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High Mammal Diversity and Flagship Species Persist Under Community Conservation in a Forest-Savannah Transition Zone in Central Cameroon

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ABSTRACT

Community conservation areas often classified as council forests play an important role in the persistence or maintenance of diverse mammal communities. However, these areas often receive limited conservation attention and resources. The Yoko Council Forest (YCF) is among the key biodiversity hotspot areas found in Cameroon. Located in the forest-savannah transition zone, YCF is managed under a community conservation regime which is ranked low within the national Protected Area hierarchy. Although several conservation activities, including community conservation, are ongoing, the diversity of wildlife needs to be updated in order to support conservation management. To this end, we conducted systematic camera trapping in YCF to assess the status and diversity of mammals. We also used binomial generalised linear models to evaluate factors affecting species richness. From 6499 independent photographic events obtained over 17,981 camera trap days, the study documented 38 terrestrial mammal species, including flagship species identified on the IUCN Red List; the endangered Nigeria-Cameroon chimpanzee (*Pan troglodytes ellioti*), two endangered pangolin species (giant-ground pangolin (*Smutsia gigantea*) and white-bellied pangolin (*Phataginus tricuspis*)), and two vulnerable carnivore species (African golden cat (*Caracal aurata*) and crested genet (*Genetta cristata*)). The most common species were blue duiker (*Philantomba monticola*), African brush-tailed porcupine (*Atherurus africanus*) and bay duiker (*Cephalophus dorsalis*). The species rarefaction curve indicates that our efforts were sufficient to record the majority of species present in the YCF. Species richness increased during the long rainy and dry seasons, but decreased during the short dry season. The response of mammalian species to ecological and anthropogenic covariates varied. For example, canopy height was positively associated with species richness, whereas distance to villages did not. Understanding how mammals respond to these factors provides insight into developing

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conservation strategies that support both wildlife persistence and community needs. Regarding the diversity of wildlife found in YCF, this area contributes at a larger scale to the global 30 ×30 initiative through sustainable management and community engagement.

1 | Introduction

We live in the age of the sixth extinction (Turvey and Crees 2019), and Cameroon is no exception to the countries that host exceptional biodiversity where conservation remains insufficient (Awazi 2025). Environmental transformation, driven by factors such as habitat loss and fragmentation, poaching, logging, and urbanisation (Bakala and Mekonen 2021; Fotsing, Kamkeng, and Zinner 2024) has led to significant impacts on biodiversity. With increasing human population densities, farmers colonise new areas for farmlands, leading to destruction of wildlife habitats (Fa et al. 2003). The dependence of local people on forest resources for their subsistence and protein needs causes high pressure on wildlife (Njoroge et al. 2009; Wilkie et al. 2016). Terrestrial mammals, a highly diverse group including ungulates, pholidotes, rodents, primates and carnivores, play a crucial role in shaping ecosystems through diverse ecological pathways (Lacher et al. 2019). However, the overexploitation of these species has far-reaching effects on ecosystems, altering the forest structure and environmental functioning (Abernethy et al. 2013; Rosin and Poulsen 2016). This highlights the profound consequences of biodiversity loss.

Cameroon has currently designated 6.80% of its land as National Park (NP), which is the strictest form of Protected Areas (PAs), where conservation is the primary objective (Annuaire statistique MINFOF 2018). However, achieving the global target of 30% PAs by 2030 (30 ×30), poses a significant challenge particularly if community conservation areas as Other Effective area-based Conservation Measures (OECMs) (Alves-Pinto et al. 2021; Jonas et al. 2024; Robinson et al. 2024) are not clearly adopted and implemented on the ground across the country. These community conserved areas are crucial not only for meeting the 30 ×30 target but also for achieving goals outlined in National Action Plans for charismatic species. In Cameroon, legislation allows for the classification of community lands as council forests, providing a framework for community-led conservation initiatives. In this country, council forests are part of the formal PA architecture, but almost any subsistence activity is allowed and they are therefore functionally equivalent to community conservation areas that are internationally seen as IUCN category 6. In such areas, community conservation aims to find a balance between preserving biodiversity and supporting local livelihoods. However, effective monitoring is essential to ensure that this balance is maintained and conservation objectives are met (Alves-Pinto et al. 2021).

Forest-Savannah Mosaic (hereafter, FSM) habitats, also known as transition forests, are habitats that occur in all major continental tropical regions of the world, particularly in Africa, South America and Southeast Asia (Dantas et al. 2016; Pletcher et al. 2022). These mosaic habitats are characterised by the coexistence of patches of different land cover types, within which different habitat types occur, including savannah, open and closed canopy forests (Pletcher et al. 2022). FSM plays an important role in biodiversity conservation as they constitute a biodiversity hotspot (Araújo 2002) and are important areas where speciation can occur due to divergent selection (Smith et al. 1997). Wildlife monitoring in these habitats is crucial not only for assessing species distributions, but also as an important conservation tool (Hedwig et al. 2018).

Camera traps (CTs) have emerged worldwide as an essential tool in wildlife research (Rovero et al. 2013). These devices are particularly useful for studying in difficult-to-access, remote or unknown areas (e.g., YCF) (Bessone et al. 2020; Fotsing, Kamkeng, Marcel Senge, and Zinner 2024; Fotsing et al. 2025; Fotsing and Meigang 2025). Camera traps can be used to assess various ecological parameters, including occupancy (see Fotsing and Meigang 2025), species' responses to environmental conditions, which can help to develop sustainable management strategies (Nichols and Williams 2006; Rovero et al. 2010). A key parameter that can be assessed using CTs is species richness, which is useful for protected area managers as it helps them to understand local variations in species numbers and therefore facilitates adaptive management (Ghimirey et al. 2024). However, in a habitat where pressure is constantly increasing, such as in transitional landscapes, species richness can be influenced by various environmental factors (DeGroot et al. 2023; Fotsing and Meigang 2025; Teixeira-Santos et al. 2020; Vanthomme et al. 2013). It is therefore crucial to understand how species respond to environmental factors in these environments as previous studies highlight the fact that species living in transitional landscapes may become increasingly maladapted to their environment or less resilient to ongoing habitat disturbances they face (Eckert et al. 2008; Sexton et al. 2009). Similarly, environmental heterogeneity and inter-species interactions also play a key role in shaping ecological communities (Stein et al. 2014), and the use of CTs can also provide a broad overview of these dynamics.

The YCF is a community conservation area in the transitional forest-savannah landscape between the rainforest located in the south and the Guinean savannah in the north. This FSM ecosystem harbours a diverse mammal community, including the Endangered Nigeria-Cameroon chimpanzee (*Pan troglodytes ellioti*, Matschie, 1914) and the Near Threatened African buffalo *Syncerus caffer* (Kirsten et al. 2021). However, scientific knowledge about the YCF landscape is limited, particularly the wildlife which is poorly studied; therefore limiting the ability of the government and site manager to effectively implement sustainable management measures in the area. In fact, YCF faces several threats, including the expansion of agriculture, transhumance, poaching, illegal logging, fishing and the collection of Non-Timber Forest Products (NTFPs), which are increasingly unsustainable (Kamgang et al. 2023; Kirsten et al. 2021). As a starting point to support conservation efforts made in the area, a preliminary survey using line transects has been carried out to assess the abundance and distribution of mammals (Kirsten et al. 2021). However, due to the limitations of these methods, most of the wildlife in the area remains unknown. According to the literature, camera traps (CTs) are a non-invasive and efficient method of capturing most of the wildlife present in an area, as demonstrated by recent studies (Fotsing, Kamkeng, Marcel Senge, and Zinner 2024, 2025; Simo et al. 2023) exhibiting similar ecological conditions to those in the YCF. Similarly, several other studies using CTs have been applied in Cameroon specifically, in the Dja Faunal Reserve, Boumba-Bek, Nki, and Lobéké NPs (Bruce et al. 2018; Hongo et al. 2020; Poulain et al. 2023) but were mainly in the rainforest. In the forest-savannah transition zones of Cameroon, Simo et al. (2023) used the CTs to improve the effectiveness of the detection of white-bellied pangolins in Mpem & Djim and Deng-Deng NPs, while Fotsing et al. (2025) and Fotsing and Meigang (2025) used CTs, to assess mammal richness and occupancy as well as environmental factors affecting detection. Measuring biodiversity is challenging due to sampling effects, but Hill numbers offer a unified metric for diversity comparisons that account for species abundance and sample completeness. Using rarefaction and extrapolation based on sample size or coverage, modern methods produce seamless curves that standardise diversity estimates across different datasets. We conducted this study to: (a) provide a robust database on wildlife diversity; (b) improve the general understanding of the factors affecting richness; (c) help conservation managers and the government achieve robust results for adaptive conservation.

The aim of this study is therefore twofold: (i) to assess the species richness and diversity of large and medium-sized mammals, and (ii) to identify environmental factors that could affect the species richness of these mammals. We hypothesised that community-conserved areas in central Cameroon would support a high mammal species richness. We therefore predicted that our extensive camera trapping efforts made would be sufficient to capture the majority of species present in the site. Based on the findings of Fotsing et al. (2025) and Fotsing and Meigang (2025), we also hypothesised that this species richness would be shaped by several environmental factors such as habitat, human disturbance, and seasonality. We further predicted that habitat structure like forest/savannah and canopy cover as well as seasonality could be positively associated with mammal species richness, whereas human disturbance like proximity to roads and villages could be negatively associated with species richness. We discuss our findings in the context of the potential contribution of OECMs to contribute to the global target for biodiversity conservation.

2 | Methods

2.1 | Study Area

The YCF (4°53′-5°1′ N/12°5′-12°20′ E) is located in the Centre Region, Mbam et Kim Division (Figure 1). It is an ecotone area of 295 km^2 with an altitude ranging between 579 and 633 m above sea level (Massoh et al. 2024). This area is situated in the south-east of Yoko Subdivision and was classified as a Council Forest by Decree N°2011/0038/PM. This classification suggests that the area is subject to some level of conservation management, aimed at balancing human activities with ecosystem protection. The YCF benefits from local management and conservation initiatives, which contribute to its ecological integrity and biodiversity conservation. Such initiatives have been implemented in this area since 2021 by several NGOs through community-based conservation, education and outreach programs (CWCI; GIZ-Cameroon, unpublished report). The Yoko landscape is characterised by a transitional climate between tropical and equatorial with four seasons: a long dry season (LDS) lasting four months from mid-November to mid-March, a short rainy season (SRS) from mid-March to the end of June, a short dry season (SDS) from July to August and a long rainy season (LRS) from September to mid-November

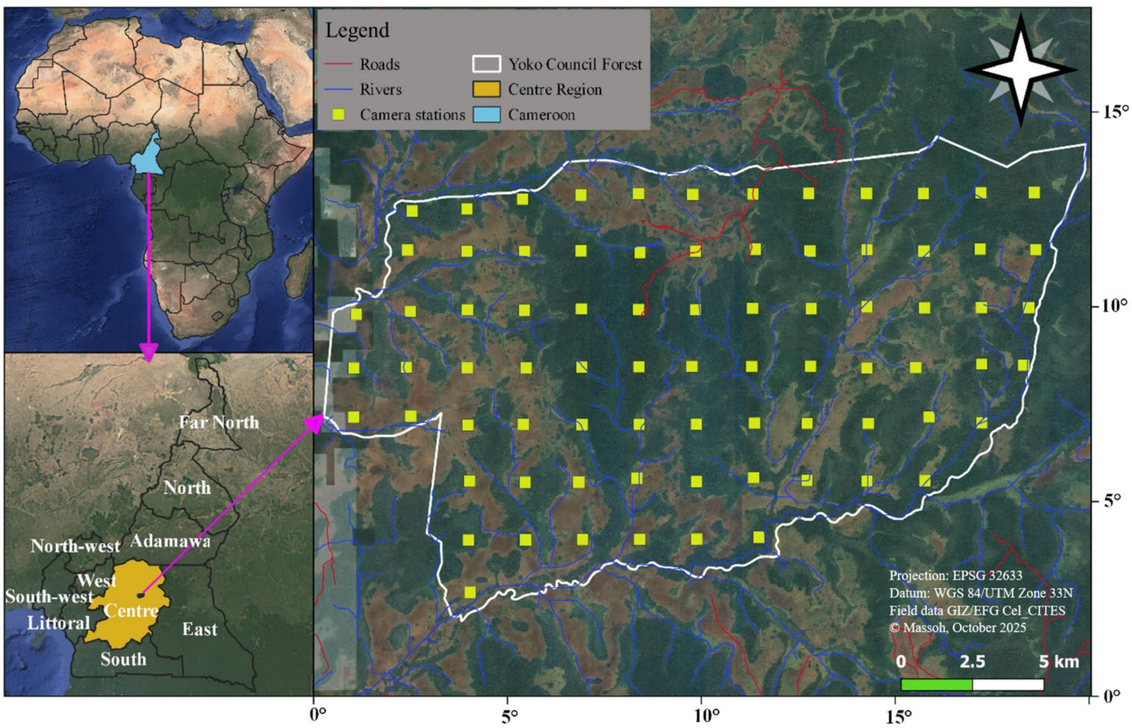

FIGURE 1 | Map showing the camera trap locations in the Yoko Council Forest, Cameroon.

(Suchel 1988; Tsalefac et al. 2000). Average annual rainfall varies between 1800 and 2000 mm, while the temperatures range from 22°C to 29°C (Tsalefac et al. 2003). This forest is characterised by a mosaic of forest-savannah habitats; it presents phytogeographical units belonging to two subsets: the Guinean- Sudanese savannah sector, and the semi-deciduous forest sector (Letouzey 1985). Subsistence agriculture, hunting, fishing and harvesting of NTFPs are among the main activities of the local people living around the study area.

2.2 | Survey Design

From November 2021 to November 2022, we conducted a camera trap survey aimed at detecting large and medium-sized terrestrial mammal species. We deployed 78 Victure HC300 Trail camera traps based on a systematic grid across the entire 295 km^2 with a grid cell size of 2×2 km and covering all habitat types of the YCF (Figure 1). Each location of CTs was recorded using a handheld Global Positioning System (GPS) receiver. Along with the camera coordinates, various environmental covariates were also documented, including, habitat, visibility, anthropogenic activity around the camera, and season. This was then documented on the camera setup/recovery sheet created for this purpose. Each camera was placed at 2 km intervals based on the estimated home ranges of most large and medium-sized mammal species (Ahumada et al. 2011). We searched for good sites for camera establishment and suitable locations for camera deployment were identified within a 200–300 m radius of the centroid of each grid cell. Once a good place was found, CTs were set to target animal trails showing clear signs of current use by any wild animals such as the presence of fresh droppings or footprints, feeding sites marked by the recently flattened grass on either side of the trail. Cameras were attached to trees at a height of 30–40 cm above the ground level suitable for capturing large and medium-sized terrestrial mammals (Bruce et al. 2018) and set perpendicular to animal trails at a distance of 4–8 m from the target, to obtain full-body lateral images of the animal (Ahumada et al. 2011). Cameras were oriented either north–south or south–north to minimize the effect of sunrise and sunset on the image quality (Bruce et al. 2018). All grass in the camera field was removed to avoid false triggers. Cameras were set up to operate 24 h/day and programmed to take three photos per trigger event, with a 5 s delay between successive triggers. The cameras remained in the field for three months and were checked at the end of the three months to replace batteries, and SD memory cards, and to retrieve faulty CTs if necessary. At the end of this activity, the CTs were left in the field for the next 3 months of data collection. This process was thus repeated every three months during the 13 months of sampling.

2.3 | Species Identification and Data Processing

The Wild ID software was used to extract metadata (GPS coordinate, date, time of each image) from each picture for identification. Animal images were identified to species level using the Kingdon field guide to African mammals (Kingdon 2013, 2015). We introduced a threshold delay of 60 min to address the problem of multiple non-independent events (McPhee 2015; Tobler et al. 2008), and to avoid potential bias from recounting. If identification was not possible, the photo was recorded as 'unidentified'. All blank photographs were recorded as 'blank'. Due to technical problems, three CTs were removed, and data from 75 CTs were used for analysis.

2.4 | Data Analysis

2.4.1 | Rarefaction and Sample Completeness Curves

Interpreting the sample size-based rarefaction and extrapolation curves will help to understand how metrics of species diversity (such as species richness, Shannon diversity or Simpson diversity) change with increasing effort or sample size. Species diversity metrics were estimated, sample-size-based (including rarefaction and extrapolation curves), coverage-based rarefaction and sample completeness curves were generated using the iNEXT package (Chao et al. 2014; Hsieh et al. 2016), version 3.0.0 with R version 4.3.2 (Core Team 2022). This package is used to estimate and compare species diversity using Hill numbers or the effective number of species (Chao et al. 2014), covering species richness, Shannon diversity and Simpson diversity. The package integrates rarefaction and extrapolation techniques based on sample size or coverage, allowing more accurate diversity comparisons between different assemblages.

The features (see table 1 in Hsieh et al. (2016)) of the iNEXT package have been widely used in many research fields, including Eren et al. (2016) for archaeological data; Kendrick et al. (2015) for ant data; Uchida and Ushimaru (2015) for grassland plant and herbivorous insect data; and Mateo- Tomàs et al. (2015) for vertebrate scavenger data. Specifically, we estimated species richness ($q = 0$), Shannon diversity ($q = 1$) and Simpson diversity ($q = 2$) for rarefied and extrapolated samples with 95% confidence intervals. To ensure robust comparisons of sample completeness, we used coverage-based rarefaction and extrapolation sampling curves, to plot diversity estimates against sample coverage, which ranges up to twice the reference sample. We used sample-size-based curves to show the estimated diversity as a function of the number of individuals, while the rarefaction curve represents what the diversity would look like if we only had a smaller sample than the one actually collected (which generally allows a fair comparison between samples of different sizes in general), and the extrapolation part predicts what the diversity would look like if we sampled more or up to a larger sample size (often up to twice or three times the original sample size, which is based on an asymptote estimate). We further investigated whether our samples are likely to have captured most of the diversity by assessing the trend of the sample completeness curves. To assess species richness in relation to sampling effort, we also constructed a species accumulation curve using camera-trap days as the measure of effort. This approach accounts for variation in sampling duration across sites and allows a direct evaluation of how species richness accumulates with increasing camera-trap effort. To achieve this task, we used the specaccum() function from the vegan R package (Oksanen et al. 2020) to compute species accumulation curves. This function calculates the expected cumulative species richness as sampling units are added randomly, with 100 permutations to estimate standard deviations of richness. The resulting curves were plotted with the *x*-axis representing cumulative camera-trap days (total effort) and the *y*-axis representing accumulated species richness, allowing us to evaluate the adequacy of sampling and the rate at which new species were detected with increasing effort.

2.4.2 | Assessing of Diversity Indices

Alpha diversity was assessed using three commonly used indices (species richness (S), Shannon index (H) and equitability index (J)), to compare diversity between vegetation types and seasons. We first applied a normality test using the Shapiro–Wilk test and performed ANOVA and/or Kruskal–Wallis test at the 5% threshold to compare the alpha diversity indices between seasons and vegetation types. We considered N as the total number of species.

2.4.3 | Relative Abundance Index

The relative abundance index (RAI) was calculated as the number of events of a given species divided by sampling effort (the total number of trapping days), multiplied by 100. RAI value ranges from 0 with no upper bound. The species was considered most frequently recorded when its RAI value was high, RAI ≥ 0.5 and least commonly detected when its RAI value was low, RAI < 0.5 (Hedwig et al. 2018).

2.4.4 | Environmental and Ecological Variables Used for Modelling

To model the species richness within the study area, we used 13 biotic and abiotic covariates (see Appendix S1 for covariates description), including season, cameras days (i.e., survey effort per cameras), distance to the nearest anthropogenic activity, habitat, and visibility, obtained during data collection and slope, elevation, topographic position index (TPI), terrain ruggedness index (TRI), canopy height, distance to river, distance to road and distance to village, derived from various sources (as described below) using remote sensing. First, we obtained canopy height data for each CTs location using the Global Canopy Height dataset, which is derived from the Global Ecosystem Dynamics Investigation (GEDI) mission (Potapov et al. 2021). Second, we obtained topographic data in GeoTIFF format by calculating TPI, TRI and slope in R using elevation data from a Shuttle Radar Topography Mission 30 m resolution Digital Elevation Model (Jarvis 2008). We then projected the Coordinate Reference System (CRS) of each raster image mentioned above to the CRS of our CTs locations and extracted the corresponding pixel value for each location using R (Core Team 2022). Finally, we used Google Earth images to digitise roads, rivers and villages, and then measured the distance to each feature as a Euclidean distance using Google Engine code after which we georectified the Google Earth images using GPS points collected from recognisable locations in the field, ensuring better alignment with ground data (see Fotsing and Meigang 2025). Normalised Difference Vegetation Index (NDVI), a widely used vegetation index in ecological studies due to its correlation with biodiversity (Gould 2000; Pettorelli et al. 2011) was computed to measure vegetation greenness. To achieve this, GeoTIFF Landsat Collection 2 Surface Reflectance satellite images for the dry season (22 December 2023) were obtained from the United States Geological Survey (USGS), ensuring no cloud contamination (Vermote et al. 2016) (while ensuring that each band downloaded contained less than 5% of cloud-affected pixels, cloud-affected pixels were later identified and removed using the quality assessment (QA_Pixel) band, ensuring only clear-sky observations were retained for analysis). The NDVI calculation was therefore performed in Python v3.12.0 using the formula available in Borowik et al. (2013).

2.4.5 | Modelling of the Species Richness

We fitted a Generalised Linear Model (McCullagh and Nelder 1989) to model the species richness (number of species at each camera's trap location) as a function of several predictor variables. In these models, we used species richness as a response variable and a set of key predictors as described above in Section 2.4.4. Slope, elevation, TPI, and TRI were included as environmental predictors to evaluate their influence on species richness while accounting for their potential confounding effects as underlying topographic gradients. Prior to the analysis, predictors such as vegetation type, season, and visibility were converted into categorical factors (reference levels: savannah, SRS and close respectively) to allow the model to estimate differences between groups and to include interaction effects. Continuous predictors were retained as numeric variables, which allows the exploration of potential non-linear relationships through transformations or spline-based approaches if needed. We also applied the log transformation to effort and the square root to slope, TRI, TPI, distance to river, distance to village and distance to road to achieve an approximately symmetrical distribution as these covariates were skewed. Similarly, all continuous predictors were z-transformed to a mean of zero and a standard deviation of one to facilitate model convergence and achieve easier interpretable coefficients (Schielzeth and Forstmeier 2009). To avoid fitting correlated variables in the model, we first used the 'corr' function from the corrplot package (version 0.95) Wei and Simko (2021) to assess the Pearson correlations between pairs of covariates with a correlation threshold set at 0.7 as in (Schober et al. 2018). Correlated covariates (Dem in our case only) were removed (we calculated the Variance Inflation Factor (VIF) for each variable and predictors with VIF > 5 were sequentially removed, starting with the variable contributing least to ecological interpretability, until all remaining variables had VIF values below the threshold (Zuur et al. 2010)) until we achieved a value under 0.7 for all pairs of covariates left (see Appendix S2, Figure S2b).

Once overdispersion, multicollinearity, normality and z.transformation were achieved, we fitted a Poisson regression model with all non-correlated (hereafter. NCP) predictors but found evidence of overdispersion ($\hat{C} > 1$) after an overdispersion test performed using the function AER from the package AER (version 1.2.9 (Kleiber and Zeileis 2018)). To address this and evaluate whether the effect of one predictor on species richness depended on another predictor (e.g., habitat × visibility), we tested biologically meaningful interactions using Negative Binomial regression using the function glm.nb from the package MASS (version 7.3-60.2 (Venables and Ripley 2022). Three interaction models were fitted: (1) all NCPs plus the interaction between habitat and visibility, (2) all NCPs plus the interaction between habitat and NDVI, and (3) all NCPs plus the interaction between visibility and NDVI. Each interaction model was compared with a reduced model containing only main effects to assess whether the interaction significantly improved model fit. This stepwise approach allowed us to detect potential context-dependent effects while maintaining ecological interpretability. We then used a Likelihood Ratio Test (LRT) to compare each interaction model vs. reduced model (see Fotsing and Meigang 2025). The result showed that the inclusion of the interaction term in the full model did not significantly (with p-value always > 0.05) improve the model fit compared to the reduced model, indicating that the interaction was not significant. We then applied model selection using the dredge function from the MuMin Package (version 1.48.11, Bartoń 2025) on the reduced model that included all NCP and selected the top-ranked model based on Akaike's Information Criterion (AIC) (Akaike 1973) using the AIC function of the package AICcmodavg (Mazerolle 2025). After calculating the Delta AIC (ΔAIC) and AIC weights (AICω) from the calculated AIC values, we considered the model with the lowest Delta AICc as the top-ranked model (Burnham and Anderson 2002). We then assessed model performance by applying the goodness-of-fit Chi-squared test (Gof) to compare the top-ranked model with the null model which assumes no predictors but only intercept. To quantify how changes in key covariates affect the rate of species detections, we exponentiate the coefficient of the top-ranked model to obtain incidence rate ratio (IRR) which can offer an interpretable measure of effect size (IRRs; Kirkwood and Sterne 2003; Zou 2004). We then calculated 95% confidence intervals (CIs) for the IRRs using the standard errors of the coefficients. Similarly, we obtained 95% confidence intervals of model estimates and fitted values by means of a parametric bootstrap ($N = 1000$ bootstraps). For model diagnostic, we checked model stability using DFBetas which revealed the model to be of good stability. Also, we assessed the Generalised Variance Inflation Factors (Fox and Monette 1992) with a threshold set at 10 (O'brien 2007) using the function 'vif' of the package 'car' (version 3.0-13; Fox and Weisberg 2019) and this revealed no collinearity issues (max (GVIF^(1/(2*Df))) = 1.045). Residual plots, including Q-Q plots and residual versus fitted values plots, confirmed that the model fit assumptions were reasonably met.

All data analysis was performed using R version 4.3.2 (Core Team 2022) and Python version 3.12.

3 | Results

3.1 | Effect of Sampling-Size: Rarefaction and Extrapolation Curves

Data was collected over a total of 17,981 days/nights at the 75 independent CT locations which yielded 6499 independent events. The analysis of the sample-size-based curve (the estimated diversity as a function of the number of individuals or samples collected, see Figure 2a) for all diversity orders shows that the curve is rising towards an asymptote suggesting that the diversity estimate is approaching the true or maximum diversity. Since all the curves flatten out, it indicates that most species or diversity components have likely been captured with the current sampling effort. Similarly, the coverage-based curve (Figure 2b) which plots the diversity metrics as a function of sample coverage, estimates the proportion of the total community diversity captured in the sample and indicates how complete the sampling is. For all diversity orders, the sample coverage value is close to 1 and plateaus suggesting that, our sample likely captures most of the diversity, indicating adequate sampling. In our case, the coverage curve approaches 1 (or very high values), indicating that the sample is quite comprehensive, and further sampling is unlikely to reveal many more species, providing confidence that diversity estimates are reliable.

3.2 | Species Richness

Overall, 38 species of large and medium-sized terrestrial mammals belonging to six taxonomic orders and 16 families were recorded (Table 1). The most diverse order was Cetartiodactyla (34.2% of the total number of species), which included three families and 13 species. This was followed by the order Carnivora (23.7%), which was represented by four families and nine species, the order Primates (18.4%) with three families and seven species and the order Rodentia (15.8%), with four families and six species. The least diversified taxa were Pholidota (5.3%) represented by a single family and two species and Tubulidentata (2.6%) with a single family and one species.

3.3 | Relationship Between Species Richness, Habitat and Season

The mean species richness varied significantly between the vegetation types ($H = 35.06$; $df = 4$; $p < 0.001$) (Table 2). Mammal species richness was highest in secondary forests (23.09 ± 1.63 species), with swamps supporting slightly fewer species (19.55 ± 1.90) (Table 2). Regarding the seasons, the mean species richness has varied between the different seasons but this variation was not statistically significant ($F = 0.94$; $p = 0.45$) (Figure 3a). A similar observation was found between forest and savannah (Figure 3b, when we summarised species richness obtained in different types of habitats in the two habitat types (Forest vs. Savannah).

3.4 | Community Assemblage and RAI

The community of mammals in the YCF was mainly composed of four guilds. The most abundant guild was herbivores (RAI = 25.6; 15 species), followed by omnivores (RAI = 7.7; 17 species) and these guilds had the highest richness. Carnivores (RAI = 1.5; 3 species), and insectivores (RAI = 1.0; 3 species) were the least abundant and had the lowest richness. In all guilds, RAI was higher for species with lower body mass, compared to species with higher body mass.

The most frequently captured was the blue duiker (*Philantomba monticola*), followed by the African brush-tailed porcupine (*Atherurus africanus*) and the bay duiker (*Cephalophus dorsalis*). The least commonly detected species were the red-legged sun

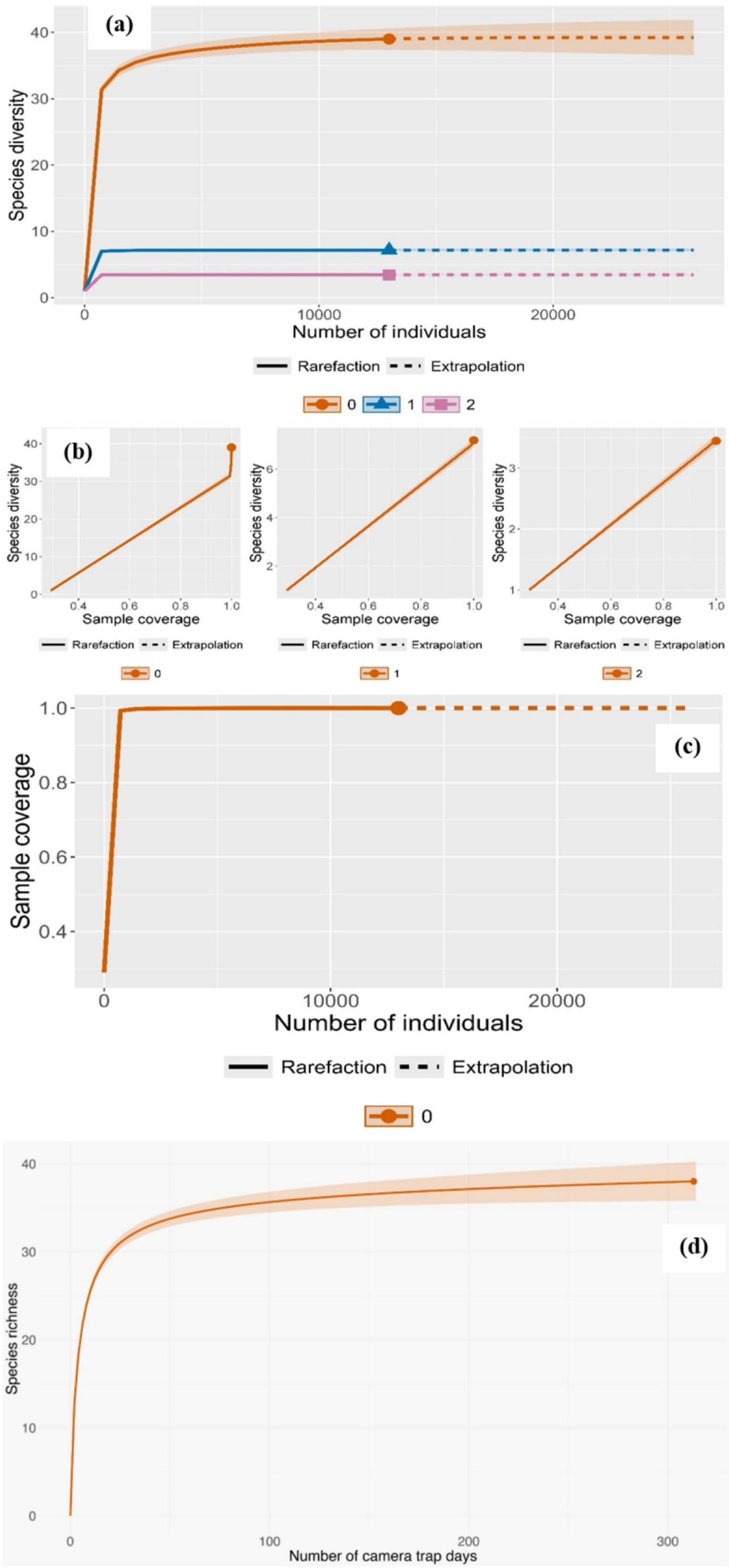

FIGURE 2 | Legend on next page.

FIGURE 2 | (a) Sample-size-based and (b) coverage-based rarefaction (solid lines segments) and extrapolation (dotted lines segments) sampling curves with 95% confidence intervals (orange shaded areas for the YCF mammal's data separated by diversity order: $q=0$ (species richness, (a) orange line), $q=1$ (Shannon diversity, (a) blue line) and $q=2$ (Simpson diversity, (a) pink line). The solid dot/triangle/square represent the reference sample (up to double). (c) Sample completeness curves linking curves (a) and (b). (d) The figure illustrates the species accumulation curve derived from camera-trap records collected during the sampling period. The orange line represents the mean cumulative number of species detected as camera-trap sampling effort (expressed as the number of camera-trap days) increases. The orange shaded areas indicate the ±1 standard deviation around the mean, showing the variability in species richness among random sampling permutations. The curve approaches an asymptote, suggesting that most of the detectable species within the study area were recorded with the given sampling effort.

squirrel (*Heliosciurus rufobrachium*), the African giant squirrel (*Protoxerus stangeri*), the common warthog (*Phacochoerus africanus*), the Mona monkey (*Cercopithecus mona*), and the Milne-Edwards's potto (*Perodicticus potto*) (Table 1).

Among the carnivores, the African civet (*Civettictis civetta*) was the most common species, followed by the long-nosed mongoose (*Xenogale naso*) and the servaline genet (*Genetta servalina*). Among the ungulate species with a high RAI (RAI ≥ 0.5), duikers (*Philantomba monticola* and *Cephalophus* spp.), sitatunga (*Tragelaphus spekii*) and red river hog (*Potamochoerus porcus*) were the most common. Among the primate species, the Tantalus monkey (*Chlorocebus tantalus*) and the Nigeria-Cameroon chimpanzee were the most common. African brush-tailed porcupine and Emin's pouched rat (*Cricetomys emini*) were the most common rodent species. Five threatened species were recorded, including the chimpanzee, and the giant pangolin (*Smutsia gigantea*), the white-bellied pangolin (*Phataginus tricuspis*), African golden cat (*Caracal aurata*) and the crested genet (*Genetta cristata*), with a low RAI value for the latter four species (Table 1 and Figure 4).

3.5 | Effect of Environmental Covariates on Mammal Species Richness

The modelling process with 13 NCP (i.e., $2^{13}-1=8192-1=8191$) possible models showed that the 75th model (ΔAICc$=0$, see Table 3) fits well compared to all other models fitted together. This model includes season, canopy height and distance to village as the main covariates affecting species richness. It also showed a better fit to the data than the null model ($p<0.001$). Species richness was significantly influenced by some key predictors such as seasons, distance to villages and canopy height (Table 4). Indeed, compared to SRS, LRS (GLM: B 0.292, $p=0.0365$) and LDS (GLM: B 0.354, $p=0.022$) were positively correlated with species richness, while SDS (GLM: B −0.351, $p=0.0243$) was negatively correlated. Similarly, canopy height was also positively correlated with species richness (GLM: B 0.088, $p=0.043$) and distance to villages was negatively associated with species richness (Table 4). The relationship between the response variables and the predictors was also plotted (Figure 5).

Intercept was estimated at 2.225 (this is the log-expected value of richness when all predictors are at their reference level (e.g., Season is baseline season, and z-scored covariates are zero)), suggesting that we have $e^{2.25}=9.26$ species. However, compared to the reference season (SRS), species richness increased by 34% in the long rainy season (LRS, IRR$=1.34$, $p=0.03$) and 42% in the long dry season (LDS, IRR$=1.42$, $p=0.021$), while it decreased by 30% in the short dry season (SDS, IRR$=0.7$, $p=0.024$). Similarly, a one standard deviation increase in canopy height was associated with a 9% increase in species richness (IRR$=1.09$, $p=0.04$), whereas increasing distance from villages led to an 8% decrease in species richness (IRR$=0.91$, $p=0.04$) (Table 5 and Figure 6).

4 | Discussion

Species richness is an essential status indicator of regional biotic communities. Given the rapid pace of environmental change around the world, it is crucial to establish baselines to efficiently monitor ecological communities, which we have done here for the YCF.

As hypothesized, the species accumulation curve demonstrates that sampling effort was sufficient to detect the majority of species within the study area (with a sampling coverage value of approximately 1). This suggests that we were able to compile a maximum diversity for our sampled community. Combined with the rarefaction, extrapolation and sample coverage curves analysis, this highlighted the efficiency of CTs as an effective technique for sampling elusive species. We were able to compile a complete inventory of 38 mammal species, consistent with the observations of Hedwig et al. (2018). This similarity may be attributed in part, to the comparable length of the sampling periods and potentially similar mammal community composition, given that both studies were conducted in forest-savannah transition zones that likely share analogous ecological characteristics and species assemblages. However, Fotsing et al. (2025) recorded 27 species, in Mpem and Djim NP, located in a similar landscape in Cameroon. The differences in species richness can be attributed to several factors, including differences in sampling effort, seasonality, and survey design. Specifically, the previous study was limited to a 3-month period in the eastern part of MDNP, covering approximately 20% of the park's area, whereas our study spanned a long period (13 months), encompassing all four seasons of the year and surveyed the entire study area.

Consistent with the study by Derebe et al. (2022) and Diriba et al. (2020) and the prediction we made, forest habitat had a higher species richness than savannah habitat, partially explained by the heterogeneity and understory cover, which provides shelter for most mammals (Agebo and Tekalign 2022; Geleta and Bekele 2016). Grassland savannahs contain homogeneous plant species that provide few foraging opportunities for large mammals (Geleta and Bekele 2016), limiting specialisation and competition among generalists (Cramer and Willig

TABLE 1 | IUCN conservation status, number of events (NE), and the relative abundance index (RAI) of large and medium-sized mammals recorded in the Yoko Council Forest.

Taxonomic order	Family	Common names	Scientific names	Number of events	Relative abundance index	IUCN status
Carnivora	Felidae	African golden cat	*Caracal aurata* (Temminck, 1827)	34	0.19	VU
Carnivora	Herpestidae	Marsh mongoose	*Atilax paludinosus* (Cuvier, 1829)	65	0.36	LC
Carnivora	Herpestidae	Black-legged mongoose	*Bdeogale nigripes* (Pucheran, 1855)	78	0.43	LC
Carnivora	Herpestidae	Long-nosed mongoose	*Xenogale naso* (de Winton, 1901)	169	0.94	LC
Carnivora	Nandiniidae	African palm Civet	*Nandinia binotata* (Gray, 1830)	50	0.28	LC
Carnivora	Viverridae	African civet	*Civettictis civetta* (Schreber, 1776)	202	1.12	LC
Carnivora	Viverridae	Crested genet	*Genetta cristata* (Sanborn, 1940)	20	0.11	VU
Carnivora	Viverridae	Blotched genet	*Genetta maculata* (Gray, 1830)	76	0.42	LC
Carnivora	Viverridae	Servaline genet	*Genetta servalina* (Pucheran, 1855)	104	0.58	LC
Cetartiodactyla	Bovidae	Peter's duiker	*Cephalophus callipygus* (Peters, 1876)	118	0.66	LC
Cetartiodactyla	Bovidae	Bay duiker	*Cephalophus dorsalis* (Gray, 1846)	527	2.93	NT
Cetartiodactyla	Bovidae	Red flanked duiker	*Cephalophus rufilatus* (Gray, 1846)	294	1.64	LC
Cetartiodactyla	Bovidae	Yellow- backed duiker	*Cephalophus silvicultor* (Afzelius, 1815)	229	1.27	NT
Cetartiodactyla	Bovidae	Defassa waterbuck	*Kobus ellipsiprymnus* (Ogilby, 1833)	19	0.11	LC
Cetartiodactyla	Bovidae	Kob	*Kobus kob* (Erxleben, 1777)	8	0.04	LC
Cetartiodactyla	Bovidae	Blue duiker	*Philantomba monticola* (Thunberg, 1789)	2377	13.22	LC
Cetartiodactyla	Bovidae	African buffalo	*Syncerus caffer* (Sparrman, 1779)	15	0.08	NT
Cetartiodactyla	Bovidae	Bushbuck	*Tragelaphus scriptus* (Pallas, 1766)	31	0.17	LC
Cetartiodactyla	Bovidae	Sitatunga	*Tragelaphus spekii* (Speke, 1863)	358	1.99	LC

(Continues)

Taxonomic order	Family	Common names	Scientific names	Number of events	Relative abundance index	IUCN status
Cetartiodactyla	Suidae	Common warthog	*Phacochoerus africanus* (Gmelin, 1788)	2	0.01	LC
Cetartiodactyla	Suidae	Red river hog	*Potamochoerus porcus* (Linnaeus, 1758)	130	0.72	LC
Cetartiodactyla	Tragulidae	Water chevrotain	*Hyemoschus aquaticus* (Ogilby, 1841)	14	0.08	LC
Pholidota	Manidae	White-bellied pangolin	*Phataginus tricuspis* (Rafinesque, 1820)	46	0.26	EN
Pholidota	Manidae	Giant pangolin	*Smutsia gigantea* (Illiger, 1815)	46	0.26	EN
Primate	Cercopithecidae	Tantalus monkey	*Chlorocebus tantalus* (Ogilby, 1841)	137	0.76	LC
Primate	Cercopithecidae	Mona monkey	*Cercopithecus mona* (Schreber, 1774)	4	0.02	NT
Primate	Cercopithecidae	De Brazza's monkey	*Cercopithecus neglectus* (Schlegel, 1876)	27	0.15	LC
Primate	Cercopithecidae	Greater spot-nosed guenon	*Cercopithecus nictitans* (Linnaeus, 1766)	50	0.28	NT
Primate	Cercopithecidae	Anubis baboon	*Papio anubis* (Lesson, 1827)	54	0.30	LC
Primate	Hominidae	Nigeria Cameroon chimpanzee	*Pan troglodytes* (Blumenbach, 1776)	117	0.65	EN
Primate	Lorisidae	Milne-Edwards's Potto	*Perodicticus edwardsi* (Bouvier, 1879)	6	0.03	NT
Rodentia	Hystricidae	African brush-tailed porcupine	*Atherurus africanus* (Gray, 1842)	557	3.10	LC
Rodentia	Nesomyidae	Emin's pouched rat	*Cricetomys emini* (Wroughton, 1910)	350	1.95	LC
Rodentia	Sciuridae	Red-legged rope squirrel	*Funisciurus pyrropus* (Cuvier, 1833)	80	0.44	LC
Rodentia	Sciuridae	Red-legged sun squirrel	*Heliosciurus rufobrachium* (Waterhouse, 1842)	1	0.01	LC
Rodentia	Sciuridae	African giant squirrel	*Protoxerus stangeri* (Waterhouse, 1843)	2	0.01	LC
Rodentia	Thryonomyidae	Marsh cane rat	*Thryonomys swinderianus* (Temminck, 1827)	19	0.11	LC
Tubulidentata	Orycteropodidae	Aardvark	*Orycteropus afer* (Pallas, 1766)	83	0.46	LC

Abbreviations: EN, endangered; LC, least concern; NT, near threatened; RAI, relative abundance index; (number of independent events of a species divided by the number of trapping days and multiplied by 100); VU, vulnerable.

TABLE 2 | Averages of species richness and mammal diversity indices per vegetation type and season in the YCF.

Parameters		Observed species richness	Number of events	Shannon–Weawer (H')	Pielou equitability (E)
Vegetation types	Grassland savannah	14 (4.83 ± 1.54)	56 (9.33 ± 3.18)	2.35 (1.16 ± 0.33)	0.89 (0.91 ± 0.01)
	Secondary forest	35 (23.09 ± 1.63)	3387 (307.91 ± 52.63)	2.17 (1.98 ± 0.11)	0.61 (0.64 ± 0.03)
	Shrubland savannah	27 (9.2 ± 1.44)	316 (31.6 ± 7.17)	2.47 (1.79 ± 0.14)	0.75 (0.86 ± 0.03)
	Swamp	32 (19.55 ± 1.90)	2403 (218.45 ± 47.54)	2.58 (2.32 ± 0.08)	0.74 (0.80 ± 0.02)
	Woodland savannah	33 (11.18 ± 2.29)	337 (30.64 ± 7.80)	2.97 (1.96 ± 0.18)	0.85 (0.89 ± 0.02)
Statistical test	Test	$H = 35.06$	$H = 38.6$	$H = 23.83$	$H = 19.04$
	p-value	$p < 0.001$	$p < 0.001$	$p < 0.001$	$p = 0.001$
Seasons	Long rainy season	32 (25 ± 4.16)	720 (240 ± 80.31)	2.49 (2.36 ± 0.11)	0.72 (0.74 ± 0.01)
	Long dry season	35 (22.75 ± 7.49)	2646 (661.5 ± 294.09)	2.65 (1.95 ± 0.65)	0.75 (0.57 ± 0.19)
	Short rainy season	36 (30 ± 1.08)	2291 (572.75 ± 94.33)	2.47 (2.41 ± 0.21)	0.69 (0.71 ± 0.06)
	Short dry season	34 (29.5 ± 3.5)	842 (421 ± 116)	2.31 (2.22 ± 0.13)	0.65 (0.66 ± 0.02)
Statistical test	Test	$F = 0.94$	$F = 2.04$	$H = 3.98$	$H = 4.19$
	p-value	$p = 0.45$	$p = 0.16$	$p = 0.26$	$p = 0.24$

Note: Averages are presented as follows [global indices value (mean index ± standard error)].
Abbreviations: H, Kruskal–Wallis test; F, one-way ANOVA test.

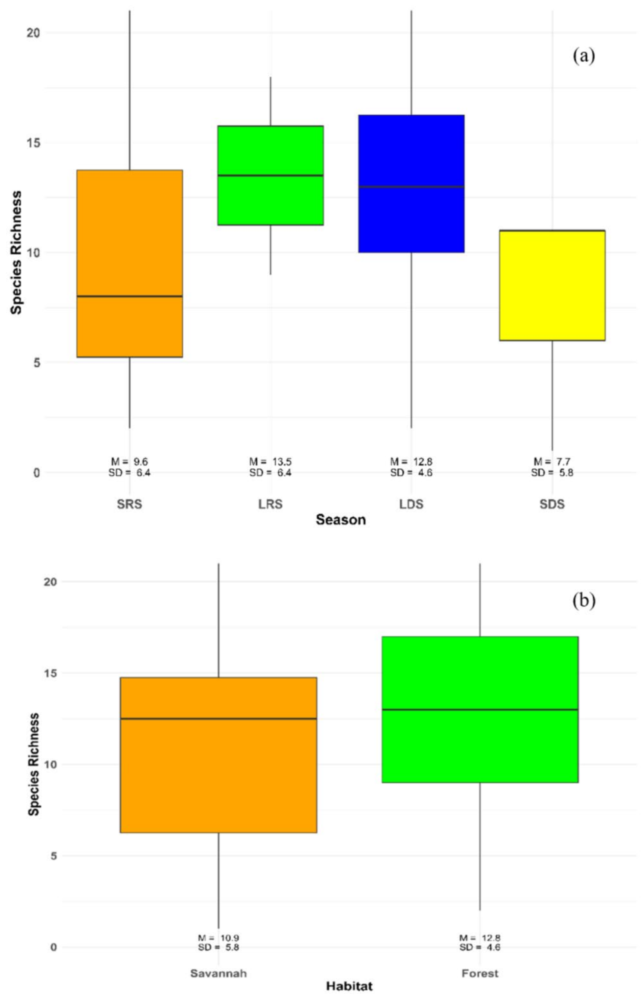

FIGURE 3 | Variation of species richness in each season (a) and habitat (b).

2002). No significant variation in species richness was observed between the seasons, in line with the findings of Desalegn et al. (2023) in the Debre Libanos Monastery Forest (central Ethiopia) and Mirghani et al. (2025) in the Dindefelo Nature Reserve (south-east Senegal) who did not find any difference in species richness between the dry season and wet season. This suggests a homogeneity of species richness across seasons, which is typical in transition forests with a habitat mosaic.

Duikers and other ungulates, as well as some rodents were the most frequently detected of all taxa recorded, corroborating other findings obtained in Central African forests (Bruce et al. 2018; Hedwig et al. 2018; Poulain et al. 2023). This can be explained by the high abundance and distribution, habitat preference and high activity pattern of these species as recorded by Simo (2024) in the Deng Deng and Mpem et Djim NPs, Cameroon. The frequent detection of these species observed in our study may also be related to their adaptability to living in forest–savannah mosaics and their ability to thrive in this area which provides a mix of forest and grassland habitats, and where they can exploit the vegetation and shelter offered by these habitats (Estes 2012; Happold 2013; Kingdon 2013). As these species are known to be abundant and widespread in mosaic landscapes (East 1988), they can occur in high densities, making them more likely to be detected. The high detection of these species could also be attributed to predator–prey dynamics. Indeed, the presence in low densities of their major predator, the African golden cat, makes them insufficient to exert strong predation pressure on their prey's populations, although they have a wide dietary range that includes smaller duiker species, rodents, and primates (Bahaa-el-din et al. 2015). Nevertheless, blue duiker and many other small mammals are a prized bushmeat species, and the high detection rates are indicative of the relative effectiveness of conservation in YCF. Some species such as squirrels and arboreal primates had low detection rates, possibly due to

FIGURE 4 | Camera trap images of the threatened species recorded in Yoko Council Forest: (a) *Pan troglodytes ellioti*, (b) *Caracal aurata*, (c) *Smutsia gigantea*, (d), and *Phataginus tricuspis*, (e) *Genetta cristata*.

TABLE 3 | Model selection using Akaike's Information Criterion (AIC) here the top 5 models.

Model number	Model structure	K	logLik	AICc	ΔAICc	AICω
75	**Richness ~ Season + z.s.Cheigh + z.s.DistVillage**	**7**	**−225.07**	**465.82**	**0.00**	**0.19**
11	Richness ~ Season + z.s.Cheigh +1	6	−226.63	466.49	0.67	0.14
9	Richness ~ z.s.Cheigh +1	3	−230.24	466.82	1.01	0.12
67	Richness ~ Season + z.s.DistVillage +1	6	−227.05	467.33	1.51	0.09
1	Richness ~1	2	−231.64	467.45	1.63	0.09

Note: The best-performing models (75) having the delta AICc value of zero. Columns indicate the number of parameters in the model (K), negative log-likelihood (logLik), the difference between the AICc values and the top-ranked model (ΔAICc), and the AICc model weights (AICω). The best models are in bold and the max GVIF^(1/(2*Df)) is 1.045 confirming no collinearity.

TABLE 4 | Summary of all significant predictor variables from the best models.

GLM output							DFBeta	
Fixed effect	***B***	**SE**	***t*.value**	***p***	**conf.low**	**conf.high**	**Min**	**Max**
Intercept	2.225	0.147	15.181	0[a]	1.934	2.514	2.085	2.315
SeasonLRS	0.292	0.321	0.907	$p<0.001$	−0.348	0.933	−0.244	0.664
SeasonLDS	0.354	0.154	2.295	$p<0.001$	0.051	0.66	0.262	0.49
SeasonSDS	−0.351	0.301	−1.167	$p<0.001$	−0.955	0.235	−0.893	−0.174
Cheigh	0.088	0.043	2.027	$p<0.001$	0.002	0.174	0.071	0.118
DistVillage	−0.088	0.049	−1.798	$p<0.001$	−0.185	0.009	−0.116	−0.073

Abbreviations: *B*, estimate; Cheigh, canopy height; DistVillage, distance to village; LDS, long dry season; LRS, long rainy season; SDS, short dry season; SE, standard error.
[a]Mean that the intercept is statistically significantly different from zero.

our methodology approach (terrestrial camera trapping), which was designed to target ground-dwelling mammals as in Mpem et Djim NP, Cameroon (Fotsing et al. 2025). Indeed, these species usually use microhabitats that are not well represented by the survey design such as canopy, and liana, which favour the species' movements through the area (Li et al. 2022). This is consistent with the findings of Beukes et al. (2025) and Marion et al. (2024) which highlight the inefficiency of camera traps to detect species if they are positioned in areas with low activities of these species. The arboreal camera trap method could be used to improve the detection of these arboreal species in the future, as suggested by Difouo et al. (2023).

Herbivores were most abundant among all guilds, similar to previous work (Ahumada et al. 2011; Bruce et al. 2018; Hedwig et al. 2018). As the first consumers in the food chain, herbivores are an important link in shaping the structure and composition of forest plant species (Owen-Smith 1988). Through their dispersal capacity, these ecological engineers participate in the regeneration process of forests and degraded landscapes through the process of seed dispersal (Stoner et al. 2007). This is the case for frugivores, particularly the chimpanzee which was recorded in our study. Across all guilds small-bodied mammals had higher capture rates than large-bodied mammals, as found by Bruce et al. (2018). In fact, unlike the large mammals (chimpanzee, buffalo, sitatunga, Peters' duiker, and red river hog) that are the target of commercial hunting (Fa et al. 2005; Kamgang et al. 2023), small-bodied species such as the blue duiker, brush-tailed porcupine, Emin's pouched rat, red-flanked duiker, etc. are known to be more resilient (van Vliet et al. 2023).

Several species of high conservation concern have been recorded, highlighting the importance of the YCF as a habitat for rare and endangered species, and making this area a priority site for conservation. Among these species, the presence of the Nigeria–Cameroon chimpanzee is of critical importance, as it is classified as Endangered on the IUCN Red List of Threatened Taxa, with a declining population estimated at only 3500–9000 individuals across its range (Humle et al. 2016; Morgan et al. 2011), and under threat elsewhere in Cameroon (Fotang et al. 2021). Among great apes, the Nigeria–Cameroon chimpanzee (RAI = 0.65) was the most commonly detected of all other threatened taxa. This is close to that obtained in the Batéké Plateau NP in Gabon (Hedwig et al. 2018), but lower than that obtained in the community zone of the Lobéké NP in Cameroon (Poulain et al. 2023) for the Central chimpanzee, *Pan troglodytes troglodytes* (Blumenbach, 1775). Small population size in combination with human pressures can explain this intermediate RAI (Kamgang et al. 2023). However, the low RAI values for some threatened species, such as the African golden cat, the crested genet, the white-bellied pangolin and the giant pangolin, suggest that these species may be rare or vulnerable to human disturbance such as hunting and habitat fragmentation, and require areas of intact habitat to survive (Crooks 2002; Ripple et al. 2014). This is consistent with the low RAI value obtained for the African golden cat in Mpem & Djim NP, Cameroon (Fotsing, Kamkeng, Marcel Senge, and Zinner 2024), suggesting the low density typical for this species (Ray et al. 2005). Contrary, different results were observed by Simo et al. (2023) in two protected areas in the forest–savannah transition zone of Cameroon, where the capture rate of the white-bellied pangolin was higher. This difference could be due to our survey design, where our CTs were targeted animal tracks on the ground, whereas Simo et al. (2023) targeted logs (fallen trees). Additionally, we used the relative abundance index (RAI) to quantify species activity calculated as the number of independent detections per 100 camera-trap days. We would like to point out that RAI is an index of relative activity rather than absolute abundance, and its interpretation may be influenced by species detectability, camera placement, and behavioural differences among species. RAI may therefore not accurately reflect true population density or abundance, particularly for elusive or rare species and should be interpreted as an index of relative activity, not absolute abundance.

As predicted, we were able to identify important environmental covariates affecting species richness of large and medium-sized mammals in the YCF. Our study showed that SRS, canopy height and distance to villages negatively affected species richness. This suggests that as distance to villages increases, species richness decreases, implying that camera trap stations closer to villages detected a large variety of species. Similar results have been reported in previous studies in Central Africa (Vanthomme et al. 2013; DeGroot et al. 2023). Some species' adaptability to human presence and reliance on human-modified landscapes for their survival may underlie this result (Kitchen et al. 2010; Vanthomme et al. 2013). However, Koerner et al. (2017) showed that the effect of human-related factors varied significantly

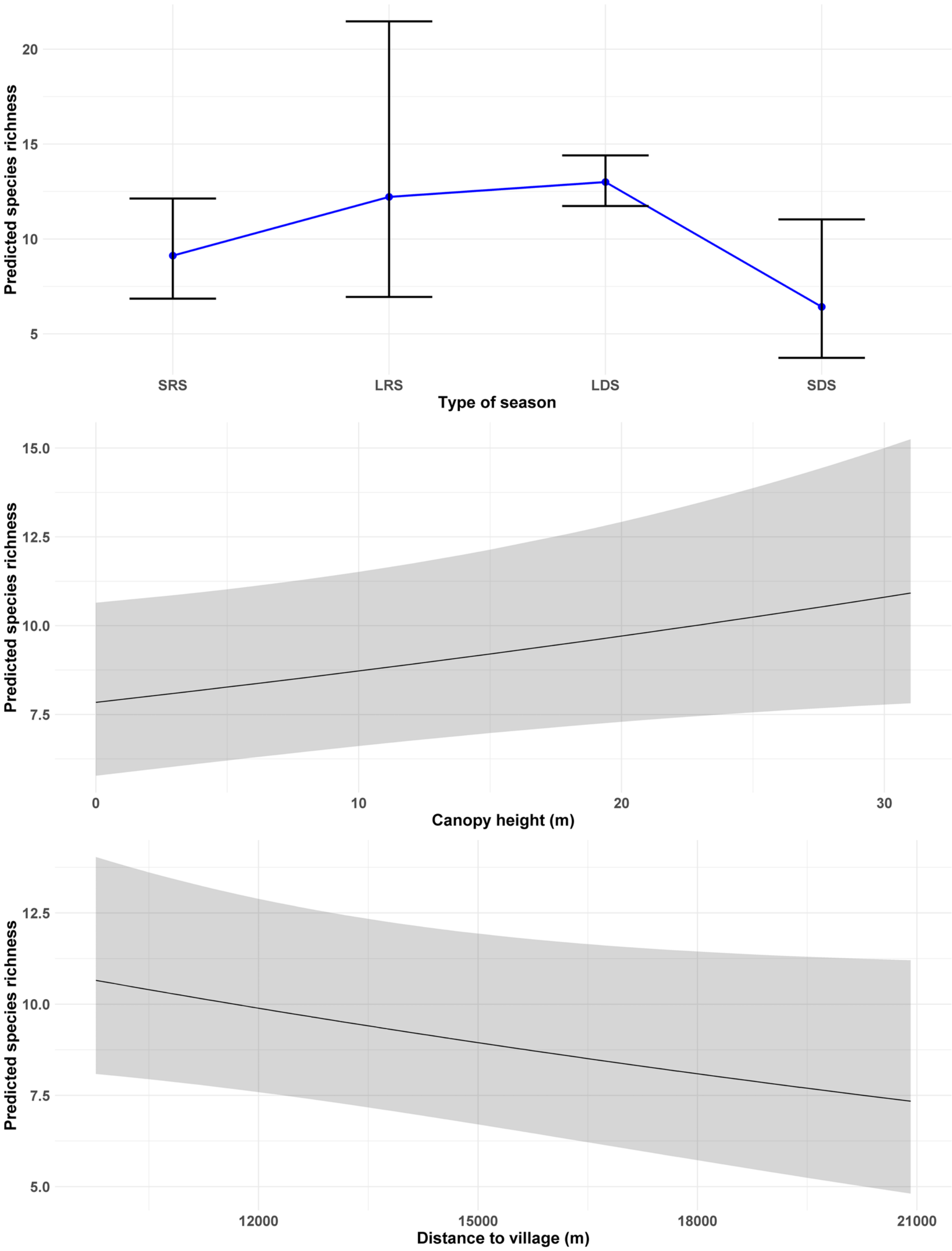

FIGURE 5 | Effects of covariates on species richness. LDS, long dry season; LRS, long rainy season; SDS, short dry season; SRS, short rainy season.

between species, with the abundance of some taxa, such as rodents, decreasing with distance to human settlements, while the abundance of others such as primates and ungulates, increased with distance to human settlements. We also found that species richness increased during the LRS, possibly related to the abundance of food which reduces competition among species, increasing therefore their detectability (Fotsing et al. 2025). The decrease in species richness during the SDS may be due to seasonal bush fires and increased hunting pressure during the dry season (Hassan 2007) or to food scarcity due to the vegetation drying up (Mohammed et al. 2024). We predicted many other covariates to be correlated with species richness, but this was not the case unfortunately. This observation may justify a more in-depth analysis probably suggesting the necessity to use a multispecies occupancy approach to assess the impact of this large set of covariates on detectability and species richness to further confirm the output of the modelling approach used.

Our findings confirm the presence of five IUCN-listed terrestrial mammal species of conservation concern, including the charismatic Nigeria-Cameroon chimpanzee. As a species indicator of forest intactness (Bahaa-el-din et al. 2015), these threatened species are highly sensitive to human activities such as poaching and illegal trade and may therefore require

TABLE 5 | Summary of incidence rate ratios (IRR).

Fixed effect	*B*	IRR	Lower_95_CI	Upper_95_CI	*p*
Intercept	2.23	9.3	6.93	12.48	0
SeasonLRS	0.29	1.34	0.71	2.5	0.036
SeasonLDS	0.35	1.42	1.06	1.9	0.020
SeasonSDS	−0.35	0.7	0.39	1.27	0.024
Canopy heigh	0.09	1.09	1.01	1.18	0.040
Distance to village	−0.09	0.91	0.83	1.01	0.010

Abbreviations: *B*, estimate; IRR, incidence rate ratio; Lower_95_CI, lower value of the confidence interval to 95%; Upper_95_CI, upper value of the confidence interval to 95%.

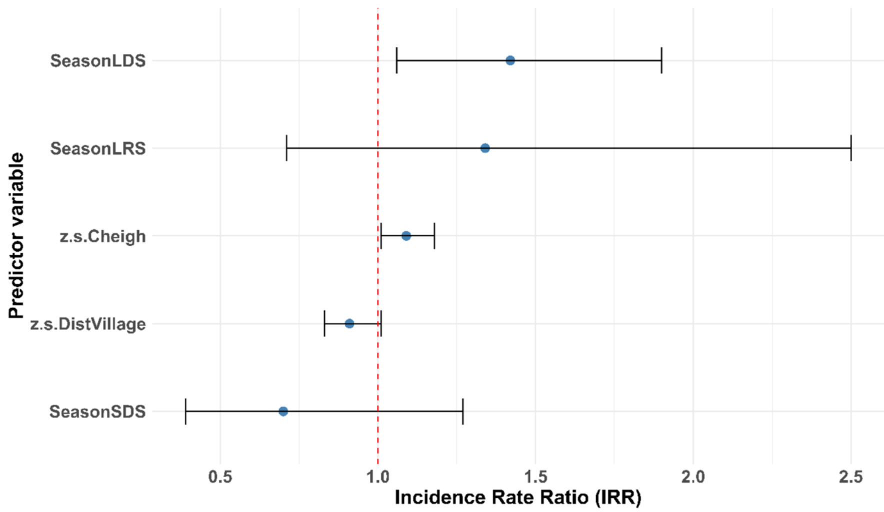

FIGURE 6 | Incidence rate ratios (IRR) plotted with standardised covariates. Cheigh, canopy height; DistVillage, distance to villages; LDS, long dry season; LRS, long rainy season; SDS, short dry season.

habitat protection during forest logging. Nevertheless, their presence in YCF indicates the potential of community conservation areas to contribute to biodiversity conservation as OECMs. Remarkably, the YCF does so with a host of charismatic and flagship species that are often characteristic of areas with strict protection and high investments. Understanding the key environmental and anthropogenic factors influencing species richness enables local communities to prioritise conservation actions such as protecting critical habitats, managing human access and monitoring sensitive areas thereby supporting sustainable forest management and contributing to international conservation targets.

Author Contributions

All authors conceived the ideas and designed the methodology; G.D.M., F.T.S., G.F.D., A.C.W. and A.M. collected the data; G.D.M. and E.D.B.F. analyzed the data; G.D.M., H.B. and E.D.B.F. led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

All data and code used in this work can also be made available on a reasonable request to the authors.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Appendix S1:** Covariates, definitions, and measurements used for modelling species richness in YCF. **Appendix S2:** Figure a Correlation plot between covariates. Here we show that correlation between digital elevation model (Dem) and Topographic position index (TPI) ($r<0.6$) and also between Dem and Distance to village (DistVillage) ($r<0.73$) is high even if our threshold was set to 0.7. Figure b Correlation plot between covariates. Here we show that after removing Dem no pairs of covariates are corelated anymore and we can therefore use them in the model.