# Relationship between deadwood structural diversity and carbon stock along environmental and disturbance gradients in Tropical dry forests

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**Abstract**

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Deadwood is a substantial component of forest ecosystems playing a vital role in maintaining ecosystem health and functioning. However, there is limited information on deadwood stand structure which encompasses attributes such as type, quantities and distribution of deadwood pieces and how it is related to its biomass. This study examined the relationship between deadwood species structural diversity and carbon stock along different environmental and disturbance factors in forest and woodland ecosystems. An agglomerative hierarchical clustering analysis was used to identify species communities, followed by indicator species analysis which was done to determine the species significantly associated with each community. Species richness, evenness and Shannon-Wiener diversity index were calculated to determine deadwood species diversity in both ecosystems. Multimodel inference approach was used to analyse the relationship between deadwood carbon stock and diversity indices, soil properties, climate and proximity to roads and settlements. Three communities were identified from forest ecosystems while four communities were from woodland. Multimodel analysis found a positive significant relationship between deadwood carbon stock and species abundance, Shannon-Wiener diversity, soil moisture and proximity to roads in both ecosystems. These findings provide insights into conservation strategies that prioritize protection and restoration of ecosystems as carbon reservois.

**Keywords**

deadwood carbon, deadwood species composition, diversity, NAFORMA - Tanzania

## Introduction

Worldwide, deadwood has emerged as a significant component of terrestrial carbon pools (Moreno-Fernández et al., 2020) and potentially important biodiversity indicators (Humphrey et al., 2005). Deadwood is one of the structural components of forest ecosystems playing a vital role in maintaining ecosystem health and functioning (Brockerhoff et al., 2017). The study of deadwood carbon stocks and their variability across different ecosystems has gained considerable attention due to its crucial role in understanding global carbon cycling and mitigating climate change (Bauhus et al., 2018).

Diverse ecosystems including tropical dry forests and woodlands exhibit distinct vegetation compositions and disturbance regimes. Forest ecosystems generally feature

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a higher density of tall trees with closed canopies, while woodlands have sparser tree distributions with more open canopies, supporting distinct community dynamics. Forests occupy 31% of the global land area and are among the richest ecosystems in terms of biodiversity and carbon pool (FAO, 2020). However, forest areas and their carbon pool have dramatically decreased due to the increasing rate of deforestation and changes in land use caused by agricultural expansion (Curtis et al., 2018).

The amount of carbon stored in a forest depends on various factors including forest structural diversity such as species composition, community assemblages, tree density and size distribution (Arasa-Gisbert et al., 2018; Gogoi et al., 2022) and are influenced by environmental factors such as temperature, precipitation and soil properties (Toledo et al., 2012). Soil properties such as organic carbon content, texture and moisture, affect the growth and distribution of plant species which in turn influences the overall structure and composition of forests (Sharma et al., 2018).

Human disturbances such as logging, agriculture expansion and infrastructure development can alter forest structure, consequently affecting deadwood dynamics. Areas close to roads and human settlements are more prone to human-induced disturbances compared to interior forests (Alamgir et al., 2017). Human disturbance may add to more deadwood hence carbon stock by increasing tree debris, but also may result in removal of standing and fallen deadwood, thus decreasing deadwood carbon stocks (Seidl et al., 2014).

In East Africa, Tanzania is one of the richest countries in terms of natural resources and biodiversity comprising of diverse vegetation types including extensive forest and woodland ecosystems (MNRT, 2015). There have been various efforts to link deadwood biomass with forest structural diversity (Hezron and Nyahongo, 2021) along with environmental and land use gradients (Komposch et al., 2022). However, increasing human pressure on natural resource extraction has increased at an alarming rate leading to an increase in forest deforestation and degradation rates. Tanzania has experienced forest loss of 469,000 hectares per year between 2002 to 2013 (URT, 2017) which also influence forest structural diversity and deadwood carbon stock. Understanding the influence of forest structure on deadwood carbon stock along environmental and disturbance gradients will provide insights into the management of dry forest ecosystems.

Therefore, this study assessed the influence of forest structure on deadwood carbon stock along environmental and disturbance gradients by exploring (1) deadwood species diversity and community composition in dry forests ecosystems (2) the relationships between environmental (soil and climate) and disturbance (distance to the nearest road and human settlements) factors and deadwood carbon stock. While living trees play a critical role in forest dynamics, this research does not specifically examine the relationships between deadwood characteristics and those of living trees. Instead, it emphasizes understanding deadwood as a standalone carbon pool and biodiversity component.

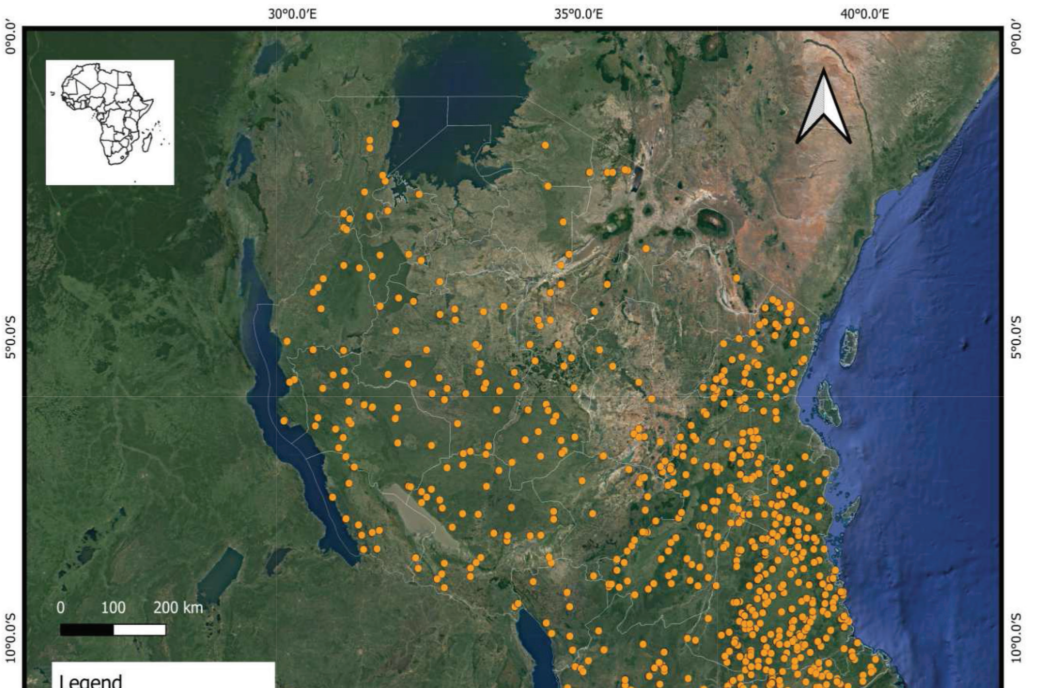



Fig. 1. Distribution of deadwood sample plots in Tanzania mainland.

## Materials and methods

### Study area

Tanzania is located between 1º00'S and 12º00'S and between 30º00'E and 41º00'E (Fig. 1) at an altitude between 358 m asl and 5,950 m asl. It lies on the east coast of Africa and is bordered by Kenya and Uganda to the north, Rwanda, Burundi, and the Democratic Republic of the Congo to the west, Zambia, Malawi, and Mozambique to the south and the Indian Ocean to the east.

Tanzania mainland has a mainly tropical climate but has regional variations due to topography. Temperature ranges between 10 °C and 20 °C during cold and hot seasons respectively. The mean annual rainfall ranges from 200 mm to over 2,000 mm per annum, with short rains from October to December and long rains from March to May.

The country's major forest types include deciduous miombo woodlands in the western, central, and southern parts, Acacia-Commiphora woodlands in the north, mangrove forests along the Indian Ocean coast and closed canopy forests that grow on the ancient mountains of the Eastern Arc. These forests and woodlands are subjected to various disturbances, such as selective logging, slash and burning for agricultural activities and wildfires, which can affect their structure and composition.

### Sampling design and data collection

Deadwood was surveyed across a range of forest and woodland ecosystems to assess its carbon stock, species richness and diversity. Data collection was designed to capture attributes directly related to deadwood, such as deadwood decay status, volume and associated environmental factors. Although living tree characteristics were documented during broader forest inventory, these data were not analysed in the context of this study.

This study employed the use of the National Forest Resources Monitoring and Assessment of Tanzania (NAFORMA) sampling design which was double sampling for stratification and optimal allocation of plots (Tomppo et al., 2014; MNRT, 2015). The National Forest Resources Monitoring and Assessment (NAFORMA) is the first forest inventory conducted to the entire Tanzania mainland and it was conducted between 2009 and 2014. The first phase sample consisted of clusters of plots laid at distances of 5 km × 5 km over mainland Tanzania. The country was divided into 18 strata based on predicted growing stock, accessibility and slope. With different sampling intensities in each stratum, the second phase samples were systematically selected from the first phase sample using optimal allocation. Higher sampling intensity was allocated to strata with high variation and high predicted growing stock while low sampling intensity was allocated to strata with low variation and low predicted growing stock.

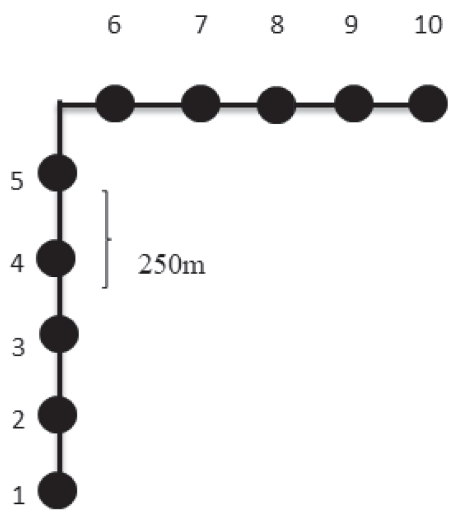


Fig. 2. NAFORMA cluster design (black solid circles = plot).

Concentric circular plots of 15 m radius were used as the sampling units and plots were grouped into clusters as a measurement unit. The distribution of deadwood sample plots within the entire mainland Tanzania is presented in Figure 1 whereby, the number of plots in a cluster varied from six to ten, depending on the accessibility of the plots. The distance between plots within a cluster was 250 m (Fig. 2) while the distance between clusters varied by stratum, from 10 to 45 km. Measurements were taken for fallen deadwood and large branches which were within the radius of 15 m plot. These included length and diameter (top and bottom) for deadwood with diameter equal or greater than 10 cm. Deadwood diameters were measured by using tree calliper and lengths were measured by using tape measure. Identification of deadwood species was done using forest botanists. Each deadwood decay status class (either solid or rotten) was detected using a knife test.

### Data analysis

All data were encoded in MS Excel and analysed by using R software, R v.4.1.1.

#### Deadwood carbon

Deadwood biomass was estimated as the product of deadwood volume and species-specific wood density. Volume was computed using Smalian formula and species-specific wood density values were sourced from the Global Wood Density database (Chave et al., 2009; Zanne et al., 2009), using the function getWoodDensity() in R. For those species-specific wood density values that were missing from the database, a default wood density value of 500 kg m$^{-3}$ was used (IPCC, 2006). Irrespective of species, a default wood density reduction factor of 0.97 was used for solid woods and 0.45 for rotten deadwood (IPCC, 2006). Deadwood biomass was converted into carbon stocks by multiplying with a carbon conversion factor of 0.47 (IPCC, 2006) and later aggregated into carbon stock density i.e. per ha for each plot.

#### Species community composition and deadwood diversity

All recorded deadwood species were identified, counted and sorted to obtain their abundance values and species composition. Deadwood species dominance was determined by estimating Species Importance Value Index (SIVI). SIVI was computed as the sum of relative density, relative dominance and relative frequency of the species per plot (Beals, 1984). Cluster analysis was done to identify distinct deadwood species communities across each forest and woodland ecosystem. We computed Bray–Curtis's distance matrix using deadwood species matrix data and then, hierarchical Ward's minimum variance clustering was performed using Ward's algorithm. Several clusters were identified based on Silhouette validation technique using function fviz_nbclust from the 'factoextra' R package (Thinsungnoena et al., 2015). Indicator species were identified using package 'labdsv', and then each community was named after the two most dominant species based on high synoptic cover-abundance values (Teshome et al., 2020). Shannon-Wiener diversity index (H'), evenness (J) and richness (S) were also calculated for each plot using the 'vegan' package.

Species Importance Value Index (SIVI) = Relative Density + Relative Dominance + Relative Frequency (Eqn. 1),whereby Relative Density is the number of individuals of a species divided by the total number of individuals of all species in the plot, Relative Dominance is the basal area of a species divided by the total basal area of all species in the plot, and Relative Frequency is the number of plots in which a species occurs divided by the total number of plots sampled.

**Acquisition of environmental data**

Climatic data (annual mean temperature and annual mean precipitation) were downloaded from WorldClim site (https://www.worldclim.org) with a 30 arc seconds resolution. For consistency with the NAFORMA, we used historical climate data from WorldClim version 2.1 for the period 1970–2000 and this version was released in January 2020. This period provides a long-term historical baseline, which is valuable for examining species distribution patterns relative to climate. The soil data (soil organic carbon, soil moisture, and soil texture) were extracted from the Re-gridded Harmonized World Soil Database: ISRIC Data (International Soil Reference and Information Centre) https://data.isric.org at a spatial resolution of 30 arc-seconds (approximately 1 km at the equator), which matches the resolution of the WorldClim climatic data we used. Consistent spatial resolution between soil and climate datasets allowed for accurate integration and comparison of environmental variables in our analyses. Values for all spatially interpolated climate and soil variables were extracted using QGIS software, sampled using the coordinates of the plot and averaged across plots for each forest and woodland ecosystem.

**Acquisition of disturbance data**

This study focused on disturbance gradients related to human proximity, using the distance to roads and settlements as proxies for human-induced disturbance levels. The rationale is that areas closer to roads and settlements are often subject to higher human activities (such as logging, forest clearing) that impact deadwood accumulation and carbon dynamics. We obtained spatial data for roads and villages as shapefiles using road and settlement vector data from OpenStreetMap (http://download.geofabrik.de/africa/tanzania.html). Values for all spatially interpolated distances (the minimum distances between each deadwood plot and the nearest road or settlement) were extracted using QGIS software and sampled using the coordinates for each forest and woodland ecosystem. To quantify the relationship between disturbance gradients and deadwood carbon stock, we used the minimum distance from the plot to the nearest road as well as the minimum distance from the plot to the nearest settlement, assuming that shorter distances indicate higher disturbance intensity.

**Associations between deadwood carbon stock and forest structure along environmental and disturbance gradients**

We determined ccollinearity among predictor variables i.e. forest structural variables (species abundance, evenness and Shannon-Wiener diversity index), environmental variables (annual mean precipitation, annual mean temperature, soil organic carbon and soil moisture) and disturbance variables (distance to the nearest road and settlement) by using the Variance inflation factor (VIF). Only predictor variable with VIF less or equal to 10 were retained for model fits (Chahouki and Zare Chahouk, 2010). Species richness and soil texture (sand, silt and clay) were removed from the model analysis due to high collinearity with other predictor variables. We used Multimodel inference approach to test the best variable combination out of all possible combinations and the final model was obtained after model averaging (Grueber et al., 2011). The predictor variables were categorized into 3 groups (Table 1), encompassing various environmental variables, disturbance gradients and forest structural diversity variables.

We fitted a global model using generalized additive model (gam) regression model with deadwood carbon

Table 1. Variable groups, variables within group and number of variables in each group

| Group | Variables within Group | Number of Variables |
|---|---|---|
| Forest structural variables | Species abundance, evenness and Shannon-Wiener diversity index | 3 |
| Environmental variables | Annual mean precipitation, annual mean temperature, soil organic carbon and soil moisture | 4 |
| Disturbance gradients variables | Distance to the nearest road and distance to the nearest settlement | 2 |

stock as a response variable against forest structural variables (deadwood species abundance, evenness and Shannon-Wiener diversity index), environmental variables (annual mean precipitation, annual mean temperature, soil organic carbon and soil moisture) and disturbance variables (distance to the nearest road and settlement) as predictor variables. Function dredge, implemented in the package 'MuMIn' (Barton, 2009) was used to generate a set of sub-models from the global model using the function get.models. Backward-forward stepwise model selection based on the Akaike Information Criterion (AIC) was done to identify optimal models from the global models (Zuur et al., 2009). A confidence set at a 95% cumulative weight was defined using the function *get.models* and the best final model was determined by model averaging using the function *model.avg* (Grueber et al., 2011). Estimation of variable importance when using this approach does not affect the accuracy of the model predictions (Koskikala et al., 2020). Thereafter, generalized linear models (GLMs) were employed to test for significant association between environmental variables and species community composition across the two ecosystems.

## Results

### Deadwood species composition, richness and diversity

A total of 528 deadwood species belonging to 224 genera and 77 families were recorded across all ecosystems. The most dominant families were Fabaceae (equivalent to number of recorded individuals 3,989), Caesalpiniaceae (equivalent to number of recorded individuals 2,717), Combretaceae (equivalent to number of recorded individuals 1,975), Phyllanthaceae (equivalent to number of recorded individuals 590) and Euphorbiaceae (equivalent to number of recorded individuals 514). The most common genera were *Brachystegia* (equivalent to individuals 2,237), *Combretum* (equivalent to individuals 1,071), *Pterocarpus* (equivalent to individuals 759), *Julbernardia* (equivalent to individuals 752), *Terminalia* (equivalent to individuals 536) and *Acacia* (equivalent to individuals 508). The most dominant deadwood species in terms of SIVI were *Brachystegia* sp. (6.66), *Julbernardia globiflora* (4.65), *Pterocarpus angolensis* (4.02), *Brachystegia spiciformis* (3.32) *Dalbergia melanoxylon* (2.63) and *Brachystegia bussei* (2.58), Table S1. These species contributed to 47.1% of the total weighted deadwood carbon stock (Table S1). In terms of each ecosystem, 199 deadwood species were unique to woodlands only, 128 deadwood species were found only in the forest ecosystem and 201 deadwood species occurred in both ecosystems. Higher deadwood species evenness was also observed in woodland (0.71) compared to the forest (0.66). Also, woodland had a higher Shannon-Wiener diversity value (4.47) compared to forest (4.12) (Table 2).

### Community composition

Three communities were identified from the forest ecosystem (Fig. 3A) while four communities were identified from woodlands (Fig. 3B), and most of the deadwood species were shared across communities. Species communities were named based on the two most important deadwood species that occurred in the community, using species indicator value (Table S2; Table S3). The identified communities from forests were *Anacardium occidentale-Rhizophora mucronata* (AR), *Brachystegia sp.-Pterocarpus angolensis* (BPa) and *Pteleopsis myrtifolia-Milleti*a sp. (PM), Fig- 3A. The four communities identified from the woodlands included *Brachystegia* sp.-*Diplorhynchus condylocarpon* (BD), *Dalbergia melanoxylon-Pteleopsis myrtifolia* (DP), *Julbernardia globiflora-Pterocarpus angolensis* (JP) and *Brachystegia spiciformis-Parinari excelsa* (BPe), Fig. 3B.

From the forest ecosystem, the highest weighted deadwood carbon stock (0.0088 t C ha$^{-1}$) was obtained from *Pteleopsis myrtifolia-Milletia* sp. (PM) community while the lowest deadwood carbon stock (0.000016 t C ha$^{-1}$) was from *Anacardium occidentale-Rhizophora mucronata* (AR) community (Fig. 4A). In woodland ecosystem the highest weighted deadwood carbon stock (0.01447 t C ha$^{-1}$) was obtained from *Brachystegia spiciformis-Parinari excelsa* (BPe) community while the lowest deadwood carbon stock (0.0000157 t C ha$^{-1}$) was from *Brachystegia* sp.-*Diplorhynchus condylocarpon* (BD) community (Fig. 4B).

### Associations between deadwood carbon stock and forest structure along environmental and disturbance gradients

Multimodel results showed that, species abundance, Shannon-Wiener diversity index, soil moisture and the proximity to roads had positive significant relationship with deadwood carbon stock in both ecosystems (Table 3; Fig. 5). Results also showed that in forest, deadwood carbon stock of *Brachystegia* sp.-*Pterocarpus angolensis* (BPa) community was positively significant associated with soil moisture (Table 4; Fig. 6A). However, in woodlands, deadwood carbon stock of *Julbernardia globiflora-Pterocarpus angolensis* (JP) community was negatively significant associated with soil organic carbon (Table 4; Fig.

Table 2. Species evenness and Shannon-Wiener diversity index in each ecosystem

| Ecosystem | Area (ha) | Number of plots | Species richness (S) | Species evenness (J) | Shannon-Wiener diversity index (H') |
|---|---|---|---|---|---|
| Forest | 3,364,400 | 173 | 128 | 0.66 | 4.12 |
| Woodland | 47,257,200 | 553 | 199 | 0.71 | 4.47 |
| Forest × woodland | | | 201 | | |

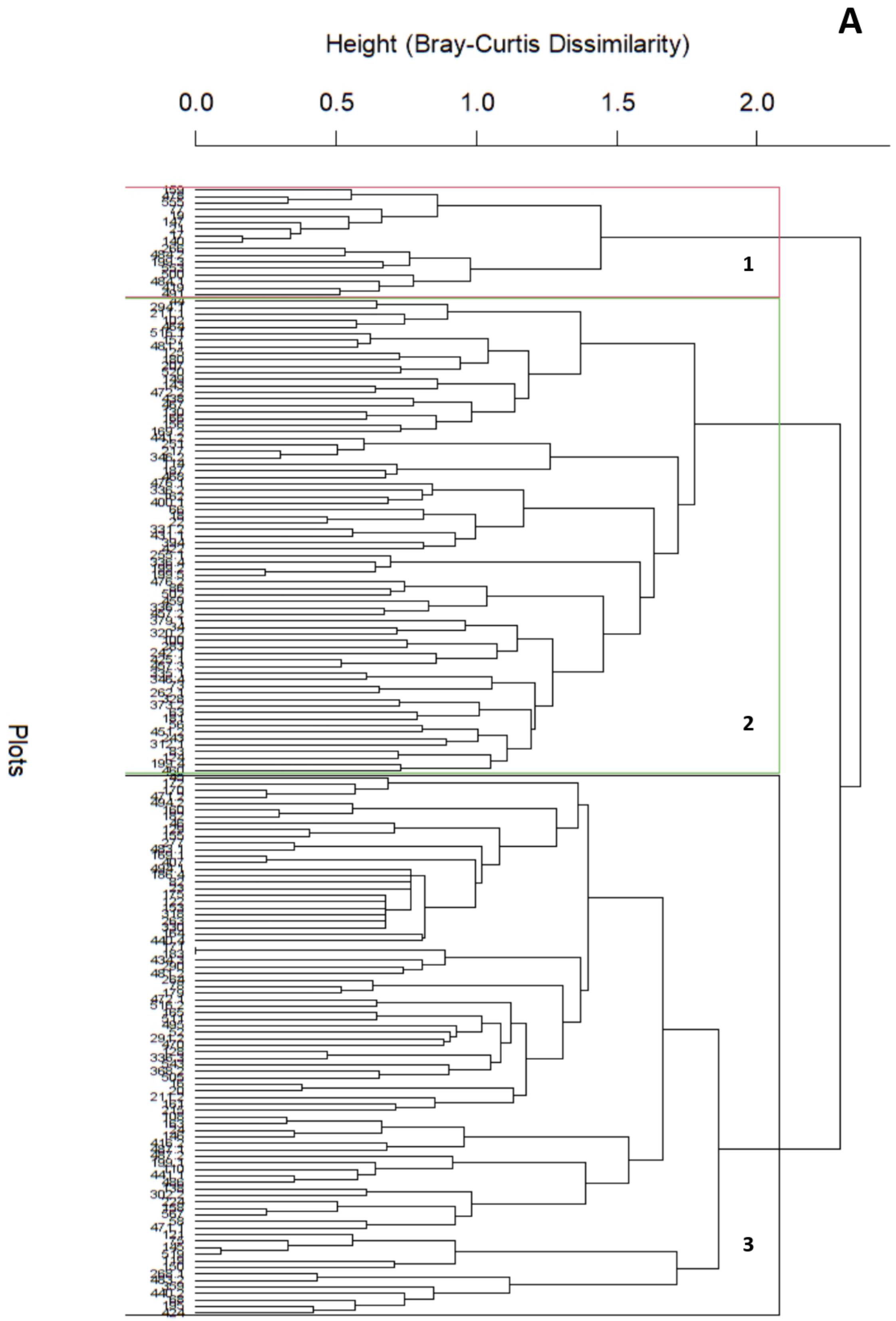
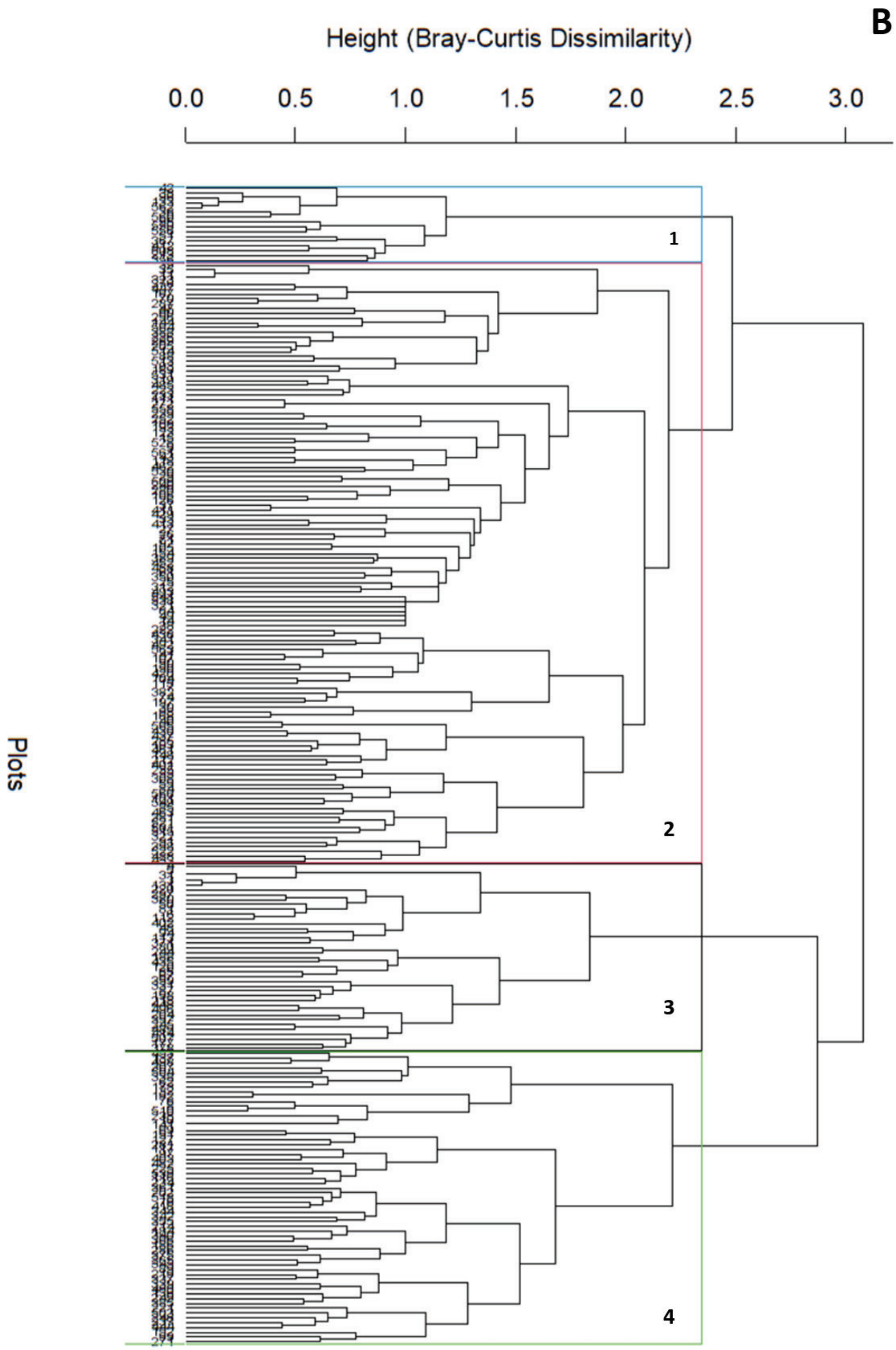



Fig. 3. Hierarchical dendrograms showing forest and woodland communities identified through clustering analysis. The x-axis represents plot codes while y-axis shows the dissimilarity index used for clustering. Figure 3A coloured rectangles indicate forest communities, 1. *Anacardium occidentale-Rhizophora mucronata* (AR), 2. *Brachystegia* sp.- *Pterocarpus angolensis* (BPa) and 3. *Pteleopsis myrtifolia-Milletia* sp. (PM). Figure 3B coloured rectangles indicate woodland communities, 1. *Brachystegia* sp.-*Diplorhynchus condylocarpon* (BD), 2. *Dalbergia melanoxylon-Pteleopsis myrtifolia* (DP), 3. *Julbernardia globiflora-Pterocarpus angolensis* (JP) and 4. *Brachystegia spiciformis-Parinari excels* (BPe).

Fig.3. – Continued



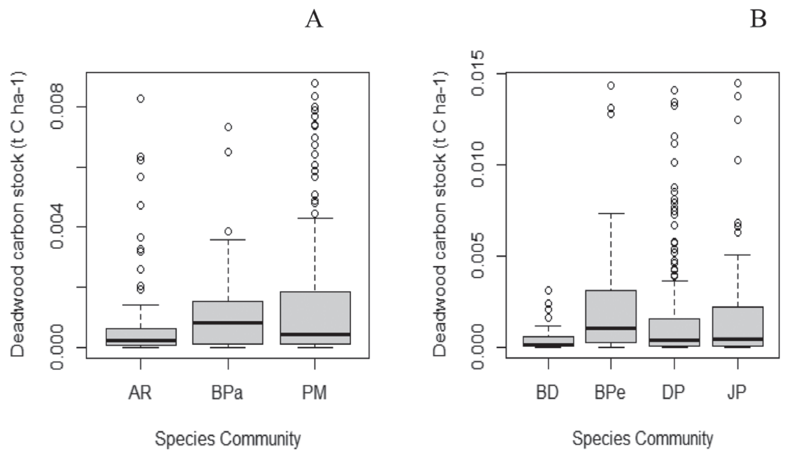



Fig. 4. Distribution of deadwood carbon stock among the identified species communities. Figure 4A from forests; *Anacardium occidentale-Rhizophora mucronata* (AR), *Brachystegia* sp.-*Pterocarpus angolensis* (BPa) and *Pteleopsis myrtifolia-Milletia* sp. (PM). Figure 4B from woodlands; *Brachystegia* sp.-*Diplorhynchus condylocarpon* (BD), *Dalbergia melanoxylon-Pteleopsis myrtifolia* (DP), *Julbernardia globiflora-Pterocarpus angolensis* (JP) and *Brachystegia spiciformis-Parinari excelsa* (BPe).

Table 3. A summary of averaged model estimates using multimodal inference approach

| Estimates | Value | Std.Error | z-value | p-value |
|---|---|---|---|---|
| (Intercept) | 0.002129 | 0.000595 | 3.566 | 0.00036 |
| Species abundance | 0.000126 | 0.000021 | 5.981 | <0.05 |
| Shannon-Wiener diversity index | 0.002381 | 0.000829 | 2.864 | ≤0.00419 |
| Soil moisture | 0.00048 | 0.000243 | 1.971 | ≤0.04876 |
| Distance to the nearest road | 0.00009 | 0.000051 | 1.762 | ≤0.07812 |

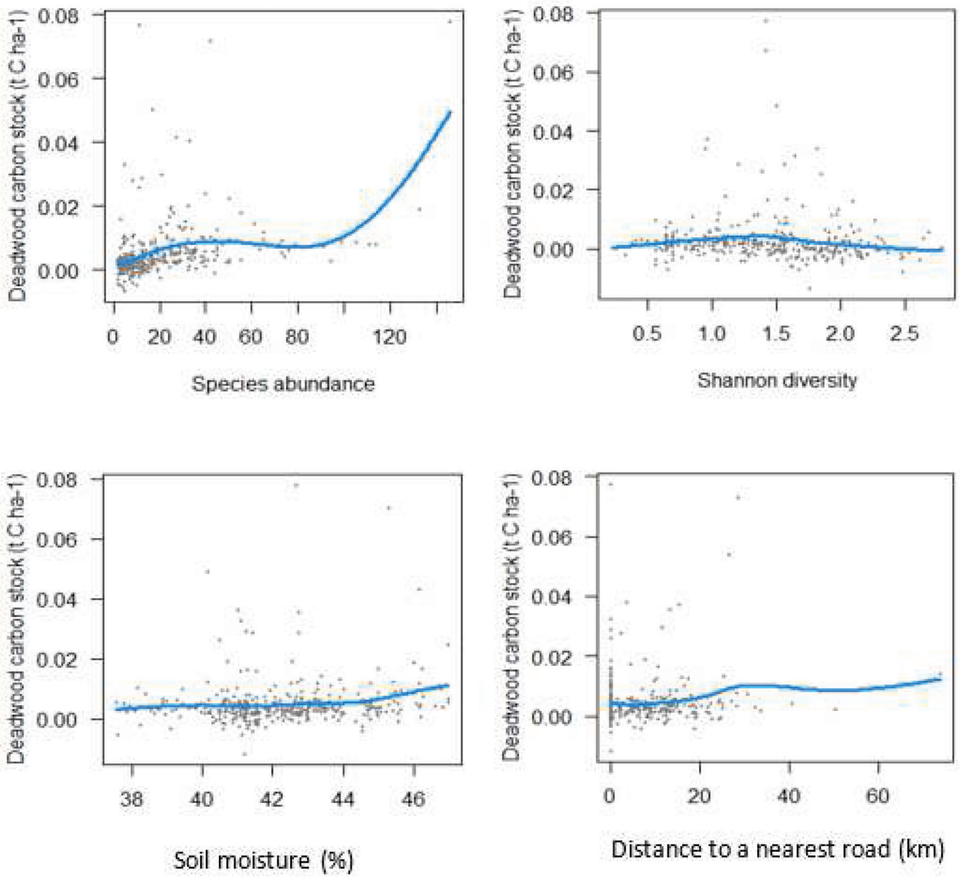



Fig. 5. Associations between deadwood carbon and species abundance, richness and soil organic carbon and soil moisture. Scatter points are raw data and the lines are predictions from the optimal averaged generalized additive model when other predictors are kept constant.

Table 4. A summary of models showing association between environmental variables and deadwood species communities across different vegetation types

| Vegetation type | Parameter | Estimate | Std.Error | t-value | p-value |
|---|---|---|---|---|---|
| Forest (soil moisture) | (Intercept) | 43.009 | 0.152 | 238.4 | <0.05 |
| | Community BPa | 0.701 | 0.368 | – 1.906 | <0.05 |
| | Community PM | 0.031 | 0.263 | – 0.119 | 0.905 |
| Woodland (soil moisture) | (Intercept) | 41.987 | 0.164 | 256.4 | <0.05 |
| | Community BPe | 0.044 | 0.272 | – 0.163 | 0.871 |
| | Community DP | 0.404 | 0.185 | – 2.181 | ≤0.03 |
| | Community JP | 0.369 | 0.211 | – 1.752 | 0.08 |
| Woodland (soil organic carbon) | (Intercept) | 1.835 | 0.0701 | 26.179 | <0.05 |
| | Community BPe | – 0.0073 | 0.1165 | – 0.632 | 0.528 |
| | Community DP | – 0.0436 | 0.0794 | – 0.549 | 0.584 |
| | Community JP | – 0.175 | 0.0902 | – 1.944 | <0.05 |

6B) while deadwood carbon stock of *Dalbergia melanoxylon-Pteleopsis myrtifolia* (DP) community was positively significant associated with soil moisture (Table 4; Figure 6C).

## Discussion

The assemblages of species in ecological communities reflect interactions among organisms as well as between organisms and the abiotic environment. Species composition, especially the dominance of species from the families Fabaceae and Caesalpiniaceae, is a characteristic feature of woodlands in tropical ecosystems, including Tanzania, similar to the findings reported by (Giliba et al., 2011; Mwakalukwa et al., 2014). The higher values of species richness observed in this study, compared to previous studies in similar tropical woodland and forest ecosystems (Girmay et al., 2020; Jew et al., 2016), could be attributed to the greater sampling effort employed as well as ecological processes influencing species richness in these ecosystems. Additionally, the disturbance gradient quantified through proximity to roads and settlements may have influenced deadwood diversity, as human activities and environmental pressures can create heterogeneous conditions favourable for diverse deadwood substrates. However, we acknowledge that other ecological processes, such as tree mortality patterns and decomposition dynamics, though relevant, were not directly assessed in this study and warrant further investigation in future research. The difference in deadwood species richness observed between forests and woodlands where forests exhibited higher species richness compared to woodlands can be attributed to several factors. Forest ecosystems generally have more complex habitat structures, providing diverse microhabitats and supporting a wider variety of species. In contrast, woodlands, characterized by more open canopies and less structural complexity, may support fewer species overall. Additionally, environmental factors such as soil moisture and organic carbon content, which were found to differ significantly between ecosystems in this study, likely contributed to these variations. Similar patterns have been reported by Mwakalukwa et al. (2014), highlighting the role of environmental heterogeneity in shaping species richness across different ecosystems.

The Shannon-Wiener diversity values presented in this study are in line with other studies by (Girmay et al., 2020) who reported Shannon-Wiener diversity values ranging from 3.25 to 4.21 but, they are much higher than those of Shirima et al. (2011) who reported the Shannon-Wiener diversity values ranging from 1.05–1.25. High Shannon-Wiener diversity values reported in this study could be attributed to the country-wise coverage of this study as it included a very large sample size. Normally, Shannon-Wiener diversity values range between 1.5 and 4.5 and rarely exceed 5 (Magurran, 2013). A threshold value of 2 has been indicated as minimum value, above which an ecosystem can be regarded as medium to highly diverse (Magurran, 2013). Therefore, the Shannon-Wiener diversity values obtained in this study implies that woodland and forest in Tanzania are highly diverse ecosystems. Nevertheless, the observed positive significant relationship between the Shannon-Wiener diversity index and deadwood carbon stock across ecosystems as illustrated in Fig. 5, highlights a general trend where ecosystems with more diverse species tend to have higher deadwood carbon stocks. However, this relationship may not be consistently positive or linear in every instance. This apparent inconsistency could be due to ecosystem-specific variations such as differences in species composition, disturbance regimes, environmental conditions and management practices which might dampen the observable trend in the figure (Schuldt et al., 2023). For instance, some ecosystems might have dominant species with low wood density or fast decomposition rates, reducing deadwood carbon stock despite high diversity.

The identification of distinct communities provides a deeper understanding of the ecological dynamics and their unique characteristics. The variation in deadwood carbon stock among forest and woodland communities underscores the importance of different species as carbon sinks (Fig. 4A and Fig. 4B), standing out as a particularly significant carbon reservoir. The cause of variation in deadwood carbon stock among species communities could be attributed to different micro-climates (Shirima et al., 2011), site-specific environmental conditions and disturbance history (Garbarino et al., 2015; Woodall et al., 2008). Moreover, the study by (Błońska et al.,

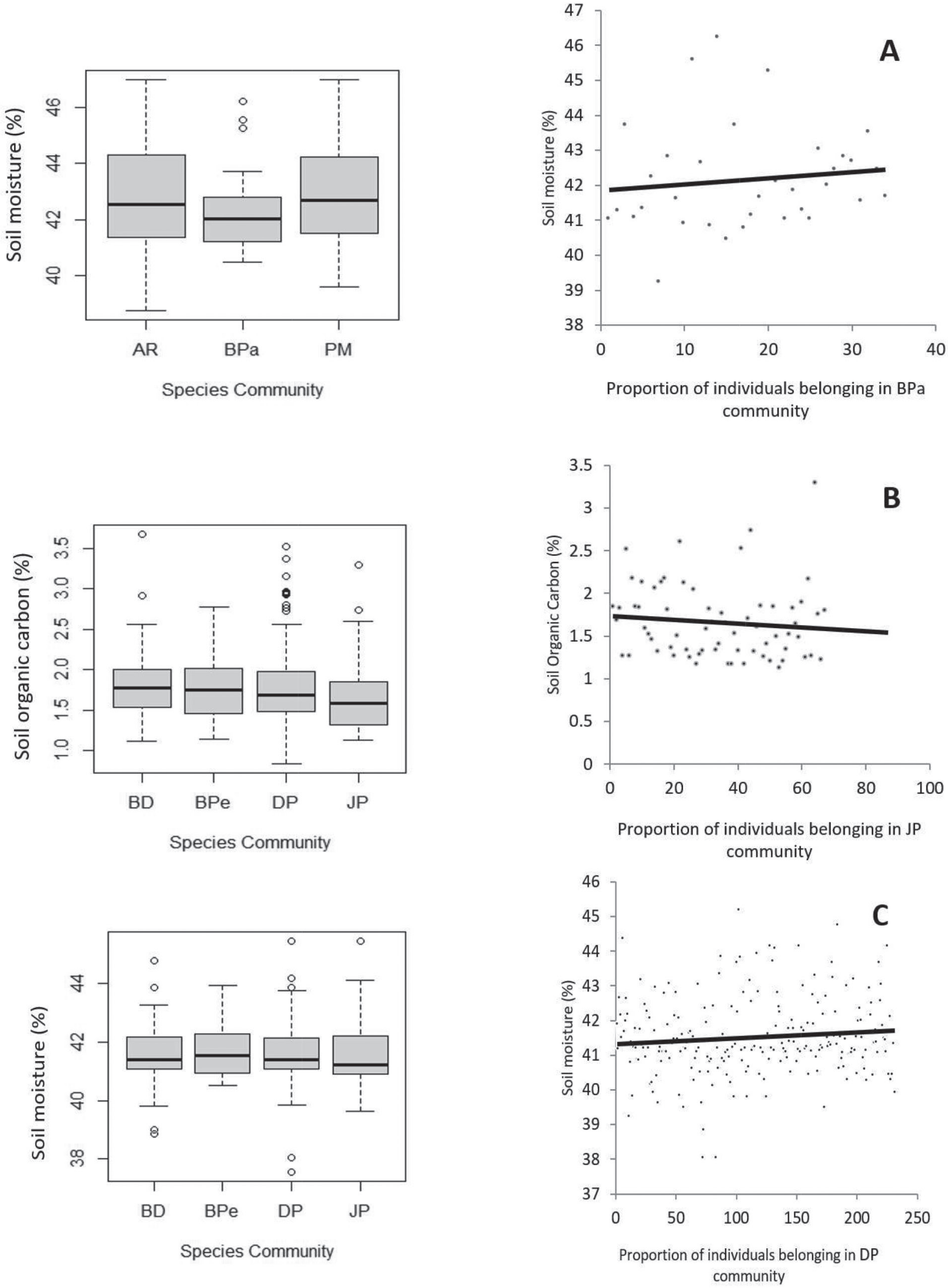



Fig. 6. A) association between *Brachystegia* sp.-*Pterocarpus angolensis* (BPa) community and soil moisture; B) association between *Julbernardia globiflora-Pterocarpus angolensis* (JP) community and soil organic carbon; C) association between *Dalbergia melanoxylon-Pteleopsis myrtifolia* (DP) community and soil moisture.

2019) discussed that, different species may lead to variations in resistance traits and nutrient limitations which slow down the rate of decomposition.

The findings also revealed contrasting associations between deadwood carbon stock and soil properties across different forest and woodland communities. In forests, higher soil moisture was positively associated with deadwood carbon stock, likely because increased moisture slows microbial decomposition, allowing deadwood to persist for longer periods (Garbarino et al., 2015). Additionally, the study by (Brockerhoff et al., 2017), showing that structurally heterogeneous forest stands, characterized diverse size classes and species mixtures provide more niches for deadwood accumulation. In contrast, within woodland communities, deadwood carbon stock showed negative relationships with soil organic carbon, and positive relationships with soil moisture. These contrasting relationships highlight how the more open canopy and resource-limited conditions in woodlands influence the decomposition and persistence of deadwood differently than in forests (Błońska and Lasota, 2017; Błońska et al., 2019). During the decomposition of deadwood, a portion of the carbon is released into the atmosphere as carbon dioxide through microbial respiration, while the remaining carbon is incorporated into the soil organic matter. This process results in a gradual reduction in the total amount of deadwood present in the ecosystem and a corresponding increase in soil organic carbon content.

Nevertheless, there was a notable pattern of higher deadwood carbon values in proximity to roads in both ecosystems (Fig. 5). This pattern could reflect the influence of increased tree mortality rates along transport corridor edges due to anthropogenic disturbances (Austin, 2002). However, it is important to note that the history and management status of forests in this study were not explicitly documented, making it challenging to differentiate between unmanaged forests and those subject to prior anthropogenic activities. In unmanaged forests, illegal harvesting, selective logging and firewood collection would likely reduce deadwood amounts by directly removing biomass (Masek et al., 2011). In contrast, managed forests or areas with historical disturbances could exhibit an accumulation of fine woody debris following timber extraction or felling operations (Kechagioglou et al., 2022). These dynamics suggest that the observed pattern near roads may result from a combination of human activities and legacy effects of past disturbances, which could contribute to both coarse and fine deadwood fractions depending on the intensity and type of disturbance (Mwakosya and Mligo, 2014). Additionally, the edge effects associated with roads can create microclimatic conditions that may stress trees, making them more susceptible to insect infestations, pathogens or drought, further contributing to increased tree mortality and deadwood accumulation near transportation corridors (Kacholi, 2014).

## Conclusion

Understanding species distribution patterns and community structures within forest and woodland ecosystems highlights their ecological significance as potential indicators of ecosystem health and carbon sequestration capacity. The distribution of deadwood carbon stocks, species diversity and communities were shown to be associated with soil moisture, soil organic carbon and proximity to roads. These findings have implications for ecosystem management and conservation emphasizing the need to maintain biodiversity and suitable habitat conditions to sustain deadwood carbon stocks. Proper management strategies including buffer zones or controlled access can promote the long-term health and resilience of these valuable ecosystems and their vital ecosystem services.

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## Supplementary material

The Supplementary material for this article can be found online at: https://goo.su/rzNPk6.

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