The effects of *Dichrostachys cinerea* (sickle bush) encroachment on herbaceous vegetation attributes in the grazing lands of Monduli district, Tanzania

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Abstract

*Reports indicate a global increase in bush encroachment, transforming open grassy systems into dense thickets of woody plants. Dichrostachys cinerea, commonly known as sickle bush or endundulu by Maasai communities, is one of the encroaching shrubs in Africa that tends to form dense thickets, which are difficult to eliminate due to its invading ability. Recently, this plant has been observed to increase tremendously in the grazing lands of Monduli district, although little is known about its impact on herbaceous vegetation attributes in the district. This study aimed to assess the effects of this encroachment on herbaceous vegetation attributes in the grazing lands***.** *Three villages (Mswakini chini, Mswakini juu and Naitolia) were purposively selected and involved in this study. Vegetation surveys and measurements, involving transect lines, plots, sub-plots, and quadrats, were conducted in encroached and adjacent non-encroached sites in the grazing land of each village. A total of 72 (20m × 20m) plots were demarcated, and in each plot, there were five (5m × 5m) sub plots, from where the biomass readings were taken (using a pasture disc meter), vegetation cover estimated (using a pin quadrat), and individual species were identified and counted within 0.25m² quadrats. A total of 64 herbaceous species*

were obtained during this study, of which 37.5% were found only in non-encroached areas, 25% only in encroached areas, and 37.5% were common to both sites. The study found that encroached areas had significantly lower biomass (409.604 Kg DM/ha) and a lower cover of desirable vegetation (10%), with greater bare ground cover (56%) compared to non-encroached sites, which had 890.72 Kg DM/ha biomass, 48% desirable vegetation cover, and 19% bare ground cover. The condition of the grazing lands was generally poorer in encroached sites than in non-encroached sites. The findings of this study show the negative effects of Dichrostachys cinerea encroachment on herbaceous vegetation attributes in grazing lands, which challenges sustainable livestock production. These results are relevant to local pastoralists, land managers, and policymakers in Monduli district and other similar areas facing sickle bush encroachment. It is recommended that strategies such as mechanical bush control, controlled grazing, and the use of prescribed burning be implemented, with the aid of advanced studies on the bush, active involvement and training of pastoralists to manage the bush effectively. Policymakers should provide frameworks and resources to promote sustainable forage production.
Keywords: Encroachment, Forage, Grazing lands, Herbaceous, Sickle bush.

1.0 Introduction

Bush encroachment is described as the process of thickening and/or expansion of woody plant species in open grasslands and savannahs (Leitner et al., 2018). This process, in turn, leads to a reduction in the grazing carrying capacity of the encroached areas (Eldridge et al., 2011). It is important to consider how these changes directly affect grazing lands, especially in Africa. For instance, the loss of palatable plants (Ward, 2005) means less forage for livestock, which can seriously affect the livelihoods of pastoral communities.

This is a common problem affecting many grassland and savannah ecosystems worldwide (Mudzengi et al., 2014). Bush encroachment primarily impacts arid and semi-arid rangelands, which make up about 40% of the world's pastoral lands (Huang et al., 2018). These encroachments are increasing globally, transforming open grassy systems into dense thickets of woody plants (Gillson et al., 2012; Parr et al., 2012). This degradation of grasslands affects a vital primary resource for livestock production, especially in sub-Saharan Africa (de Klerk, 2004; Nesongano, 2018). Notably, although bush encroachment is a global issue, African rangelands are considered more susceptible (Osborne et al., 2018).

Dichrostachys cinerea (sickle bush) is one of the many thorny, encroaching woody plants in Africa (Kraaij and Ward, 2006). In Swahili, it is known as *Mkula jembe* or *Mvunja shoka* (Orwa et al., 2009), and among the Maasai in the Monduli district, it is commonly referred to as *Endundulu* (TTPC, 2010). Its high encroachment potential leads to a significant increase in density making it invasive with negative effects on grazing value of invaded rangelands. Furthermore, the plant forms dense thickets that are challenging to eliminate (Roques et al., 2001). Its invasive qualities are further supported by its ability to produce numerous viable seeds, regardless of the plant's age (Mudzengi et al., 2014). Additionally, the high propagule pressure, aided by the longevity of the seeds (which can remain in the soil for up to five years), contributes to its resilience (Bussa and Shibru, 2020; Mudzengi et al., 2014). Seed dispersal mechanisms are often supported by animals, such as goats, which feed on the pods (Tjelele et al., 2012). The species also possesses a deep taproot and numerous lateral roots (US Forest Service, 2011), along with prolific reproduction from root suckers (Mudzengi et al., 2014; Wakeling and Bond, 2007), facilitating rapid growth and natural regeneration (Zeid et al., 2009; Bussa and Shibru, 2020). Furthermore, according to Blaser et al. (2014), the reproduction of this plant is likely influenced by its strong ability to fix nitrogen symbiotically. Sickle bush is tolerant to disturbances, thriving particularly well in overgrazed areas (Mudzengi et al., 2014). Currently, there is a notable increase in sickle bush in the Monduli district, mostly attributed to factors such as fire suppression, climate change, shifts in land use practices, and overgrazing (Oba et al., 2000; Smit, 2004; Tews and Jeltsch, 2004; Yusuf et al., 2011). Its spread in Monduli district, where grazing lands are important for local communities, shows the need for strategies to manage it effectively without affecting the local communities.

There are conflicting reports on the effects of bush encroachment on the herbaceous plant community growing underneath. For instance, some researchers have reported a positive impact on species richness and diversity (Belay et al., 2013), while others have found a negative effect on understory vegetation variables (Mudzengi et al., 2014). Yet others have reported a neutral impact on understory productivity (Belay et al., 2013). Studies suggest that the impact of woody encroachment on herbaceous vegetation may depend on the type of encroacher species involved (Osborne et al., 2018). This means the impact depends on many factors, such as the type of encroaching plant and local conditions. For this reason, studying this issue in Monduli district can help provide clearer answers.

There is a scarcity of information on the impacts of the encroachment of sickle bush in the Monduli district. Most of the people in this district are Maasai pastoralists, who depend on available grazing lands for their livestock (Homewood et al., 2006). Therefore, the management of these lands is very important, and effective management requires appropriate information (Fuhlendorf et al., 2012). This study aimed to assess the effects of sickle bush encroachment on herbaceous vegetation attributes in the grazing lands of Monduli district. Specifically, the study addressed the following questions (i). Does sickle bush affect the herbaceous species composition and diversity? (ii).

Does sickle bush affect herbaceous vegetation cover, biomass production and desirability?

2.0 Materials and methods

2.1 The study area

This research was conducted in Monduli district, involving the grazing lands of three pastoral villages which are Mswakini juu, Mswakini chini and Naitolia (Figure 1). The study sites were purposively selected due to the presence of sickle bush encroachment. Monduli district is found in Arusha region, Northern Tanzania located at latitude 3°20′S and longitude 36°15′E, mainly inhabited by Maasai pastoralists (97%) (Kimaro et al., 2018) whose income mainly come from livestock keeping (Homewood *et al.*, 2006).

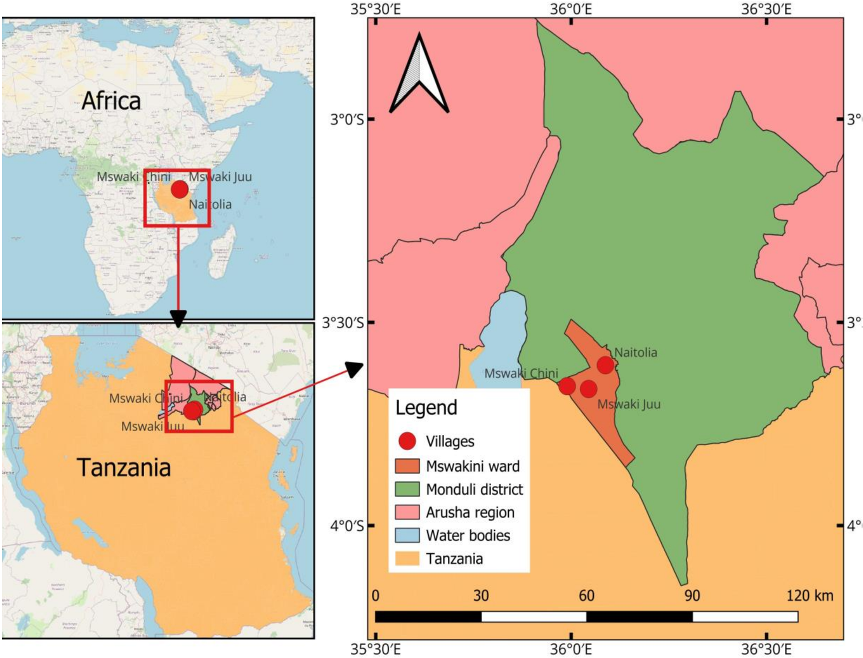

Figure 1. Map of the study area with selected study villages shown in red dots (Source: Mguluka et al., 2023)

2.2 Research design

The research design used was a comparative field-based vegetation survey employing a stratified systematic sampling approach, preceded by a reconnaissance survey that involved interviewing local people to identify the encroached and non-encroached areas in each village. Thereafter, sickle bush encroached and non-encroached adjacent sites were selected in the grazing areas of each selected village. In each site three transect lines of 300 meters were laid out. On each transect, four 20 m × 20 m plots were pegged systematically on the ground. Then, 5 m ×5m sub plots were established at four corners and one at the centre of each plot making a sum of five sub-plots on each plot. The total number of plots in each village was 24 (12 from each site). The number of sub plots was 120 per village. The plots were established at 50 m from the roads to reduce the edge effects.

2.3 Data collection

Data were collected after the first rains in the district (from January – March, 2023) where most of the herbaceous plants were at their blooming stage which made easy the identification of different plant species.

2.3.1 Species composition and desirability

Species identification and counting were done in five 5 m × 5 m sub plots in each plot using 0.5m × 0.5m quadrat which were randomly thrown twice in each sub plot. All the herbaceous species inside these quadrants were identified and counted. Then, names, numbers, life spans, life forms, and desirability of the species by the grazing animals were recorded. Desirability was assessed based on the literature and local people's grazing experience. The species were grouped into four categories which are (i) highly desirable (plants frequently grazed and preferred by livestock due to their nutritional quality and palatability), (ii) moderately desirable (plants that are consumed by livestock but less preferred compared to highly desirable species),(iii) less desirable (plants that are grazed only under certain conditions, typically when more preferred species are unavailable),(iv) undesirable (plants that are rarely or never grazed by livestock due to their low palatability, toxicity, or other factors that make them unsuitable for grazing).Species identification was done by using the plant Net mobile application. This application identifies plants by using visual recognition technology and the Tanzanian forage species identification guide which was developed by Kayombo *et al.* (2016). Species which were not identified in the field were collected (both pictures and samples) for further identification.

2.3.2 Above-ground biomass

Data for above-ground biomass was collected by using the pre-calibrated pasture disc meter in the sub plots. The calibration was done by relating the height of the disc and the biomass of samples that were randomly collected from each study site to cover the variations in the study site. A total of 30 samples were clipped from under the disc plate and their respective heights were recorded. The sample size was chosen to ensure the data captured the variation in vegetation across the sites while being manageable within the available time and resources. The choice of 30 samples was done after effective visual assessment and was influenced by logistical constraints, like the time required clipping, process, and analyzing each sample. Given the need to oven-dry samples, the selected samples were manageable within the study's available timeframe and resources without compromising the quality of the calibration process. The samples were then taken to Tanzania Agricultural Research Institute (TARI) Selian Laboratories located in Arusha City, where they were oven-dried at 70◦C for 48 hours. The final dry weights in Kilograms per hectare were plotted against the respective heights in metres to obtain the best-fit equation which was then inserted in the pasture disc meter. The obtained equation for calibration was $Y=0.151X+0.0001$(R^2=0. 54), where X is bulk height in metres.

2.3.3 Vegetation cover (cover of soil by plants)

Vegetation cover was determined by using a pin quadrat which was inserted by six pins to purposively point to a specific cover type in each sub plot. Four cover categories are litter (dead plants cover or freshly fallen plant parts), bare ground (points with no plants or litter materials), desirable plants cover (points with plants that are edible and preferred by the grazing animals) and undesirable plants cover (points with plants that are not edible and not preferred by the grazing animals), were recorded depending on the point that the pins intercepted. Number of pins hitting each cover category was recorded for calculations of vegetation cover in percentages. To minimize bias, the pin quadrat was placed randomly within each sub plot, ensuring that all cover types had an equal chance of being intercepted. Additionally, the same procedure was consistently followed across all sub plots and data was collected by the same person to reduce variation in how the method was applied.

2.4 Data analysis

Data were checked for normality, outliers and homogeneity of variances before analyses as proposed by Zuur *et al.* (2010). Independent sample t-test at 95% confidence interval was used to test for significance differences in measured vegetation variables between encroached and non-encroached sites. Statistical analyses were performed using SAS software (SAS, 2014). Species composition in both encroached and non-encroached sites was determined by using the following formula, which was described by Jawuoro *et al.* (2017), the formula uses the species' relative densities $\left(\frac{ni}{N}\right) X\ 100$

Where n_i is the number of individual species per plot and N is the total quantity of species within the same plot. Species diversity was determined by using Shannon-Wiener's diversity index (H′) as described by Jawuoro *et al.* (2017). This is calculated by using the following formula.

$$H' = -\sum [\left(\frac{ni}{N}\right) X \, In \left(\frac{ni}{N}\right)]$$

Where N is the total number of individuals (or amount) for the site, n_i is the number of individuals of each species, and "ln" is the natural logarithm of the number.

Species richness (S) was calculated as the total number of species per plot.

Species evenness (E) was calculated as;

$$J = \frac{H'}{H\text{max-}}$$

Where; **H′** is Shannon Weiner's diversity index for the plot; Hmax is the natural log of species richness (S) =lnS.

Additionally, Jaccard's similarity index was used to compare the species in encroached and non-encroached sites. This measure of similarity is known to be unbiased and strong compared to other similarity indices (Ludwing and Reyonlds, 1988). The Jaccard's similarity index between encroached (i) and non-encroached area (j), was calculated by the following formula: $Sij = \frac{a}{a+b+c}$

Where: a= the number of species present in both i and j; b= the number of species in i but not in j; and c= represents the number of species present in j but not in i. This coefficient has a value from 0 to 1, where 1 reveals complete similarity, a similarity index close to 1 indicates high similarity, an index close to 0 indicates low similarity and 0 indicates complete dissimilarity (Musese et al., 2020).

3.0 Results

3.1 Herbaceous species compositions

A total of 64 herbaceous species were observed during this study of which 16 (25%) were in encroached sites only, 24 (37.5%) were in non-encroached sites only, and 24 (37.5%) were common to both sites. The life forms of these plants were, 47 forbs (73%), 11 grasses (17%), 5 shrubs (8%) and I sedge (2%). Their life spans include 26 annuals (40.6%) and 38 perennials (59.4%). The Jaccard's similarity index for species between the two sites category was found to be 0.375.

3.2 Species' desirability comparison between encroached and non-encroached sites

The number of highly desirable and moderately desirable herbaceous species was higher in non-encroached sites. Species that were unique to encroached sites were all undesirable or less desirable. Majority of encountered plants were undesirable to livestock (36) which is 56%, 11 plants (17%) were less desirable, 9 (14%) were moderately desirable and only 8 plants (13%) were highly desirable (Table 1). The common desirable herbaceous species observed in both sites were *Cenchrus stramineum*, *Cynodon dactylon*, and *Urochloa mosambicensis*.

Table 1. Comparison of species in terms of desirability between the two sites categories in Monduli Tanzania

Sites	Number of species				Total
	Undesirable	Less desirable	Moderately desirable	Highly desirable	
Encroached site	13	3	0	0	16
Non-encroached sites	12	4	4	4	24
Both sites	11	4	5	4	24
Total	36	11	9	8	64

Legend: the species found in both sites, were present in encroached and non-encroached sites

3.3 Herbaceous species richness, evenness and diversity

Plant species diversity, richness, and evenness were analyzed between the encroached and non-encroached grazing areas. The findings are summarized in Table 2. Shannon Wiener diversity index (H′) showed no significant difference (p = 0.4418) between the encroached areas (1.58 ± 0.07) and the non-encroached areas (1.66 ± 0.07). Similarly, species evenness did not differ significantly (p = 0.2073) between the encroached areas (0.75 ± 0.02) and the non-encroached areas (0.72 ± 0.02). However, species richness was significantly higher in the non-encroached areas (10.17 ± 0.5) compared to the encroached areas (8.78 ± 0.5), with a p-value of 0.0471, indicating a notable difference in the number of species present between the two categories.

Table 2. Plant species diversity, richness, and evenness between the two grazing areas categories in the study area, Monduli Tanzania

Parameters	Site category		p-values
	Encroached areas	Non-encroached areas	
Diversity (Shannon Weiner's index (H′)	1.58±0.07	1.66±0.07	0.4418
Species evenness	0.75±0.02	0.72±0.02	0.2073
Species richness	8.78±0.5	10.17±0.5	0.0471

3.4 Aboveground plant biomass

The mean aboveground plant biomass in the study area varied significantly between the encroached and non-encroached sites. Encroached sites had a substantially lower mean biomass ($p<0.0001$) compared to the non-encroached sites Figure 2.

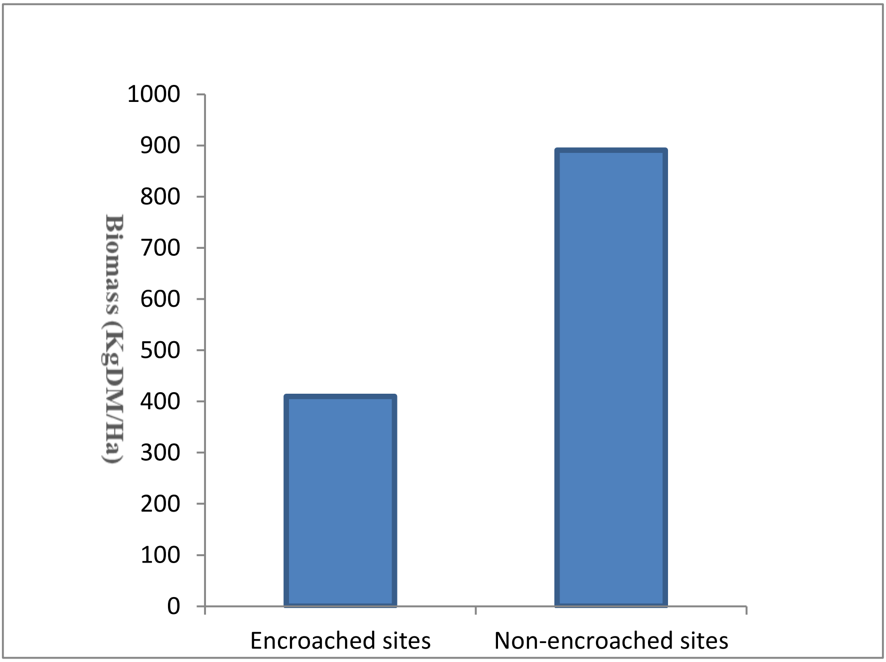

Figure 2. Mean aboveground plant biomass (kgDmHa^{-1}) among sickle bush encroached and non-encroached sites in Monduli Tanzania

3.5 Herbaceous plants cover, litter cover and bare land

The results indicate that the percentage of bare soil was significantly higher and the desirable plant cover was significantly lower in encroached sites compared to non-encroached sites. Litter (%) and undesirable plant cover (%) were statistically not significant (Table 3). When considering cover (%) in specific sites, encroached sites had higher percentages of bare soil (55.38%) compared to other cover categories. In non-encroached sites, desirable plant cover was higher (approximately 48%) compared to other cover categories.

Table 3. Live plant cover, dead plant cover (litter), and bare land cover of the two site categories in Monduli Tanzania

Cover type	site category		p-values
	Encroached sites	Non-encroached sites	
Desirable plants cover (%)	10±2.74	48±2.74	<.0001
Litter cover (%)	25±3.05	26±3.05	0.7200
Undesirable plants cover (%)	9±1.65	7±1.65	0.2787
Bare land (%)	56 ±3.1	19±3.1	<.0001

Legend: For desirable plants percent cover; 76-100=Excellent, 51-75=Good, 26-50=Fair, 1-25=Poor 0=Very poor. Vice versa is true for percentage bare land

4.0 Discussion

The presence of sickle bush changes the species composition in terms of desirability of species due to the presence of highly and moderately desirable species that were only found in the non-encroached sites and there were none unique to encroached sites. These species included *Eragrostis supebra*, *Panicum infestum*, *Digitaria ciliaris*, *Neonotonia wightii*, and *Dactyloctenum aegyptium* among others (Appendix 1). Among the 64 species that were identified during this study, 24 (38%) were not found in the encroached sites. Of all 16 species that were unique to encroached areas, 13 (81%) were undesirable and 3 (19%) were less desirable plants. This shows that sickle bush encroachment causes a reduction of desirable plants. These results agree with what has been observed by Yassin (2019) who reported fewer desirable plants in bush-encroached areas. Presence of many undesirables and less-desirables in encroached sites shows the changes in the nutritive quality of the plants growing under the bushes. These plants are mainly shade tolerant (Pang et al., 2017). According to Koerner and Collins (2014), the presence of many undesirable forbs and grasses in the grazing lands is an indicator of land degradation. Therefore, this change in species composition may be a result of land degradation that has been brought about by the presence of *Dichrostachys cinerea*.

This degradation is further highlighted by the low similarity in species composition between encroached and non-encroached sites. This shows that there is a fundamental change in species composition within the study area (Musese et. al., 2020). The low similarity between encroached and non-encroached sites might be due to the vigorous growth habit of sickle bush and its ability to alter the surrounding environment (Randle et al., 2018). Therefore, with time if this plant is not managed, more changes in species composition within the area are expected to occur.

The changes in species composition are accompanied by reductions in species richness; a key indicator of grazing land health (Wagg et al., 2022). Species richness was significantly lower in the encroached sites than in the non-encroached sites, similar to what has been reported by Bussa and Shibru (2020). This may be due to the competitive advantage sickle bush has over other plants in terms of important resources of light, water and space. Also, its vigorous growth habit and extensive root system enable it to grow densely over other plants co-occurring with it (Mudzengi et al., 2014). This finding highlights the adverse effects of sickle bush encroachment which likely reduce the establishment of less competitive species. Bush encroachment can lead to soil compaction, reduced seed germination of herbaceous plants, and habitat degradation, which collectively diminish species richness (Christian, 2010). In contrast, the non-encroached sites, likely experiencing lower grazing pressure, provide more favorable conditions for a diverse array of plant species to thrive.

Beyond species richness, the encroachment also significantly affects aboveground biomass, a critical component of grazing systems. The findings revealed that mean aboveground plant biomass was significantly lower in encroached sites compared to non-encroached sites. This suggests that sickle bush encroachment directly reduces the availability of forage, which is vital for maintaining healthy grazing systems. This reduction in biomass has important ecological and management implications for the grazing systems. The reduction in biomass production may be due to the strong shading effect that is brought about by the shrubby growth form of sickle bush (Randle et al., 2018). The results align with this finding, highlighting how shading by sickle bush may limit the growth of herbaceous plants, thus diminishing their biomass production. Shading reduces photosynthetically active radiation for the herbaceous layer below bushes and subsequently reduces the growth of these plants in the grass layer (Eldridge et al., 2011). Also, the increased competition for nutrients between the woody plants and herbaceous forages (Mudzengi et al., 2014) likely intensifies the reduction in biomass in encroached sites. Biomass production determines the amount of forage available for the livestock (Abule et al., 2007). This reduced biomass production significantly limits the forage supply for grazing animals, which may compromise livestock productivity and grazing management strategies. From an ecological point of view, reduced aboveground biomass may hinder important ecosystem functions such as carbon sequestration, soil stabilization, and nutrient cycling (Tessema et al., 2012). These ecosystem functions are essential for the overall health of the rangeland, and the reduction in biomass caused by bush encroachment may further promote environmental degradation. Small above-ground biomass yield in sickle bush encroached areas has also been reported by Smit (2004), Randle et al. (2018), and Bussa and Shibru (2020), which emphasizes the broader concern regarding the impacts of bush encroachment on land productivity.

The distribution of vegetation cover types further illustrates the impacts of bush encroachment on grazing lands. There were more desirable plants than other cover categories in non-encroached sites, which may be due to lack of competition for soil nutrients and water (Mudzengi et al., 2014), absence of allelochemicals (Carmenate, 2019), absence of shade effect and other disturbances from an encroacher. The same reasons may explain the low percentages (7% and 19% respectively) of non-desirable plants and bare lands in the non-encroached sites. The small bare ground and non-desirable plants cover with a high desirable plants cover in non-encroached sites, has also been reported by Bussa and Shibru (2020). Moreover, the percent of bare ground cover and desirable plants cover are used in assessing the range condition (Sangeda and Maleko, 2018). Therefore, condition was generally poor in the encroached sites than in non-encroached sites as indicated by higher percent (56%) of bare land cover and smaller percent of desirable plants cover (10%) in these encroached sites.

The high percentage of bare lands in encroached sites may be due to high sickle bush demands of water, nutrients and light, causing the death of herbaceous plants growing underneath similar to the reports by Angassa and Oba (2007). Additionally, according to Christian (2010), the water consumption and evapotranspiration rates are higher in woody plants (in this case sickle bush) than in herbaceous plants. Therefore, encroached soils tend to be dry and less nutritive. This leads to the death of many herbaceous plants and the creation of bare grounds. Both desirable and undesirable plant cover was also lower which may be due to large space being occupied by an encroacher.

5.0 Conclusions and recommendations

Sickle bush is a native plant which can suppress other plants as revealed through the findings of this study. It affects the majority of herbaceous vegetation attributes including, desirable vegetation cover, above-ground biomass production and species richness. These effects of sickle bush on the herbaceous plants if not controlled, will in the long term affect the carrying capacity of the grazing lands of the Monduli district and make them more susceptible to invasion by other invading plants. These findings are relevant to local pastoralists, land managers, and policymakers in Monduli district and other similar areas facing bush encroachment. It is recommended that strategies such as mechanical de-bushing, controlled grazing, and the use of prescribed burning be implemented, with the aid of advanced studies on the bush, active involvement and training of pastoralists to manage the bush effectively. Policymakers should provide frameworks and resources to promote sustainable forage production.

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References
Abule, E., Snyman, H.A., and Smit, G.N. (2007). Rangeland evaluation in the middle Awash valley of Ethiopia: II. Woody vegetation. Journal of Arid Environments 70: 272-292.

Angassa, A., and Oba, G. (2007). Effects of management and time on mechanisms of bush encroachment in southern Ethiopia. African Journal of Ecology 46: 186-196.

Belay, T.A., Totland, Ø., and Moe, S.R. (2013) Ecosystem responses to woody plant encroachment in a semiarid savanna rangeland. Plant Ecology 214: 1211-1222.

Blaser, W. J., Shanungu, G.K., Edwards, P.J., and Olde Venterink, H. (2014). Woody encroachment reduces nutrient limitation and promotes soil carbon sequestration. Ecology and Evolution 4: 1423-1438.

Bussa, B., and Shibru, S. (2020). Effects of Sicklebush (Sickle bush (L) wight and arn. shrub) Encroachment on Floristic and Vegetation Structure in Semi-arid Savannah of Southern Ethiopia. August. 01.

Carmenate Figueredo, O., Pupo Feria, C., and Herrera Toscano, J. A. (2019). “Proposal of actions for the agroecological reconversion of a farm in Las Tunas municipality” pp. 264-274.

Christian, C. (2010). Groundwater Resources in Namibia: A Desktop Study. Namibia Agricultural Union. Windhoek.

De Klerk, J. N. (2004). Bush encroachment in Namibia. Report on Phase 1 of the Bush encroachment, monitoring and management project. Windhoek, Namibia: Ministry of Environment and Tourism.

Eldridge, D.J., Bowker, M.A., Maestre, F.T., Roger, E., Reynolds, J.F., and Whitford, W.G. (2011). Impacts of shrub encroachment on ecosystem structure and functioning: Towards a global synthesis. Ecology Letters 14: 709-722.

Fuhlendorf, S.D., Engle, D.M., Elmore, R.D., Limb, R.F., and Bidwell, T.G. (2012). Conservation of pattern and process: developing an alternative paradigm of rangeland management. Rangeland Ecology and Management 65: 579-589.

Gillson, L., Midgley, G.F., and Wakeling, J.L. (2012). Exploring the significance of land cover change in South Africa.

Homewood, K., Trench, P., Randall, S., Lynen, G., and Bishop, B. (2006). Livestock health and socio-economic impacts of a veterinary intervention in Maasailand: infection-and treatment vaccine against East Coast fever. Agriculture Systems 89: 248-271.

Huang, C.Y., Archer, S.R., McClaran, M.P., and Marsh, S.E. (2018). Shrub encroachment into grasslands: End of an era? Peer-reviewed Journal 6: e5474.

Jawuoro, S.O., Koech, O.K., Karuku, G.N., and Mbau, J.S. (2017). Plant species composition and diversity depending

on piospheres and seasonality in the southern rangelands of Kenya.
Kayombo, C.J., Giliba, R.A., Sangeda, A.Z., and Maleko, D.D. (2016). Tanzania Forage Species Identification Guide. Vilex Enterprises (Dar es Salaam, Tanzania).
Kimaro, E.G., Mor, S.M., and Toribio, J.A.L. (2018). Climate change perception and impacts on cattle production in pastoral communities of northern Tanzania. Pastoralism 8: 1-19.
Koerner, S.E., and Collins, S.L. (2014). Small-scale patch structure in North American and South African grasslands responds differently to fire and grazing. Landscape Ecology 28: 1293-1306.
Leitner, M., Davies, A.B., Parr, C.L., Eggleton, P., and Robertson, M.P. (2018). Woody encroachment slows decomposition and termite activity in an African savanna. Global Change Biology 24: 2597-2606.
Leticia, J.M., Samora, M.A., Deo, D., Shirima, A.W., and Kilewa, R. (2020). "Effects of the Abundance of Parthenium Hysterophorus on the Composition and Diversity of Other Herbaceous Plant Species in Simanjiro Rangeland, Tanzania." International Journal of Engineering Technologies and Management Research 7: 11-20.
Mguluka, E.E., Sangeda, A.Z., and Kimera, S.I. (2024). Pastoralists' Perception and Attitude on Invasion of Dichrostachys cinerea (sickle bush) in the Grazing Lands of Monduli District, Tanzania. Pastures and Pastoralism 02: 1-19 Doi: https://doi.org/10.33002/pp0201.
Mudzengi, C., Kativu, S., and Dahwa, E. (2014). Effects of Sickle bush (l.) Wight and Arn (Fabaceae) on herbaceous species in a semi-arid rangeland in Zimbabwe. Nature Conservation 7: 51-60. https://doi.org/10.3897/natureconservation.7.526460.
Nesongano, W. (2018). The effects of climate change, land-use and elevated carbon dioxide on tree-grass interactions in Southern African savannas. Universät Tübingen, Germany.

Orwa, C.A., Mutua, K.R., Jamnadass, R., and Anthony, S. (2009). Agro-forestry Database: a tree reference and selection guide version 4.0.

Osborne, C.P., Charles-Dominique, T., Stevens, N., Bond, W.J., Midgley, G., and Lehmann, C.E.R. (2018). Human impacts in African savannas are mediated by plant functional traits. New Phytologist 220: 10-24.

Parr, C.L., Gray, E.F., and Bond, W.J. (2012). Cascading biodiversity and functional consequences of a global change-induced biome switch. Diversity and Distributions 18: 493-503.

Randle, M., Stevens, N., and Midgley, G. (2018). Comparing the differential effects of canopy shading by Sickle bush and Terminalia sericea on grass biomass. South African Journal of Botany 119: 271-277.

Tessema, Z., K, de Boer W.F., and Baars R.M.T. (2012). Influence of grazing on soil seed banks determines the restoration potential of aboveground vegetation in a semi-arid savanna of Ethiopia. Biotropica 44: 211-219.

Tjelele, J.T., Dziba, L.E., and Pule, H.T. (2012). Recovery and Germination of Sickle bushSeeds Fed to Goats (*Capra hircus*). Rangelands 65: 105-108.

TTPC: Top Tropicals Plant Catalogue (2010). *Dichrostachys cinerea*. www.toptropicals.com.

US Forest Service, (2011). Sickle bush (L.) R. Wight and Arnott. Pacific Island Ecosystems at Risk (PIER).

Utaile, Y.U., Honnay, O., Muys, B., Cheche, S.S., and Helsen, K. (2021). Effect of Sickle bushencroachment on plant species diversity, functional traits and litter decomposition in an East-African savannah ecosystem. In Journal of Vegetation Science 32:1.

Wagg, C., Roscher, C., and Weigelt, A. (2022). Biodiversity–stability relationships strengthen over time in a long-term grassland experiment. Nature Communications 13: 7752.

Wakeling, J., and Bond, W. (2007). Disturbance and the frequency of root suckering in an invasive savanna shrub,

Dichrostachys cinerea. African Journal of Range and Forage Science 24: 73-76.
Ward, D. (2005). Do we understand the causes of bush encroachment in African savannas? African Journal of Rangeland and Forage Science 22: 101-105.
Yassin, I. M. (2019). Bush Encroachment in Borana Rangeland in the Case of Southern Ethiopia: Causes, Impacts and Management Implications. International Journal of Agriculture Innovations and Research 7: 2319-1473.
Yusuf, H., Treydte, A.C., Demissew, S., and Woldu, Z. (2011). Assessment of woody species encroachment in the grasslands of Nechisar National Park, Ethiopia. African Journal of Ecology 49: 397-409.
Zeid, A. H., Hifnawy, M. S., and Mohammed, R.S. (2009). Phenolic Compounds and Biological Activities of Sickle bushL. Medicinal and Aromatic Plant Science and Biotechnology 3: 42-49.

Appendix 1: List of plant species recorded during the vegetation survey in the study area, Monduli, Tanzania

S/n	*spp* name	Growth form	Life span	Desirability	Frequency of occurrence	
					Encroached sites	Non-encroached sites
1	***Acalypha indica***	forb	annual	Undesirable	0	41
2	***Acanthospermum hispidum***	forb	annual	Undesirable	0	36
3	***Achyranthes aspera***	forb	annual	less desirable	0	51
4	***Albuca bruteata***	forb	perennial	Undesirable	0	2
5	***Allium spp***	forb	perennial	Undesirable	0	2
6	***Alternanthera laguroides***	forb	perennial	Undesirable	109	0
7	***Amaranthus muricatus***	forb	annual	Undesirable	13	189
8	***Amaranthus viridis***	forb	annual	less desirable	2	0
9	***Arachis hypogea***	forb	annual	less desirable	0	11
10	***Arachis glabrata***	forb	perennial	highly desirable	0	3
11	***Aristida adoensis***	Grass	perennial	less desirable	86	790
12	***Asparagus officinalis***	forb	perennial	less desirable	2	2
13	***Bidens pilosa***	forb	annual	Less desirable	0	2
14	***Bothriochloa pertusa***	Grass	perennial	Moderately desirable	16	632
15	***Brachiaria deflexa***	Grass	perennial	highly desirable	70	470
16	***Cenchrus stramineum***	Grass	perennial	highly desirable	735	1854
17	***Cerastium fontanum***	forb	annual	Undesirable	3	0
18	***Commelina spp***	forb	perennial	Moderately desirable	2	164
19	***Cucumis sativus***	forb	annual	Undesirable	0	1
20	***Cynodon dactylon***	Grass	perennial	Moderately desirable	81	125
21	***Cyperus rotundus***	Sedge	perennial	less desirable	12	2
22	***Dactyloctenum aegyptium***	Grass	perennial	Moderately desirable	0	42
23	***Desmodium triflorum***	forb	annual	highly desirable	4	495
24	***Digitaria ciliaris***	Grass	perennial	Moderately desirable	0	16
25	***Dipcadi montanum***	forb	perennial	Undesirable	5	0
26	***Dyschoriste spp***	forb	perennial	Undesirable	346	181

S/n	*spp* name	Growth form	Life span	Desirability	Frequency of occurrence	
					Encroached sites	Non-encroached sites
27	***Ecballium elaterium***	forb	perennial	Undesirable	2	0
28	***Elephantopus mollis***	forb	perennial	Undesirable	22	0
29	***Eragrostis minor***	Grass	annual	Moderately desirable	15	31
30	***Eragrostis superba***	Grass	perennial	Moderately desirable	0	15
31	***Frankenia pulverulenta***	forb	annual	undesirable	2	0
32	***Galeopsis angustifolia***	forb	annual	undesirable	3	133
33	***Gnaphalium spp***	forb	annual	undesirable	0	1
34	***Gomphrena celesiodes***	forb	perennial	undesirable	3	0
35	***Gomphrena globosa***	forb	annual	less desirable	0	31
36	***Grewia kakothamnos***	shrub	perennial	less desirable	2	0
37	***Heliotropium indicum***	forb	annual	undesirable	0	4
38	***Hypericum humifusum***	forb	annual	undesirable	0	2
39	***Ipomoea mombassana***	forb	perennial	undesirable	5	12
40	***Justicia calyculata***	forb	perennial	undesirable	4	68
41	***Lantana camara***	shrub	perennial	undesirable	5	0
42	***Launaea cornuta***	forb	perennial	Moderately desirable	0	2
43	***Metha longifolia***	forb	perennial	undesirable	0	32
44	***Macroptilium atropurpureum***	forb	perennial	Highly desirable	0	2
45	***Misopates orontium***	forb	annual	undesirable	2	0
46	***Mollugo nudicaulis***	forb	annual	undesirable	35	204
47	***Mollugo verticillata***	forb	annual	undesirable	173	792
48	***Myosotis verna***	forb	annual	undesirable	0	3
49	***Neonotonia wightii***	forb	perennial	highly desirable	0	17
50	***Ocimum spp***	forb	annual	undesirable	0	8
51	***Panicum infestum***	Grass	perennial	highly desirable	0	392
52	***Satureja hortensis***	forb	perennial	undesirable	3	2
53	***Senna obsitufolia***	Shrub	perennial	undesirable	92	14
54	***Sida spp***	forb	perennial	Less desirable	46	57
55	***Solanum Incanum***	shrub	perennial	undesirable	34	40

S/n	*spp* name	Growth form	Life span	Desirability	Frequency of occurrence	
					Encroached sites	Non-encroached sites
56	***Solanum nigrum***	shrub	perennial	undesirable	2	0
57	***Sphagneticola trillobata***	forb	perennial	undesirable	0	4
58	***Tapinanthus belvisii***	forb	perennial	undesirable	2	0
59	***Tephrosia purpurea***	forb	perennial	undesirable	187	0
60	***Tribulus terrestris***	forb	annual	undesirable	76	1156
61	***Trifolium repens***	forb	annual	Moderately desirable	23	4
62	***Urochloa mosambicensis***	Grass	perennial	highly desirable	1252	3755
63	***Zinnia haageana***	forb	annual	undesirable	4	0
64	***Zornia glochidiata***	forb	annual	less desirable	408	0