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Soil characteristics within vegetation patches are sensitive indicators of savanna rangeland degradation in central Namibia

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ABSTRACT

Savanna rangelands are threatened by increased grazing pressure, with the risk of desertification and woody plant encroachment. In this context, sensitive early-warning indicators for irreversible shifts toward degraded rangelands are urgently needed. We hypothesize that soil properties, in particular soil organic carbon (SOC) and carbon isotopes (δ^{13}C) are suitable to detect degradation in different land management systems. We conducted a natural experiment on arid savanna rangelands on Arenosols in Namibia. As treatments, we considered two land management systems (freehold farms with rotational grazing and systematic bush thinning, and communal rangelands with free grazing without systematic bush thinning), with different grazing intensities (grazing gradients), and four vegetation patch types (dominance of bare soil, annual plants, perennial grasses, or woody plants) with four replications per land management system. Clay was considered a system-inherent confounder. SOC stocks were lower on communal rangelands (8.2 ± 2.8 Mg ha^{-1}) than on freehold farms (9.7 ± 2.6 Mg ha^{-1}). Besides clay content, SOC stocks correlated with perennial grass cover ($r = 0.6$) in freehold farms, and with woody plant cover ($r = 0.54$) in communal rangelands. δ^{13}C values were higher by 2.15 ‰ on freehold farms, indicating greater origin of C_4 grass derived SOC. Distance to the nearest water point – as an estimate of local grazing intensity – had a small effect on SOC stocks compared to clay. SOC was a sensitive degradation indicator in savanna rangelands when considered together with the patchy character of savanna vegetation. Under current land management and tenure, communal rangelands faced higher degradation threats than freehold farms, but were also more vulnerable due to lower clay contents.

1. Introduction

Savannas cover about 15% of the world's land surface, with half of their area used as rangelands for livestock production (Asner et al., 2004). Consequently, savannas play a key role in livestock forage production and carbon storage (Millennium Ecosystem Assessment, 2005; Ahlström et al., 2015). However, intensifying droughts and increasing grazing pressure endanger savanna rangelands, which can lead to long-term degradation of the ecosystems and carbon loss (D'Odorico et al., 2012; Hoover et al., 2020). The subsequent cutbacks on essential ecosystem services are tremendous (Rietkerk et al., 2021), affecting many people's livelihoods directly through livestock production and indirectly through the global carbon cycle.

Degradation of arid rangelands manifests itself in soil and vegetation changes (van der Westhuizen et al., 2022), which can lead to desertification (Millennium Ecosystem Assessment, 2005; Dakos et al., 2019) or woody plant encroachment (Peters et al., 2009; D'Odorico et al., 2012; Dakos et al., 2019). Both phenomena are associated with a loss of perennial grasses (D'Odorico et al., 2012) and altered soil nutrient distributions (Schlesinger et al., 1996). Low soil fertility can impede the productivity and recovery of the ecosystem and maintain the degraded state (e.g. D'Odorico et al., 2012; Hoover et al., 2020). However, finding leading soil-based indicators of degradation is challenging (Kosmas et al., 2014), because soil properties vary over different timescales; while some soil properties, such as texture, remain relatively fixed (Karssenberg et al., 2017), others, such as nutrient contents and

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stoichiometry, are more likely to indicate qualitative changes (Froese et al., 2023). Soil degradation includes long-term loss of soil productivity caused by physical, chemical, and biological processes.

In arid rangeland ecosystems, physical soil degradation can be reflected in higher bulk densities, mainly caused by livestock trampling, and soil erosion, depending on topography, texture, vegetation, and rainfall (Sandhage-Hofmann, 2022). Chemical soil degradation can be reflected in changes of soil organic matter (SOM) and nutrient concentrations. Under high grazing pressure, inputs from livestock urine, dung and woody plant encroachment enrich SOM and can counteract direct losses of SOM from erosion and indirect losses mediated by a reduced herbaceous vegetation cover (Sandhage-Hofmann, 2022). The parameter most often considered in degradation studies is soil organic carbon (SOC, Byrnes et al., 2018; de Moraes Sá et al., 2018; Lai and Kumar, 2020). SOC is positively associated with critical soil functions in rangeland production, including nutrient and pH buffering, water retention, and aggregate stabilization (see Sandhage-Hofmann, 2022 and references therein). Therefore, SOC and the carbon-to-nitrogen (CN) ratio are well-established indicators for assessing changes in rangeland productivity in savannas (Eldridge et al., 2017; Byrnes et al., 2018).

Rangeland degradation goes along with changes in vegetation composition (Guuroh et al., 2018) and a shift in the ratio of perennial grasses to woody plants (e.g., Gonnet et al., 2003; Di Virgilio et al., 2019; van der Westhuizen et al., 2022). Soil carbon isotopes can help to track these changes (Boutton et al., 1998). In subtropical ecosystems, perennial grasses typically undergo C_4 photosynthesis, resulting in a less negative $\delta^{13}C$ value of about −13‰ compared to woody plants with C_3 photosynthesis, and have a $\delta^{13}C$ value of about −26‰ (Boutton et al., 1998; Sandhage-Hofmann et al., 2020).

Although vegetation is an integrative indicator of overall soil quality (Kotzé et al., 2020), our understanding of how arid rangeland soils respond to environmental stress is still limited. Herbaceous vegetation in semi-arid savannas is highly variable between successive seasons, depending on the total amount and distribution of precipitation (van Oudtshoorn, 2016). Many soil properties are less dynamic than vegetation properties (Karssenberg et al., 2017), in which case it can help distinguish between fast and slow degradation processes in savanna systems and reveal potentially irreversible changes in the rangeland system (Bestelmeyer et al., 2006). Thus, combining vegetation and soil data can provide a more comprehensive view of changes in ecological states (Tongway and Hindley, 2004; Bestelmeyer et al., 2006). Previous studies with this approach often distinguish the patchy character of savannas between inter- and subcanopies (Linstädter et al., 2016; López-Sánchez et al., 2016), or within patches of intercanopy habitats only (Maestre et al., 2009). However, holistic approaches that combine soil properties with vegetation characteristics of sub- and intercanopy habitats and within intercanopy habitats in rangelands are still rare (Ochoa-Hueso et al., 2017; Maestre et al., 2022).

In scientific literature, savanna degradation is increasingly discussed in the context of tipping point phenomena (e.g., Dakos et al., 2019; Berdugo et al., 2020; Hoover et al., 2020; Roberts et al., 2021). For social-ecological systems like savanna rangelands, a tipping point, also referred to as regime shift, critical transition or alternative stable states, "is a threshold at which small quantitative changes in the system trigger a non-linear change process that is driven by system-internal feedback mechanisms [leading] to a qualitatively different state of the system, which is often irreversible" (Milkoreit et al., 2018). Thus, a tipping point is based on four principles: multiple stable states, non-linear change, feedbacks as driving mechanisms, and limited reversibility (Milkoreit et al., 2018). Although tipping points are rather loosely defined in the scientific literature (van Nes et al., 2016), there is broad consensus that tipping point transitions mathematically correspond to saddle-node or fold bifurcation points (Dakos et al., 2019).

Assessing the state of heavily used rangelands provides a possibility to qualitatively assess the status quo of rangelands (Pringle and Landsberg, 2004). The spatial distribution of grazing intensity, which is largely determined by permanent water provision for livestock, serves as a key driver of rangeland degradation (Pringle and Landsberg, 2004). Water point gradient approaches can explain site-specific soil and vegetation patterns in savannas (Hess et al., 2020). Therefore, the distance to water points is a popular proxy to capture the extent to which grazing intensity affects rangeland conditions (Kotzé et al., 2013; Sandhage-Hofmann et al., 2015). Furthermore, spatial gradients can be used as a space-for-time substitution, as they can reflect temporal dynamics, such as crossing an ecological threshold (Pickett, 1989; Scheffer and Carpenter, 2003; Kreyling et al., 2014).

Many studies have shown that rangeland management affects soils and vegetation. Rangeland management includes grazing systems, like rotational and free grazing (Kotzé et al., 2013; Linstädter et al., 2014; Sandhage-Hofmann et al., 2015; Di Virgilio et al., 2019), and factors such as controlling stocking rates and the managing woody vegetation to control the rangeland's carrying capacity. Comparing rotational and free grazing, it is not possible to conclude which management system is generally more sustainable (Sandhage-Hofmann, 2022). Observations vary with climate and other environmental factors (Di Virgilio et al., 2019; Chabala et al., 2022; Maestre et al., 2022), and management options are often limited to specific land tenure systems (Sayre et al., 2017). Furthermore, it remains unclear when degradation of rangeland management systems is equivalent to crossing a threshold or tipping point (Jamsranjav et al., 2018).

Accordingly, the main objective of this study was to identify early-warning indicators and potential thresholds of rangeland degradation in arid savannas, based on soil characteristics. This was done by analyzing soil properties along local grazing gradients in two tenure systems with different land management practices on Arenosols developed from Kalahari sands. Our hypotheses were that (i) SOC and $\delta^{13}C$ are suitable and sensitive indicators of savanna rangeland degradation, as they correlate with land management and grazing intensity. Because savanna rangelands are characterized by large spatial heterogeneity of soil and vegetation characteristics, we additionally assume that (ii) the response of SOC and $\delta^{13}C$ to degradation processes is also influenced by vegetation patch types and vegetation cover. Finally, we expect that (iii) SOC and $\delta^{13}C$ show non-linear behavior along grazing gradients, indicating a threshold behavior. To test our hypotheses, we collected soil samples (0–10, 10–20 cm, supplemented by soil cores up to 1 m) and assessed vegetation cover along grazing gradients in communal rangelands with free grazing and no bush thinning and freehold farms with rotational grazing and systematic bush thinning in the Greater Waterberg Landscape of Namibia. To account for the patchy character of savanna ecosystems, soil samples were collected in subcanopy and intercanopy habitats in a paired sampling design. Within intercanopy habitats, we additionally stratified our sampling into the vegetation patch types of bare soil and perennial grasses, where available. We analyzed basic soil properties with a focus on SOC, and CN ratios. In addition, we analyzed $\delta^{13}C$ isotopes as indicator for the origin of SOC and to detect long-term vegetation changes.

2. Material and methods

2.1. Study area

Namibia is the driest country in sub-Saharan Africa (Atlas of Namibia Team, 2022) and local people's livelihoods are primarily based on livestock farming (Republic of Namibia, 2014). The study was conducted on rangelands in the southeast of the Waterberg Plateau in the Otjozondjupa region of Namibia on the fringe of the Kalahari (latitude 20°33′ to 21°13′ S, longitude 17°13′ to 18°39′ E, elevation of 1264 to 1478 m, Fig. 1). The climate of the study area is arid (UNEP aridity index 0.14–0.16) with an average annual temperature of 20 to 21 °C, and an average annual precipitation of 350 to 400 mm. Rainfall is erratic and seasonal from November to April. Average annual evaporation ranges from 2400 to 2500 mm, resulting in a water deficit of up to 1700 mm.

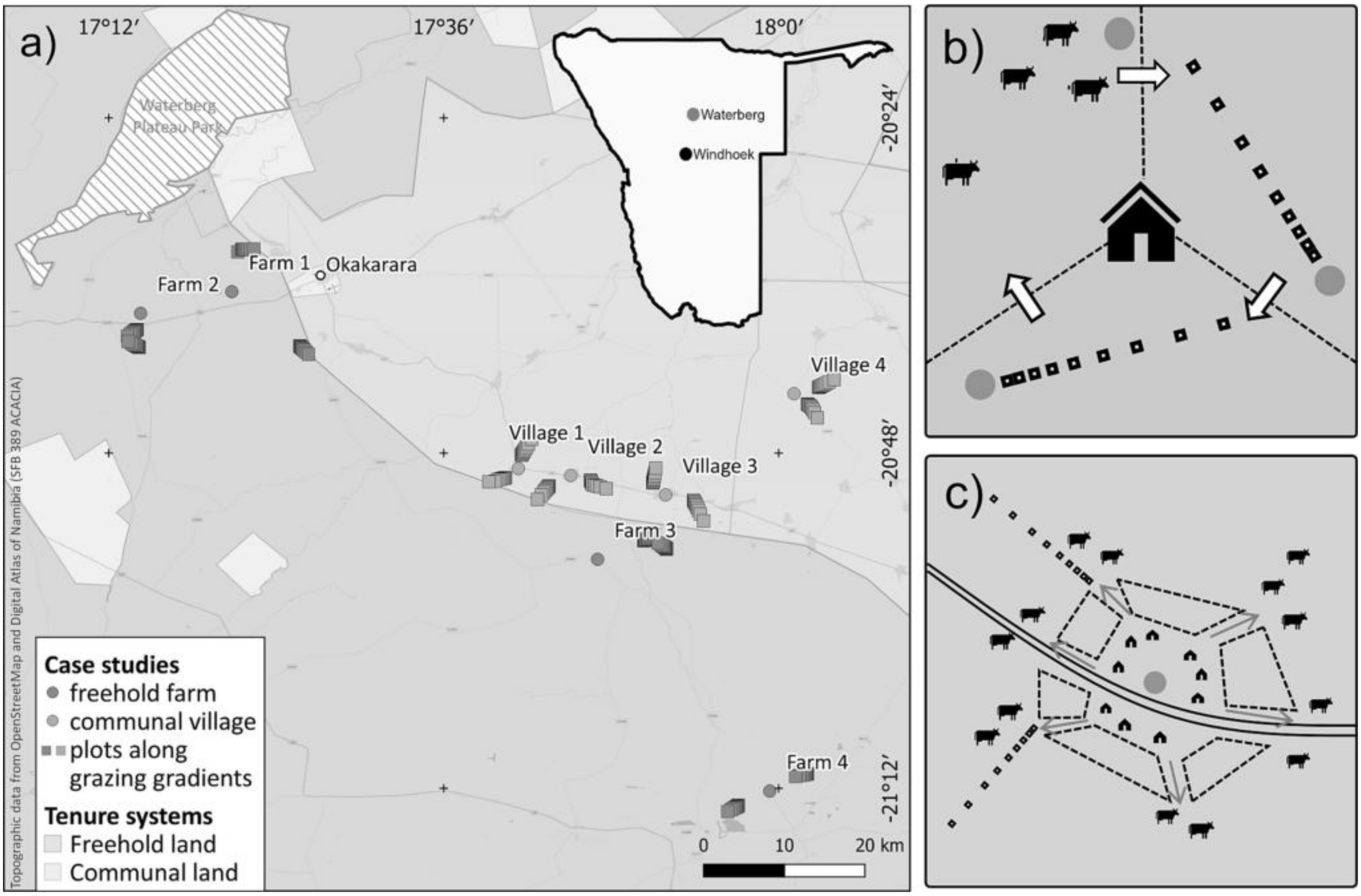

Fig. 1. (a) Location of the study sites southeast of the Greater Waterberg Plateau, Namibia, and schematic representation of land management strategies (b) for freehold farms with rotational grazing and bush thinning, and (c) for communal rangelands with free grazing and no bush thinning. Fences are indicated by dashed lines, cattle trails between fenced areas or along traffic roads are indicated by red arrows, the nine squares extending from the water points (blue) represent the plots spaced logarithmically along grazing gradients within the respective land tenure system.

The soils were classified as Arenosols (IUSS Working Group WRB, 2022; Fig. S1) derived from Kalahari sands and scattered calcareous pans (Atlas of Namibia Team, 2022).

The study area is located in the Acacia Tree-and-Shrub Savanna biome, where the Central Kalahari transitions into the Northern Kalahari and Thornbush Shrubland. Here the vegetation consists mainly of more or less open shrubland (Atlas of Namibia Team, 2022). The tree layer is dominated by *Senegalia mellifera* (black-thorn acacia) and *Terminalia sericea* (silver terminalia), while the grass layer is characterized by perennial C_4 grasses such as *Eragrostis rigidior* (curly-leaved love-grass), *Eragrostis trichophora* (hairy love-grass), *Aristida congesta* (tassel three-awn), and *Stipagrostis uniplumis* (silky Bushman-grass). From individual farmer interviews (Brinkmann et al., 2021) and the interpretation of satellite imagery comparing >30 years, it appears that the Otjozondjupa region has become heavily woody plant encroached, mainly by *Senegalia mellifera* and *Terminalia sericea*. Inappropriate rangeland management practices that have led to overgrazing, suppression of bush fires, and increased atmospheric CO_2 levels are considered to be the main causes of this phenomenon (Atlas of Namibia Team, 2022).

2.2. Tenure-based rangeland management practices

Half of Namibia's area is privately owned, mostly in large freehold farms, while about one-third of the country is communal land (as of 2020, Atlas of Namibia Team, 2022). In the Greater Waterberg Landscape, freehold farms verge on communal land. Prior colonialism, the Greater Waterberg Landscape was pastoral territory of the Ovaherero tribe with grazing systems similar to extensive modern rotational grazing systems (Menestrey Schwieger and Mbidzo, 2020). Land allocations and hence grazing systems changed during the German and South African colonial interventions from 1911 onwards (Menestrey Schwieger and Mbidzo, 2020). In 1937, communal land was allocated as it is today, while freehold land was not allocated as it is today until 1955. Today, these two tenure systems greatly differ in terms of their land management practices. Freehold farms apply rotational grazing with rangelands structured in fenced camps. Each camp has its own water point or has access to one through short fenced passages. The camps are grazed for about five days (information provided by the farmers), followed by a resting period of several weeks to allow the herbaceous vegetation and perennial grasses to recover and reestablish from grazing impacts. Additionally, most freehold farms do cost-intensive large-scale bush thinning. The earliest systematic bush thinning was presumably done around 1980 within the freehold sites (personal communication by the farmers). Bush thinning, i.e. the removal of woody plants by aerial application of chemicals, or manual cutting or stem burning, counteracts woody plant encroachment (Nghikembua et al., 2023).

The rangelands of the communal villages are common property, where free grazing is practiced and livestock graze around and away from the villages. For a long time, it was not easily possible to fence single camps. However, it has become common practice that private individuals fence off immediate areas surrounding their homes on the edges of villages to secure grazing land and avoid cattle theft (Menestrey Schwieger and Mbidzo, 2020). Central water points for cattle are located close to the village centers (Fig. 1b), which leads to rather free grazing regimes around the villages, but only after the fenced-off camps. In part, communal farmers herd their cattle to seasonal cattle posts when rainfall is insufficient (Menestrey Schwieger and Mbidzo, 2020). On communal land, property rights and the related laws (see 2.1) restrict bush thinning. Thus, communal farmers have fewer legal options to efficiently intervene in the competition between woody and herbaceous plants for water, light and space.

The overall carrying capacity of these rangelands is low, at about 18 ha per livestock unit for the region (Stehn, 2011). In addition, the actual carrying capacity is dynamic between successive seasons and varies with annual precipitation and its distribution (Atlas of Namibia Team, 2022). On freehold farms, stocking rates and grazing-resting phases can be adapted to annual conditions, whereas on communal rangelands, overgrazing is common (Kotzé et al., 2013) due to limited flexibility (Menestrey Schwieger and Mbidzo, 2020) and already advanced rangeland degradation (Menestrey Schwieger, 2022). Freehold farms practicing rotational grazing and systematic bush thinning have been shown to produce more grasses than communal rangelands with free grazing (Sandhage-Hofmann et al., 2020); This was also visible in our study area, where freehold farms with rotational grazing had more herbaceous vegetation cover than communal farms (Table S1).

2.3. Study design

We conducted a natural experiment with land management, grazing intensity, and vegetation patch type as treatments, clay as a covariate and four replications per land management system, namely four freehold farms, and four communal villages (Fig. 1a). At each site, we established two transects ($n = 16$) with nine plots (10×10 m^2) along grazing gradients leading away from water points (plot 1) to a location at the maximum distance from this and, in the case of communal rangelands, any other water point (plot 9). Plots were arranged in a logarithmic order, starting at the water points where plots were placed closer together to account for the higher grazing pressure close to water points (Manthey and Peper, 2010). Transect lengths varied between 1.2 km and 1.5 km on freehold farms, according to camp size, and between 2 km to 3 km in communal villages, according to the proximity of neighboring villages and water points. In both tenure systems, the immediate vegetation-free area around the water points – the so-called sacrifice zones (Sandhage-Hofmann et al., 2015) – was excluded (Fig. 3). Since the villages were surrounded by fenced camps, open rangelands only commenced thereafter, mostly connected to the water points by narrow, highly trampled pathways or traffic roads. Consequently, the sacrifice zone was extended to a radius of 1.8 to 4.0 km from the water point (Fig. 1b), and communal transects started thereafter.

Fieldwork was conducted during the rainy season from February to May of 2021, preceded by a prolonged drought (Shikangalah, 2020). Ground cover data in the plots, which were used to upscale and explain our soil properties, were visually estimated as described in Amputu et al. (2023), where, we recorded the cover of bare soil, plant litter, herbaceous plants, and woody plants at the plot level. To calculate the mean perennial grass cover per plot, we recorded all vascular plant species and visually estimated the percent ground cover of each species within two quadrats (1 m^2) within the intercanopy habitat (Fig. 3) of each plot.

Ground cover values differed greatly between freehold and communal rangelands (Fig. 2). On average, herbaceous vegetation cover was 12% higher on freehold farms, with perennial grasses accounting for a 9% difference. Woody plant cover ranged from 0 to 30% on freehold farms, except for eight plots on one farm that had never practiced large-scale bush thinning, resulting in a woody cover of up to 60%. Woody plant cover on communal rangelands ranged from 0.5 to 75% and was on average 7.5% higher than on freehold rangelands (Table S1).

We stratified our soil sampling into intercanopy and subcanopy habitats (Fig. 3) to account for these two cardinal habitat types in savanna ecosystems (Sandhage-Hofmann et al., 2021). In this study, intercanopy habitats include the entire mosaic of bare soil patches, patches with plant litter, and herbaceous plants accessible to livestock grazing. In contrast, subcanopy habitats comprise woody plants; soil samples were collected only under shrubs and trees inaccessible to livestock grazing, to avoid capturing livestock shading hotspots under trees. In each plot, we collected two soil samples (0–10 cm; 10–20 cm) in the herbaceous vegetation of intercanopy habitats ($n = 243$). One of the intercanopy samples per plot was paired with an additional soil sample in an adjacent subcanopy habitat ($n = 143$). To better capture the patchy character of the savanna ecosystem, the intercanopy soil samples were on some plots supplemented with samples of bare soil patches that were clearly distinguishable from the matrix intercanopy vegetation ($n = 23$), and with samples of perennial grass patches that were in at least their second year of growth, if present ($n = 114$, Fig. 3). Bulk density (BD, 0–10 cm; 10–20 cm) was determined at the levels of intercanopy and subcanopy habitats. In addition, we collected soil cores up to 1 m in an intercanopy habitat on the first plot of each transect ($n = 16$, Table S2).

2.4. Soil analyses

All soil samples were air-dried and sieved (<2 mm). Subsequent analyses were performed on the corresponding samples listed in Table S2. Soil texture analysis was carried out via the combined sieve and pipette method (ISO 11277, 2002). Bulk density was determined by gravimetric analysis of three 0.1 dm^3 steel cylinder samplers. The maximum water holding capacity (WHC_{max}), which is critical in arid environments for the storage of water (Francis, 2019), was determined

Fig. 2. Exemplary vegetation cover on freehold farms (left) with a characteristic open savanna (bush thinned) and a dense cover of perennial grasses, and communal rangelands (right) with more woody plant encroachment and less grass cover.

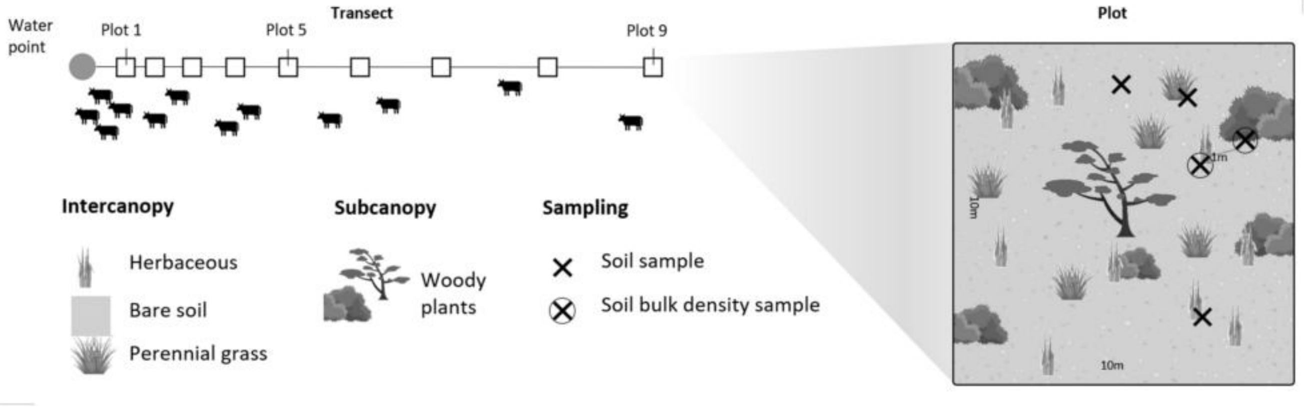

Fig. 3. Schematic study design of transects along grazing gradients with nine observation plots of 100 m^2, and schematic sampling design within observation plots with two samples in intercanopy habitats, of which one was paired with a sample in a subcanopy habitat, and, if present, selective samples within patches of bare soil and perennial grasses.

according to Forster (1995). The pH value was measured in a 1:2.5 soil-water suspension.

Total carbon and nitrogen (N) concentrations were determined in duplicate by dry combustion and elemental analysis (ISO 10694, 1995; vario EL cube; Elementar, Hanau, Germany). Since inorganic carbon was absent, total carbon was equated to organic carbon, hereafter referred to as soil organic carbon (SOC). SOC stocks (C_t in Mg ha^{-1}) were calculated proportionally to the area of the intercanopy habitats (A_i in %) and the area of the subcanopy habitats (A_s in %) of each plot as follows:

$$C_t = A_i(SOC_{Di} \times BD_{Di} \times D) + A_s(SOC_{Ds} \times BD_{Ds} \times D) \quad (1)$$

where *SOC* is carbon concentration (g C 100 g^{-1} soil), *BD* is soil bulk density (g cm^3), *D* is soil sampling depth (cm), A is the area and *i* and *s* are intercanopy and subcanopy habitats, respectively.

The $\delta^{13}C$ of the samples used to measure SOC was analyzed using an Isotope Ratio Mass Spectroscope (Delta V Advantage IRMS, Thermo Electron Corporation, Germany) according to Eq. (2):

$$\delta^{13}C = \frac{R_{sample} - R_{standard}}{R_{standard}} \times 100 \quad (2)$$

where R_{sample} is the $^{13}C/^{12}C$ isotope ratio of the sample and R_{standard} is the $^{13}C/^{12}C$ isotope ratio in respect to the V-PDB standard for carbon isotopes (Coplen et al., 2006).

2.5. Data analyses

Statistical analyses were conducted with R version 4.2.2 (R Core Team, 2022).

First, we used one-way analysis of variance (ANOVA) to test for confounding systematic differences between treatment levels of land management, vegetation patch type, and grazing intensity, with clay as the response variable, which was considered system inherent. For ANOVA we used the lme4 package (Bates et al., 2015) based on a full random slopes model (Arnqvist, 2020), to account for unbalanced data, blocking, and pseudo-replication within the hierarchical sampling structure. We considered random intercepts for different sites, and transects nested within them, and random slopes for the tested factors, vegetation patch type or grazing intensity, there included as plot number along the transects. Significance of factors was assessed using Type II sums of squares with the Kenward-Roger's degrees of freedom method. The residuals for each variable were checked graphically for homoscedasticity and normality by inspecting QQ plots.

Clay and land management were not independent (Table 1), and therefore both could not be included in a model to assess the effects of land management while controlling for clay. We then summarized soil properties (refer to Table S2 for sample sizes) in strata of land management, and vegetation patch type either as means and standard deviations ($\bar{x} \pm s$) or as box plots, and analyzed them descriptively.

Table 1
Results of one-way ANOVA for respective dependencies between clay and the factors land management, vegetation patch type, and grazing intensity in topsoil (0-10 cm).

Effect	Numerator degrees of freedom	Denominator degrees of freedom	F	P
Land management	1	6	6.01	<0.05
Veg. patch type	1	7	0.18	0.69
Grazing intensity	1	7	0.00	0.96

To evaluate the presence and magnitude of thresholds along the grazing gradients, we compared the Akaike information criterion (AIC) of linear, quadratic, cubic, and exponential regression models (Swift and Hannon, 2010). Response variable was SOC stocks in topsoil (0–20 cm), and predictor variables were length of grazing gradients (as proxy for the grazing impact) and clay content. Clay and grazing intensity were standardized. We predicted SOC would decline exponentially from water points (Manthey and Peper, 2010), and be positively correlated with clay content. We included random effects, where we considered each transect nested within site as random intercept and included clay as random slope as there was evidence at $p < 0.05$ that the effect of clay differs across transects nested within sites. To visualize outcomes of mixed effects models, we used the sjplot package (Lüdecke, 2023) and the MuMIn package (Bartoń, 2023). Next, following our hypothesis, we used quadratic, cubic, and exponential models to determine whether the SOC stocks responded in a non-linear manner to grazing pressure – while controlling for clay.

Pearson correlation coefficients were computed respectively for freehold farms and communal rangelands between upscaled SOC stocks, clay content, WHC_{max}, $\delta^{13}C$, grazing intensity, and ground cover stratified by bare soil, herbaceous vegetation, perennial grass cover thereof, and cover of woody plants with a subset of data according to the WHC_{max} ($n = 80$, Table S2).

3. Results

3.1. Dependency of treatment factors

As part of the natural experiment, we tested for dependencies between the considered treatment factors with system-inherent properties, here clay. While the factors vegetation patch type and grazing intensity were independent, land management was confounded by clay (F(1, 6) = 6.01, $p < 0.05$; Table 1).

3.2. Physico-chemical soil properties

The Arenosols contained between 78.4 and 96.3% of sand. The communal areas had higher sand contents and lower silt and clay contents, which summed up to an average of 5% lower values than those on freehold farms (Table 2). In both tenure systems, clay was slightly higher in 10 to 20 cm. Texture showed no differences between inter- and subcanopy habitats, consequently we did not look further for signs of erosion. The textural differences between tenure systems corresponded with an overall higher WHC$_{max}$ of 3.8% in freehold farms. Here, also differences between vegetation cover became visible, where the WHC$_{max}$ was respectively greater in subcanopy habitats than the adjacent intercanopy habitats, with a difference of 3.8% in communal rangelands (Table 2). Bulk density increased with soil depth and was higher in intercanopy habitats (1.69 g cm^{-3}) than in subcanopy habitats (1.49 g cm^{-3}, Table 2), with no difference between tenure systems. The pH value did not differ notably between tenure systems or between vegetation habitats (Table 2).

3.3. SOC, CN, and total N

The soil core data from 1 m soil depth showed that SOC concentrations and CN ratios gradually decreased with increasing soil depth. The gradual decrease of SOC concentrations and CN ratios was more pronounced in freehold farms than in communal rangelands. In the topsoil (0–10 cm), SOC concentrations were about twice as high in freehold farms (5.82 ± 2.40 g C kg^{-1} soil) than in communal rangelands (2.57 ± 0.69 g C kg^{-1} soil), but differences between tenure systems diminished below 30 cm, where mean values ranged from 2.41 ± 0.41 to 1.43 ± 0.18 g C kg^{-1} soil (SOC stock results follow). CN ratios were also higher in the topsoil (0-10 cm) of freehold farms (10.5 ± 1.4, $\bar{x} \pm s$) than communal rangeland (9.2 ± 1.0, $\bar{x} \pm s$). Below 10 cm, CN ratios were rather comparable between the tenure systems, ranging from a maximum of 11.5 in the topsoil to a minimum of 6.1 at 100 cm depth (Fig. 4). Overall, differences in carbon concentrations and CN ratios between the tenure systems were most pronounced in the topsoil (20 cm).

When we distinguished the SOC concentrations of our data set between the four vegetation patch types, we found that SOC concentrations in the topsoil (0-10 cm) were higher with longer-lived vegetation patch types. Specifically, the woody plants had the highest concentration, followed by perennial grasses, herbaceous vegetation, and then bare soil. The SOC concentrations tended to be lower in intercanopy habitats of communal rangelands compared to freehold farms. However, these differences were not observed in subcanopy habitats. Similar to SOC, the CN ratio increased as the plant-life form became more perennial. The differences in CN ratio were more pronounced in communal rangelands compared to freehold farms (Fig. 5). For total N, the differences between tenure systems and vegetation cover corresponded to those we found for SOC concentrations.

3.4. Origin of carbon – δ^{13}C isotopes

δ^{13}C isotopes from intercanopy soil cores of 1 m increased with soil depth and had a wider range from −26.1 to −16.4‰ in communal rangelands compared to freehold farms where the range was −20.8 to −15.0‰ (Fig. 4). The differences between the two tenure systems were most pronounced in the upper 30 cm, where δ^{13}C isotopes were less negative in freehold farms than in communal rangelands (Fig. 4).

A similar trend was observed when examining the topsoil (0-10 cm) within specific patch types. In freehold farms, the δ^{13}C value correlated with perennial grass cover ($r = 0.4$, Fig. 8), unlike communal rangelands. On average, each δ^{13}C value under a given vegetation patch type had lower values in communal rangelands than in freehold farms (Fig. 5). According to the C$_3$-photosynthetic cycle of woody vegetation, δ^{13}C isotopes were lower in subcanopy habitats than in the intercanopy habitats in both tenure systems. Contrary to this, the highest ratio of δ^{13}C isotopes (−14.8‰) was found under perennial grasses in freehold farms, again matching their C$_4$-photosynthetic pathway (Fig. 5).

3.5. SOC stocks in topsoil (0–20 cm) according to vegetation cover on plot level

In freehold farms, mean SOC carbon stocks (Fig. 6, 'total') were similar in inter-, and subcanopy habitats, amounting to 9.7 ± 2.6 Mg ha^{-1} upscaled proportionally to the vegetation cover. In communal rangelands, however, SOC stocks were lower in intercanopy than in subcanopy habitats, resulting in overall lower SOC stocks when upscaled (8.2 ± 2.8 Mg ha$^{-1,}$ Fig. 6).

When broken down to the SOC stocks for each habitat type, SOC stocks were comparably highest in subcanopy habitats of freehold farms (10.5 ± 2.8 Mg ha^{-1}) and communal rangelands (10.7 ± 4.6 Mg ha^{-1}). Intercanopy habitats of freehold farms were slightly lower (9.5 ± 2.7 Mg ha^{-1}). However, intercanopy habitats of communal rangelands showed the lowest values with 7.5 ± 2.7 Mg C ha^{-1} (Fig. 6).

3.6. SOC stocks along grazing gradients

When we examined the grazing gradients, there were no distinct patterns in topsoil SOC stocks (0–20 cm, Fig. 7) in both tenure systems, only a small decrease in SOC stocks along the grazing gradients could be observed. However, the scatterplot across all plots again shows that freehold rangelands had higher SOC stocks than communal rangelands, mainly due to clay according to the steeper slope of the regression pane (Fig. 7). We also observed no trends for the other below ground parameters we assessed, which were SOC concentrations, total N, CN ratio and δ^{13}C isotopes in the intercanopy habitats (Table S3). It is worth mentioning that plot 1 (nearest to the water point) in the freehold farms had higher SOC and total N concentrations than the other plots in the topsoil (0–10 cm, Table S3).

To explain SOC stocks in response to grazing intensity, and to detect potential non-linear changes along grazing gradients that could indicate

Table 2
Physico-chemical soil properties ($\bar{x} \pm s$) in relation to tenure system, habitat type (with int = intercanopy and sub = subcanopy) and soil depth (for n see Table S2).

Depth [cm]	Tenure	Habitat	Bulk density [g cm^{-3}]	Sand [%]	Silt [%]	Clay [%]	WHC$_{max}$ [%]	pH [H$_2$O]
0–10	Freehold	int	1.67 ± 0.05	86.7 ± 4.3	4.7 ± 1.5	7.0 ± 2.3	28.7 ± 3.5	6.1 ± 0.4
		sub	1.49 ± 0.09	87.5 ± 4.8	4.1 ± 1.7	7.6 ± 3.2	30.8 ± 3.2	6.2 ± 0.6
	Communal	int	1.69 ± 0.04	91.1 ± 2.6	2.9 ± 0.9	4.6 ± 1.3	24.9 ± 2.1	6.2 ± 0.5
		sub	1.50 ± 0.07	91.9 ± 2.1	2.7 ± 0.6	4.4 ± 1.1	28.6 ± 3.9	6.3 ± 0.6
10–20	Freehold	int	1.68 ± 0.05	87.5 ± 3.7	4.1 ± 1.3	8.2 ± 2.1		
		sub	1.59 ± 0.06					
	Communal	int	1.68 ± 0.04	91.1 ± 2.7	2.9 ± 0.8	5.9 ± 1.3		
		sub	1.57 ± 0.05					

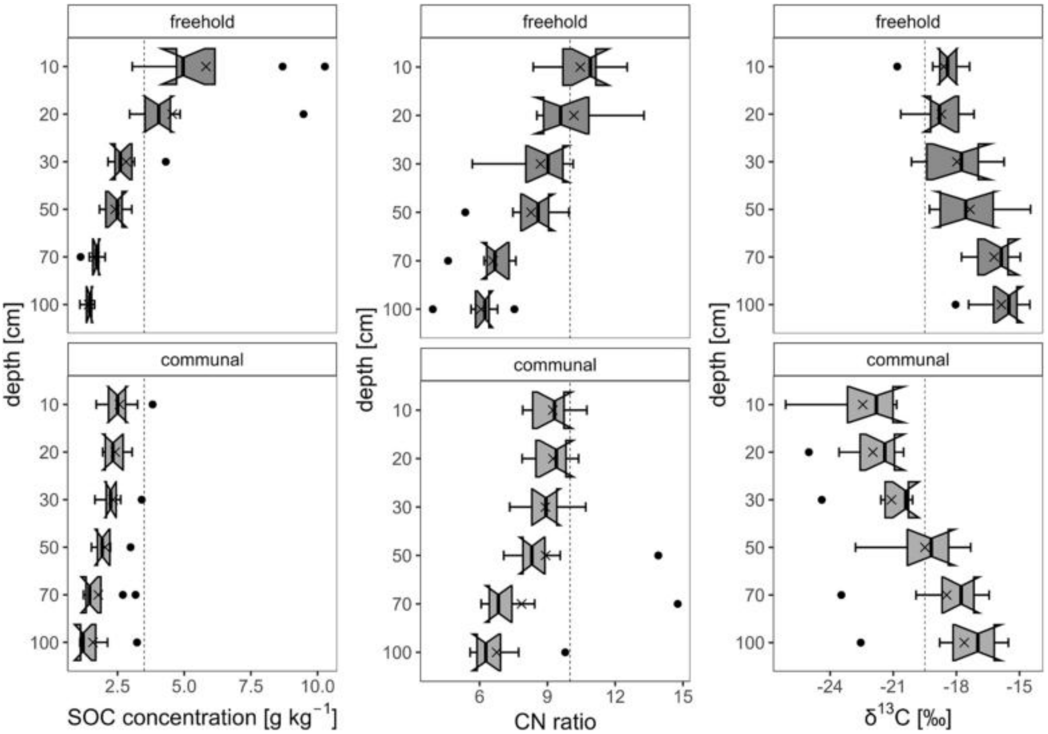

Fig. 4. SOC concentrations, CN ratios, and δ^{13}C isotopes in respect to tenure system in 1 m soil depth of intercanopy habitats ($n = 16$). The median is shown as a bold line, the mean as a cross and the 95% confidence interval as a notch.

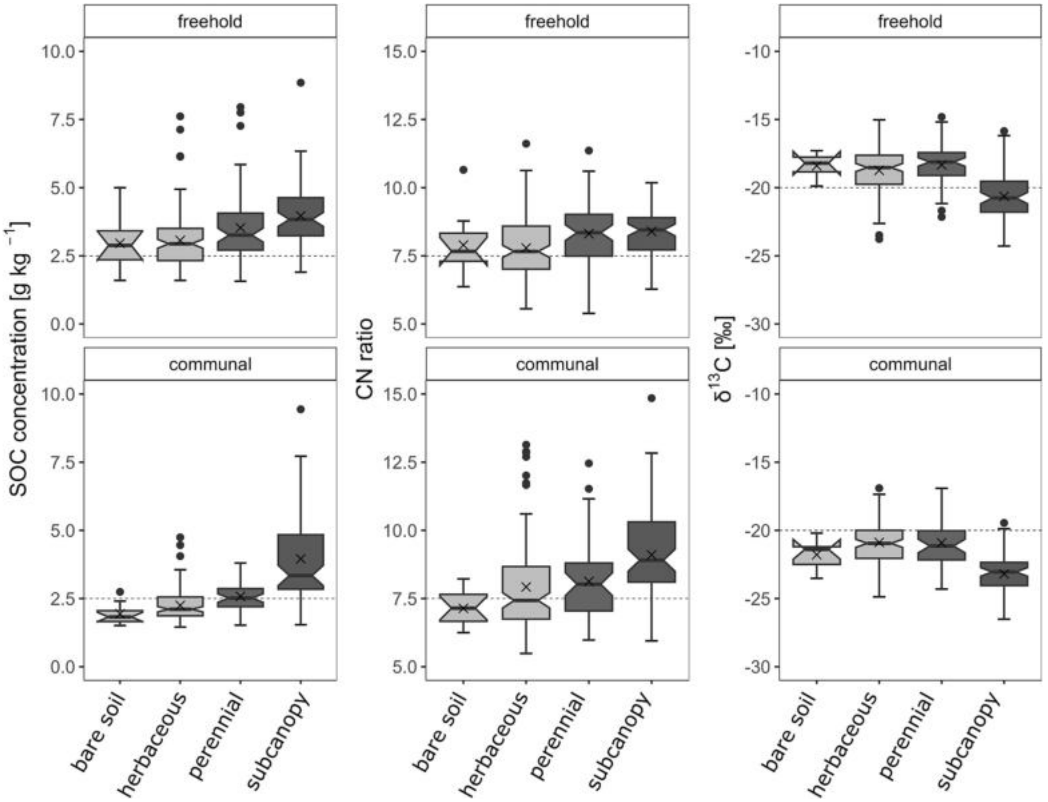

Fig. 5. SOC concentrations, CN ratios and δ^{13}C isotopes in topsoil (0–10 cm) in respect to tenure system and vegetation patch type distinguishing bare soil ($n = 23$), herbaceous vegetation ($n = 243$), perennial grass ($n = 113$), and subcanopy ($n = 143$, Table S2). The median is shown as a bold line, the mean as a cross, and the 95% confidence interval as a notch.

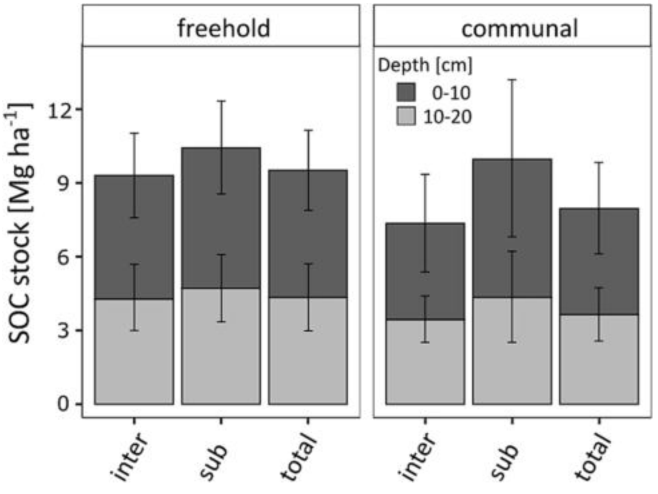

Fig. 6. SOC stocks ($\overline{x} \pm s$) in respect to tenure systems, soil depth, and habitat type differentiated between 'inter' for intercanopy habitats, 'sub' for subcanopy habitats and 'total' upscaled for the actual proportion of habitats on plots according to Eq. (1) ($n = 144$).

tipping points, we fitted several linear mixed effects models with clay content as an inherent determinant of the system (Table S4). The log-transformed distance from water points served as a proxy for the linear decrease in grazing intensity. According to the AIC, the linear model performed best in comparison to quadratic, cubic, and exponential models. In the linear model, the effect of clay was almost 4 times greater than grazing (Table S4). However, the contribution of grazing to the marginal R^2 of 0.26 was only 12% (Table S4). Considering that the 95% confidence intervals (−0.68, −0.03) are close to zero, we cannot rule out that the effect of grazing is much weaker, in line with the visual observations (Fig. 7). In addition, we found no evidence of a non-linear response of soil parameters toward grazing pressure, which is often perceived to be an indicator of tipping point behavior.

3.7. Other soil- and vegetation-related factors influencing rangeland properties

About half of the variance in SOC stocks was explained by linear random effects model by grazing intensity, clay and the random effects (conditional $R^2 = 0.53$, Table S4). To unravel some of the remaining dynamics of SOC stocks in freehold farms and communal rangelands, we created separate correlation matrices (Fig. 8) with clay as a system-inherent soil property, WHC_{max}, $\delta^{13}C$ values, and according to our hypotheses, grazing intensity, cover of bare soil, herbaceous vegetation, perennial grasses (a proxy for more short-term soil-plant interactions) and woody plants (a permanent vegetation determinant). On freehold farms, SOC stocks were positively correlated with clay content ($r = 0.37$), WHC_{max} ($r = 0.55$), herbaceous vegetation cover ($r = 0.43$), and the perennial grass cover in intercanopies ($r = 0.6$), and negatively with woody plant cover ($r = -0.33$). On communal rangelands, SOC stocks were positively correlated with clay content ($r = 0.49$), WHC_{max} ($r = 0.5$), and cover of woody plants ($r = 0.44$). Grazing intensity negatively correlated with SOC stocks in freehold farms ($r = -0.39$) but not communal rangelands.

4. Discussion

Savanna rangelands are threatened by degradation that is likely to result in irreversible shifts in ecosystem conditions. This is evident in the rangelands of the southeast of the Waterberg Plateau in Namibia, where the loss of perennial grasses and the process of woody plant encroachment are reducing the carrying capacity of the rangelands for livestock. Rangelands are more vulnerable to degradation when SOC is lost (Schwinning et al., 2008; Hoover et al., 2020) and when the plant-life history of the herbaceous layer is dominated by annuals rather than perennials (Ruppert et al., 2015). We found that communal rangelands with free grazing and without bush thinning were more degraded than freehold rangelands with rotational grazing and bush thinning, based on communal rangelands having overall lower perennial grass cover, higher woody plant cover, and lower SOC stocks (Table S1, Fig. 6).

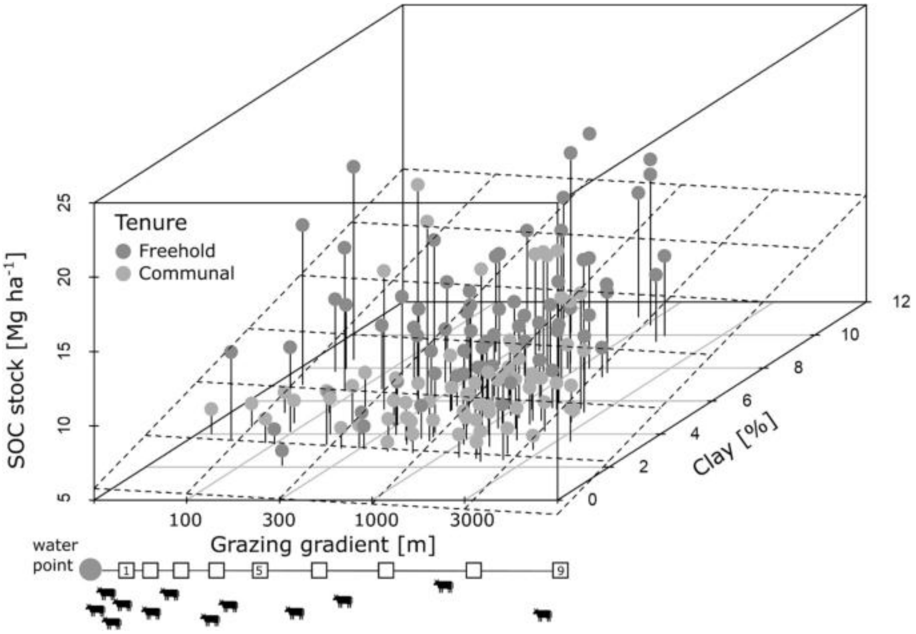

Fig. 7. Visualization of effects of grazing intensity (x-axis), and clay (z-axis) on SOC stocks in topsoil (0-20 cm) across both tenure systems (n = 144). Dashed pane indicates the linear least square regression fit to all data points. Detailed regression outcomes can be found in Table S4.

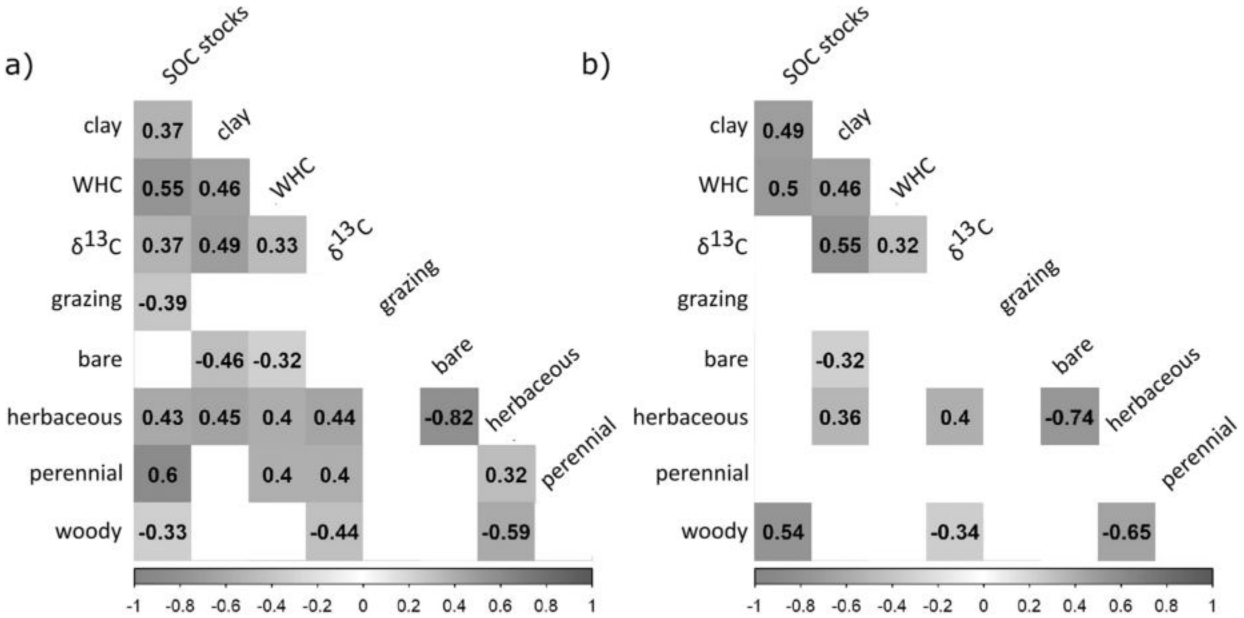

Fig. 8. Correlation matrices in freehold farms (a) and communal rangelands (b), including for SOC stocks (0-20 cm), clay content, WHC_{max}, δ^{13}C values, grazing intensity, and cover of bare ground, herbaceous vegetation, perennial grasses in intercanopy habitats, and woody plants at a significance level of $p < 0.05$. Shades of red show negative correlations and blue positive correlations. Intensity of color indicates strength of correlation.

4.1. Physical soil properties

Our study area is located in the Kalahari Basin, characterized by deep Arenosols with high sand content. We found no variation in physical soil properties along the gradients, but significant differences in texture when comparing the land management systems, with higher clay and silt contents in freehold farms than in communal areas. Wind erosion in savannas can lead to a decrease in clay content and an increase in fine sand content (van der Westhuizen et al., 2022). However, we found no evidence of erosion along the gradients, such as exposed woody roots, despite a slight increase in clay content with soil depth in both tenure systems (Table 2). Inherent textural differences between the two systems are more likely to be the cause for the observed variations in clay and silt. The clay content of Arenosols can vary between 0 and 15% (IUSS Working Group WRB, 2022). Communal rangelands were at a lower part of this range than freehold farms. In addition, the same initial geological substrate, the color of the sands varied, from reddish-brown on the freehold farms to yellowish-brown on the communal rangelands. Such color differences can indicate different grain size distributions in Kalahari sands despite the same wind-blown origin (Du Bothma and Du Toit, 2018), which we consider to be true in our case. It is not uncommon in southern Africa for freehold farms to have more ideal soil conditions for plant growth than communal villages. With better land historically allocated to freehold farms (Sandhage-Hofmann et al., 2015; Menestrey Schwieger and Mbidzo, 2020), soils on freehold farms are likely to have had better baseline conditions, which are associated with a higher WHC_{max} (Table 2) and hence drought resilience than soils on communal rangelands, likely exacerbated by different land management strategies.

Soil degradation in arid rangelands is often associated with soil compaction from cattle trampling (Kotzé et al., 2020; Sandhage-Hofmann, 2022). However, in our study area, mean bulk density was not affected by grazing intensity or overall land management (Table 2), which is in line with the sandy texture of the Arenosols (Hartemink and Huting, 2008; Sandhage-Hofmann et al., 2015). In respect to ground cover, we found that bulk density was significantly lower by 0.2 g cm^{-3} in subcanopy habitats compared to adjacent intercanopy habitats. This agrees with Tate et al. (2004). One reason for lower bulk densities in subcanopy soils can be a more active soil fauna there due to different environmental conditions (litter, roots, and microclimate), with more biopores leading to a general loosening of the soil under shrubs and trees (Marquart et al., 2020). In addition, woody plants of a certain size form a physical barrier for cattle, which is why these areas are less prone to trampling and better preserve biopores (Marquart et al., 2020). In our sandy soils, bulk density was not a sensitive indicator of livestock-related degradation.

4.2. SOC as degradation indicator

Sandy savanna soils with little rainfall generally have low SOC contents (Jones, 1973). Mean SOC values were 2.3 ± 1.0 g C kg^{-1} soil in communal rangelands and 3.1 ± 1.0 g C kg^{-1} soil in freehold farms across topsoil (0–10 cm) in intercanopy habitats (Fig. 5, ‘herbaceous’). Similar to Sandhage-Hofmann et al. (2015), we found the highest values in freehold farms closest to water points, where cattle excreta were most abundant, and in subcanopy soils, where plant litter input is also elevated. Our results are consistent with other studies in southern Africa, and the Greater Waterberg Landscape (Nghalipo et al., 2019; Nghikembua et al., 2021; Sandhage-Hofmann et al., 2015).

Several factors, including land management, can influence the SOC concentrations, especially at smaller spatial scales of observation (Wiesmeier et al., 2019). Comparing 64 rangeland studies, Byrnes et al. (2018) found that free grazing can reduce SOC, CN ratios, and total N, while rotational grazing maintains higher SOC concentrations, because resting periods allow the recovery of plant biomass and hence meliorate soils. Meta-analyses have some limitations (Fohrafellner et al., 2023), and the effects of grazing management practices may vary by ecosystem type (Di Virgilio et al., 2019) and site-specific environmental factors (Byrnes et al., 2018). However, sustainable land management requires adaptive grazing strategies with flexible decision making (Di Virgilio et al., 2019). In Namibian rangelands, there is an ongoing shift from communal to private land (Atlas of Namibia Team, 2022). Rotational rangeland management is usually withheld to freehold farms, while resources on communal rangelands are available on a first-come, first-served basis. Tenure limits the flexible and sustainable use of forage on communal rangelands (Atlas of Namibia Team, 2022) and hinders the potential for SOC storage provided by the presumed positive SOC response to rotational grazing (Byrnes et al., 2018), especially given the spatial extent of arid savannas and their almost exclusive use as rangeland.

Besides land management, texture, particularly clay content, is considered an important predictor of SOC storage in soils (Nichols, 1984; Amelung et al., 1998). Recent studies have shown that the potential to store SOC in subtropical and tropical soils can be more complex than correlations with the total clay content (von Fromm et al., 2021). Consistent with these findings, we observed no differences in SOC stocks in the subcanopy habitats between the tenure systems despite differences in clay content (Fig. 6, Table 2), but variations between and within intercanopy habitats. Clay roughly determines the overall potential for SOC storage, but in this arid system, other factors such vegetation also play an important role.

In addition to clay, in both land management systems, SOC stocks correlated with the WHC. In freehold farms the correlation of WHC and SOC stocks was stronger than that with clay, likely because WHC benefits from SOM (Bagnall et al., 2022) that possibly comes from the perennial grasses there. Clay and SOC both contribute to WHC. It was shown that 1% in SOM can increase about 1.5 to 1.7% of WHC, especially in sandy soils (Libohova et al., 2018), indicating a positive feedback between perennial grasses and the WHC.

In rangelands, vegetation influences soil-chemical properties and vice versa (Kotzé et al., 2020), which in savannas is characterized by the coexistence of woody plants and grasses (Malongweni and van Tol, 2022). There are several studies comparing subcanopy and intercanopy soils. For example, in the Kruger National Park, South Africa, Malongweni and van Tol (2022) found that concentrations of SOC and total N were almost twice as high in subcanopy soils. Similar trends were observed in Kuruman, South Africa, by Sandhage-Hofmann et al. (2020) who reported higher SOC, N and pH between woody, herbaceous and bare soil dominated patches. This is consistent with our results, although the differences between vegetation patch types were more pronounced on communal rangelands than on freehold farms. However, we found that the higher woody plant cover and the higher SOC concentrations in communal areas did not translate to greater SOC stocks likely because of lower bulk densities in the subcanopy habitats, which resulted in a leveling effect on the overall SOC stocks in the topsoil (0-20 cm). This finding contrasts with other studies that focused on woody plant encroachment (Eldridge et al., 2011; Sandhage-Hofmann et al., 2020), showing the benefits of woody plant encroachment on SOC stocks. However, inconsistent patterns were emphasized in a review of studies quantifying changes in SOC due to woody plant encroachment by Archer et al. (2017).

Unlike previous studies, we not only compared subcanopy and intercanopy soils, but also differentiated within intercanopy habitats between patches with perennial grasses, other herbaceous plants, and bare soil. We found that SOC concentrations, CN, and total N concentrations increased with the longevity of the present plant-life form, from bare soil to herbaceous vegetation to perennial grasses and woody plants. This was true for both tenure systems, although communal rangelands generally had lower SOC values for each vegetation patch type, except for the subcanopy soils. In addition, the differences between the soil cover types were greater in communal rangelands. Intercanopy habitats are highly sensitive to grazing impacts (Malongweni and van Tol, 2022). This was also supported by the lower mean diameter and abundance of perennial grass tufts, which was 3.6 ± 2.9 cm ($n = 44$) on communal rangelands compared to 7.5 ± 4.0 cm ($n = 69$) on freehold farms (data not shown, Fig. 2). It appears that not only the presence of herbaceous cover, but especially its composition from C_4 grasses is crucial for SOC storage, with perennial grasses increasing the SOC content. Overall, this approach confirms the feasibility of SOC as indicator of rangeland degradation, when considered separately for different types of vegetation patches. Habitat-specific monitoring approaches can hence improve the accuracy and precision of predicting changes in the spatial distribution of carbon stocks (Dimobe et al., 2018).

4.3. ^{13}C isotopes in soils as indicator for SOC origin

Stable ^{13}C isotopes helped to identify the origin of SOC. This is particularly useful in ecosystems that are undergoing a vegetation shift up to crossing a threshold (Boutton et al., 1998), e.g., from grass-dominated vegetation to woody-dominated vegetation. In a recently published study conducted within our research area, rangeland change trends were analyzed using high- to medium-resolution images from satellite sensors dating back 55 years (1965–2020; Brinkmann et al., 2023). Both freehold farms and communal areas experienced woody plant encroachment from 1965 to a comparable extent. However, between 1986 and 2001, these trends ceased in freehold farms, presumably due to bush thinning measures. In savannas, characterized by a mixture of woody plants (forbs and trees, $\sim -25.7\ \delta^{13}$C ‰, Codron, 2005) and C_4 grasses ($\sim -11.7\ \delta^{13}$C ‰, Codron, 2005), both contribute to SOC. In our study area, the mean carbon isotope ratio varied between the two tenure systems, with −18.5 ‰ and − 21.0 ‰ in the topsoil (0-10 cm) of freehold farms and communal areas, respectively. According to Boutton et al. (1998), the vegetation composition is considered stable when the δ^{13}C values of SOC in the upper 20 cm are similar to the plant community. This was observed across all vegetation patch types, where isotope ratios (Fig. 5) were clearly influenced by the present vegetation, but with consistently less negative δ^{13}C values in freehold farms. We assume that the higher cover of perennial grasses, and the lower cover of woody vegetation on freehold farms could be responsible for these results. In communal rangelands the calculated contribution of woody-derived carbon is 64%, compared to 45% in freehold farms with little change observed along the grazing gradient. These results indicate that the current herbaceous vegetation remains the dominant input source for SOC in freehold farms, while woody plants were the main input source in communal areas, corresponding to the higher canopy cover of woody plants in communal areas. We interpret the differences in ^{13}C isotopes between tenure systems as plant community shifts due to bush thinning on freehold farms. Bush thinning enhances the competitive advantage of perennial grasses for space, water, and light compared to woody plants.

Woody plant encroachment is perceived to be enhanced by many factors, including increasing atmospheric CO_2 (Devine et al., 2017), but also the influence of management on vegetation-herbivore interactions (Koch et al., 2022) at a local-scale (Devine et al., 2017). Livestock has long been of cultural importance to the Ovaherero (the communal land users in the study area), contributing to food security, sustainability, and social status in communal farming systems, but state interventions, policies and human population growth have substantially altered their local social-ecological processes (Menestrey Schwieger and Mbidzo, 2020). The current land allocation of our study sites was established between 1911 and 1955, so the two different management strategies have been practiced in rough forms for >60 years (see 2.1), whereas systematic bush thinning was first applied in the 1980s on freehold farms. Our analyses confirmed, the presence of C_4 grasses in deeper soils, suggesting that the establishment of woody plants has occurred mainly in recent decades. The combination of bush thinning and rotational grazing on freehold farms appears to have prevented woody plant encroachment and created a stable ecological condition under the given management interventions, with C_4 plants as the dominant SOC input source. In contrast, soil isotope analyses revealed that communal rangelands experienced woody plant encroachment, indicating a threshold of limited reversibility without management interventions. We conclude that ^{13}C isotopes serve as an indicator to reveal ongoing vegetation shifts, such as the transition from an open savanna dominated by C_4 grasses to a woody-dominated system as seen in the communal rangelands, combined with a loss of ecosystem functions as rangeland.

4.4. Tipping points along water point gradients?

Along the grazing gradients, we found high variation in physical and chemical soil properties and high variation (see 3.5). Grazing effects are relatively weak compared to the background noise (Arndt et al., 2022). Analyzing spatial patterns around water points is not trivial and highly site- and context-dependent (Hess et al., 2020), as quantifying heterogeneity remains a major challenge in rangeland studies (Li and Reynolds, 1995; Fuhlendorf et al., 2017). The identification of such tipping points requires that there are clear degradation gradients up to complete desertification. Hence, we conclude that, at least along our transects, tipping points for increased rangeland degradation either did not exist or were not reached under the prevailing grazing pressure.

In the Omaheke region, 160 km east of our study area, Katjiua and Ward (2012) also observed no changes in SOC, total N, available P, and soil moisture with increasing distance from water points in communal rangelands. There, beyond 200 m soil and vegetation changes were rather uniform (max. 5 km; Katjiua and Ward, 2012). Our communal gradients were comparable, both in their lengths ranging from 2 to 3 km, and also in their SOC values.

The absence of tipping points matches with the findings of Hillebrand et al. (2020), who could not detect thresholds along spatial gradients in a dataset of 36 meta-analyses (comprising 4601 studies of global change effects on natural communities, e.g., soil respiration) as well. In their study, sensitivity analyses revealed that noise in the response variables due to complex interactions of multiple factors hindered the identification of tipping points. Hillebrand et al. (2020) therefore doubt that tipping points can be defined with empirical data along gradients of environmental stress. Nevertheless, we cannot rule out the existence of tipping points (Hillebrand et al., 2020).

4.5. Natural experiment: texture and tenure as confounder

Understanding how grazing interacts with local environmental factors is crucial to predict what will happen to dryland ecosystems under climate change and increasing human pressure (Maestre et al., 2022). Differences in SOC concentrations and their origins between land management systems, alongside vegetation cover data, and varying grazing intensity, can be sensitive indicators of degradation in soils of arid environments (de Moraes Sá et al., 2018), but the application of a natural experiment introduces certain constraints. The systematic differences in clay content and land management systems made it challenging to establish causal relationships.

Our study gives valuable insights into the status quo of soil conditions in freehold farms and communal rangelands of the Greater Waterberg Landscape. SOC and isotope values varied within vegetation patch types and between land management systems. We attribute some of the observed differences to targeted land management interventions, such as grazing regime and bush thinning. These practices are known to affect the growth of perennial grasses and subsequently SOC. This observation aligns with findings from other studies that compare SOC over larger areas (Kotzé et al., 2013; Sandhage-Hofmann et al., 2015; Chabala et al., 2022). While land management interventions contribute to the observed variation in SOC, differences in SOC and isotope ratios between tenure systems may also imply differences between sites in fertility and the overall capacity to grow grasses.

5. Conclusion

SOC was a reliable indicator of rangeland condition when soil samples were stratified by vegetation patch type, especially in the intercanopy habitats, and should therefore be included in future rangeland studies. Additionally, $\delta^{13}C$ isotopes proved useful to understand the input sources of SOC. In savannas, SOC storage and forage production are closely linked and sensitive to land management. However, in the context of a natural experiment, interpreting differences in SOC between land management systems proved challenging because controlling for the influence of confounding factors (e.g., tenure and clay) was not straightforward.

There is a compounding of problems that increase the risk of rangeland degradation in the Greater Waterberg Landscape. In particular, the most vulnerable areas (e.g. least fertile soils with little clay) are those where management (e.g. continuous grazing and no bush thinning) is most likely to degrade rangeland conditions. The presence of perennial grasses correlated with SOC storage in the soils and the water holding capacity, implying a positive feedback that can enhance the resilience to drought and anthropogenic use in savannas.

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Ethics approval

The National Commission on Research, Science, and Technology in Namibia approved the research for this paper (Permit Number RPIV01162020).

CRediT authorship contribution statement

Katrin Zimmer: Conceptualization, Data curation, Formal analysis, Validation, Visualization, Writing – original draft. **Vistorina Amputu:** Data curation, Writing – review & editing. **Lisa-Maricia Schwarz:** Data curation, Writing – review & editing. **Anja Linstädter:** Conceptualization, Funding acquisition, Writing – review & editing. **Alexandra Sandhage-Hofmann:** Conceptualization, Data curation, Funding acquisition, Supervision, Validation, Writing – review & editing.

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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Appendix A. Supplementary data

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