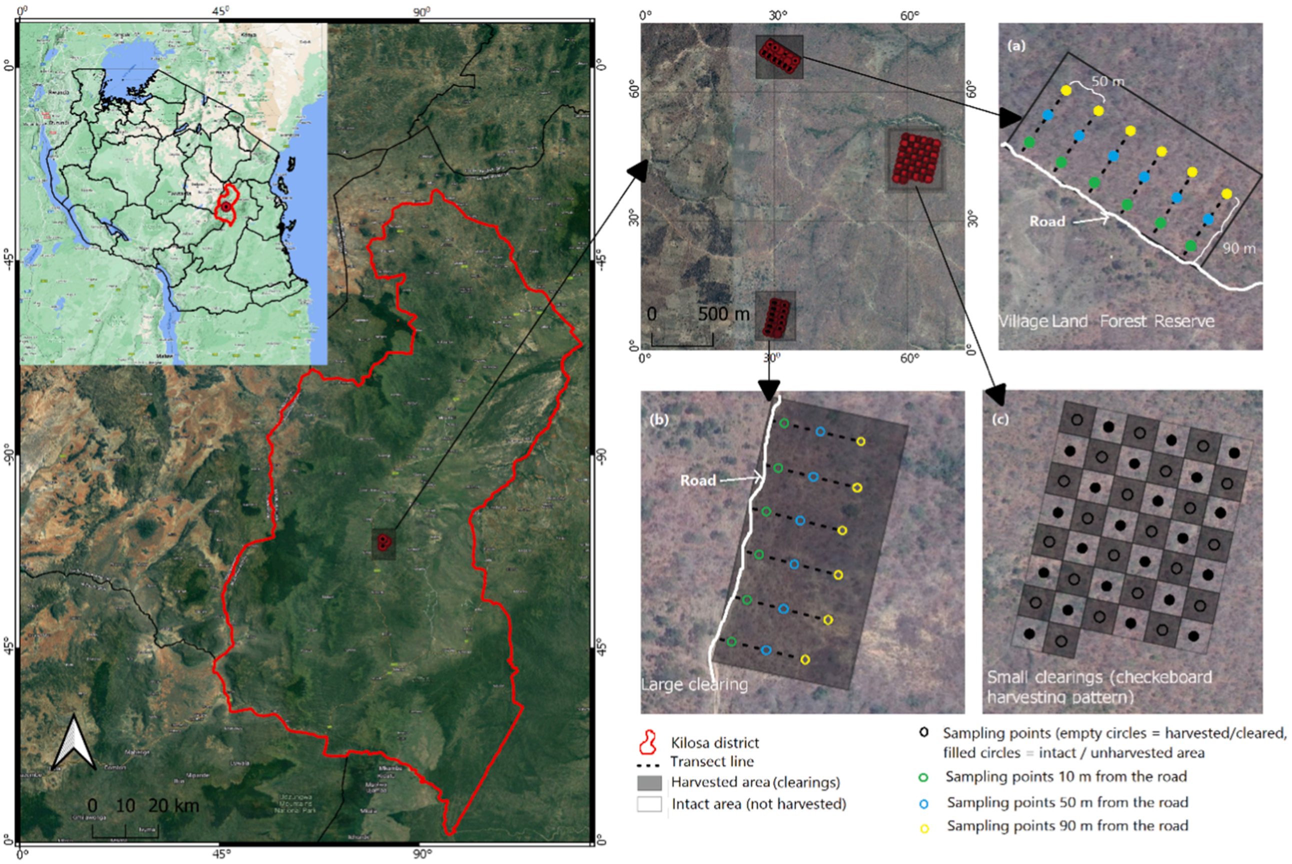

Fig. 1. Map showing the location of the study site in Kilosa district (Morogoro, Tanzania) and the sampling design.The site had four study treatments. (a) village land forest reserve, (b) large clearing (300 × 300 m), (c) small clearings and alternate intact plots (50 × 50 m) with a checkerboard pattern. In the large clearing and in the village forest reserve, six parallel 90 m-long transects 50 m apart were established starting from a secondary road. Along each transect, sampling plots were located at distances of 10, 50, and 90 m from the road. In the checkerboard-pattern site with small clearings and adjacent intact plots, we established a total of 44 sampling plots at the center of both small clearings and small intact plots.

understory consisting of grasses and shrubs (Kajembe et al., 2013; Sangeda & Maleko, 2018). Soils in our study area are mostly loamy sand (Table 1).

Ulaya Mbuyuni is one of the 30 villages where the Sustainable Charcoal Project (SCP) is being implemented as a pilot project within TTCS. The SCP in Ulaya Mbuyuni started in 2012 and, among other things, helped to develop a land use plan for proper resource management at the village level (Huggins, 2018). This involved the designation of the village land forest reserve and Forest Management Units (FMU). FMUs are forest areas set aside from the rest of the village land forest reserve for sustainable charcoal production. They cover approximately 15 % of the entire village land forest reserve, the remaining 85 % being dedicated to protection and beekeeping, with occasional low-intensity selective logging. Although grazing lands have been set aside, livestock keepers often graze in these FMUs and the reserve, especially during the dry season, although this is illegal under village by-laws (Mabele, 2019).

Table 1
Mean (standard error, SE) soil sand (%), silt (%), clay (%) of the topsoil (0–20 cm), and woody vegetation basal area (m^2/ha) of different plot status and harvesting systems within the sustainable charcoal project forest in Ulaya Mbuyuni village, Kilosa-Morogoro, Tanzania.

		Forest Management Units (FMU)		
	Village land forest reserve (n = 18)	Large clearing (n = 18)	Small clearings (n = 22)	Small Intact plots (n = 22)
Sand (%)	81.6 (0.2)	79.2 (0.5)	79.1 (0.2)	79.9 (0.2)
Silt (%)	7.3 (0.1)	8.7 (0.1)	7.5 (0.1)	7.7 (0.1)
Clay (%)	11.1 (0.1)	12.2 (0.6)	13.4 (0.2)	12.3 (0.1)
Basal area (m^2/ha)	13.9 (0.2)	3.5 (0.1)	7.4 (0.2)	13.9 (0.2)

2.2. *Charcoal harvesting process*

Within the FMUs, a forest inventory is carried out to help determine harvesting areas and annual harvest volumes. Once the areas to be harvested have been identified, they are marked using GPS (Ishengoma et al., 2016). The original large clearing (300 × 300 m) was clear-felled. In the case of small clearings, a total of 3024 plots of 50 × 50 m grids have been established in the entire FMU of Ulaya Mbuyuni. Of these, only 126 plots, totaling 31.5 ha, are harvested annually. The harvesting follows a checkerboard pattern of alternating squares, half of which are harvested in the first 12-year cycle (126 per year) and other half left intact until the second 12-year cycle. This means that all 3,024 plots are harvested within two cycles, after which the process starts again. According to the harvesting guidelines, only trees with dbh > 15 cm should be harvested for charcoal production but, in practice, trees with dbh as small as 5 cm are often harvested. All trees are harvested except for timber species, those that support ecosystem biodiversity (e.g., nest trees, fruit trees), those on steep slopes, and those within 60 m of water sources. All wood harvested for charcoal production is then gathered in one area of the site, where the charcoal is made with improved earth kilns. Following that, the charcoal is packed and transported to the closest road by either motorcycle or bicycle, where it is picked up by a tractor.

2.3. Site selection and sampling design

We conducted our study in the charcoal FMUs and the wider Village Land Forest Reserve. Within the FMUs, we identified several small clearings (50 × 50 m) of varying ages in a checkerboard pattern, and the one large clearing (300 × 300 m) harvested at the beginning of the project in 2013. The checkerboard harvesting style started being implemented in 2016. Among the identified areas with small clearings, we selected one that was first harvested back in 2016, comprising 22 clearings and 22 alternating intact plots 50x50 m in size where forest had not been harvested in the first harvesting cycle (Fig. 1). For large clearings, the fieldwork was carried out in 2018 when we collected data both in the village land forest reserve as a control and in the large clearing that had, by then, last been harvested 5 years ago. In 2021, we collected data from small plots (50 × 50 m), including both clearings harvested in 2016 and their adjacent intact plots. By then, the small clearings were the same age as the large clearing had been at the time of sampling (5 years). This allowed us to compare the two harvested sites at the same age since harvest. We assume that no major changes in land use occurred during this five–year period and that, therefore, it is possible to compare these two clearing sizes in different years (i.e., 2018 and 2021) given that the age since harvest was the same during sampling. In total, we had four distinct treatments – large clearing, small clearings, small intact plots (not harvested plots 50 × 50 m in size), and village land forest reserve (intact forest of larger dimensions).

In the large clearing and in the village forest reserve, we established six parallel 90 m-long transects 50 m apart, starting from a secondary road. Along each transect, sampling plots were located at distances of 10, 50, and 90 m from the road, resulting in 18 sampling plots per site ($\mathbf{n}_{\mathrm{VLFR}} = 18$, $\mathbf{n}_{\text{large clearing}} = 18$). The sampling plots had a radius of 6 m and were centered along transects. In the checkerboard-pattern site with small clearings and adjacent intact plots, we established a total of 44 sampling plots 6 m in radius at the center of both small clearings ($\mathrm{n}_{\text{small clearings}} = 22$) and small intact plots ($\mathrm{n}_{\text{small intact plots}} = 22$) plots.

2.4. Soil infiltration capacity

We measured soil infiltration capacity at the center of each sampling plot using a single ring infiltrometer with a height of 27 cm and inner diameter of 30 cm. At each measurement point, we inserted the ring 5 cm into the soil and pre-wetted it. The pre-wetting process involved pouring two liters of water and letting it completely infiltrate before starting the infiltration capacity measurements. We then filled the ring with water up to 20 cm and recorded the water level every 5 min. Immediately after each reading, the ring was refilled to the original water level. We monitored the water level at 5 min intervals for the first half hour and 10-minute intervals thereafter, until we observed stable infiltration rates (approximately the same reading for three consecutive measurements). Steady-state infiltration capacity was then estimated by using the *SSphilip* function in the “HydroMe” package in R (Omuto, 2013).

2.5. Soil and vegetation sampling

In each sampling plot a 20 cm deep soil pit was dug, from which different soil samples were collected. Topsoil samples were collected at 0 – 20 cm soil depth. Bulk density samples were collected with a stainless-steel cylinder of 98.17 cm^3 volume (with 5 cm for height and inner diameter), at the center of the topsoil layer (0 – 20 cm), in one of the pit walls. All soil samples were weighed, packed, and labeled in the field and then taken to the laboratory for analysis. In the laboratory, bulk density was determined by the oven-dry method (Blake & Hartge, 1986); SOC by the Walkley-Black chromic acid wet oxidation method (Bremner & Jenkinson, 1960); and soil texture by the hydrometer method (Bouyoucos, 1936). We then calculated SOC stock for the topsoil in tonnes per hectare using the formula:

$$SOC(tonnes ha^{-1}) = SOC\%/100 \times BD\left(gcm^{-3}\right) \times Thickness(m) \times 10,000 \qquad (1)$$

where: SOC = percent soil organic carbon (g per 100 g of dry soil), BD = bulk density in g cm^{-3} and Thickness = Thickness of the soil layer in meters. Coarse fragments were negligible in the topsoil and are therefore not considered in the equation.

In all sampling plots, all woody plants with dbh > 5 were counted and measured for basal area calculation.

2.6. Data analysis

We performed all statistical analyses in R software (R Core Team, 2019). We checked for data normality using the Shapiro-Wilk normality test (shapiro_test function) and homogeneity of variances using Levene’s test (LeveneTest function) across our data groups before choosing the appropriate analysis. We used ANOVA (aov function in R) to test for differences in steady-state infiltration capacity (mm h^{-1}), SOC (% and tonnes ha^{-1}), and bulk density (g cm^{-3}) across the four different study treatments (village land forest reserve, large clearings, small clearings, and small intact plots). Due to the low number of observations per treatment (<30), we used the Mann-Whitney test for between-groups pairwise comparison. We used a one-way ANOVA to check the differences in steady-state infiltration capacity, bulk density, basal area, and SOC stock at different distances into the village forest reserve from the road, and a Mann-Whitney test for pairwise comparison of these measures. Given the nature of our study area and treatments, our sampling points were clustered. Hence, we applied mixed-effects models using the lmer() function from the package “lme4″ (Bates et al., 2015) to establish a relationship between infiltration capacity (dependent variable) and SOC stock (tonnes ha^{-1}), basal area (m^2/ha), and bulk density (g cm^{-3}) as covariates, while having treatments as random intercept and slope.

3. Results

Mean soil steady-state infiltration capacity was 2.7 times higher ($p < 0.001$) in small clearings (337, SE ± 4 mm h^{-1}; Fig. 2) than in the large clearing (121 ± 3 mm h^{-1}), and was highest in the village land forest reserve (400 ± 5 mm h^{-1}) and in small intact plots (385 ± 2 mm h^{-1}), although we observed no significant differences between these two treatments ($p = 0.7$). In small clearings, mean steady-state infiltration capacity was significantly lower than in small intact plots ($p = 0.02$; Fig. 2), but only by 12 %.

SOC stock followed a similar trend to steady-state infiltration capacity (Fig. 3): mean SOC stock was highest in small intact plots (19 ± 0.2 tonnes ha^{-1}), followed by the village land forest reserve (16 ± 0.2 tonnes ha^{-1}), small clearings (15 ± 0.2 tonnes ha^{-1}), and the large clearing (12 ± 0.2 tonnes ha^{-1}). No significant difference in mean SOC stock was observed between the village land forest reserve and small clearings ($p = 0.3$). Differences in mean SOC stock between small and large clearings were marginally significant ($p = 0.08$).

Bulk density was approximately within the same range across the four treatments with the exception of the large clearing, where the mean bulk density was significantly higher (1.6 ± 0.005 g cm^{-3}) than that in the village land forest reserve (1.4 ± 0.005; $p < 0.001$), small intact plots (1.5 ± 0.004; $p < 0.001$), or small clearings (1.4 ± 0.003; $p < 0.001$) (Fig. 4).

We observed that plots within the village land forest reserve located 10 m from the road exhibited lower values of steady-state infiltration capacity and SOC stock than plots located further away (Figs. 2 and 3). A closer look revealed that steady-state infiltration capacity, SOC stock, and basal area increased with increasing distance from the road, but bulk density did not (Fig. 5). Mean steady-state infiltration capacity (Fig. 5a) was lowest in plots located 10 m from the road (308 ± 6 mm h^{-1}), intermediate at 50 m (424 ± 15 mm h^{-1}), and highest at 90 m from

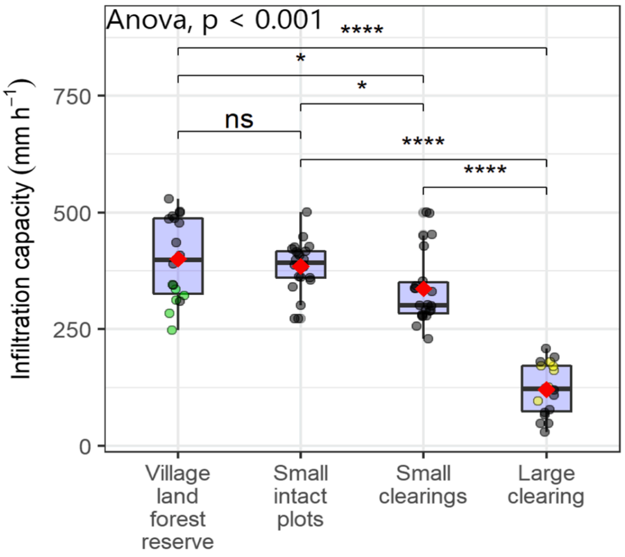

Fig. 2. Means (red diamond) and boxplot (median, first and third quartile) of soil steady-state infiltration capacity (mm h^{-1}) for the four different study treatments within the study area in Ulaya Mbuyuni village, Kilosa, Morogoro region, Tanzania. Green dots show steady-state infiltration capacity (mm h^{-1}) of plots at 10 m from the road within the village forest reserve. Yellow dots show steady-state infiltration capacity (mm h^{-1}) of plots at 10 m 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). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

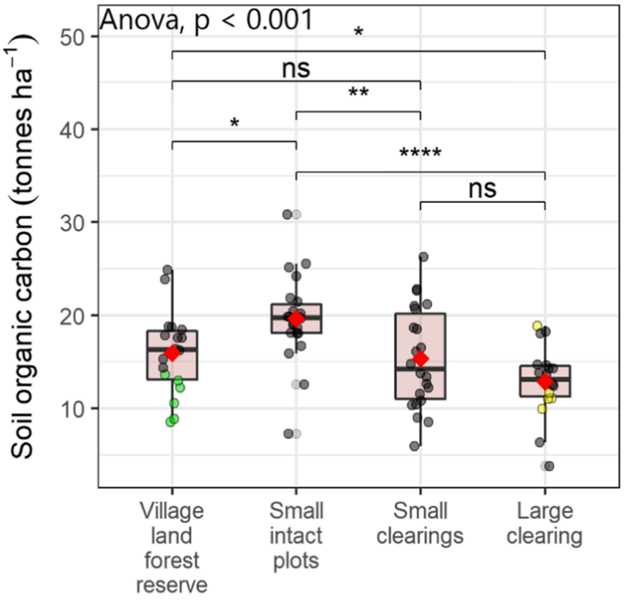

Fig. 3. Means (red diamond) and boxplot (median, first and third quartile) of soil organic carbon (tonnes ha^{-1}) for the four different study treatments within the study area in Ulaya Mbuyuni village, Kilosa, Morogoro region, Tanzania. Green dots show soil organic carbon (tonnes ha^{-1}) of plots at 10 m from the road within the village forest reserve. Yellow dots show soil organic carbon (tonnes ha^{-1}) of plots at 10 m from the road within the large clearing. Other plots are shown in gray. Asterisks denote significance levels (‘***’ = < 0.001, ‘**’ = 0.001–0.01, ‘*’ = 0.01–0.05, ‘ns’ > 0.05). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

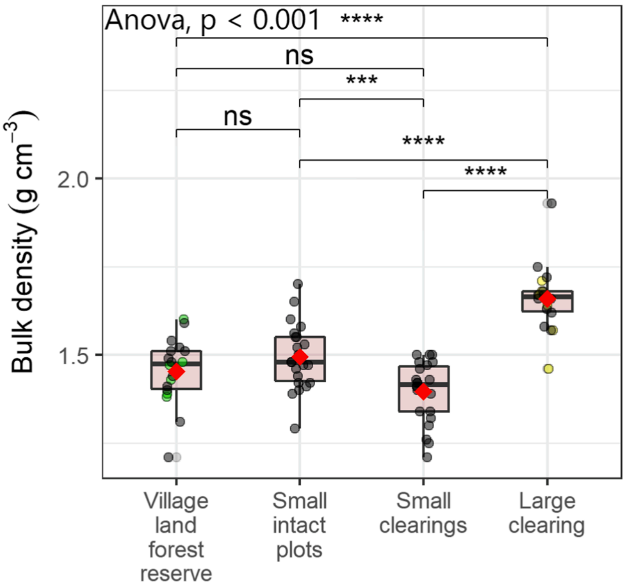

Fig. 4. Means (red diamond) and boxplot (median, first and third quartile) of soil bulk density (g cm^{-3}) for the four different study treatments within the study area in Ulaya Mbuyuni village, Kilosa, Morogoro region, Tanzania. Green dots show soil bulk density (g cm^{-3}) of plots at 10 m from the road within the village forest reserve. Yellow dots show soil bulk density (g cm^{-3}) of plots at 10 m from the road within the large clearing. Gray dots show soil bulk density (g cm^{-3}) in different treatments within the study area. Asterisks denote significance levels (‘***’ = < 0.001, ‘**’ = 0.001–0.01, ‘*’ = 0.01–0.05, ‘ns’ > 0.05). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the road (469 ± 6 mm h^{-1}). There was a significant difference in steady-state infiltration capacity between plots located at 10 and 50 m ($p = 0.04$) and between 10 and 90 m ($p = 0.009$), but not between 50 and 90 m ($p = 0.4$). Similarly, mean SOC stock (Fig. 5c) was lowest in plots at 10 m from the road (11 ± 0.4 tonnes ha^{-1}) and increased with an increase in distance to 50 m (18 ± 0.6; $p = 0.002$ tonnes ha^{-1}), but not with a further increase from 50 to 90 m (18 ± 0.4; tonnes ha^{-1}, $p = 0.6$). Mean tree basal area was also lowest in plots located 10 m from the road (11 ± 0.3 m^2/ha) and increased at 50 m (15 ± 0.4 m^2/ha; $p = 0.008$), but no further increase was observed from 50 to 90 m (15 ± 0.4 m^2/ha, $p = 0.9$) (Fig. 5e). Mean soil bulk density was 1.46 ± 0.01 g cm^{-3} in plots at 10 m from the road, 1.48 ± 0.01 g cm^{-3} at 50 m and 1.42 ± 0.02 g cm^{-3} at 90 m (Fig. 5g). There was no difference in mean soil bulk density between 10 and 50 m ($p = 0.34$), 50 and 90 m ($p = 0.57$), or 10 and 90 m from the road ($p = 0.94$). In the large clearing no clear pattern was observed regarding distance from the road, for any of the parameters measured (Fig. 5b, d, f, and h).

We found a significant positive relationship between steady-state infiltration capacity (mm h^{-1}) and basal area (m^2/ha) (Fig. 6, Table 2).

4. Discussion

As we hypothesized, larger clearings in miombo woodlands have a more negative impact on soil bulk density, SOC content, and steady-state infiltration capacity than smaller clearings. Steady-state infiltration capacity in the large clearing was 30 % of that in the village forest reserve, while in small clearings it was 84 %, indicating that it is not just the cutting of trees that matters but also the size of the area being cleared. Similar patterns were observed for soil bulk density and SOC stock. We also found that the presence of roads affects the properties of the nearby forest and its soils, as indicated by lower tree basal area, infiltration capacity, and SOC stock on plots close to the road compared to those further into the forest.

Across all treatments, soil steady-state infiltration capacity and SOC

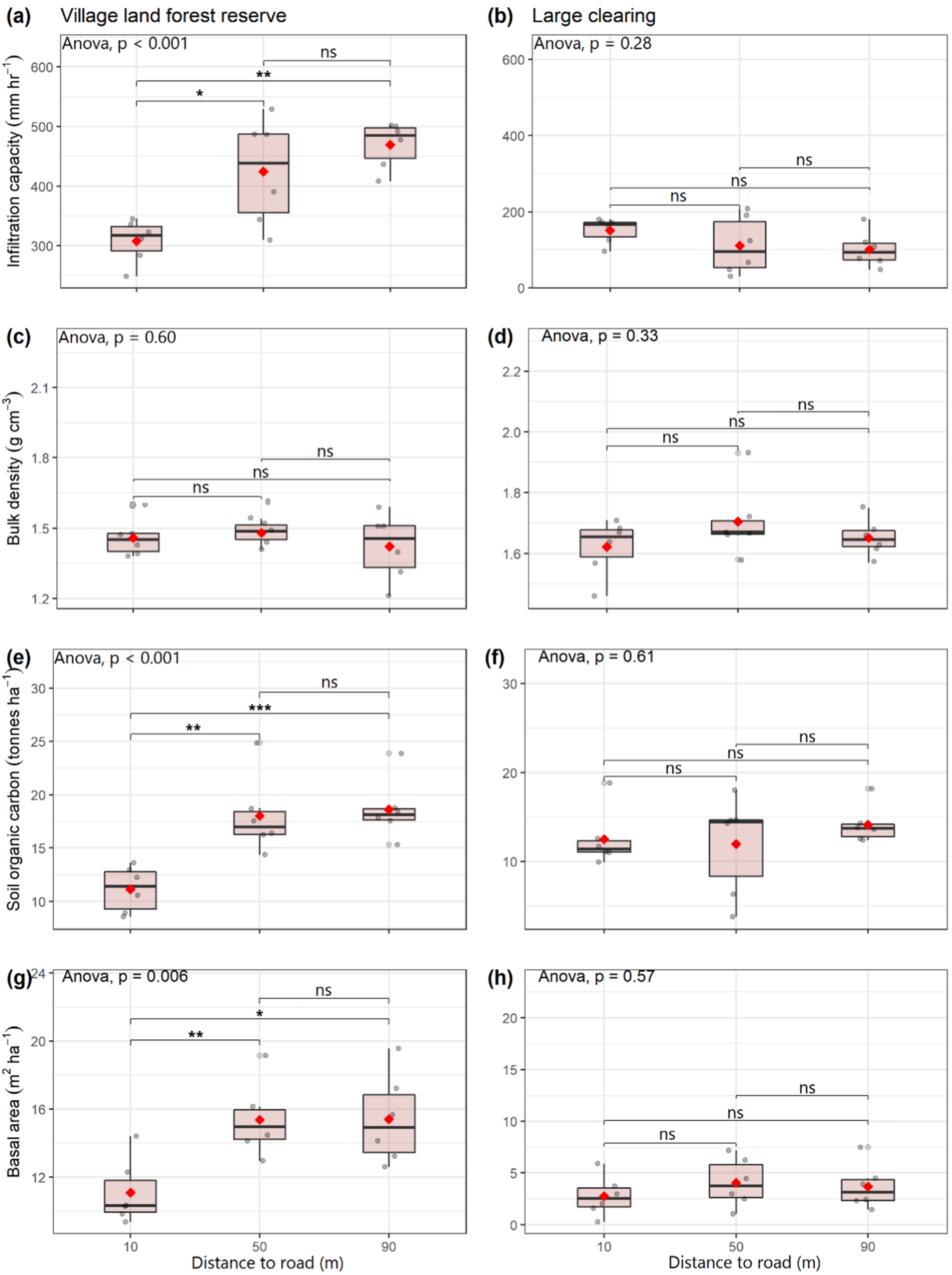

Fig. 5. Means (red diamond) and boxplot (median, first and third quartile) of; (a, b) topsoil steady-state infiltration capacity (mm h^{-1}), (c, d) topsoil bulk density (g cm^{-3}), (e, f) topsoil soil organic carbon (tonnes ha^{-1}) (g, h), and basal area m^2/ha), against the distance from the sampling plots to the road (m) within the village land forest reserve (left column) and large clearing (right column) in the study area in Ulaya Mbuyuni village, Kilosa-Morogoro, Tanzania. Red diamonds show the mean values per treatment. Gray dots are the individual observations. Signif. codes. '***' = < 0.001, '**' = 0.001–0.01, '*' = 0.01–0.05, 'ns' > 0.05. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

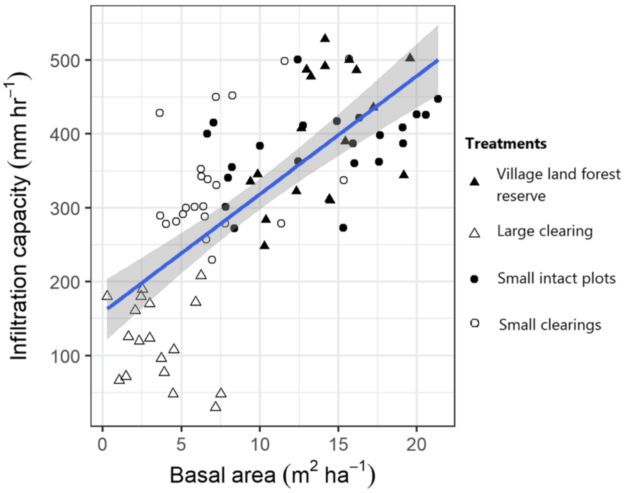

Fig. 6. Scatter plot showing the relationship between steady-state infiltration capacity (mm h^{-1}) and basal area (m^2/ha) in the study area in Ulaya Mbuyuni village, Kilosa-Morogoro, Tanzania. Different symbols indicate different treatments. The regression line is shown in blue, and the 95 % confidence interval in gray. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 2

Regression coefficients and *p*-values for the mixed-effect linear model showing the relationship between infiltration capacity (mm h^{-1}) as the dependent variable and basal area (m^2/ha), SOC stock (tonnes ha^{-1}), and bulk density (g cm^{-3}) as covariates, with treatments (village land forest reserve, large clearing, small intact plots, and small clearings) as a random effect in the study area in Ulaya Mbuyuni village, Kilosa-Morogoro, Tanzania. Marginal and conditional R^2 are also indicated (Nakagawa et al., 2017).

Parameter	Coefficients	p-value	Marginal R^2	Conditional R^2
Intercept	224.79	0.12	0.12	0.71
Basal area	6.99	0.002		
SOC stock	0.91	0.59		
Bulk density	2.49	0.98		

stock increased with basal area (Fig. 6), indicating that tree cover positively affects these two soil properties. Trees have been widely documented to improve soil's physicochemical properties (Lal, 1996; Rhoades, 1997; Shen et al., 2019), primarily through increased addition of organic matter (Barros & Fearnside, 2016; Devi, 2021; Omoro et al., 2013; Stockmann et al., 2015) which enhances the presence of both flora and soil fauna around them (Korboulewsky et al., 2016; Shirima et al., 2015). Trees also influence the amount of sunlight and wind speed, and moderate the heat and moisture around them (De Frenne et al., 2021), protect the ground from excessive heat from direct sunlight and from the force of rainfall (Vasconcelos & Sacht, 2020), and create hotspots of water input through stemflow (Metzger et al., 2021 & 2019). Because of this, trees create a favorable microclimate for local ecosystem niches, including those for soil microbes and soil fauna that further positively influence soil properties around them (Barrios et al., 2012). Furthermore, the activity of soil fauna and roots enhances soil aggregation and macroporosity, leading to improved soil water conductivity and drainage (Belsky et al., 1993; Benegas, 2013). Our findings support previous studies which have shown increased soil infiltration capacity with tree cover in various tropical ecosystems (Bargués-Tobella et al., 2014; Benegas, 2013 & 2018; Ilstedt et al., 2016; Lulandala et al., 2021). Tree removal typically reduces the quantity and quality of organic matter input and increases soil surface exposure to rainfall and sunlight (Bhuyan & Laskar, 2020), which can result in higher impact from raindrops and crust formation at the soil surface, leading to lower infiltration and increased surface runoff and erosion (Aber et al., 2010). In addition, during the dry season, the absence of tree cover means less protection against direct sunlight, which may result in increased soil evaporation (Ellison et al., 2017) and photodegradation of soil organic matter (Barnes et al., 2012) which can, in turn, lead to increased soil bulk density.

Mean steady-state infiltration capacity in small clearings was almost three times higher than in the large clearing, supporting the conclusions of previous studies on the hydrological impacts of gaps in the tree canopy in other systems such as agroforestry parklands (Bargués-Tobella et al., 2014 & 2020). The difference in soil hydrological properties between large and small clearings in our study can be explained in part by a tree proximity effect (Kirchhoff et al., 2021; Yadessa et al., 2001; Zinke, 1962). As well as the influence trees have on the area under their canopy, they may also impact neighboring open areas through the activity of their roots, which extend beyond the canopy edge (Bargués-Tobella et al., 2014; Benegas, 2013; Dunkerley, 2000; Lyford & Qashu, 1969). The ratio of canopy to root system radius for trees and shrubs in drylands can be as small as 1:10 (Lejeune et al., 2004), suggesting that the influence of trees on the surrounding soil extends to a considerable distance (Benegas, 2013; Dunkerley, 2000; Lyford & Qashu, 1969; Metzger et al., 2021). In our study, in small clearings the additive effect of tree roots from all sides of the clearings enhanced the soil properties more effectively than in the larger clearing. This phenomenon has been reported in other forest gap studies, which have shown improvements in soil quality - especially soil properties such as SOC stock, bulk density, and porosity - from the center to the edge of the gap, as well as in smaller gaps compared to large ones (He et al., 2015). Findings from a study in an agroforestry parkland in Burkina Faso also indicate that small open areas had higher preferential flow and more deep water drainage than larger ones (Bargués-Tobella et al., 2014 & 2020). In our study area small clearings sometimes contained retained trees which had been spared from the initial harvesting, which may have further contributed to better soil hydrology. The observed differences in infiltration capacity between large and small clearings are particularity interesting given the high soil sand content (around 80 %) in our study area. Sandy soils typically exhibit high infiltration rates (Basset et al., 2022) and one would a priori expect that they are less susceptible to changes in land use and land cover. However, our findings show that this is not the case, highlighting the importance of land use and land cover as drivers of soil hydrological functioning.

Within the village land forest reserve, basal area, steady-state infiltration capacity, and SOC stock were lower near the road than further into the forest (>50 m). Soil properties are often disturbed where a road or path crosses or comes close to forested areas (Deljouei et al., 2018; Strömquist & Backéus, 2009). Forest roads have been shown to negatively affect numerous properties of nearby forests, including forest species composition and structure (Zamani et al., 2020), the physical and chemical properties of soils (Olander et al., 1998), and the spatial–temporal distribution of forest fauna (Boston, 2016). However, the effect depends on the size and usage of the road (TRB & NRC, 2005). In our study area, forest roads are mainly used by people, motorcycles, livestock, and sometimes tractors coming and going from the forest to transport charcoal and timber. The decrease in infiltration capacity near the road could be explained by loss of tree cover, which may be due to illegal tree cutting for building material (poles) or firewood collection (Manyanda et al., 2021), frequent movement of livestock, and grazing (Lulandala et al., 2021). Both illegal cutting and livestock grazing lead to biomass removal, which may, in turn, decrease soil organic matter and SOC stock (Devi, 2021).

Our findings show that tree harvesting for charcoal production negatively affects soil hydrological properties and SOC stock. However, large clearings appear more detrimental than small clearings. The deterioration of soil hydrological properties and SOC stock in large clearings can lead to increased surface runoff and soil erosion (Haghnazari et al., 2015), consequently reducing soil and groundwater recharge (Bargués-Tobella et al., 2014) and leading to an even drier system. On the other hand, small clearings showed promising potential for increased underground water recharge through relatively high infiltration capacity and soil protection compared to large clearings. In small clearings, water loss by transpiration will be reduced compared to an intact forest, while maintaining a comparatively high infiltration capacity (Ilstedt et al., 2016). This may significantly affect the groundwater budget, making small clearings a better alternative for improved groundwater recharge than the unharvested forest. The findings of this study could play a critical role in informing the proper management of Miombo woodlands, but further research is needed to determine optimum gap sizes and retention of trees in gaps with respect their impact on groundwater recharge and other ecosystem services.

5. Conclusion

Sustainable tree harvesting schemes are essential to reduce the degradation of dry forests and woodlands such as Miombo. Our findings indicate that excessive gap sizes (90,000 m^2) negatively affect soil hydrological functioning, and highlight the importance of considering such impacts when designing sustainable wood harvesting schemes. Results from this study also suggest that livestock grazing in combination with charcoal production has a further negative impact on the hydrological properties of soil. Therefore, where other land uses co-occur in the harvested forest, management must take such interactions into consideration. Finally, we have shown that small clearings, while not completely unaffected, experience relatively small changes in terms of soil hydrology and SOC stock compared with unharvested forests and this, combined with lower transpiration losses, may even contribute to higher levels of groundwater recharge. Taken together, our findings suggest that small clearing size (2,500 m^2) minimizes the negative impacts of tree harvesting for charcoal production on soil hydrological functioning and that controlling livestock grazing can further minimize soil degradation.

CRediT authorship contribution statement

L. Lulandala: Conceptualization, Methodology, Investigation, Validation, Data curation, Formal analysis, Project administration, Writing - original draft, Writing, reviewing and editing. **A. Bargués-Tobella:** Writing - original draft, reviewing and editing. **C.A. Masao:** Resources, Project administration, Writing - original draft, Writing, reviewing and editing. **G. Nyberg:** Conceptualization, Funding acquisition, Project administration, Supervision, Vizualization, Writing, Reviewing and editing. **U. Ilstedt:** Conceptualization, Project administration, Supervision, Vizualization, Writing, Reviewing and editing. All authors read and approved the final draft.

Declaration of Competing Interest

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

Data availability

Data will be made available on request.

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