Interactions of soil properties and dominant tree canopy species with vegetation recruitment within a mine wasteland of the Zambian Copperbelt

Arthur A. Owiny [a 1,*], Paxie W. Chirwa [b 2], Jules Christian Zekeng [c d 3], Theodore Mulembo Mwamba [c e 4], Stephen Syampungani [b c 5]

[a]School of Natural Resources, Department of Plant and Environmental Sciences, Copperbelt University, P.O Box 21692, Kitwe, Zambia

[b]University of Pretoria, Department of Plant and Soil Sciences, Plant Sciences Complex, Pretoria, 0002, South Africa

[c]Chair of Environment and Development, Oliver R. Tambo African Research Chair Initiative Project, Copperbelt University, P.O. Box 21692, Kitwe, Zambia

[d]University of Douala, Advanced Teachers Training School for Technical Education, Department of Forest Engineering, P.O. Box 1872, Douala, Cameroon

[e]Faculty of Agriculture, University of Lubumbashi, Kasapa Road, P.O. Box 1825, Lubumbashi, Congo

* Corresponding author. Email: owinyiarthur@yahoo.com

Highlights

- Heavy metals reduce biodiversity under dominant tree canopy species.
- Tailing dam shows low variability in soil properties.
- Random Forest analysis reveals how soil variables influence biodiversity.
- Calcium and Magnesium positively influence herbaceous richness and grass abundance, respectively.
- Dominant species have a greater influence on biodiversity than soils.

Abstract

Mine wastelands can affect soil's properties, leading to stunted plant growth. However, little is known about the effect of different categories of soil parameters on plant recruitment under harsh conditions such as mine wastelands. The study aimed to determine how various soil properties contribute to vegetation recruitment around dominant tree canopy species within a Cu mine wasteland of the Zambian Copperbelt province. Soil and understory tree vegetation were sampled from 8 m radius circular plots beneath the canopies of dominant tree species. Grass and herb samples were collected from smaller, nested plots with a 1 m radius. Soil properties were analysed and biodiversity metrics (i.e. species richness, abundance, Shannon-Wiener and Simpson indices) were calculated. Data analysis included Kruskal-Wallis tests to assess soil properties variations among canopy species, a Weighted Linear Mixed Model (W-LMM) to evaluate the fixed effects of dominant tree canopy species on the biodiversity metrics and Random Forest models to identify key soil variables influencing biodiversity metrics. Results showed minimal variation in soil properties among canopy species. Dominant tree canopy species significantly influenced biodiversity, with *Ficus capensis* and *Ficus craterostoma* enhancing richness and diversity in tree and herb layers, while grass layers showed minimal responses. Heavy metals (e.g. Zr, Sn, Pb) reduced diversity, whereas nutrients like Fe, Ca, and Mg positively affected recruitment, explaining up to 40.5 % of variance in tree richness. These results emphasize the importance of soil nutrient management and species selection in improving plant recruitment, offering key considerations for enhancing ecological restoration efforts in mine tailings dams.

Keywords: Biodiversity dynamics; Ecological restoration; Heavy metal toxicity; Phytoremediation; Soil-Plant interaction; Soil contamination; Species recruitment

1. Introduction

The Zambian Copperbelt region, endowed with abundant copper deposits, has historically experienced extensive extraction and processing activities, thus resulting in substantial amounts of copper and other metals in the environment (Sillitoe et al., 2017). This has contributed to water, soil, and air contamination, thus posing serious threats to biodiversity and human health in the region (Kosgei and Mukuwa, 2020, Muimba-Kankolongo et al., 2022). Such risk of pollution is exacerbated by the presence of minerals containing sulfurs, which, when exposed to air and water, bring about the phenomenon known as acid mine drainage (AMD), resulting from chemical reactions that produce sulfuric acid (Park et al., 2019). The acidic runoff and the metal release into the nearby ecosystems harm water, soil and air quality (Franks et al., 2011, Lèbre et al., 2017).

Various approaches address pollution from mine wastelands, including physical and chemical techniques like soil capping, washing, and chemical precipitation, depending on the contaminant of interest (Dermont et al., 2008, Dhaliwal et al., 2020, Koul and Taak, 2018). However, these methods come with high costs, potential secondary pollution (Anawar and Chowdhury, 2020), and limited sustainability, as they do not restore the ecosystem to its natural state. Addressing environmental pollution requires a holistic and multifaceted approach considering the type of mining activity, contaminants, and environmental aspects such as soil conditions. Phytoremediation has been observed to be a sustainable and cost-effective approach to mitigating pollution and promoting ecological restoration by utilizing native plant species to naturally clean the environment of contaminants (Peco et al., 2021). It can enhance biodiversity, improve soil quality, and provide long-term environmental benefits, thus making it a compelling choice for addressing pollution associated with mine wastelands.

Several studies have been conducted in various regions to identify plant species suitable for phytostabilization purposes (e.g., Asia, North Africa, North America) precisely from e.g., China, Egypt, and Mexico, respectively (Galal and Shehata, 2015, Li et al., 2007, Santos et al., 2017). For instance, Santos et al. (2017) identified five native perennial plant species with potential for phytostabilization of the Nacozari mine tailings (Mexico). In the Democratic Republic of Congo, Ilunga et al. (2015) studied plant functional traits that are desirable for plants for ecological restoration of the Katanga copper mining area and recommended that their choice be based on desirable annual life cycle, growth phenology in the wet season, root depth of 0–10 cm underground system, bud bank by seeds, and dispersal size of < 2 mm × 2 mm. In addition, Festin et al. (2019) compared tree species suitable for phytoremediation between copper-contaminated soils and the neighboring natural soils in the Copperbelt Province of Zambia and found that the biological traits that favour colonization of tailings dams include ability to control erosion, fast growth in high light level, moderate tolerance to copper as well as nitrogen fixing ability, mycorrhizal symbiosis and ability to regulate copper uptake through accumulation or exclusion. In a follow-up study, Matakala et al. (2023) evaluated the phytostabilization potential of indigenous tree species in the tailings dams of the Zambian Copperbelt province for restoration purposes. While these studies provided insightful information, identifying at least 10 tree species suitable for the phytoremediation of heavy metals, the precise impact of soil properties on species recruitment in such harsh conditions remains unclear. Understanding the intricate relationship between soil properties and native tree species recruitment is vital for restoring ecosystems and making them capable of thriving in mine wastelands (Wang et al., 2021a). Similarly, there is a particular need to gain insights into the influence of soil properties on understory plant species recruitment, which has received little attention to date. Moreover, the partitioning influence of soil properties such as texture, organic matter content, and mineral composition can influence the behavior of contaminants in soil and vice versa, which in turn significantly impacts understory plant species recruitment and the success of ecological restoration on mine wastelands (Fu et al., 2020).

Soil conditions play a critical role in the successful establishment and growth of these plant species in mine wastelands. In areas where mining activities have caused severe soil degradation, such as the Copperbelt province of Zambia, recruitment of plant species can be extremely challenging due to elevated levels of metals. These high concentrations of metals can be toxic to many plant species because they can disrupt nutrient uptake, impair physiological processes, and induce plant toxicity symptoms (Singh et al., 2016). When present at elevated levels, heavy metals can limit the availability of essential soil nutrients (Perkovic et al., 2022, Qaswar et al., 2020), nutrient acquisition, and the general performance of plants (Goyal et al., 2020). Likewise, the acidic nature of the soil resulting from acid mine drainage can directly impact the availability of essential nutrients. It can alter soil pH, which creates unfavourable conditions for native tree species (Venkateswarlu et al., 2016). The presence of elevated copper and other metal levels in the soil negatively affects the germination, survival, and health of trees, thus impeding the restoration process (Skousen et al., 2019). A comprehensive mine reclamation such as soil management practices and site management e.g., soil amendment, fertiliser and organic matter, can be applied to improve mine wastelands conditions.

Understanding the relationship between vegetation recruitment and soil properties is crucial for restoring mine wastelands. Therefore, our study focused on understanding this relationship by answering the following questions: i) How do soil properties vary within and among the dominant tree canopy species within the mine wasteland? It was hypothesised that soil properties, including textural, chemical, and heavy metal concentrations, vary significantly among different dominant tree canopy species within the mine wasteland, reflecting the influence of each species on soil composition and nutrient availability; ii) How do species richness, abundance and diversity metrics vary between dominant tree canopy species within the tailing dam? It was hypothesised that species richness, abundance, and diversity metrics differ significantly between dominant tree canopy species within the tailings dam, with certain canopy species exerting more substantial effects on these metrics in the tree and herb layers than in the grass layer (iii) How do soil variables shape species richness, abundance, and diversity metrics? It was hypothesised that soil properties influence varies in species richness, abundance, and diversity metrics.

2. Materials and methods

2.1. Study area

The study was conducted on Tailings dam (TD 10) (12°32'08.0"–12°31'22'' S and 28°14'31.0"–28°14'10'' E), in Mufulira District, Copperbelt Province of Zambia (Fig. 1) between February and April 2022. TD10 was decommissioned in 1988 and is currently under the management of Mopani Copper Mine. It is the most recently decommissioned tailings dam in the Copperbelt Province and appears to be at the early stages of vegetation succession, making it a suitable area for this study. The tailings dam is predominantly contaminated with copper, and at some point, cobalt, barium, nickel, arsenic, zinc, lead, chromium, vanadium and cadmium (in order of decreasing concentration) are also present (Chileshe et al., 2019). Mufulira is located approximately 19 Kilometers Southwest of Mokambo boarder with the Democratic Republic of Congo at an average elevation of 1274 m above sea level. It lies within the medium rainfall belt of Zambia and receives an average of 1000 mm of rainfall annually

(Festin et al., 2019). The mean monthly temperature is between 16 °C in June and July and 26 °C in October. It comprises three seasons: a wet season between October and April, a cool and dry season between May and July, and a hot and dry season between August and October (Sracek et al., 2018).

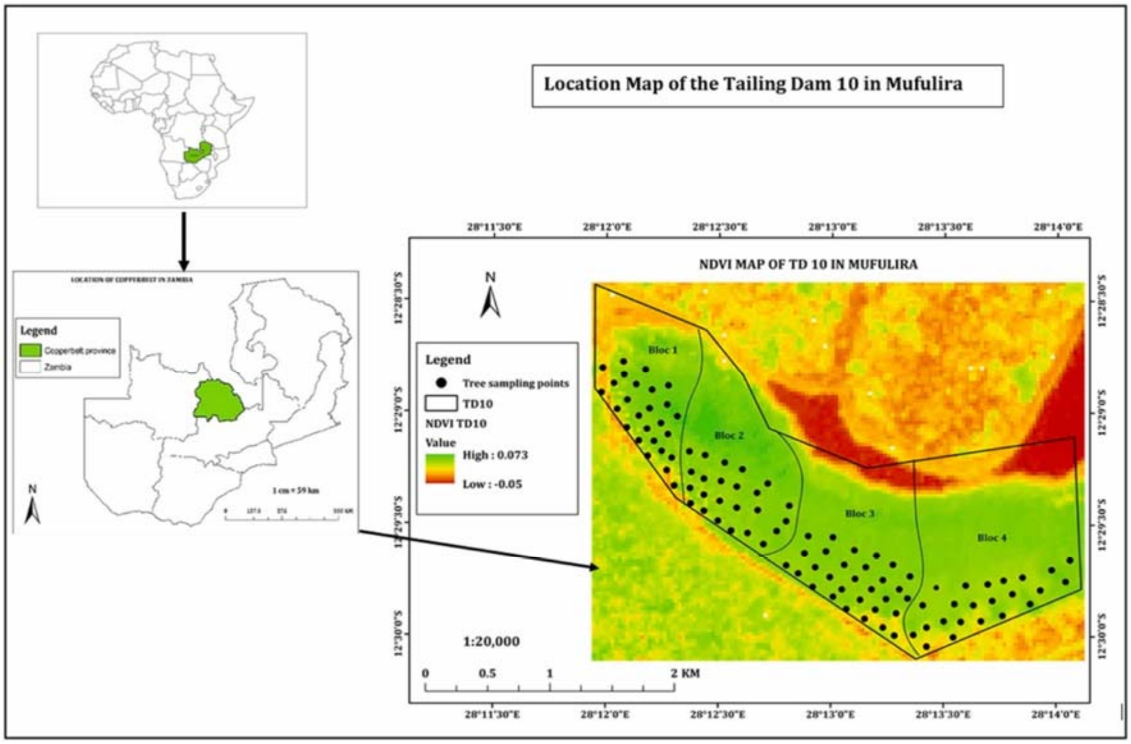

Fig. 1. Map of the Tailings dam (TD 10) and the location of sampling plots in different blocks.

2.2. Sampling plot design and plot selection

The selection of dominant tree canopy species in this study was informed by earlier studies conducted in the same area (Festin et al., 2019, Matakala et al., 2023), which provided essential insights into the biological traits of a broad range of mine wastelands-native tree species. Festin at al. (2019) examined growth patterns, root structures, and nutrient uptake capacity and identified species with the potential to adapt to the harsh conditions of mine wastelands., while Matakala et al. (2023) singled out tree species' ability to stabilize soil and tolerate heavy metals, underscoring their interest in the prospect of ecological restoration of mine wastelands. Results from these studies provided for categorising the inventoried tree species in terms of dominance on mine wastelands across a large area of the Zambia Copperbelt Province. Based on this, the ten most dominant tree canopy species were identified as the target tree species for this study i.e., *Acacia sieberiana, Acacia polyacantha, Bauhinia petersiana, Dichrostachys cinerea, Ficus capensis, Ficus craterostoma, Gmelina arborea, Rhus longipes, Syzygium guineense,* and *Terminalia mollis*. The dominance of these tree species plausibly implies their better resilience to environmental disturbances and potentially a greater likelihood to effectively contribute to ecological restoration of the mine wastelands by allowing recruitment, diversity and growth of understory vegetation in mine wastelands. In plots with more than one dominant tree canopy species, mixed plots were considered, with the most significant species by basal area highlighted in the designation of the plot, such as (i) Mixed species plots with *Acacia polyacantha,* hereafter MsAp; (ii) Mixed species plots with *Bauhinia petersiana,* hereafter MsBp (Table 1). Considering these mixed plots in our experimental design allows us to ensure that our data are generated in a scenario that captures the natural complexity of the investigated ecosystem.

Table 1. Distribution of plots belonging to the dominant tree canopy species inventoried within the TD 10.

	Blocks			
Dominant tree canopy species	**B1**	**B2**	**B3**	**B4**
Mixed plots with *Acacia polyacantha* (MsAp)	3		3	3
Acacia polyacantha	3	9	4	2
Acacia sieberiana		2	1	
Mixed plots *Bauhinia petersiana* (MsBp)		1	1	2
Bauhinia petersiana		2		
Dichrostachys cinerea		2	1	1
Ficus capensis	1		1	1
Ficus craterostoma	5	1	4	4
Gmelina arborea	3	1	2	
Rhus longipes	1		2	2
Syzygium guineense		1	1	1
Terminalia mollis	1	4		5

The stratification and map of the TD 10 were done using a normalized difference vegetation index (NDVI) color, permitting the exclusion of the area without vegetation (Fig. 1). Based on the topography map, the remaining vegetated area was divided into four blocks (B) according to slope orientation: Northward (B1; plots 1–23), Westward (B2; plots 24–47), Southward (B3; plots 48–77) and Eastward (B4; plots 78–100) slopes. In each block, a transect line of 1 km was established, and along the transect line, each selected dominant tree canopy species identified was used as the center of the circular sample plot. Each circular plot had a radium of 8-meter and was located at least 50–100 m from any other plot centroid (Robertson et al., 2007). The botanical inventory and measurement of tree species ≥ 5 cm dbh and soil sample collection were done within the 201.062 m^2 area covered by the dominant tree canopy species. In addition, four 1 m radius subplots were nested within the 8 m radius plots (Place in the North, West, South, and East directions of the dominant tree canopy species) to inventory the grass and herbs layers.

2.2.1. Soil parameters

At the same sampling points (plots) of grass and herbs, soil samples were collected from a depth of (0–20 cm) using a 7 cm diameter soil auger (Bufebo and Elias, 2020). Soil samples were collected at the same time as the botanical inventory. In each plot, four soil samples (approx. 450 g) were collected from the North, West, South, and East directions and then a composite sample was made for each plot. Samples were stored in polythene bags and taken to the laboratory, where the soil samples were sieved through a 2-mm screen to remove stone particles and roots (Wang et al., 2021b). Samples were air-dried to halt biological transformation before soil textural analysis. Soil properties analyzed were grouped into three categories: (i) soil textural properties (i.e., sand, clay, silt); (ii) chemical parameters herein as chemical parameters (i.e., pH, Soil organic carbon, K, Na, Ca, Mg); and (iii) heavy metals (i.e., Ba, Cr, Cu, Fe, Pb, Mn, Rb, Sb, Sn, Sr, Ti, Zr and Zn) chosen among known toxic metals, were selected based on their significance in assessing the ecological impact on biodiversity, persistence, and trends of heavy metal contamination.

The soil pH was determined using a pH meter, while soil organic carbon (SOC) was analyzed using the Walkley-Black method (Nelson and Sommers, 1982). The exchangeable cations (Ca^{2+}, Mg^{2+}, K^{+}, and Na^{+}) were analyzed using an atomic absorption spectrophotometer (AAS). Exchangeable cations were extracted by heating 5 g of soil sample with HNO_3 at pH 7.0 (Vadivel, 2020).

The soil texture was determined by estimating the percentage of sand (particle size 0.05–2.0 mm), silt (0.002–0.05 mm), and clay (<0.002 mm) obtained by using the hydrometer methods (Miller and Schaetzl, 2011). This study analyzed heavy metal concentrations (Ba, Cr, Cu, Fe, Pb, Mn, Rb, Sb, Sn, Sr, Ti, Zr and Zn) from soil samples with XRF, X-MET7000 eXpress Mining package (Tavares et al., 2019). Samples were subjected to further grinding using a Rock-lab milling machine and passed into a sieve of particle diameter of < 73µm to maintain sample homogeneity (Desroches et al., 2018, Hokura, 2021, Ravansari et al., 2020, Tavares et al., 2020a, Tavares et al., 2020b, Tavares et al., 2020c) (Supplementary materials 1).

2.3. Plant surveys

2.3.1. Tree species layer

All tree species were identified for each plot, and their diameter was measured and recorded. Dominant tree canopy species with (dbh > 5 cm) were considered as the center point for each plot (i.e., sampling plot). For each plot, all trees with (dbh > 5 cm) were measured at 1.3 m aboveground level, while all individual species with diameter < 5 cm were measured at the root cap. Tree species encountered were primarily identified in the field by observing the (leaves, flowers, habit and other diagnostic characters) but in a few cases, specimens were collected and identified at the Kitwe-Zambian Forestry Herbarium. All the tree species were identified at the species level.

2.3.2. Undergrowth herbaceous and grass layer

The abundance of undergrowth herbs and grass species was recorded using four circular plots of 1 m radius according to the modified coefficient of Blaun-Blanquet (Wikum and Shanholtzer, 1978, Bonham et al., 2004). The abundance of herb and grass was recorded as follows: + =< 1 %, 1 = (1–5 %), 2a= (6–12 %), 2b= (13–25 %), 3 = (26–50 %), 4 = (51–75 %), 5 = (76–100 %). Similarly, specimens of unidentified herb and grass species were collected for further identification at the Kitwe-Zambian Forestry Herbarium. All herbs and grass species except one were identified at the species level.

2.4. Taxonomic diversity

This study calculated diversity and dominance indices and species richness of trees, grass and herbs to characterize the vegetation. Species richness refers to the number of species present without particular regard for the exact area or number of individuals examined (Magurran, 2004). However, it is helpful to distinguish between numerical species richness and the number of species present in a collection containing a specified number of individuals (Hurlbert, 1971). The Shannon-Wiener index assumes that individuals are randomly sampled or distributed from an indefinitely large population and that all species are represented in the sample. The Shannon-Wiener (Shannon and Weaver, 1949) and Simpson indices (Simpson, 1949) were computed for diversity indices. The Shannon-Wiener index is computed from the following equation:

$H^1 = -\Sigma\, p_i \ln p_i$;

where p_i is the proportion of individuals belonging to the ith species found in a sample.

The Simpson index (Simpson, 1949) measures the number of species present and considers the relative abundance of each species. It represents the probability that two randomly selected individuals in the area belong to the same species (Maurer and McGill, 2011). Simpson index measures how individuals are distributed among species of a population, and it is given by:

$\gamma = \Sigma_i P_i^2;$

where P_i is the proportion of individuals belonging to the *i*th species found in a sample. Due to its reciprocal character, it starts with the value of 1 and raises up to the total number of species.

2.5. Statistical analysis

To mitigate issues associated with the unbalanced design of the plots across the four blocks, due to the lower number of plots in B2 and more for *Acacia polyacantha*, the Kruskall-Wallis test was used to determine how soil properties vary between the dominant tree canopy species, following the Shapiro-Wilk test, which indicated non-normality ($p < 0.05$). As a non-parametric method, the Kruskal-Wallis test is less sensitive to imbalances, reducing the likelihood of skewed results. When a difference between the mean of soil variables was significant, a turkey HSD post-hoc test was used to separate the dominant tree canopy species.

The Weighted Linear Mixed Model (W-LMM) was used to determine the fixed effects of dominant tree canopy species on the species richness, abundance, diversity and dominance metrics of various layers while accounting for random effects at the bloc level. The Bloc and dominant tree canopy species were weighted to mitigate their imbalance. Marginal and conditional determination coefficients assessed the variance explained by fixed and random effects. To ascertain the specific influences of dominating species, 95 % confidence intervals and significance tests (*p*) have been used (Littell et al., 2006, Nakagawa and Schielzeth, 2013).

The Random Forest (RF) was used to determine how each soil variable shapes species richness, abundance and diversity metrics of various layers. RF is an ensemble learning technique that generates multiple decision trees using random data subsets and features (Breiman, 2001). Each tree was trained on a bootstrap sample, and then a random subset of traits was evaluated for splitting at each node (Liaw and Wiener, 2001). Various indicators, such as Mean Decrease Accuracy (MDA) and Mean Decrease Gini (MDG), were used to assess the model's performance. These measures assess each feature's importance based on its effect on accuracy and Gini impurity, respectively (Archer and Kimes, 2008). The variable relevance scores show how much each predictor contributes to the overall prediction accuracy. Furthermore, we assessed statistical significance using the p-values for each feature's influence on the model's predictions (Altmann et al., 2010). Accuracy for classification tasks was quantified for model performance evaluation, and the importance of each feature in splitting nodes was assessed using p-values for Gini. These measures offer a thorough assessment of the RF model's performance. All analyses were performed in R.3.5.1 (R Development Core Team, 2022).

3. Results

3.1. Soil variation between and among the dominant tree canopy species

3.1.1. Soil textural parameters

The soil analyses revealed that the tailings dam has a sand texture in most areas and does not vary within the tailings dam. Of the 100 plots, 45 % have a loamy sand texture, 36 % have a sandy sand texture, and 19 % have a sandy loam texture. Regarding the texture analyses around each plot, the granulometric composition (% of clay, silt, and sand) does not vary significantly ($p > 0.05$; Fig. 2 and Table S1). The granulometric textural composition across the tailings dam had an average and standard deviation of sand (82.89 ± 7.19), clay (6.58 ± 1.94), and silt (10.15 ± 6.33).

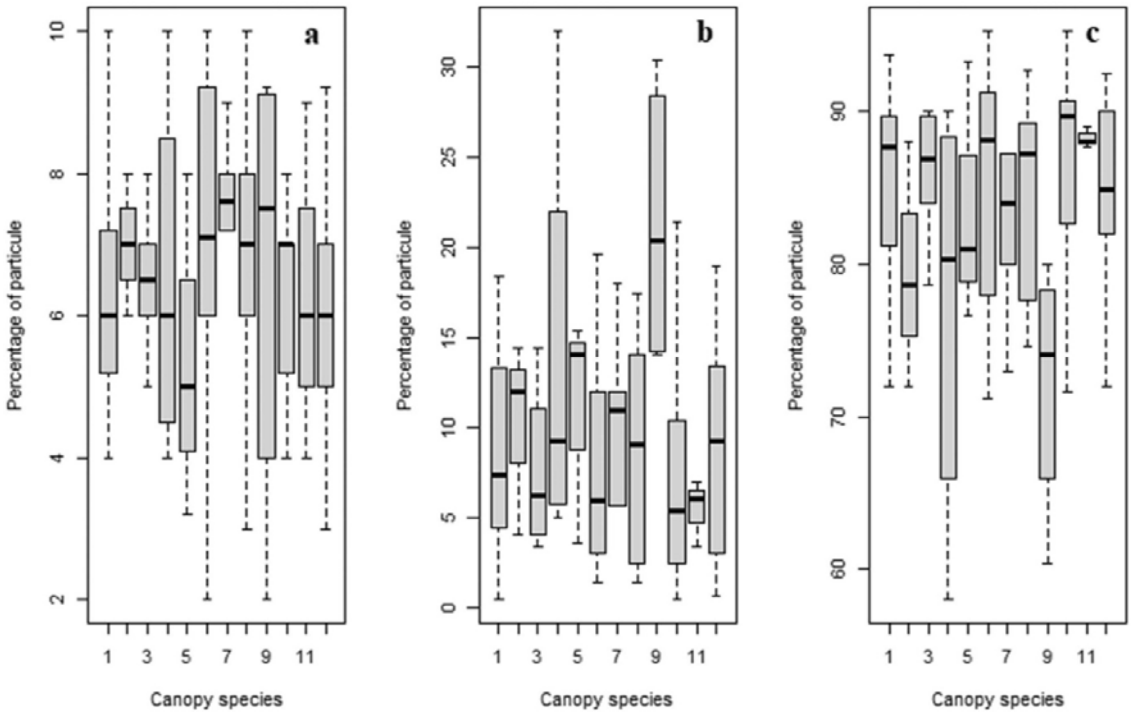

Fig. 2. Boxplot of variation of soil texture parameters between the dominant tree canopy species stands: a) Clay; b) Silt; and c) Sand. Dominant tree canopy species stands: 1. *Acacia polyacantha*, 2. *Acacia sieberiana*, *3.Bauhinia petersiana, 4.Dichrostachys cinerea, 5.Ficus capensis, 6.Ficus craterostoma, 7.Gmelina arborea, 8.Rhus longipes, 9.Syzygium guineense, 10.Terminalia mollis;* 11.Mixed species with *Acacia polyacantha; 12.*Mixed species with *Bauhinia petersiana.*

3.1.2. Soil chemical properties

The chemical properties of the soil samples under the ten dominant tree canopy species are presented in Fig. 3 and Table S1. The results indicated that there is no significant variation (Kruskall-Wallis χ^2= 14.82; *p* = 0.19) for soil organic carbon (SOC) under the dominant tree canopy species (Fig. 3a). For SOC, *Dichrostachys cinerea* dominant tree canopy species recorded the high average concentration (2.23 ± 1.39 %), while *Ficus capensis* dominant tree canopy species recorded the lower average concentrations (0.92 ± 0.38 %). *Ficus capensis* dominant tree canopy species recorded lower concentrations for SOC (Fig. 3). Soil pH around the 10 dominant tree canopy species was basic (8.55 ± 0.4). In particular, the soil pH value around *Acacia polyacantha* dominant tree canopy species (08.34 ± 0.47) was significantly lower than the value around *Terminalia mollis* dominant tree canopy species (08.92 ± 0.28) (Kruskall-Wallis χ^2= 21.26; *p* = 0.03) but similar to the values found around the other dominant tree canopy species (Fig. 3b; Table S1). Regarding the soil macroelements (i.e., Na, K, Ca and Mg), a significant difference was only observed in Ca (Kruskall-Wallis χ^2= 22.48; *p* = 0.02) and specifically around *Ficus craterostoma* dominant tree canopy species (27.432,50 ± 4.23 mg/g) and Mixed species plots with *Acacia polyacantha* (18.46 ± 5.83 mg/g) which showed high and lower concentrations, respectively. However, there were also variations for the other macroelements, although they were insignificant under the different dominant tree canopy species (Fig. 3c–f; Table S1).

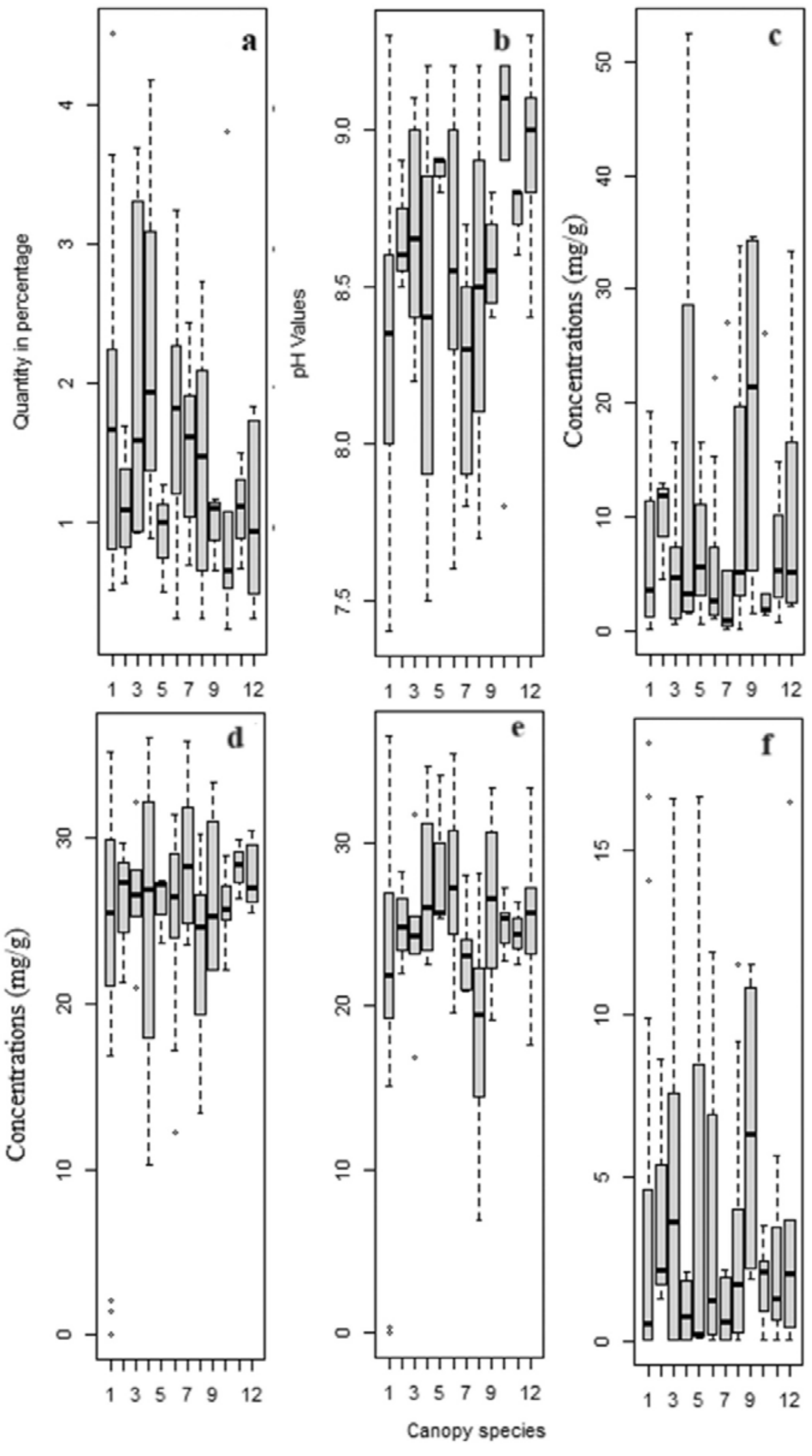

Fig. 3. Boxplot of physico-chemical parameters variations between the dominant tree canopy species stands: a) Soil organic carbon; b) pH; c) Na; d) K; e) Ca and f) Mg. Dominant tree canopy species: 1. *Acacia polyacantha*, 2. *Acacia sieberiana*, 3.*Bauhinia petersiana,* 4.*Dichrostachys cinerea,* 5.*Ficus capensis,* 6.*Ficus craterostoma,* 7.*Gmelina arborea,* 8.*Rhus longipes,* 9.*Syzygium guineense,* 10.*Terminalia mollis;* 11.Mixed species with *Acacia polyacantha;* 12.Mixed species with *Bauhinia petersiana.*

3.1.3. Soil heavy metal concentrations

The heavy metal concentrations of the soil samples from the dominant tree canopy species within the tailings dam are displayed in Fig. 4. The results showed that only Sr (Fig. 4i) varied significantly under dominant tree canopy species (Kruskall-Wallis χ^2= 26.48; $p = 0.005$). The significant difference was between Mixed species plots with *Acacia polyacantha* dominant tree canopy species and *Ficus craterostoma* dominant tree canopy species, which exhibited high content (0.066 ± 0.015 mg/g) and low content (0.042 ± 0.011 mg/g) of Sr, respectively. There was no significant difference between the other heavy metals studied, however, different concentrations were observed depending on the dominant tree canopy species and the heavy metal in consideration (Fig. 4a–h, j–m; Table S1; $p < 0.05$). Only Fe concentrations significantly varied under the *Ficus capensis* dominant tree canopy species. High concentrations of Cu and Mn were observed only for *Dichrostachys cinerea* and *Gmelina arborea* dominant tree canopy species, respectively (Fig. 4c–d; Table S1).

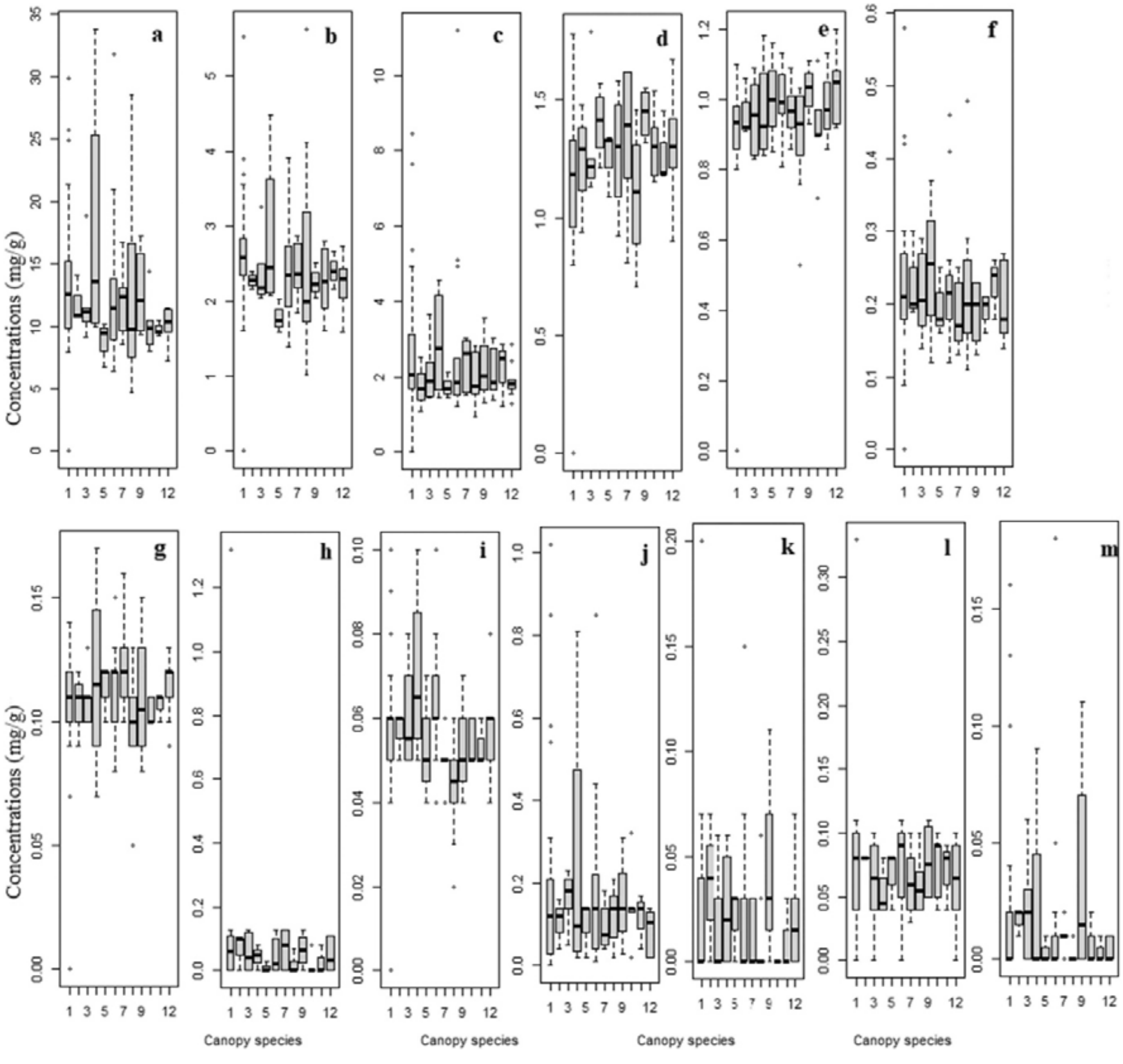

Fig. 4. Boxplot of heavy metal parameters variation between the dominant tree canopy species stands: a) Fe; b) Ti; c) Copper; d) Mn; e) Ba; f) Zr; g) Rb; h) Cr; i) Sr; j) Zn; k) Sn; l) Sb; m) Pb. Dominant tree canopy species: 1. *Acacia polyacantha*, 2. *Acacia sieberiana*, 3. *Bauhinia petersiana,* 4. *Dichrostachys cinerea,* 5. *Ficus capensis,* 6. *Ficus craterostoma,* 7. *Gmelina arborea,* 8. *Rhus longipes,* 9. *Syzygium guineense,* 10. *Terminalia mollis;* 11. Mixed species with *Acacia polyacantha;* 12. Mixed species with *Bauhinia petersiana.*

3.2. Species richness, diversity and dominance metrics variations and influence of the dominant tree canopy species on their recruitment

Across layers, the weighted Linear Mixed Models indicated that canopy species significantly affect recruitment patterns in the tree and herb layers but have a limited impact on the grass layer (Table 2, Table 3, Table 4). The results of the weighted linear mixed models revealed that across layers, the models demonstrated the ability to explain variation in richness, abundance, diversity and dominance indices (Table 2, Table 3, Table 4). The performance of the weighted Linear Mixed Models varied across the tree, herb, and grass layers in terms of species richness, abundance, diversity and dominance metrics.

For the tree layer, the models showed that all metrics were shaped by both Marginal R^2 (fixed effects) with values varying between 15.5 and 28.8 and the Conditional R^2 (random effects representing block-level differences) with values between 24.8 and 38.5 (Table 1). The random effects, remarkably the block-level variability, played a more prominent role in the tree layer (ICC = 26 %), reflecting spatial heterogeneity in recruitment. The models performed moderately well, especially in explaining species richness. Among the fixed effects, *Ficus capensis* (Estimate = 2.59, 95 % CI: 0.71–4.48, $p = 0.008$) and *Ficus craterostoma* (Estimate = 1.13, 95 % CI: 0.11–2.16, $p = 0.030$) significantly increased richness. *Gmelina arborea* (Estimate = 31.23, 95 % CI: 18.32–44.14, $p < 0.001$) and *Bauhinia petersiana* (Estimate = 14.07, 95 % CI: 1.04–27.10, $p < 0.035$) had a strong effect on tree abundance. *Ficus craterostoma* significantly increased Shannon-Wiener diversity (Estimate = 0.39, 95 % CI: 0.09–0.70, $p = 0.012$) and Simpson diversity (Estimate = 0.18, 95 % CI: 0.02–0.34, $p = 0.030$), indicating its role in supporting diverse and dominance tree communities.

For the herb layer, the weighted Linear Mixed Models indicated that marginal R^2 (fixed effects) explained 7.1–9.9 % of the variance, while conditional R^2 (random effects) added little to the models, with values between 1.6–12.2 (Table 3). This suggests that fixed impacts, particularly the dominant tree canopy species, played a more critical role in explaining variation in herbaceous recruitment. However, the random effect of the block was negligible (ICC = 0), indicating minimal spatial heterogeneity at this scale. However, specific canopy species still showed notable fixed effects. *Ficus capensis* significantly increased species richness (Estimate = 2.12, 95 % CI: 0.94–3.31, $p = 0.003$), while *Gmelina arborea* enhanced herb abundance (Estimate = 19.67, 95 % CI: 11.02–28.32, $p < 0.001$). For herbaceous diversity, *Ficus craterostoma* exhibited positive effects on Shannon-Wiener diversity (Estimate = 0.42, 95 % CI: 0.12–0.72, $p = 0.005$) and Simpson diversity (Estimate = 0.21, 95 % CI: 0.03–0.38, $p = 0.022$), implying that it promotes a more even and varied herb layer.

The models showed very low explanatory power for the grass layer, with marginal R^2 values around 1.0–1.1 % across all metrics (Table 4). Conditional R^2 and random effects were nearly negligible, with ICC values approximating 0, reflecting little spatial variability across blocks. None of the fixed effects were statistically significant for richness, abundance, or diversity metrics. No canopy species had significant fixed impacts on species richness or abundance in the grass layer (Table 3). However, *Rhus longipes* exhibited a marginal impact on the Shannon-Wiener index (Estimate = 0.57, 95 % CI: −0.09–1.23, $p = 0.088$).

3.3. Influence of soil variable on species richness, abundance and diversity metrics across vegetation layers growing under the dominant tree canopy species

3.3.1. Random forest models performance

The Random Forest analysis revealed that soil variables influenced the species metrics across different layers (Table 5). The table summarises the performance metrics—RMSE, R^2, and MAE—achieved for each combination of vegetation layer (Tree, Grass, Herbs) and response variable (Richness,

Table 2. Influence of dominant tree on the tree species composition within the tailing dam 10 in the Copperbelt province of Zambia.

Predictors	**Species richness**				**Abundance**				**Shannon-Wiener index**			**Simpson index**		
	Estimates	**CI**	***P***	**Post-hoc**	**Estimates**	**CI**	***P***	**Post-hoc**	**Estimates**	**CI**	***P***	**Estimates**	**CI**	***P_values***
Intercept	2.84	1.75 – 3.93	**< 0.001**		10.54	3.70 – 17.39	**0.003**		0.7	0.41 – 0.99	**< 0.001**	0.39	0.25 – 0.53	**< 0.001**
Fixed effects														
Acacia polyacantha	1.13	0.10 – 2.16	0.201	ab	2.38	−08.18 – 8.50	0.578	abcd	0.40	0.09 – 0.80	0.112	0.25	0.06 – 0.38	0.08
Acacia sieberiana	0.05	−1.86 – 1.96	0.961	ab	−3.59	−21.13 – 13.95	0.685	abcd	−0.05	−0.61 – 0.52	0.873	−0.07	−0.37 – 0.23	0.649
Bauhinia petersiana	−0.18	−1.60 – 1.24	0.806	ab	14.07	1.04 – 27.10	**0.035**	abcd	−0.22	−0.64 – 0.21	0.312	−0.14	−0.36 – 0.08	0.207
Dichrostachys cinerea	1.65	−0.02 – 3.32	0.053	ab	11.96	−3.41 – 27.34	0.126	abcd	0.43	−0.07 – 0.93	0.089	0.19	−0.07 – 0.45	0.155
Ficus capensis	2.59	0.71 – 4.48	**0.008**	b	15.63	−1.74 – 33.00	0.077	abcd	0.34	−0.22 – 0.90	0.225	0.13	−0.16 – 0.43	0.373
Ficus craterostoma	1.13	0.11 – 2.16	**0.03**	ab	−0.75	−10.18 – 8.67	0.874	cd	0.39	0.09 – 0.70	**0.012**	0.18	0.02 – 0.34	**0.03**
Gmelina arborea	1.4	−0.00 – 2.80	0.050	ab	31.23	18.32 – 44.14	**< 0.001**	ab	0.25	−0.16 – 0.67	0.229	0.13	−0.09 – 0.35	0.254
Mixed species with *A. polyacantha*	0.96	−0.26 – 2.19	0.122	ab	3.18	−8.05 – 14.41	0.574	ac	0.33	−0.03 – 0.70	0.072	0.17	−0.02 – 0.36	0.087
Mixed species with *B. petersiana*	1.28	−0.42 – 2.97	0.137	ab	18.25	2.72 – 33.79	0.22	abcd	0.12	−0.38 – 0.63	0.625	0.04	−0.22 – 0.31	0.743
Rhus longipes	1.33	−0.19 – 2.85	0.085	ab	10.58	−3.40 – 24.56	0.136	abcd	0.11	−0.35 – 0.56	0.642	0	−0.23 –0.24	0.979
Syzygium guineense	1.25	−0.64 – 3.14	0.193	ab	11.98	−5.43 – 29.38	0.175	abcd	0.18	−0.38 –0.74	0.522	0.08	−0.21 – 0.38	0.579
Terminalia mollis	−0.63	−1.81– 0,55	0.294	a	−3.82	−14.62 – 6.98	0.483	ac	−0.24	−0.59 – 011	0.182	−0.14	- 0.32 – 0.05	0.138
Random Effects														
σ^2	2.39				203.96				0.21			0.06		

τ_{00}	0.85_{Bloc}			16.83_{Bloc}			0.05_{Bloc}			0.01_{Bloc}		
ICC	0.26			0.08			0.2			0.16		
N	4_{Bloc}			4_{Bloc}			4_{Bloc}			4_{Bloc}		
Observations	94			94			94			94		
Marginal R^2/Conditional R^2	0.167/0.385			0.288/0.343			0.155 / 0.327			0.154 / 0.288		
AIC	300.078			721.393			160.496			54.43		
ANOVA Mixed Models	Chisq	Df	Pr(>Chisq)	Chisq	Df	Pr(>Chisq)	**Chisq**	Df	Pr(>Chisq)	Chisq	Df	Pr(>Chisq)
	23.049	11	**0.017**	39.937	11	**3.67E−05**	**19.472**	11	0.053	18.273	11	0.075

Table 3. Influence of dominant tree on the herbs species composition within the tailing dam 10 in the Copperbelt province of Zambia.

Predictors	**Richness**				**Abundance**			**Shannon**				**Simpson**			
	Estimates	**CI**	***P***	**Post-hoc**	**Estimates**	**CI**	***P***	**Estimates**	**CI**	***P***	**Post-hoc**	**Estimates**	**CI**	***P***	**Post-hoc**
Fixed effects															
Intercept	3.01	1.85 – 4.16	**< 0.001**		0.38	−0.11 – 0.86	0.125	**0.58**	0.21 – 0.94	**0.002**		0.31	0.10 – 0.53	**0.004**	
Acacia sieberiana	−1.13	−3.83 – 1.58	0.409	ac	−0.23	−1.40 – 0.94	0.696	−0.23	−1.09 – 0.64	0.606	ab	−0.11	−0.62 – 0.40	0.672	a
Bauhinia petersiana	−0.72	−2.43 – 0.99	0.406	ab	−0.25	−0.99 – 0.48	0.495	−0.06	−0.61 – 0.48	0.817	ab	−0.01	−0.33 – 0.31	0.957	a
Dichrostachys cinerea	−1.7	−3.48 – 0.09	0.062	b	0.35	−0.42 – 1.12	0.372	−0.46	−1.03 – 0.11	0.114	ac	−0.23	−0.56 – 0.11	0.178	a
Ficus capensis	−1.12	−2.97 – 0.72	0.228	ab	−0.01	−0.80 – 0.79	0.989	−0.29	−0.88 – 0.30	0.325	ab	−0.13	−0.48 – 0.21	0.443	a
Ficus craterostoma	−0.23	−1.53 – 1.07	0.728	ab	−0.08	−0.64 – 0.48	0.774	0.02	−0.39 – 0.44	0.921	ab	0.03	−0.21 – 0.28	0.791	a

Gmelina arborea	−0.45	−2.00 – 1.09	0.56	ab	−0.15	−0.81 – 0.52	0.662		−0.18	−0.67 – 0.32	0.478	ab	−0.09	−0.38 – 0.20	0.554	a
Mixedspecies with Acacia polyacantha	−0.68	−1.98 – 0.61	0.297	b	0.07	−0.48 – 0.62	0.802		−0.22	−0.63 – 0.20	0.303	a	−0.09	−0.33 – 0.15	0.476	a
Mixedspecies with Bauhinia	−1.2	−2.71 – 0.31	0.117	b	−0.21	−0.85 – 0.43	0.51		−0.26	−0.74 – 0.22	0.287	ab	−0.09	−0.37 – 0.19	0.511	a
Rhus longipes	−0.21	−1.52 – 1.11	0.754	ab	−0.08	−0.64 – 0.49	0.789		0.05	−0.37 – 0.47	0.801	ab	0.04	−0.20 – 0.29	0.724	a
Syzygium guineense	1.56	0.11 – 3.02	**0.036**	a	0.64	0.02 – 1.27	**0.045**		0.43	−0.04 – 0.90	0.07	b	0.19	−0.09 – 0.46	0.18	a
Terminalia mollis	0.59	−0.65 – 1.83	0.343	ab	−0.01	−0.54 – 0.51	0.956		0.22	−0.17 – 0.62	0.267	bc	0.15	−0.08 – 0.38	0.196	a
Random Effects																
σ2	21.34				3.98				2.19				0.75			
τ00	0.08 Bloc				0.00 Bloc				0.00 Bloc				0.00 Bloc			
ICC	0								0				0			
N	4 Bloc				4 Bloc				4 Bloc				4 Bloc			
Observations	97				97				97				97			
Marginal R2 / Conditional R2	0.099 / 0.122				0.0970 / NA				0.085 / 0.016				0.071 / 0.022			
AIC	373.812				229.173				179.499				88.472			
ANOVA Mixed Models	Chisq	Df	Pr(>Chisq)		Chisq	Df	Pr(>Chisq)		Chisq	Df	Pr(>Chisq)		Chisq	Df	Pr(>Chisq)	
	35.151	11	**0.00023**		14.963	11	0.1842		**28.315**	11	**0.002896**		21.454	11	**0.02896**	

Table 4. Influence of dominant tree on grass species within the tailing dam 10 in the Copperbelt province of Zambia.

	Richness			**Abundance**			**Shannon**			**Simpson**		
Predictors	**Estimates**	**CI**	**p**	**Estimates**	**CI**	**p**	**Estimates**	**CI**	***P***	**Estimates**	**CI**	***P***
Fixed Effects												
(Intercept)	1.84	0.33 – 3.34	**0.018**	0.43	−0.49 – 1.34	0.355	0.32	−0.26 – 0.89	0.276	0.21	−0.11 – 0.53	0.202
Bauhinia petersiana	0.96	−1.71 – 3.64	0.474	0.73	−0.88 – 2.34	0.368	0.31	−0.71 – 1.33	0.541	0.17	−0.40 – 0.73	0.557
Dichrostachys cinerea	−0.09	−2.61 – 2.43	0.945	−0.28	−1.80 – 1.23	0.708	0.19	−0.77 – 1.15	0.693	0.15	−0.38 – 0.69	0.563
Ficus capensis	0.16	−5.00 – 5.33	0.95	−0.14	−3.24 – 2.97	0.93	0.37	−1.60 – 2.34	0.706	0.29	−0.81 – 1.39	0.598
Ficus craterostoma	0.42	−1.28 – 2.12	0.623	0.04	−0.99 – 1.06	0.943	0.11	−0.54 – 0.76	0.743	0.04	−0.32 – 0.40	0.834
Gmelina arborea	0.88	−1.27 – 3.02	0.416	0.16	−1.12 – 1.45	0.801	0.35	−0.47 – 1.16	0.401	0.18	−0.28 – 0.63	0.441
Mixedspecies with *Acacia polyacantha*	0.4	−1.31 – 2.11	0.641	−0.03	−1.06 – 1.00	0.957	0.2	−0.46 – 0.85	0.549	0.1	−0.26 – 0.47	0.576
Mixedspecies with *Bauhinia*	−0.12	−2.13 – 1.88	0.903	−0.26	−1.46 – 0.95	0.674	−0.08	−0.85 – 0.68	0.825	−0.09	−0.52 – 0.33	0.658
Rhus longipes	1.56	−0.17 – 3.29	0.076	−0.14	−1.18 – 0.90	0.789	0.57	−0.09 – 1.23	0.088	0.29	−0.07 – 0.66	0.114
Syzygium guineense	0.94	−0.89 – 2.77	0.308	0.91	−0.19 – 2.01	0.104	0.23	−0.47 – 0.92	0.52	0.06	−0.33 – 0.45	0.747
Terminalia mollis	1.24	−0.38 – 2.85	0.13	0.47	−0.50 – 1.45	0.337	0.5	−0.12 – 1.11	0.11	0.26	−0.08 – 0.61	0.127
Random Effects												
σ2	24.37			8.75			3.54			1.1		
τ00	0.00 Bloc			0.02 Bloc			0.00 Bloc			0.00 Bloc		
ICC				0								
N	4 Bloc			4 Bloc			4 Bloc			4 Bloc		
Observations	71			71			71			71		
Marginal R2 / Conditional R2	0.011 / NA			0.010 / 0.012			0.011 / NA			0.011 / NA		
AIC	281.345			220.801			165.644			95.226		
ANOVA Mixed Models	Chisq	Df	Pr(>Chisq)	Chisq	Df	Pr(>Chisq)	Chisq	Df	Pr(>Chisq)	Chisq	Df	Pr(>Chisq)
	11.43309	10	0.324786	12.82244	10	0.23377	11.8508	10	0.295166	12.85384	10	0.231959

Abundance, Shannon, and Simpson). For the tree layer, richness had an RMSE of 1.486, a variance explained R^2 of 40.5 %, and an MAE of 1.264. The tree abundance response showed a higher RMSE (14.594) but explained 21.7 % of the variance. Shannon and Simpson metrics for this layer revealed lower RMSE and MAE values, with R^2 values of 32.9 % and 21.3 %, respectively. Species richness was shown to have an RMSE of 1.342 for grass layers, accounting for 13.9 % of the variance. The RMSE values for the abundance and Shannon-Wiener index were 0.645 and 0.491, respectively, and they explained 18.7 % and 16.0 % of the variations. On the contrary, in this layer, the Simpson index showed the highest variation, with a variance of 22.9 %. Species richness in the herbs layer had an RMSE of 1.431, meaning that 6.7 % of its variance could be explained. The Shannon-Wiener and Simpson indices exhibited R^2 values of 11.8 % and 9.0 %, respectively, whereas abundance had an RMSE of 0.571 and explained 6.8 % of its variance.

Table 5. Performance metrics statistics for species response variables across vegetation layers.

Layers	Response	RMSE	R^2	MAE
Tree	Richness	1.486	0.405	1.264
	Abundance	14.594	0.217	11.335
	Shannon	0.482	0.329	0.405
	Simpson	0.257	0.213	0.223
Grass	Richness	1.342	0.139	1.152
	Abundance	0.645	0.187	0.516
	Shannon	0.491	0.160	0.418
	Simpson	0.282	0.229	0.251
Herbs	Richness	1.431	0.067	1.208
	Abundance	0.571	0.068	0.390
	Shannon	0.437	0.118	0.380
	Simpson	0.245	0.090	0.224

3.3.2. Influence of soil variable on species richness across tree, grass and herb Layers

The results of the Random Forest analysis showed that soil properties influence species richness across tree, grass, and herb layers. This influence varies from one soil variable to another and across vegetation layers (Table 6; Appendices S2-S4).

Concerning the influence of soil properties on species richness, the relative importance of the impact of soil variables differs across tree, grass and herb layers (Table 6; Appendices S2–S4). Indeed, heavy metal iron (Fe) was the only soil variable that significantly influenced the specific richness of the tree layer (MDA = 10.44 & *p* accuracy = 0.01; MDG = 14.94 & *p* Gini = 0.04; Importance value = 16.86). Tree species richness increases significantly with Fe content up to a certain threshold (0.15), where it begins to decrease (Appendix S2). The soil textural variable sand content also showed a meaningful influence, even if not significantly (Importance value = 15.45). It was found that tree species richness increased rapidly when sand content was higher than 90 % (Appendix S2). The heavy metal Zinc (Zn), nutrient Calcium (Ca) and the heavy metal Manganese (Mn) are the third, fourth and fifth soil variables with a high influence on tree species richness, respectively (Tian et al., 2015) (Table 6). The heavy metals Zn and Mn were found to increase tree species richness at lower concentrations, while tree species richness increased with soil concentration of Ca (Appendix S2).

Table 6. Relative importance and performance tests statistics for random forest analysis of the soil variable influence on species richness of the different layers (tree, grass and herb).

Variables	**Trees layer**					**Grass layer**					**Herbs layer**				
	MDA	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**
Fe	**10.44**	**14.94**	**16.86**	**0.01**	**0.04**	−0.16	3.90	3.92	0.54	0.57	−1.50	8.88	8.04	0.96	0.19
Sand	0.62	14.45	15.45	0.46	0.16	−2.93	2.25	2.37	0.89	1.00	0.78	10.38	9.92	0.44	0.12
Zn	3.07	15.65	15.19	0.17	0.20	1.85	5.28	5.90	0.19	0.20	2.04	5.81	5.95	0.36	0.77
Ca	−1.32	13.35	14.67	0.72	0.50	−0.97	4.33	4.09	0.59	0.57	**8.37**	**11.36**	**11.65**	**0.01**	**0.04**
Mn	3.92	15.29	14.56	0.06	0.25	−0.92	4.29	3.98	0.70	0.58	3.23	8.32	8.43	0.13	0.43
Na	4.08	13.00	12.76	0.09	0.50	−1.29	6.86	7.56	0.66	0.19	−0.47	9.42	9.28	0.54	0.24
Cu	2.46	11.12	10.53	0.21	0.64	−0.33	5.10	4.92	0.51	0.43	−0.46	8.49	8.85	0.61	0.38
Zr	1.11	9.55	10.20	0.41	0.92	−0.34	5.07	5.25	0.55	0.31	−1.03	4.22	4.67	0.76	1.00
K	−3.75	11.12	10.18	1.00	0.67	0.21	5.01	5.17	0.47	0.38	−0.25	7.39	8.45	0.60	0.50
Sr	−2.40	9.46	9.83	0.96	0.72	−4.68	3.09	3.14	0.99	0.77	1.99	5.45	5.42	0.24	0.79
Rb	2.43	8.76	9.69	0.27	0.72	−0.16	3.60	3.35	0.50	0.55	−2.81	4.66	4.49	0.99	0.92
SOC	−0.80	9.84	9.66	0.70	0.81	−1.91	4.07	4.27	0.76	0.64	2.99	6.66	6.59	0.12	0.62
Ba	0.40	10.17	9.53	0.42	0.91	0.67	4.36	4.25	0.35	0.52	−1.04	5.48	6.14	0.69	0.94
Ti	2.54	9.31	9.24	0.29	0.87	−2.04	4.55	4.41	0.85	0.43	1.30	9.24	8.43	0.43	0.22
Mg	3.46	9.09	8.42	0.07	0.74	−1.25	4.05	4.15	0.61	0.36	1.02	9.16	10.28	0.25	0.13
Clay	−0.02	7.16	7.69	0.45	0.69	−2.29	2.63	2.62	0.80	0.59	−1.96	4.30	4.49	0.84	0.63
Sb	−1.65	6.79	7.68	0.67	1.00	0.62	5.76	5.33	0.26	0.13	−1.87	5.61	5.07	0.83	0.79
pH	2.80	7.52	7.14	0.21	0.80	−1.40	2.90	3.26	0.73	0.68	0.56	4.92	4.49	0.37	0.52
Silt	0.84	6.82	6.65	0.47	0.99	−2.46	3.13	2.55	0.88	0.79	2.68	9.60	9.20	0.20	0.16
Sn	−0.66	5.55	5.19	0.56	0.45	0.75	2.58	2.60	0.29	0.27	−1.66	2.47	2.31	0.78	0.78
Cr	−1.81	4.82	4.73	0.90	0.86	0.05	2.10	2.34	0.35	0.68	−1.39	2.82	2.73	0.82	0.85
Pb	1.03	4.69	4.11	0.25	0.68	−0.50	3.40	3.37	0.66	0.26	−3.46	2.11	1.91	0.95	0.96

Contrary to tree and herb layers, no significant influence of soil on tree species richness was found for grass (Table 6). For grass, the five soil variables with relative importance higher than 5.00 where Na (Importance value = 7.56), Zn (Importance value = 5.90), Zr (Importance value = 5.25), Sb (Importance value = 5.33) and K (Importance value = 5.01). Grass species richness decreases with higher concentrations of heavy metals Zn and Sb, as well as soil nutrient K. However, grass species richness increased with soil Na Concentrations while it fluctuated with the increase of Zr concentrations (Appendix S3).

Regarding the herb layer, it was found that soil nutrients variables Ca (MDA = 8.37 & *p* accuracy = 0.01; MDG = 11.36 & *p* Gini = 0.04; Importance value = 11.65), Na (Importance value = 9.42), Mg (Importance value = 9.16), and soil texture variables sand (Importance value = 9.92), and silt (Importance value = 9.60) were the five variables influencing the most herb richness with the Ca influence being significant (Table 6). Herb species richness negatively responded to the increasing proportion of sand, suggesting a decline in diversity in sandier soils. Conversely, soil nutrients Na and Ca positively influenced herb richness, with notable increases as their concentrations rose. Silt and Mg have mixed effects, with slight variations in herb richness at different concentration levels, indicating a less consistent influence (Appendix S4).

3.3.3. Influence of soil variables on species abundance across tree, grass, and herb layers

Like metric species richness, it was found that soil variables influence species abundance across vegetation layers even though the soil variables influencing it were not the same (Table 7). That influence varies from one variable to another across the vegetation layers (Table 7; Appendices S5-S7).

The abundance of the tree layer was found to be influenced mainly by soil variables K (Importance value = 1837.08), Sb (Importance value =1698.84), pH (Importance value = 1389.62), Cu (Importance value = 938.84) and Cr (Importance value = 818.23). Specifically, the random forest analyses showed that K has a non-linear positive influence on the abundance of tree species, with a noticeable increase when K levels reach a particular threshold. The Sb on its side showed a complex relationship with tree species abundance. The abundance of trees is strongly positively influenced by the soil pH, increasing gradually as the pH becomes less acidic. However, there are trends in heavy metals Cu and Cr where the abundance of trees decreases initially as their concentrations rise before stabilising (Appendix S5).

Regarding the abundance of grass layer, it was found that Mg (MDA = 8.42 & *p* accuracy = 0.04; MDG = 3.92 & *p* Gini = 0.04; Importance value = 3.77) and Fe (MDA = 5.88 & *p* accuracy = 0.04; MDG = 3.01 & *p* Gini = 0.02; Importance value = 3.36) were the most significant influencing soil variables (Table 7). These soil variables were followed by Ti (Importance value = 2.51), Ca (Importance value = 2.31), and Mn (Importance value = 2.09), respectively. After the random forest analysis, the specific analysis of each soil variable showed that Mg appears to have a positive marked effect, with the abundance of grass increasing significantly as Mg concentrations above a particular threshold. Conversely, heavy metals Fe and Ti showed an inverse trend, with the abundance of grass decreasing significantly as their concentrations rose. While Mn exhibits a negative trend with a sharp decline in the abundance of grass at high levels, Ca has a complex relationship with graminoids abundance (Zhang et al., 2023) (Appendix S6).

Furthermore, the soil variables influencing the abundance of the herb layer were Ba (Importance value = 2.44), Cu (Importance value = 2.07), Na (Importance value = 1.51), Ca (Importance value = 1.44), Ti (Importance value = 1.36) and Mg (Importance value = 1.27). The analyses of how various soil factors affect the abundance of herbs showed that Ba and Cu were negative, showing that when these latter two increase, the presence of herbs decreases. Conversely, Ca and Na were beneficial, indicating an

Table 7. Relative importance and performance tests statistics for random forest analysis of the soil variable influence on species abundance of the different layers (Tree, grass and herbs).

Variables	**Trees layer**					**Grass layer**					**Herbs layer**				
	MDA	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**p.Accuracy**	**p.Gini**
K	4.10	2039.41	1837.08	0.14	0.06	0.31	0.95	1.12	0.62	0.67	2.49	0.69	0.75	0.51	0.74
Sb	0.48	1600.91	1698.84	0.41	0.10	2.73	1.48	1.38	0.14	0.22	2.50	0.83	0.88	0.20	0.47
pH	4.30	1316.19	1389.62	0.08	0.15	0.10	0.45	0.47	0.59	0.97	−0.87	0.52	0.50	0.93	0.74
Cu	2.57	860.86	938.84	0.30	0.51	−0.89	1.19	1.15	0.76	0.46	3.98	2.16	2.07	0.12	0.06
Cr	2.46	926.85	818.23	0.16	0.12	1.45	0.37	0.36	0.29	0.94	2.38	0.54	0.79	0.19	0.36
Sr	−0.85	758.61	798.10	0.77	0.50	−1.60	1.29	1.16	0.84	0.28	1.55	0.58	0.63	0.37	0.67
Ti	1.00	747.78	757.45	0.72	0.72	2.22	2.47	2.51	0.22	0.06	1.22	1.06	1.36	0.70	0.41
Ca	−1.49	757.64	741.29	0.87	0.62	1.62	2.16	2.31	0.26	0.20	3.65	1.44	1.21	0.13	0.26
Zr	2.39	707.22	694.43	0.19	0.68	0.39	1.04	1.06	0.46	0.64	2.93	1.04	0.95	0.21	0.31
Ba	−1.51	584.51	686.04	0.81	0.85	0.29	0.97	1.10	0.54	0.70	4.70	2.15	2.43	0.09	0.19
Zn	2.50	667.21	685.72	0.36	0.72	0.44	1.29	1.18	0.56	0.35	3.45	0.61	0.52	0.21	0.74
clay	0.73	728.62	683.34	0.37	0.37	−1.74	0.30	0.34	0.84	1.00	−1.10	0.59	0.50	0.68	0.53
Mn	−0.77	656.53	673.53	0.87	0.69	6.59	2.33	2.09	0.07	0.19	−1.75	0.58	0.57	0.89	0.76
Rb	2.88	870.08	669.32	0.30	0.34	1.88	0.98	0.87	0.37	0.48	2.14	0.51	0.53	0.59	0.80
sand	0.87	564.02	602.08	0.50	0.76	−0.75	1.03	1.05	0.70	0.54	−2.60	0.35	0.37	0.94	0.96
silt	0.90	595.37	578.31	0.43	0.66	−1.90	0.84	0.86	0.78	0.69	−0.19	0.30	0.35	0.69	0.98
Fe	2.17	527.59	577.00	0.60	0.96	**5.88**	**3.01**	**3.36**	**0.04**	**0.02**	3.48	0.77	0.72	0.37	0.55
Mg	0.93	463.46	498.04	0.48	0.84	**8.42**	**3.92**	**3.77**	**0.04**	**0.04**	2.58	1.38	1.27	0.18	0.11
SOC	1.56	432.97	420.89	0.48	0.97	2.40	1.28	0.99	0.18	0.50	1.58	0.89	0.76	0.46	0.54
Na	2.34	437.12	394.32	0.13	0.98	−2.19	0.75	0.76	0.84	0.85	−1.59	1.94	1.51	0.82	0.17
Pb	−2.92	305.57	344.42	0.96	0.54	−1.95	0.20	0.17	0.90	0.97	0.77	0.42	0.64	0.49	0.36
Sn	0.10	183.22	185.59	0.36	0.85	−0.47	0.39	0.40	0.56	0.51	0.35	0.13	0.16	0.37	0.87

increase in the herb's wealth with more significant concentrations of these elements. On the other hand, Mg has a generally positive effect that increases herbaceous wealth as concentration increases, but Ti has the opposite effect, causing a significant decline in herb abundance (Appendix S7).

3.3.4. Influence of soil variable on diversity metrics across tree, grass, and herb layers

Random forest analysis reveals that Na was the soil property influencing diversity indices for all vegetation layers, significantly affecting tree and grass layers (Table 8, Table 9). However, the importance of soil properties varied from layer to diversity metrics. For the Shannon-Wiener index, the most influential variables were Zn (Importance value = 1.06), Mn (Importance value = 1.05), sand (Importance value = 1.04) and Zr (Importance value = 0.98). In contrast, the Simpson index was mainly influenced by Sr (Importance value = 0.30), Zr (Importance value = 0.29), Ti (Importance value = 0.28) and Mn (Importance value = 0.24). Na showed a decreasing relationship with the Shannon-Wiener index, indicating that increasing its concentration reduces diversity. Zn and Mn, on the other hand, showed a more complex relationship, with fluctuations that could reflect variable effects depending on their levels. Sand and Zr also influenced tree diversity (Appendix S8). Higher Na concentrations were also associated with lower tree diversity for the tree Simpson index, while Sr and Ti contribute to an increase in this index at certain levels. Mn and Zr, on the other hand, show more complex relationships with the Simpson index (Appendix S11).

Similar trends were observed for herbs layers, both for the Shannon-Wiener index (Table 8) and the Simpson index (Table 9). The Shannon-Wiener index for grasses was influenced by Ca (Importance value = 0.88), Cu (Importance value = 0.84), Zr (Importance value = 0.75), Zn (Importance value = 0.71) and Fe (Importance value = 0.63). The Simpson index for grasses is affected by Ca (Importance value = 0.33), Cu (Importance value = 0.29), silt (Importance value = 0.23), Zn (Importance value = 0.23), Fe (Importance value = 0.21), Zr (Importance value = 0.21) and Ti (Importance value = 0.21). Na has a marked decreasing effect on the Shannon-Wiener index of grasses. Ca showed significant fluctuations. Cu and Zn have an inverse relationship with the Shannon-Wiener index, their increase being associated with a decrease in the index. At the same time, Fe and Zr showed more stable trends (Appendix S9). Regarding the Simpson index of grasses, high Na concentrations were also associated with decreased diversity. Zn and Cu also showed decreasing trends, while Ti and Zr showed more complex variations, with peaks and fluctuations that could reflect specific effects on the Simpson index. Silt and Fe exert a more stable effect, with moderate variations in the Simpson index (Appendix S12).

Unlike the tree and grass layers, grass species diversity was strongly influenced by Ba (Table 8, Table 9). In addition to Ba, soil properties such as Na (Importance value = 1.12), Mn (Importance value = 0.95), Ca (Importance value = 0.85), K (Importance value = 0.68) and SOC, 0.61) also influence the Shannon-Wiener index of grasses (Table 8). For the Simpson index, the most influential properties were Na (Importance value = 0.34), Mn (Importance value = 0.30), Ca (Importance value = 0.27), SOC (Importance value = 0.21), Sr (Importance value = 0.21) and Rb (Importance value = 0.21). Ba and Na favor increased diversity for grasses at higher concentrations, while Mn and Ca showed a downward trend, which could limit certain species. K and SOC showed variable effects, marked by peaks and troughs (Appendix S10). Finally, high concentrations of Ca were associated with a decrease in the Simpson index of grasses, while SOC was positively correlated with this index. In contrast, heavy metals such as Sr and Rb showed opposite trends, potentially reducing herbaceous diversity around dominant trees in tailings dam zones where these elements are present (Appendix S13).

Table 8. Relative importance and performance tests statistics for random forest analysis of the soil variable influence on Shannon-Wiener index of the different layers (Tree, grass and herbs).

Variables	**Trees layer**					**Grass layer**					**Herbs layer**				
	MDA	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**p.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**p.Accuracy**	**p.Gini**
Na	**8.67**	**1.77**	**1.77**	**0.02**	**0.10**	**5.68**	**1.37**	**1.37**	**0.04**	**0.04**	−0.25	0.44	1.12	0.58	0.98
Zn	3.00	1.08	1.06	0.16	0.29	−1.28	0.64	0.71	0.73	0.37	−3.35	0.49	0.57	0.96	0.86
Mn	−1.20	0.96	1.05	0.66	0.73	−1.70	0.51	0.52	0.74	0.79	−0.21	0.56	0.95	0.50	0.57
sand	0.81	1.00	1.04	0.37	0.35	−0.23	0.47	0.49	0.49	0.63	−1.72	0.42	0.60	0.85	0.99
Zr	0.24	1.06	0.98	0.47	0.40	0.30	0.67	0.75	0.50	0.47	−3.42	0.46	0.58	0.95	0.20
Ba	−0.94	0.86	0.93	0.55	0.85	−1.23	0.50	0.45	0.66	0.81	**4.47**	**0.66**	**1.11**	**0.04**	**0.06**
Cu	0.35	0.87	0.92	0.32	0.70	4.03	0.83	0.84	0.08	0.19	0.54	1.10	0.61	0.33	0.09
Sr	0.79	0.88	0.91	0.32	0.49	−2.38	0.42	0.48	0.85	0.81	−2.88	0.51	0.60	0.96	0.78
Ti	1.41	0.88	0.88	0.33	0.68	−2.50	0.59	0.56	0.90	0.50	−2.89	0.49	0.48	0.94	0.71
SOC	1.46	0.94	0.85	0.26	0.66	−3.83	0.44	0.41	0.96	0.89	−2.58	0.66	0.61	0.89	0.38
Fe	4.39	0.76	0.80	0.15	0.88	−0.58	0.56	0.63	0.58	0.50	0.11	0.55	0.48	0.59	0.57
K	0.86	0.77	0.77	0.47	0.92	1.26	0.57	0.59	0.26	0.61	−0.10	0.39	0.68	0.50	0.74
Ca	−1.85	0.69	0.70	0.78	1.00	1.15	0.86	0.88	0.31	0.22	3.47	0.99	0.85	0.16	0.12
Sb	−0.73	0.64	0.67	0.46	0.97	0.33	0.51	0.56	0.37	0.63	−1.81	0.34	0.48	0.78	0.52
clay	1.65	0.60	0.66	0.17	0.62	−2.62	0.33	0.35	0.92	0.79	3.21	2.54	0.35	0.16	0.13

Pb	1.47	0.61	0.64	0.18	0.18	−0.02	0.36	0.32	0.34	0.26	−0.48	0.63	0.30	0.59	0.36
Mg	−1.69	0.58	0.64	0.63	0.98	−0.63	0.47	0.44	0.51	0.64	−0.14	0.30	0.47	0.46	0.71
Sn	0.40	0.66	0.64	0.33	0.15	0.41	0.25	0.23	0.26	0.44	−2.51	0.60	0.47	0.88	0.77
silt	0.36	0.63	0.60	0.47	0.98	2.12	0.63	0.62	0.26	0.38	−1.97	0.62	0.56	0.88	0.65
Rb	−0.02	0.59	0.59	0.59	0.97	−3.17	0.40	0.38	0.96	0.82	−1.28	0.62	0.59	0.73	0.74
Cr	−2.57	0.58	0.59	0.90	0.58	−0.50	0.34	0.32	0.50	0.59	1.48	0.54	0.33	0.23	0.82
pH	−0.67	0.56	0.50	0.58	0.97	−1.44	0.45	0.43	0.78	0.59	−0.70	0.55	0.54	0.59	0.33

Table 9. Relative importance and performance tests statistics for random forest analysis of the variable soil influence on Simpson index of the different layers (Tree, grass and herbs).

Variables	**Trees layer**					**Grass layer**					**Herbs layer**				
	MDA	**MDG**	**Importance**	**P.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**p.Accuracy**	**p.Gini**	**MDA**	**MDG**	**Importance**	**p.Accuracy**	**p.Gini**
Na	**5.74**	**0.36**	**0.37**	**0.02**	**0.21**	**6.45**	**0.43**	**0.40**	**0.01**	**0.01**	2.55	0.35	0.34	0.16	0.11
Sr	1.59	0.26	0.30	0.18	0.36	−1.59	0.14	0.14	0.59	0.83	0.55	0.19	0.21	0.34	0.52
Zr	0.85	0.29	0.29	0.25	0.35	−0.08	0.20	0.21	0.43	0.55	−1.13	0.21	0.19	0.61	0.66
Ti	0.82	0.28	0.28	0.40	0.38	−2.53	0.18	0.21	0.79	0.58	−3.10	0.14	0.15	0.94	1.00
Mn	−0.26	0.24	0.24	0.43	0.69	−2.31	0.19	0.18	0.76	0.72	1.92	0.29	0.30	0.17	0.17
Ba	−0.02	0.23	0.23	0.40	0.77	−1.15	0.16	0.17	0.55	0.88	2.32	0.36	0.37	0.18	0.10
clay	3.00	0.22	0.23	0.10	0.24	−1.38	0.10	0.10	0.63	0.79	−2.60	0.12	0.13	0.88	0.76

SOC	−1.04	0.27	0.23	0.63	0.62	−4.88	0.17	0.16	0.98	0.70	−1.76	0.19	0.21	0.79	0.86
Zn	−0.66	0.22	0.23	0.60	0.76	−0.38	0.21	0.23	0.43	0.31	−2.37	0.18	0.18	0.83	0.76
Fe	2.05	0.22	0.21	0.22	0.60	−1.65	0.22	0.21	0.63	0.27	−2.84	0.15	0.14	0.90	0.98
Cu	−1.92	0.23	0.21	0.70	0.83	2.74	0.29	0.29	0.10	0.14	−2.87	0.20	0.20	0.88	0.68
Sn	−1.42	0.19	0.20	0.66	0.14	−2.34	0.07	0.07	0.86	0.65	0.48	0.19	0.18	0.24	0.04
K	1.58	0.21	0.19	0.28	0.93	−2.79	0.17	0.16	0.84	0.73	−1.07	0.21	0.20	0.65	0.76
sand	0.47	0.21	0.19	0.31	0.71	0.01	0.17	0.18	0.40	0.59	−0.66	0.18	0.19	0.52	0.64
Pb	1.24	0.18	0.19	0.21	0.15	−1.22	0.10	0.09	0.59	0.45	−0.54	0.11	0.11	0.60	0.47
Sb	−2.17	0.18	0.18	0.66	0.95	−1.19	0.15	0.15	0.51	0.79	−3.25	0.17	0.15	0.91	0.84
Ca	−1.94	0.18	0.18	0.76	0.96	1.82	0.32	0.33	0.12	0.08	2.16	0.28	0.27	0.20	0.23
Mg	−0.13	0.14	0.17	0.44	0.98	−0.20	0.15	0.15	0.42	0.67	−3.59	0.14	0.14	0.98	0.92
Cr	0.27	0.18	0.17	0.35	0.28	−1.64	0.13	0.12	0.57	0.34	−1.44	0.11	0.11	0.59	0.57
pH	−2.59	0.16	0.16	0.93	0.79	−0.85	0.14	0.16	0.50	0.69	−1.11	0.18	0.19	0.69	0.31
Rb	0.97	0.15	0.15	0.31	0.98	−3.20	0.11	0.13	0.88	0.96	2.18	0.21	0.21	0.15	0.29
silt	−1.38	0.16	0.15	0.61	0.96	2.31	0.25	0.23	0.09	0.20	−1.40	0.19	0.19	0.65	0.55

4. Discussion

4.1. Soil variables within the mine wastelands

The soil texture across the tailings dam was predominantly sandy and homogeneous, with no significant variation in granulometric composition between plots. This uniform texture throughout the tailings dam aligns with previous studies that highlighted the homogeneous nature of soil in mining and waste disposal areas (Festin et al., 2019, Lal, 2009, Chileshe et al., 2019). The relative consistency in soil texture is likely a result of the mining and ore processing methods employed, which might contribute to specific depositional or erosional characteristic across the dam. Although the study by Chileshe et al. (2020) showed different textural properties in the tailing dams of Chingola in the Copperbelt Province of Zambia, the present study found similar general trends in soil texture, abeit with a higher percentage of sand.

Afforestation is one of the important measures for ecological restoration of mining generated landscape. One of the ecosystem services expected from such an approach is the effects of tree species in maximizing soil function. Tree species can influence soil properties through a wide array of mechanisms (Yang et al., 2018). Driven by their taxonomic diversity and potentially their diversified functional traits, trees species can greatly differ in their influence on soil properties. Results of the present study revealed that the concentrations of soil organic carbon (SOC) did not exhibit significant variations across different dominant tree canopy species in general, implying that this carbon-related parameter is relatively consistent under the canopies of the various tree species studied. Normally, due to their taxonomic diversity, the studied dominant species are expected to present contrasting functional attributes of leaf, which are likely to affect litter decomposition rate and the soil carbon input such as leaf longevity, leaf mass per area, and leaf nitrogen content (Rawat et al., 2021, Takahashi and Miyajima, 2008). In the context of this study however, it is apparent that specific tree species could not influence carbon dynamics in soil.

Furthermore, the study indicates that the soil pH within the investigated area, encompassing the dominant tree canopy species exhibited an overall basic trend (8.55 ± 0.4). Notably, a significant variation was observed around *Acacia polyacantha* dominant tree canopy species (with a lower pH) in comparison with *Terminalia mollis* dominant tree canopy species, which displayed a higher pH. Plausibly, the balance between production and consumption of soil hydrogen ions significantly varied in the two mentioned canopies comparatively to other dominant canopies. Several processes are involved in shaping the dynamic of soil hydrogen ions, including nutrient cycles, quantity and types of litter inputs, exudation of ions and organic compounds and other rhizospheric transformations such as nitrogen fixation and nutrient uptake by plant (Hong et al., 2018, Kooch et al., 2017), which may also have contributed to the observed differential effects. It is possible that *A. polyacantha* has created soil conditions favorable production of relatively higher amount of hydrogen ions, while *T. mollis* exhibit the opposite. There is however little quantitative information available on the comparative effects of trees on soils properties, which warrants further investigations.

The differential influence between tree species was also apparent on soil mineral composition, particularly for Ca, as *Ficus craterostoma* exhibited a higher Ca concentration (27.43 ± 4.23 mg/g) while the Mixed species plots with *Acacia polyacantha* showed a lower concentration (18.46 ± 5.83 mg/g) compared to other studied tree canopees. This could imply species-specific influences on soil calcium levels, likely associated with distinct tree characteristics, notably the influence on nutrient cycling processes (Noble and Randall, 1999), thereby influencing soil Ca dynamics. Since Ca is essential for plant growth and soil structure stability, this variation can have considerable impact on plant recruitment and growth patterns under the canopy. Calcium is indeed a macronutrient that is essential for plant growth, as it plays a role in cell wall formation, membrane stability, enzyme activation, and signal transduction (White and Broadley, 2003). The effect of tree species on soil macroelements was however isolated, as Ca was the only macroelement affected by the effects of tree canopies.

Similarly, among soil heavy metals, only Sr was significantly influenced by tree canopy species, showing that the biogeochemistry of Sr was more sensitive to the plant-induced changes in the soil environment than other heavy metals. Intriguingly, like the case with Ca, the significant effect of tree canopy species is evident only under *Ficus craterostoma* and under Mixed species with *Acacia polyacantha,* highlighting the particularity of this species in modulating the biogeochemistry of mineral in the conditions of the investigated tailing dams. Converse to the trend with Ca, higher Sr content are recorded under Mixed species with *Acacia polyacantha* whereas *Ficus craterostoma* canopy presents lower levels of Sr. The observed contrasting behavior of *Ficus craterostoma* canopy and Mixed species with *Acacia polyacantha* towards Ca and Sr highlighting the complex interactions between the canopy tree species and the soil properties in the mine wastelands. Likewise, a contrasting pattern of Ca and Sr can be noticed, which is line with their expected antagonistic in the biogeochemical system. The two elements present similar chemical structure (Kanter et al., 2010; Gupta and Clemens, 2017) to the degree they can antagonize each other's mobility in the plant-soil system. Calcium is indeed the analogue of strontium, and their biological discrimination is somewhat difficult for their uptake by living organisms (Kobayashi et al., 1991). El-Shazly et al. (2016) also noted an antagonistic effect between Ca and Sr, as Ca application was found to decrease Sr translocation. The relationship between calcium and strontium was also established in a soil from Eastern Transbaikalia as calcium deficiency was correlated with excess of strontium to account for the occurrence of the endemic Kashin–Beck disease (Ermakov et al., 2019).

4.2. Species richness, abundance and diversity across vegetation layers recruitment under the dominant tree canopy species

The results of this study showed that dominant tree species differently influence recruitment patterns and diversity, richness and abundance metrics in the different vegetative layers, with marked impact noted for tree and grass notably (Table 6). These observations are particularly relevant in tailings dams, where plant colonization dynamics are essential for stabilizing soils.

In the tree layer, the models revealed that fixed effects (Marginal R^2) and random effects (Conditional R^2) jointly explained the variations observed, with a significant contribution from spatial variability (ICC = 26 %). The results are consistent with those of Festin et al. (2019), who found that the floristic composition and diversity of tree species naturally colonising tailings dams in Zambia varied among different canopy tree species. Current results revealed that species such as *Ficus capensis* and *Ficus craterostoma* increased tree richness and diversity of understory trees. A few studies have pointed to the place of Ficus trees as critically important components of tropical ecosystems. These tree species produce large and nutritive fruits, which is believed to attract frugivorous birds and mammals, thereby facilitating seed dispersal by accelerating seed deposition and increasing species diversity in the understory (Omeja et al., 2011). In an experiment evaluating the effectiveness of remnant trees in inducing forest recovery, Ficus trees were found to induce more sapling species around them comparatively non-Ficus trees (Cottee-Jones et al., 2016). Besides, being fast-growing, we cannot rule out the possibility of Ficus to increase species richness due to the higher canopy areas, which offering a microenvironment favorable to accommodate a diversity of trees through processes such as shading, soil cooling, or organic matter inputs, etc. These traits are cardinal in disturbed landscapes such tailings dams, where these ficus tree species could act as ecosystem engineers (Jones et al., 1996). For example, their nutrient-rich litter and extensive root systems can reduce erosion and improve water retention (Cottee-Jones et al., 2016), creating resilient ecosystems on these artificial sites, often exposed to abiotic stresses.

Likewise, the pronounced effects of *Gmelina arborea* and *Bauhinia petersiana* on tree abundance also underline their ability to rapidly dominate disturbed environments, a trait particularly advantageous in the early stages of ecological succession on tailings dams (). These results demonstrate the possibility of certain canopy species to directly or indirectly create more favorable conditions or niches for the establishment of both *Ficus* or non-*Ficus* seedlings. This suggests the interest to prioritizing large trees, and fruit-bearing trees, particularly *Ficus* trees, in prospect of effective ecological restoration of tailing dams in the region. The ecological relevance of *Ficus* trees is substantiated by further observation that *ficus species (F. capensis and F. craterostoma)* stimulated herb richness. Like the Ficus species, *G. arborea* canopy was also found to play a non negligeable role in herb recruitment by stimulating their abundance. In tailings dams, the herbaceous layer is crucial in early soil stabilisation and erosion prevention, notably by establishing a rapid vegetation cover (Duncan et al., 2015). Interactions between the canopy of dominant trees and herbs may be due to the facilitating effect of the dominant tree canopy, which modifies light, reduces water competition or enriches the soil with nutrients via organic debris (Callaway, 1995). Tree canopy species with higher water-use efficiency and hydraulic architecture may also promote microhabitats with favorable moisture conditions, allowing certain herbaceous species to thrive (Ruthrof et al., 2015). It is clear that environmental drivers of species recruitment under dominant tree canopy species operated at different scales or pathways, for which *Ficus* species i.e/ *F. capensis* and *F. craterostoma* and *G. arborea* notably, are expected to play a key role.

In contrast to the tree and herb layers, the results indicate that neither fixed nor random effects significantly explain variations in richness, abundance or diversity metrics in the grass layer. This lack of effect could reflect the low dependence of grasses on microenvironmental conditions modified by dominant trees or their ability to tolerate poor edaphic environments. Nonetheless, *Rhus longipes* seems to have a marginal effect on grass diversity, warranting further research to identify subtle but significant interactions.

4.3. The relationship between soil properties and floristic species recruitment under dominant tree canopy species

The study demonstrated that edaphic properties have a differentiated influence on species richness, depending on the vegetative layer. In the tree layer, Fe played a key role, promoting richness at moderate concentrations before becoming toxic (Kabata-Pendias, 2010). High sand content (>90 %) also enhances richness by facilitating drainage and aeration, two essential conditions for growth trees in tailings dams (Wang et al., 2021). In contrast, herb and grasses layers showed high sensitivity to nutrients such as Ca and Na, which promote richness, while heavy metals such as Zn and Sb reduce it at high concentrations (Mensah et al., 2018). These results confirm that the extreme soil conditions of tailings dams strongly modulate plant composition as a function of plant-soil interactions (Skousen et al., 2019).

Species abundance is mainly influenced by nutrients and heavy metals, with variations depending on the vegetative layer. In the tree layer, K and soil pH increase abundance by improving physiological functions and nutrient availability (Wang et al., 2021). However, heavy metals such as Cu and Cr reduce abundance at high concentrations, limiting root growth (Festin et al., 2019). For grasses and herbs, nutrients such as Mg promoted abundance, while heavy metals such as Fe and Ti showed negative effects. These observations underline the need for targeted management of heavy metals to improve plant abundance, particularly in degraded environments (Skousen et al., 2019).

Species diversity, measured by the Shannon-Wiener and Simpson indices, was influenced by nutrients and heavy metals to varying degrees depending on the layer. Na significantly reduces tree diversity, probably by limiting the ecological niches available, while Zn, Mn and sand promote it to moderate levels (Kabata-Pendias, 2010). In herbaceous and grassy layers, Ca and Cu negatively influence diversity at high concentrations, while Na and Ba increase it, reflecting differential tolerance to edaphic constraints (Festin et al., 2019; Lugo, 1997). These results demonstrated the importance of edaphic properties in structuring plant communities on tailings dams and guide ecological restoration strategies by integrating adapted soil and species management.

Soil texture does not significantly affect plant diversity in grass, herbs and tree species layers. These findings suggest that while soil properties like texture influence vegetation recruitment, their overall contribution is relatively minor within mine wastelands. This emphasizes the resilience or adaptability of certain species to suboptimal soil conditions and highlights the potential importance of other ecological, biotic, or historical factors in shaping plant communities in these disturbed areas. Mechanistically, species adapted to disturbed environments may possess traits that allow them to thrive regardless of soil texture variations, such as efficient water use, tolerance to low nutrient availability, and the ability to survive in contaminated soils. These findings align with earlier research that found no relationship between soil texture and plant species richness or diversity in different ecosystems, such as grasslands, forests, and wetlands (Rodrigues et al., 2018). However, when variation in texture occurs, it can influence plant diversity by affecting keys soil properties like moisture retention, nutrient availability, and water holding capacity (Xie et al., 2016), which directly impact plant physiological processes.

A noteworthy study finding is the negative and significant correlation between soil pH and species richness and abundance among understory tree layers. This implies that as soil pH decreases, there is a concurrent decline in diversity, species richness and the number of individual trees under the dominant tree canopy species. Such a correlation suggests that soil acidity may inhibit nutrient availability and disrupt root growth, ultimately limiting the establishment and growth of understory tree species. Acidic conditions can also alter the microbial communities essential for nutrient cycling, further restricting plant recruitment. These results align with Alsherif et al. (2022), who identified soil pH as a major determinant of plant diversity in various ecosystems, such as grasslands, forests, and wetlands. Moreover, soil pH has significant applications in nutrient cycling, plant nutrition, and soil remediation, encompassing bioremediation and physicochemical remediation efforts (Neina, 2019). The lower soil pH value observed around *Acacia polyacantha* compared to *Terminalia mollis* suggests species-specific effects on the soil chemistry, possibly through the release organic acids or alterations in microbial activity (Zou et al., 2023, Barrow and Alfred, 2023). This mechanistic insight underlines how certain tree species can modify their microenvironments, impacting soil acidity and, consequently, the diversity and structure of understory vegetation. The intricate relationship between soil characteristics and the ecological dynamics of understory vegetation, may be more pronounced under specific conditions or in combination with other soil factors, suggesting a complex interaction that warrants further investigation.

For grass and herbs layers, the study found that heavy metal contents, particularly Zr, Sn, and Pb, were negatively correlated with species richness and the Shannon-Wiener index. This indicates that increased concentrations of these heavy metals correspond with decreased species richness and diversity, suggesting that heavy metal pollution might be a critical factor limiting for species variety and abundance in these layers. Heavy metals stress cause oxidative stress that can disrupt the functioning of key metabolic processes in for plants growth and development (Isermann, 2005). These toxic effects reduce the overall fitness of plants, especially those not adapted to high heavy metal concentrations, ultimately leading to lower diversity in contaminated areas (Fischer et al., 2014). Understanding the specific physiological responses of different plant species to heavy metals could help identify species with higher tolerance and accumulation potential, which is essential for ecological restoration efforts.

In the tree species layer, only Ba significantly affected tree species richness highlighting the potential role of specific heavy metals in influencing tree recruitment and survival. Some tree species, such as *Robinia pseudoacacia*, *Populus* spp., and *Ginkgo biloba,* have been observed to tolerate and accumulate heavy metals, including Cd, Zn, Cu, and Pb (Capuana, 2020; Kim and Lee, 2005). Mechanistic insights into how these species manage heavy metal stress are warranted, which could provide valuable information to develop proxies to unveil other tree species suitable for phytostabilization and restoration of mine wastelands.

4.4. Implications for ecological restoration and sustainable management of tailings dams

The results of this study highlight that soil properties, such as texture, nutrients and heavy metal concentrations, directly influence the diversity and composition of plant communities in tailings dams. Sandy soils, while offering good drainage, limit water and nutrient retention, thus hindering plant growth (Wang et al., 2021). Proactive texture management, through the addition of organic amendments, could improve these soils and promote ecosystem stabilization. Furthermore, based on the observed association of *F. craterostoma* with Ca enrichment in the soil under the canopy, we cannot rule out the contribution of Ca in the recruitment of both tree and herbs species under the *F. craterostoma* canopy. Hence, Ca enrichment in soil can be considerable as an option to accelerate plant species recruitment under different canopies. These observations altogether showed that dominant trees act as founder species, influencing the long-term ecological trajectories and resilience of plant communities (Cottee-Jones et al., 2016).

Finally, the differences in species richness and diversity observed under different tree canopies underline the importance of selecting species that favor varied ecological niches and high ecosystem resilience. Ficus species such as *Ficus capensis* and *F. craterostoma,* and to a lesser extent G. arborea, which increase diversity in several strata, should be prioritised in reforestation programs. In parallel complementary strategies, such as the introduction of adapted grass mixtures, could also maximize vegetation cover in the early restoration phase (Mensah et al., 2018). These approaches, combined with continuous monitoring of soil and plant communities, would enable strategies to be adjusted according to evolving conditions, thus optimising the sustainability and functionality of restored ecosystems.

5. Conclusions and recommendations

This study highlights the complex interactions between soil properties and vegetation recruitment in mine wastelands, emphasizing the critical role of dominant tree canopy species in shaping understory communities. Soil texture, chemical properties, and heavy metal concentrations varied across blocks within the tailings dam. Notable findings include the positive influence of soil pH, Magnesium (Mg), and Rubidium (Rb) content on tree species richness under dominant canopy species. Calcium (Ca) also emerged as a significant factor influencing vegetation dynamics. However, the overall recovery of species richness and diversity was minimally dependent on soil parameters, with heavy metals explaining only 1.7–2 % of species recruitment variability. The Shannon-Wiener and Simpson diversity indices revealed significant variations among canopy species, with species richness declining in response to elevated concentrations of Zirconium (Zr), Tin (Sn), and Lead (Pb). These findings emphasize the importance of targeted strategies to leverage the ecological potential of dominant tree species while addressing soil and environmental challenges.

To maximize the ecological restoration of mine wastelands, restoration strategies should prioritize the selection and management of canopy species such as *Ficus craterostoma* and *Ficus capensis*, which significantly enhance diversity and richness. These species can be propagated in nurseries and incorporated into phased planting programs. Their ability to foster diverse and stable understory communities makes them valuable in rehabilitation efforts. Additionally, understanding and harnessing soil-plant interactions are critical for tailoring restoration approaches. For example, site-specific soil amendments such as calcium-based supplements can enhance species richness in areas with deficient nutrient levels. Detailed soil mapping should precede restoration to ensure interventions are matched to local soil conditions.

While soil properties play a role in the phytostabilization potential of trees, other factors such as biotic interactions, microclimatic conditions, and disturbance regimes are equally crucial. Future efforts should incorporate research into the roles of mycorrhizal fungi, pollinators, and soil biota, which could support plant recruitment and diversity. Collaborative studies involving ecologists and microbiologists can uncover how these factors interact with soil and vegetation. Experimental inoculation with beneficial fungi and monitoring pollinator networks could further enhance restoration outcomes.

Given the context-dependent nature of phytostabilization, species-specific evaluations are necessary to assess tree tolerance to heavy metals, their growth characteristics, and their ability to stabilize contaminated soils. Pilot studies can test the performance of species such as *Acacia polyacantha* and *Gmelina arborea* under varying environmental conditions, providing insights into their suitability for large-scale restoration. Furthermore, long-term monitoring programs using remote sensing and ground surveys are essential to track ecological changes, refine strategies, and adapt to evolving environmental conditions.

Finally, the specific mechanisms by which canopy trees influence understory plant communities, such as altering microclimates or soil chemistry, warrant detailed investigation. Controlled experiments can isolate these effects, providing a deeper understanding of the pathways through which canopy trees enhance recruitment and diversity. By integrating these findings into restoration strategies, mine wastelands can be transformed into resilient ecosystems, fostering biodiversity and offering long-term environmental and socioeconomic benefits.

Funding

The author Arthur A Owiny, has received a Ph.D scholarship from the Copperbelt University- African Centre of Excellence for Sustainable Mining (CBU-ACESM). In addition, the International Development Research Centre (IDRC), Canada, the National Research Foundation (NRF), South Africa and the National Science and Technology Council (NSTC), Zambia through Copperbelt University Oliver R Tambo Africa Research Chair Initiative (CBU-ORTARChI) Chair of Environment and Development also funded the project.

Consent for publication

All authors approved the submission.

CRediT authorship contribution statement

Jules Christian Zekeng: Writing – review & editing, Visualization, Supervision. **Paxie W Chirwa:** Writing – review & editing, Supervision, Software, Resources, Investigation. **Theodore Mulembo Mwamba:** Writing – review & editing, Supervision. **Arthur Owiny:** Writing – original draft, Methodology, Investigation, Formal analysis. **Stephen Syampugani:** Writing – review & editing, Supervision, Software, Project administration, Funding acquisition, Conceptualization.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests. Arthur Arnold OWINY reports financial support was provided by Copperbelt University- African Centre of Excellence for Sustainable Mining (CBU-ACESM). If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

The work was funded by The Copperbelt University-African Centre of Excellence for Sustainable Mining (CBU-ACESM) and the Copperbelt University Oliver R Tambo Africa Research Chair Initiative (CBU-OTARChI) Chair of Environment and Development supported by the International Development Research Centre, Canada (IDRC), National Research Foundation, South Africa (NRF), National Science and Technology Council, Zambia (NSTC). ORTARChI is an initiative of the International Development Research Centre, Canada (IDRC), National Research Foundation, South Africa (NRF), National Science and Technology Council, Zambia (NSTC). MOPANI Mines granted permission to conduct this study. We thank Mrs Sarah Banda and Mr. Geophat Mpautua for helping with fieldwork and plant identification and Mr. Felix Chileshe for soil analysis.

Data availability

Data will be made available on request.

References

Alsherif, E.A., Al-Shaikh, T.M., AbdElgawad, H., 2022. Heavy metal effects on biodiversity and stress responses of plants inhabiting contaminated soil in Khulais, Saudi Arabia. Biology 11, 164. https://doi.org/10.3390/biology11020164.

Altmann, A., Toloşi, L., Sander, O., Lengauer, T., 2010. Permutation importance: a corrected feature importance measure. Bioinformatics 26, 1340–1347.

Anawar, M.H., Chowdhury, R., 2020. Remediation of polluted river water by biological, chemical, ecological and engineering processes. Sustainability 12 (17), 7017–7035. doi:10.3390/su12177017.

Archer, K.J., Kimes, R.V., 2008. Empirical characterization of random forest variable importance measures. Comput. Stat. Data Anal. 52, 2249–2260.

Barrow, N.J., Alfred, E.H., 2023. The effects of PH on nutrient availability depend on both soils and plants. Plant Soil 487, 21–37. https://doi.org/10.1007/s11104-023-05960-5.

Bonham, C.D., Mergen, D.E., Montoya, S., 2004. Plant cover estimation: a contiguous daubenmire frame. Rangelands 26 (1), 17–22. https://doi.org/10.2307/4001909.Breiman, L., 2001. Random forests. Mach. Learn. 45, 5–32.

Bufebo, B., Elias, E., 2020. Effects of land use/land cover changes on selected soil physical and chemical properties in Shenkolla watershed, South Central Ethiopia. Adv. Agric. 1–8. https://doi.org/10.1155/2020/5145483.

Callaway, R.M., 1995. Positive interactions among plants. Bot. Rev. 61, 306–349.

Capuana, M., 2020. A review of the performance of woody and herbaceous ornamental plants for phytoremediation in urban areas. iForest Biogeosci. For. 13 (2), 139–151. https://iforest.sisef.org/contents/?id=ifor3242-013

Chileshe, M.N., Syampungani, S., Festin, E.S., Tigabu, M., Daneshvar, A., Odén, P.C., 2020. Physico-chemical characteristics and heavy metal concentrations of copper mine wastes in Zambia: implications for pollution risk and restoration. J. For. Res. 31, 1283–1293.

Chileshe, M.N., Syampungani, S., Sandell Festin, E., Tigabu, M., Daneshvar, A., Odén, P.C., 2019. Heavy metal concentrations and physico-chemical characteristics of copper mine wastes in Zambia: implications for pollution risk and restoration. J. For. Res. https://doi.org/10.1007/s11676-019-00921-0.

Cottee-Jones, H.E.W., Bajpai, O., Chaudhary, L.B., Whittaker, R.J., 2016. The importance of Ficus (Moraceae) trees for tropical forest restoration. Biotropica 48, 413–419.

Dermont, G., Bergeron, M., Mercier, G., Richer-Laflèche, M., 2008. Soil washing for metal removal: A Review of Physical/Chemical Technologies and Field Applications. 152(1), 1–31. doi:10.1016/j.jhazmat.2007.10.043.

Desroches, D., Bédard, L.P., Lemieux, S., Esbensen, K.H., 2018. Suitability of using a handheld XRF for quality control of quartz in an industrial setting. Miner. Eng. 126, 36–43. https://doi.org/10.1016/j.mineng.2018.06.016.

Dhaliwal, S.S., Singh, J., Taneja, P.K., Mandal, A., 2020. Remediation techniques for removal of heavy metals from the soil contaminated through different sources: A review. Environ. Sci. Pollut. Res. Int. 27 (2), 1319–1333. https://doi.org/10.1007/s11356-019-06967-1.

Duncan, C., Thompson, J.R., Pettorelli, N., 2015. The quest for a mechanistic understanding of biodiversity-ecosystem services relationships. Proc. Biol. Sci. 282, 20151348.

El-Shazly, A.A., Rezk, M.A., Abdel-Sabour, M.F., Mousa, E.A., et al., 2016. Effect of calcium levels on strontium uptake by canola plants grown on different texture soils. J. Nucl. Technol. Appl. Sci. 4, 1–10.

Ermakov, V., Bech, J., Gulyaeva, U., Tyutikov, S., Safonov, V., Danilova, V., Roca, N., 2019. Relationship of the mobile forms of calcium and strontium in soils with their accumulation in meadow plants in the area of Kashin–Beck endemia. Environ Geochem Health 42, 159–171. https://doi.org/10.1007/s10653-019-00323-5.

Festin, E.S., Salka, C., Tigabua, M., Syampungani, S., Odéna, P.C., 2019. Biological traits of tropical trees suitable for restoration of copper-polluted lands. Ecol. Eng. 138, 118–125. https://doi.org/10.1016/j.ecoleng.2019.07.010.

Fischer, C., Christiane, R., Britta, J., Nico, E., Jussi, B., Sabine, A., Stefan, S., Wolfgang, W.W., Jens, S., Anke, H., 2014. How do earthworms, soil texture and plant composition affect infiltration along an experimental plant diversity gradient in grassland? PLoS ONE 9 (6), e98987. https://doi.org/10.1371/journal.pone.0098987.

Franks, D.M., Boger, D.V., Cote, C.M., Mulligan, 2011. Sustainable development principles for the disposal of mining and mineral processing wastes. Resour. Policy 36 (2), 114–122. https://doi.org/10.1016/j.resourpol.2010.12.001.

Fu, D., Wu, X., Duan, C., Smith, A.R., Jones, D.L., 2020. Traits of dominant species and soil properties co-regulate soil microbial communities across land restoration types in a subtropical plateau region of Southwest China. Ecol. Eng. 153, 105897. https://doi.org/10.1016/j.ecoleng.2020.105897.

Galal, T.M., Shehata, H.S., 2015. Bioaccumulation and translocation of heavy metals by Plantago major L. grown in contaminated soils under the effect of traffic pollution. Ecol. Indic. 48, 244–251. https://doi.org/10.1016/j.ecolind.2014.08.013.

Goyal, D., Yadav, A., Prasad, M., Singh, T.B., Shrivastav, P., Ali, A., Dantu, P.K., Mishra, S., 2020. Effect of heavy metals on plant growth: an overview BT - contaminants in agriculture: sources, impacts and management. In: Naeem, M., Ansari, A.A., Gill, S.S. (Eds.), Contaminants in Agriculture. Springer International Publishing, pp. 79–101. https://doi.org/10.1007/978-3-030-41552-5.

Gupta, D.K., Clemens, W., 2017. Behaviour of strontium in plants and the environment. Behav. Strontium Plants Environ. 1–170. https://doi.org/10.1007/978-3-319-66574-0.

Hokura, A., 2021. X-ray fluorescence spectrometry. Anal. Sci. 37, 1–2 doi.10.2116/analsci.highlights2101.

Hong, S., Piao, S., Chen, A., Liu, Y., Liu, L., Peng, S., Sardans, J., Sun, Y., Peñuelas, J., Zeng, H., 2018. Afforestation neutralizes soil pH. Nat. Commun. 9, 520.

Hurlbert, S.H., 1971. The nonconcept of species diversity: a critique and alternative parameters. Ecology 52, 577–586. https://doi.org/10.2307/1934145.

Ilunga, E., Mahy, G., Piqueray, J., Séleck, M., Shutcha, M., Meerts, P., Faucon, M.P., 2015. Plant functional traits as a promising tool for the ecological restoration of degraded tropical metal-rich habitats and revegetation of metal-rich bare soils: a case study in copper vegetation of Katanga, DRC. Ecol. Eng. 82, 214–221. https://doi.org/10.1016/j.ecoleng.2015.04.084.

Isermann, M., 2005. Soil pH and species diversity in coastal dunes. Plant Ecol. 178, 111–120. https://doi.org/10.1007/s11258-004-2558-8.

Jones, C.G., Lawton, J.H., Shachak, M., 1996. Organisms as ecosystem engineers. In: Samson, F.B., Knopf, F.L. (Eds.), Ecosystem Management: Selected Readings. Springer New York, New York, NY, pp. 130–147.

Kabata-Pendias, A., 2010. Trace Elements in Soils and Plants, 4th ed. CRC Press. 10.1201/b10158.

Kanter, U., Hauser, A., Michalke, B., Dräxl, S., Schäffner, A.R., 2010. Caesium and strontium accumulation in shoots of Arabidopsis thaliana: genetic and physiological aspects. J Exp Bot 61, 3995–4009. https://doi.org/10.1093/jxb/erq213.

Kim, K.D., Lee, E.J., 2005. Potential tree species for use in the restoration of unsanitary landfills. Environ Manage 36 (1), 1–14. https://doi.org/10.1007/s00267-004-1089-3.

Kobayashi, E., Sugihira, N., Suzuki, K.T., 1991. Biological discrimination between calcium and strontium in kidneys and bone of young and adult rats. Biol. Trace Elem. Res. 28, 187–194.

Kooch, Y., Tarighat, F.S., Hosseini, S.M., 2017. Tree species effects on soil chemical, biochemical and biological features in mixed caspian lowland forests. Trees 31 (3), 863–872. https://doi.org/10.1007/s00468-016-1511-5.

Kosgei, H., Mukuwa, M., 2020. Environmental effects of tailings dams in the Zambian Copper Belt: A review paper.

Koul, B., Taak, P., 2018. Biotechnological Strategies for effective remediation of polluted soils. Chem. Methods Soil Remediat. 4, 77–84. https://doi.org/10.1007/978-981-13-2420-8_4.

Lal, R., 2009. Soil degradation is a reason for inadequate human nutrition. Food Sect. 1 (1), 45–57. https://doi.org/10.1007/s12571-009-0009-z.

Lèbre, É., Corder, G.D., Golev, A., 2017. Sustainable practices in the management of mining waste: A focus on the mineral resource. Miner. Eng. 107, 34–42. https://doi.org/10.1016/j.mineng.2016.12.004.

Li, M., Luo, Y., Su, Z., 2007. Heavy metal concentrations in soils and plant accumulation in a restored manganese mineland in Guangxi, South China. Environ. Pollut. 147, 168–175. https://doi.org/10.1016/j.envpol.2006.08.006.

Liaw, A., Wiener, M., 2001. Classification and regression by Random Forest. Forest 23.

Littell, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., Schabenberger, O., 2006. SAS for mixed models, 2nd ed. SAS Institute, Cary, NC.

Lugo, A.E., 1997. The apparent paradox of re-establishing species richness on degraded lands with tree monocultures. For. Ecol. Manag. 99, 9–19. https://doi.org/10.1016/s0378-1127(97)00191-6.

Magurran, A., 2004. Measuring Biological Diversity. Blackwell Publishing, Oxford, UK.

Matakala, N., Chirwa, P.W., Mwamba, T.M., Syampungani, S., 2023. Species richness and phytoremediation potential of mine wastelands-native trees across the Zambian Copperbelt Region. Heliyon 9 (3), e13585. https://doi.org/10.1016/j.heliyon.2023.e13585.

Maurer, B.A., MCGill, B.J., 2011. Measurement of species diversity. In: Magurran, A.E., McGill, B.J. (Eds.), Biological diversity. Frontiers in measurement and assessment. Oxford University Press, Oxford, pp. 55–65.

Mensah, S., du Toit, B., Seifert, T., 2018. Diversity–biomass relationship across forest layers: implications for niche complementarity and selection effects. Oecologia 187 (3), 783–795. https://doi.org/10.1007/s00442-018-4144-0.

Miller, B.A., Schaetzl, R.J., 2011. Precision of soil particle size analysis using laser diffractometry. Soil Sci. Soc. Am. J. 76, 1719–1727. https://doi.org/10.2136/sssaj2011.0303.

Muimba-Kankolongo, A., Nkulu, C.B.L., Mwitwa, J., Kampemba, F.M., Nabuyanda, M.M., 2022. Impacts of trace metals pollution of water, food crops, and ambient air on population health in Zambia and the DR Congo. J. Environ. Public Health 1–14. https://doi.org/10.1155/2022/4515115.

Nakagawa, S., Schielzeth, H., 2013. A general and simple method for obtaining R² from generalized linear mixed-effects models. Methods Ecol. Evol. 4 (2), 133–142.

Neina, D., 2019. The role of soil pH in plant nutrition and soil remediation. edited by Marco Trevisan. Appl. Environ. Soil Sci., 5794869

Nelson, D.W., Sommers, L.E., 1982. Total carbon, organic carbon and organic matter. In: Page, A.L., Miller, R.H. (Eds.), Methods of Soil Analysis, Part 2, second ed. ASA and SSSA, Madison, WI, USA, pp. 534–580 (Agronomy Monograph).

Noble, A.D., Randall, P.J., 1999. Alkalinity effects of different tree litters incubated in an acid soil of N.S.W., Australia. Agrofor. Syst. 46 (2), 147–160. https://doi.org/10.1023/A:1006299615488.

Omeja, P.A., Chapman, C.A., Obua, J., Lwanga, J.S., Jacob, A.L., Wanyama, F., Mugenyi, R., 2011. Intensive tree planting facilitates tropical forest biodiversity and biomass accumulation in Kibale

National Park. Uganda For. Ecol. Manag. 261, 703–709. https://doi.org/10.1016/j.foreco.2010.11.029.

Park, I., Tabelin, C.B., Jeon, S., Li, X., Seno, K., Ito, M., Hiroyoshi, N., 2019. A review of recent strategies for acid mine drainage prevention and mine tailings recycling. Chemosphere 219, 588–606. https://doi.org/10.1016/j.chemosphere.2018.11.053.

Peco, J.D., Higueras, P., Campos, J.A., Esbrí, J.M., Moreno, M.M., Battaglia-Brunet, F., Sandalio, L.M., 2021. Abandoned mine lands reclamation by plant remediation technologies. Sustainability 13, 6555. https://doi.org/10.3390/su13126555.

Perkovic S., Paul C., Vasic F., Helming, K., 2022. Heavy Metals in Agricultural Soils.

Qaswar, M., Yiren, L., Jing, H., Kaillou, L., Mudasir, M., Zhenzhen, L., Huimin, Z., 2020. Soil nutrients and heavy metal availability under long-term combined application of swine manure and synthetic fertilizers in acidic paddy soil. J. Soils Sediment. 20, 2093–2106. https://doi.org/10.1007/s11368-020-02576-5.

R Development Core Team, 2022. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ravansari, R., Wilson, S.C., Tighe, M., 2020. Portable X-ray fluorescence for environmental assessment of soils: not just a point and shoot method. Environ. Int. 134, 105250. https://doi.org/10.1016/j.envint.2019.105250.

Rawat, M., Arunachalam, K., Arunachalam, A., Alatalo, J.M., Pandey, R., 2021. Assessment of leaf morphological, physiological, chemical and stoichiometry functional traits for understanding the functioning of himalayan temperate forest ecosystem. Sci. Rep. 11, 23807. https://doi.org/10.1038/s41598-021-03235-6.

Robertson, C., Nelson, T.A., Boots, B., 2007. Mountain pine beetle dispersal: the spatial–temporal interaction of infestations. For. Sci. 53, 395–405. https://doi.org/10.1093/forestscience/53.3.395.

Rodrigues, P.M.S., Carlos, E.G.R.S., Jhonathan, de Oliveira S., Walnir, G.F.J., Rubens, M. dos Santos, Andreza, V.N., 2018. The influence of soil on vegetation structure and plant diversity in different tropical savannah and forest habitats. Plant Ecol. 11, 226–236 doi.org/10.1093/jpe/rtw135.

Ruthrof, K.X., George, M., Giles, E.J.H., 2015. Early differential responses of co-dominant canopy species to sudden and severe drought in a mediterranean-climate type forest. Forests 6 (6), 2082–2091. https://doi.org/10.3390/f6062082.

Santos, A.E., Cruz-Ortega, R., Meza-Figueroa, D., Romero, F.M., Sanchez-Escalante, J.J., Maier, R.M., Neilson, J.W., Alcaraz, L.D., Freaner, F.E.M., 2017. Plants from the abandoned Nacozari mine tailings: evaluation of their phytostabilization potential. PeerJ 5, e3280. https://doi.org/10.7717/peerj.3280.

Shannon, C.E., Weaver, W., 1949. The Mathematical Theory of Communication. University of Illinois Press, Urbana.

Sillitoe, R.H., Perelló, J., Creaser, Wilton, J., Dawborn, T., 2017. Age of the Zambian Copperbelt. Min. Depos. 52, 1245–1268. https://doi.org/10.1007/s00126-017-0726-8.

Simpson, E.H., 1949. Measurement of diversity. Nature 163, 688.

Singh, S., Parihar, P., Singh, R., Singh, V.P., Prasad, S.M., 2016. Heavy metal tolerance in plants: Role of transcriptomics, proteomics, metabolomics, and ionomics. Front. Plant Sci. 6, 1–36. https://doi.org/10.3389/fpls.2015.01143.

Skousen, J.G., Ziemkiewicz, P.F., McDonald, L.M., 2019. Acid mine drainage formation, control and treatment: approaches and strategies. Extr. Ind. Soc. 6 (1), 241–249. https://doi.org/10.1016/j.exis.2018.09.008.

Sracek, O., Kříbek, B., Mihaljevič, M., Ettler, V., Vaněk, A., Penížek, V., Filip, J., Veselovský, F., Nyambe, I., 2018. The impact of wetland on neutral mine drainage from mining wastes at Luanshya in the Zambian Copperbelt in the framework of climate change. Environ. Sci. Pollut. Res. 25, 28961–28972. https://doi.org/10.1007/s11356-018-2929-7.

Takahashi, K., Miyajima, Y., 2008. Relationships between leaf life span, leaf mass per area, and leaf nitrogen cause different altitudinal changes in leaf δ 13C between deciduous and evergreen species. Botany 86. https://doi.org/10.1139/B08-093.

Tavares, T.R., Molin, J.P., Javadi, S.H., Carvalho, H.W.P., Mouazen, A.M., 2020a. Combined Use of Vis-NIR and XRF sensors for tropical soil fertility analysis: assessing different data fusion approaches. Sensors 21 (1), 148. https://doi.org/10.3390/s21010148.

Tavares, T.R., Molin, J.P., Nunes, L.C., Alves, E.E.N., Melquiades, F.L., Carvalho, H.W.P., Mouazen, A.M., 2020b. Effect of X-ray tube configuration on measurement of key soil fertility attributes with XRF. Remote Sens. 12, 963. https://doi.org/10.3390/rs12060963.

Tavares, T.R., Mouazen, A.M., Alves, E.E.N., dos Santos, F.R., Melquiades, F.L., Pereira de Carvalho, H.W., Molin, J.P., 2020c. Assessing soil key fertility attributes using a portable x-ray fluorescence: a simple method to overcome matrix effect. Agronomy 10, 787. https://doi.org/10.3390/agronomy10060787.

Tavares, T.R., Nunes, L.C., Alves, E.E.N., Almeida, E., Maldaner, L.F., Krug, F.J., Carvalho, H.W.P., Molin, J.P., 2019. Simplifying sample preparation for soil fertility analysis by X-ray fluorescence spectrometry. Sensors 19, 5066. https://doi.org/10.3390/s19235066.

Tian, Q., Liu, N., Bai, W., Li, L., Chen, J., Reich, P.B., Yu, Q., Guo, D., Smith, M.D., Knapp, A.K., Cheng, W., Lu, P., Gao, Y., Yang, A., Wang, T., Li, X., Wang, Z., Ma, Y., Han, X., Zhang, W., 2015. A novel soil manganese mechanism drives plant species loss with increased nitrogen deposition in a temperate steppe. Ecology 97 (1), 65–74, 10.1890/15-0917.1.

Vadivel, V., 2020. Atomic Absorption Spectrophotometer (AAS).

Venkateswarlu, K., Nirola, R., Kuppusamy, S., Thavamani, P., Naidu, R., Megharaj, M., 2016. Abandoned metalliferous mines: ecological impacts and potential approaches for reclamation. Rev. Environ. Sci. Bio/Technol. 15, 327–354. https://doi.org/10.1007/s11157-016-9398-6.

Wang, Z., Wang, G., Ren, T., Wang, H., Xu, Q., Zhang, G., 2021b. Assessment of soil fertility degradation affected by mining disturbance and land use in a coalfield via machine learning. Ecol. Indic. 125, 107608. https://doi.org/10.1016/j.ecolind.2021.107608.

Wang, G., Zhang, Q., Du, W., Fuxun, A., Rong, J., Guo, H., 2021. Microbial communities in the Rhizosphere of different willow genotypes affect phytoremediation potential in Cd contaminated soil. Sci. Total Environ. 769, 145224. https://doi.org/10.1016/j.scitotenv.2021.145224.

Wang, R., Zhang, J., Sun, H., Sun, S., Qin, G., Song, Y., 2021a. Effect of different vegetation on copper accumulation of copper-mine abandoned land in tongling, China. J. Environ. Manag. 286, 112227. https://doi.org/10.1016/j.jenvman.2021.112227.

White, P., Broadley, M., 2003. Calcium in Plants. Ann. Bot. 92, 487–511. https://doi.org/10.1093/aob/mcg164.

Wikum, D.A., Shanholtzer, G.F., 1978. Application of the Braun-Blanquet cover-abundance scale for vegetation analysis in land development studies. Environ. Manag. 2 (4), 323–329. https://doi.org/10.1007/bf01866672.

Xie, Y., Fan, J., Zhu, W., Amombo, E., Lou, Y., Chen, L., Fu, J., 2016. Effect of heavy metals pollution on soil microbial diversity and bermudagrass genetic variation. Front. Plant Sci. 7, 755. https://doi.org/10.3389/fpls.2016.00755.

Yang, N., Ji, L., Salahuddin, Yang, Y., Yang, L., 2018. The influence of tree species on soil properties and microbial communities following afforestation of abandoned land in northeast China. Eur. J. Soil Biol. 85, 73–78. https://doi.org/10.1016/j.ejsobi.2018.01.003.

Zhang, Y., Wang, R., Sardans, J., Wang, B., Gu, B., Li, Y., Liu, H., Peñuelas, J., Jiang, Y., 2023. Resprouting ability differs among plant functional groups along a soil acidification gradient in a meadow: a rhizosphere perspective. J. Ecol 111 (3), 631–644. https://doi.org/10.1111/1365-2745.14051.

Zou, L., Bai, Y., Huang, J., 2023. Soil pH and dissolved organic carbon shape microbial communities in wetlands with two different vegetation types in Changdu area. Tibet J. Mt. Sci. 20 (3), 750–764. https://doi.org/10.1007/s11629-022-7753-3.