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Diversity, distribution, and conservation of millipedes (Myriapoda: Diplopoda) in the Douala-Edéa National Park, Littoral Region of Cameroon

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Abstract

Biodiversity inventories are essential for the management and monitoring of ecosystems, especially in protected tropical areas. In the Afrotropical region, biodiversity conservation initiatives are primarily focused on charismatic vertebrates, while invertebrates such as millipedes, which are largely endemic, have received little attention. To partly fill the gaps of knowledge relating to millipedes, this study was conducted in the Douala-Edéa National Park, one of the most important protected areas in Cameroon. A year-long field survey was conducted using the classical active search method in quadrats. Overall, 36 millipede species belonging to 22 genera and nine families were identified from 799 individuals that were collected. The Chelodesmidae was the richest family (8 species), followed by the Oxydesmidae and the Spirostreptidae (6 species each). The most species-rich habitat was primary forest with 24 species, while the lowest species richness was observed in an agroforest (4 species). Moreover, the highest millipede abundance was observed in open meadows (41.80 % of the total abundance), compared to the agroforest (2.75 % of the total abundance). The primary forest was the most diverse habitat (H' = 2.86; E = 0.73) compared to the other habitat types. *Kartinikus colonus* (Spirostreptidae) was widely distributed as it was found in all habitat types, whereas approximately half of the total species recorded (17 species) showed restricted distributions as they were confined to a single habitat. The non-metric multidimensional scaling (NMDS) analysis revealed that millipede communities in secondary forest, primary forest, mangrove, and agroforest ecosystems were similar and distinct from those in open meadows. Despite the high richness and abundance of millipede species in the Douala-Edéa National Park, anthropogenic activities such as agricultural practices and felling were identified as the main threats to this fauna. Thus, it is crucial to implement conservation initiatives to protect millipede species in this protected area.

Keywords Soil macrofauna | ecology | coastal woodland | tropical Africa

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1. Introduction

Biodiversity monitoring in protected areas is an integral part of the assessment of their performance and provides necessary information for an effective management (McGeoch et. al. 2011). Invertebrates are useful and informative indicators of other elements of biodiversity, ecosystem function and restoration (McGeoch 2007, McGeoch et al. 2011, Hamer & Slotow 2017). Although invertebrates occupy a wide range of ecological niches and exhibit many important ecological functions, they receive relatively little attention, primarily due to taxonomic problems encountered in identifying numerous taxa (Minelli 2015). It is important to determine the level and patterns of diversity in an area, including the identities and number of taxa, their distributions and community diversity. One of the main reasons for conserving and monitoring invertebrates, particularly in protected areas, is to ensure adequate protection of rare and threatened species and communities (Samways 1993).

Within this important soil component, millipedes (Diplopoda) are a highly diverse, but largely understudied class of land arthropods with over 12,000 described species (Minelli 2015). Only about 20% of the global species diversity of millipedes (estimated at 50,000 to 80,000 species) is currently known (Minelli & Golovatch 2013). Being primarily represented by forest-dwelling mesophilic detritus feeders, millipedes have long been recognized as playing important ecological roles, mostly in temperate and tropical regions where their diversity is particularly high (Golovatch & Kime 2009). The class encompasses 16 extant orders, more than 140 families, and approximately 2000 genera (Minelli & Golovatch 2013, Golovatch & Liu 2020). Being very ancient and taxonomically diverse, widely distributed on all continents except Antarctica, almost entirely terrestrial, poorly vagile and highly limited in compensatory ecological faculties, millipede have long been considered as a group exemplary for biogeographic studies and reconstructions (Shelley & Golovatch 2011, Golovatch & Liu 2020). Because millipedes have been acknowledged as key taxa in several ecological processes, studying the composition of their assemblages and their distributions is crucial (Edwards 1974). Millipedes are highly sensitive to disturbances and can be used as ecological indicators (Paoletti et al. 2007).

Despite the importance of millipedes in ecosystem functioning and their high diversity in tropical regions, the current version of the IUCN RedList comprises only ca. 200 millipede species assessments (IUCN 2023), which represents only 1% of the described taxa (Karam-Gemael et al. 2018). Several studies have shown that, directly or indirectly, anthropogenic disturbances influence invertebrate richness, abundance, and/or composition, although only very few studies on millipedes have been performed (e.g. Birang et al. 2003, David 2009, Deblauwe & Dekoninck 2007, Fotso et al. 2015, Loranger-Merciris et al. 2008, Tadu et al. 2014). Therefore, in addition to species composition, the abundance of certain species may be useful for bioindication, making them potentially useful species for assessing and monitoring anthropogenic disturbances in natural habitats (Magrini et al. 2011). Anthropogenic activities, including deforestation, are likely to be critical to forest specialists, especially in tropical areas (Spelzhausen at al. 2020). Millipede communities in old-growth forests are often species-rich, perhaps related to the high diversity of tree leaf litter (David 2009). The immediate impact of deforestation on soil macro-arthropods is strongly negative (Mathieu et al. 2005). Forest specialists are eliminated and do not re-invade sites that become grasslands (David 2009). In tropical plantations of eucalyptus, rubber, and cocoa, millipede abundance may remain high, but species richness may be significantly reduced (Bourdanne 1997).

Cameroon's coastal forests are rapidly degrading, even in protected areas, largely due to slash and burn cultivation, non-conventional exploitation of tree, and the implementation of industrial agroforest farms, which are important drivers of regional climate change in the Great Forest Belt of the Gulf of Guinea (Mahmoud et al. 2019). Therefore, it is crucial to study their soil invertebrate communities, including millipedes, which are usually very important to soil functioning (David 2015). To date, no study on the diversity, distribution, and conservation importance of millipedes in the Douala-Edéa National Park has ever been conducted. We investigated changes in millipede diversity, distribution, and community structure within various habitat types in the Douala-Edéa National Park. We also discuss the implications for millipede conservation in this protected area located in the Littoral evergreen forest zone of Cameroon.

2. Materials and Methods

2.1 Study site

This study was conducted from January 2020 to December 2020 in the Douala-Edéa National Park, located between 3°14′ and 3°50′N latitude and, 9°34′ and 10°03′E longitude (Fig. 1). Established in 1932, this protected area covers approximately 1 600 km^2 of land and water (Blaikie & Simo 2000) and is under the management of Cameroon's Ministry of Forestry and Wildlife Management (MINFOF). Situated in the Littoral evergreen forest zone of Cameroon (Letouzey 1985), the Douala-Edéa National Park extends across the lower Sanaga River and 100 km along the Atlantic coastline of Cameroon (Feka et al. 2009). The park is in a transitional zone and within a sedimentary lowland plain between 0 to 50 m altitude. Most of the northern part is subject to tides. Douala-Edéa National Park stretches 35 km inland from the Atlantic coast, with its eastern boundary running alongside the Dipombé River. (Angoni et al. 2015, Mayaka et al. 2013). It is in the Great Forest Belt of the Gulf of Guinea, which represents one of the world's largest biodiversity hotspots with a wealth of understudied and threatened species (Ajonina & Usongo 2001). Thus, is a crucial ecosystem for the future of the world, particularly in the context of climate change due to human pressure which is progressively destroying natural habitats (Angoni et al. 2015). Streams occupy about 1 % of the reserve's area; the largest water surface is Lake Tissongo. Soil types range from sandy to sandy-silty further inland. The south of the reserve is characterized by a typically Cameroonian warm and humid climate, with two seasons (unimodal distribution of rainfall). The rainy season lasts approximately nine months (Suchel 1987). Average annual rainfall ranges from 3 000 to 4 000 mm and average monthly temperature varies throughout the year from 24°C to 29°C (Feka et al. 2009). The park area is dominated by a lowland tropical equatorial forest (80 %) and covered by about 6.4 % mangrove, dominated by *Lophira alata* and *Saccoglottis gabonensis*, both of which are seriously threatened by deforestation (Mayaka et al. 2013, Takoukam Kamla et al. 2021)..

2.2 Sampling design

The sampling activities consisted of a one-year monthly field survey, from January to December 2020 in five vegetation types (primary forest, secondary forest, mangrove, open meadow, and agroforest) in the Douala-Edéa National Park. Samples were collected using active searching method (Mesibov et al. 1995, Mwabvu 2014, Means et al. 2015). Millipedes were collected in 20 movable box quadrats of 9 m^2 along two line transects of 200 m in each of the vegetation types in the Doala-

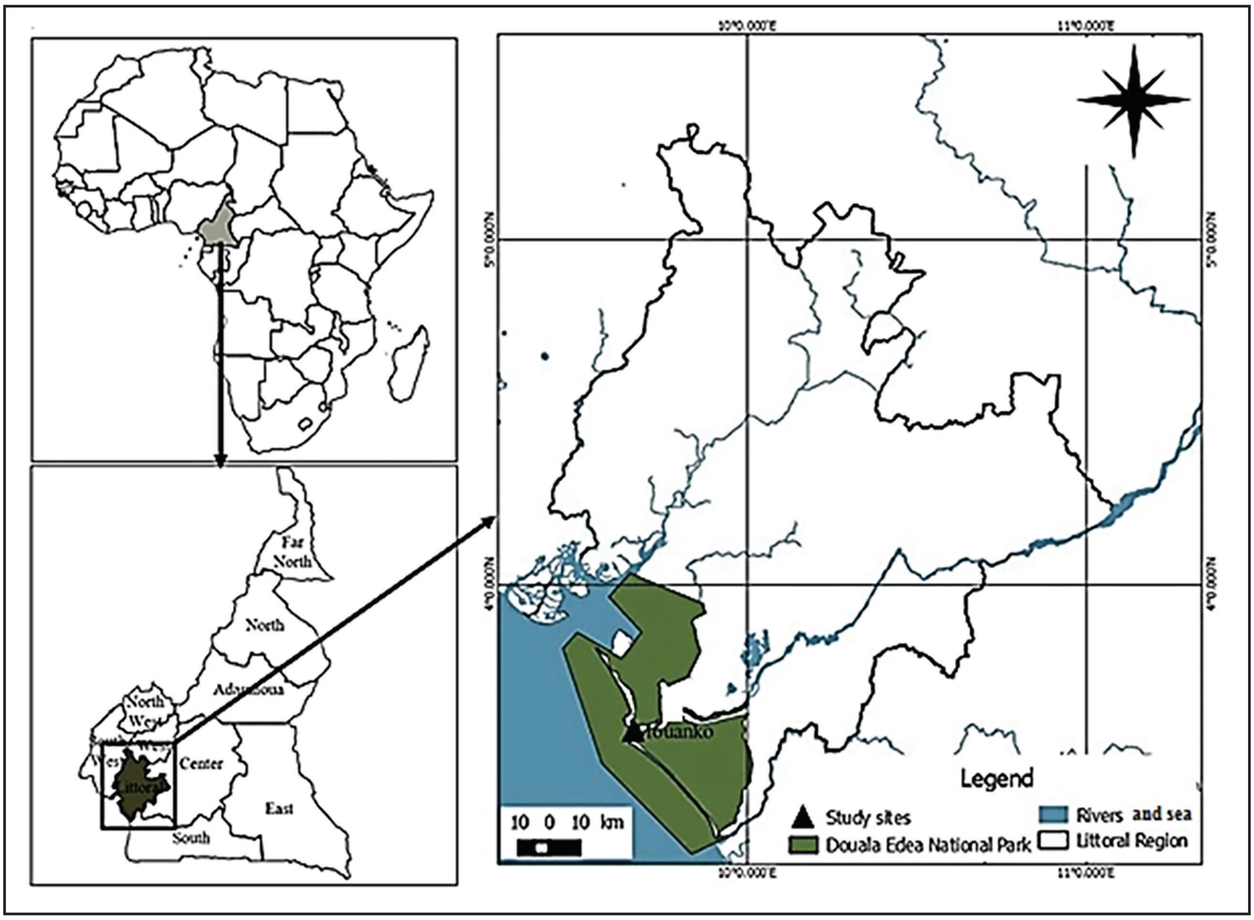

Figure 1. Study site in the Douala-Edéa National Park.

Edéa National Park. Consecutive box quadrats were 20 m apart in the two parallel and adjacent transects. The transects were 10 m apart. Millipedes that were in the litter or walking on the surface and on tree trunks were caught using forceps and preserved in tubes containing 70 % alcohol. The collected specimens were taken to the Laboratory of Zoology at the University of Yaoundé 1 for identification.

2.3 Identification of millipedes

Specimens were identified using dichotomous keys available in the literature (Kraus 1960, 1966; Demange & Mauriès 1975, Krabbe 1982, Hoffman et al. 1996, Hamer 1999, Golovatch et al. 2015, Nzoko-Fiemapong et al. 2017, Nzoko-Fiemapong & Enghoff 2018). Due to the fact that millipede identifications are mostly based on male gonopod structures, largely females and juveniles could not be formally identified and were excluded from analyses. Voucher specimens were deposited within the reference collections of the Laboratory of Zoology at the University of Yaounde 1.

2.4 Data analysis

2.4.1 Sample success

The sampling success was calculated using two non-parametric estimators of the species richness Chao1 and ACE (Abundance-based Coverage). The sampling success was estimated using the formula (S/T) x100 where S is the observed species richness and T the mean of the two calculated theoretical species richness (Marcon 2015).

2.4.2 Millipede diversity

The number of species by order, family and genus was determined, and percentages calculated from the total number of specimens collected. Relative abundances were expressed in percentages. The observed species richness S and the Margalef index Mg were determined. The Alpha diversity permitted us to characterize the species diversity by determining the Shannon–Wiever's index (H′). The t-test was used to compare H' values between two vegetations types (Hutcheson 1970). We computed the evenness of the studied communities using the Pielou's index. The Berger–Parker index was used to determine the dominance status of species. The Beta diversity index (Bray–Curtis distances) was used to visualize differences in community turnover among different habitat types. Cluster analysis was performed using Euclidean distances between rows (Species) and columns (Habitat).

2.4.3 Millipede distribution

To visualize the relationship between habitat types and millipede community, non-metric multidimensional scaling (NMDS) was performed based on Bray–Curtis dissimilarity index. The results were plotted in an NMDS ordination plane in a two-dimensional space. Differences in the millipede community among different habitats visualized with NMDS were analyzed using a permutational multivariate analysis of variance (PERMANOVA) test. After a significant PERMANOVA test ($p \leq 0.05$), a SIMPER analysis (Percent Similarity) was performed to examine which millipede groups were driving the differences in the community among the habitat types.

2.4.4 Millipede structure

The Rank–abundance plotting was used to illustrate the degree of millipede species abundance in each habitat type.

The Species Abundance Distributions models of millipede communities were compared to the Motomura's geometric distribution model, Mac-Arthur's broken stick model, and Fisher's logarithmic series model (Heip et al. 1998, Cielo Filho et al. 2012, Havyarimana et al. 2013, Marcon 2015) to find the model that best fits the data set. These models provide information about how species are distributed and how they share the available resources in the ecosystem (Havyarimana et al. 2013). PAST software (Paleontological Statistics Software Package for Education and Data Analysis) automatically generated the results from the row data. This software enabled us to compare the observed abundance distribution to the expected theoretical distributions using the Khi2 (χ^2) test.

We performed all the analysis using PAST, version 3.14 (Hammer et al. 2001) and R software version

2.4.5 Effect of habitat type on millipede abundance

Data matrices of species abundance counts in relation to habitat were constructed for each millipede species. The Kruskal-Wallis test was used to compare the abundance of different millipede species across all sampled habitats.

3. Results

3.1 Assessment of sampling effort

The global sampling success of species (97%) was considerably high in this study (Tab. 1). The Chao1 and ACE (Abundance-based Coverage) non-parametric estimator revealed that, 36 out of 37 species were collected, suggesting that only one rare species was not sampled. In the primary forest, 24 out of 26 species were collected, suggesting that two rare species were not found. In the secondary forest, 13 out of 15 species were recorded, suggesting that two rare species were not collected. In contrast, in mangrove, agroforest and open meadows, almost all species were collected (Tab. 1).

The individual rarefaction plot curves showed an asymptotic trend for primary forest, mangrove, and open meadow habitats, while the secondary forest showed that it was less sampled in comparison to other vegetation types. Thus, more effort is needed to record all millepede species in the secondary forest. The curves of primary forest, mangrove and open meadows had similar slopes. However, the curves of agroforest, open grassland and secondary forest were well below that of the primary forest, suggesting lower species richness at these sites compared to primary forest (Fig. 2).

3.2 Millipede species richness in the Douala-Edéa National Park

A total of 36 millipede species belonging to 22 genera and nine families were identified from 799 individuals collected (Fig. 2, Tab. 1). Chelodesmidae was the most common family in terms of species richness (8 species), followed by Oxydesmidae and Spirostreptidae (6 species each), and Pyrgodesmidae (5 species) and Odontopygidae (4 species). The Pachybolidae, Stemmiulidae and Trichopolydesmidae were all represented by two species, while Cryptodesmidae was monospecific (Fig. 3, Tab. 2). The two most species-rich genera were Paracordyloporus and Monachodesmus with three species each (Fig. 3, Tab. 2).

The most species rich habitat was primary forest (24 species), followed by mangrove (17 species), secondary forest (13 species), open meadows (11 species), while

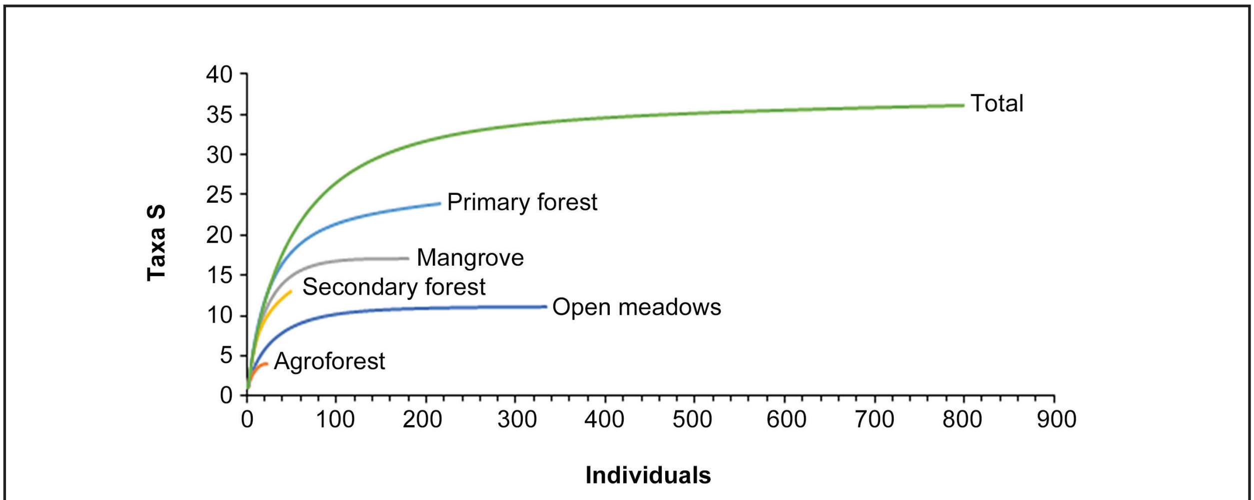

Figure 2. Species rarefaction curve among different habitat types in Douala-Edéa National Park.

Table 1: Millipede sampling effort in different vegetation types in Douala-Edéa National Park. The values in brackets represent the sampling success (%) from species richness estimators.

	Open meadows	Agroforest	Mangrove	Secondary forest	Primary forest	Total
N	334	22	179	49	215	799
Sobs	11	4	17	13	24	36
n_{max}	204	15	41	8	21	207
Margalef (Mg)	1.72	0.97	3.08	3.08	4.28	5.24
Chao1	11(100)	4(100)	17(100)	15(86.67)	26(92)	37(97)
ACE	11 (100)	4 (100)	17(100)	16 (81.3)	25(96)	37(97)

N = Numbers of sampled millipede specimens, **S_{obs}**: Observed Species richness, **nmax** = maximum abundance, **Mg** = Margalef's richness index.

agroforest was the least species rich habitat with four species (Tab. 1). Differences in species richness among habitat type combinations were highly significant ($p < 0.0001$).

3.3 Diversity of millipedes in the Douala-Edéa National Park

Primary forest (H' = 2.86, J = 0.73), mangrove (H' = 2.55, J = 0.76) and secondary forest (H' = 2.31, E = 0.77) were highly diverse while the agroforest (H' = 0.97, E = 0.66) and open meadows (H' = 1.45, J = 0.39) had lower diversity (Tab. 2). Comparisons revealed highly significant differences in millipede diversity among habitat types (see Tab. 2). In agroforest, mangrove, secondary forest, and primary forest, the Pielou Eveness is near to 1, suggesting high homogeneity of millipede communities in those habitat types. Most of the habitat types, including primary forest, secondary forest, and mangrove, had a low dominance of a particular species, and thus a very high species diversity of the communities (Tab. 2). In contrast, agroforest and open meadow had shown strong dominance of one species, and thus a very low species diversity of the communities.

3.4. Dissimilarity of millipede communities between habitat types

Figure 4 below shows the non-metric multidimensional scaling (NMDS) scatterplot of millipede faunal data in

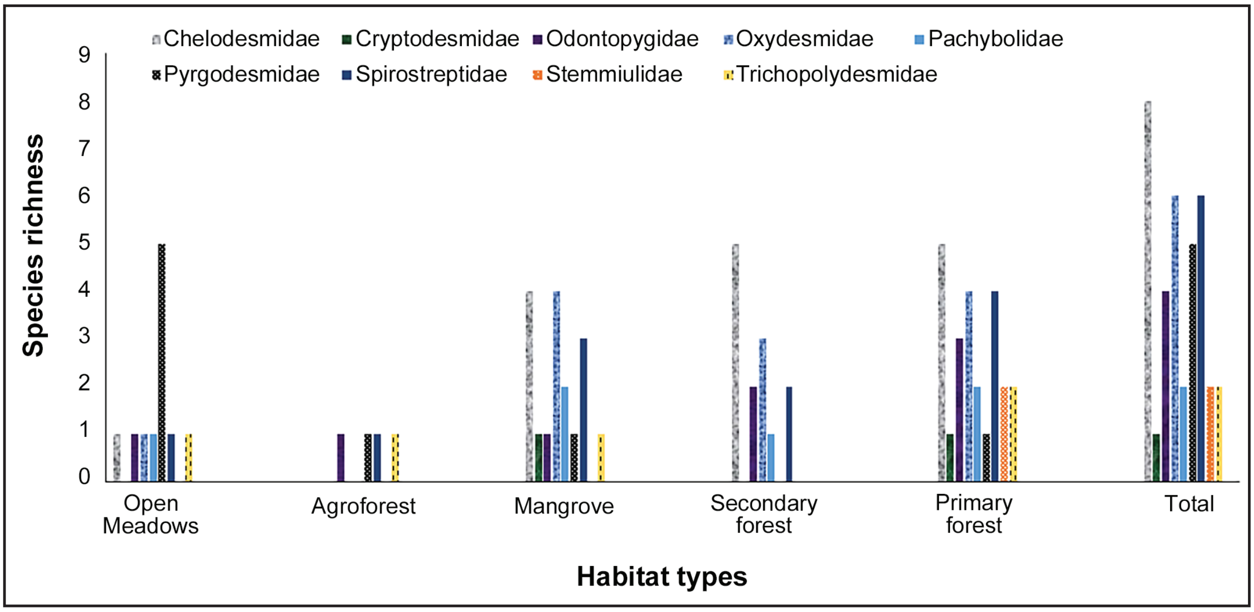

Figure 3. Millipede species richness of the different families by habitat types in Douala-Edéa National Park.

Table 2. Diversity indices in different habitat types in Douala-Edéa National Park

	Open meadows	Agroforest	Mangrove	Secondary forest	Primary forest
Simpson (λ)	0.40	0.50	0.10	0.12	0.07
H'	1.45[a]	0.97[b]	2.55[c]	2.31[d]	2.86[e]
H'_{min}	1.30	0.55	2.40	2.17	2.74
H'_{max}	1.57	1.23	2.62	2.42	2.92
Pielou (J)	0.39	0.66	0.76	0.77	0.73
N1	4	3	13	10	17
N2	3	2	10	8	14
Berger-parker (D)	0.61	0.68	0.23	0.16	0.15

λ = Simpson's diversity index, **H'** = Shannon-Weaver's diversity index, **H'max** = Shannon-Weaver's maximum diversity index, **H'min** = Shannon-Weaver's minimum diversity index, **J** = Pielou's evenness index, **SE** = sampling effort, **N1** = Hill's diversity number one = $e^{H'}$, **N2** = Hill's diversity number two, **D** = Berger-Parker's dominance index. The letter [a, b, c, d] and e represent the results of the pairwise comparison of Shannon-Weaver index in the habitat types using Fisher T test.

five habitat types within the Douala-Edéa National Park. It appears that the millipede communities changed with habitat type in the Douala-Edéa National Park, despite the low stress obtained (stress = 0.28). In addition, quantitative analyses using PERMANOVA indicated a different species composition among sampled habitats (F = 3.485, p = 0.0001). Given the significance of the test, we observed that the main difference is between primary forest and all the other vegetation types (Tab. 3).

Species contribution in terms of dominance was different across habitat types as indicated by the SIMPER analysis (Tab. 4). The six most dominant species that contributed about 50 % of the dissimilarity among the surveyed vegetations were *Trichochaleponcus* sp. (overall contribution: 18.65 %), *Kartinikus colonus* (overall contribution: 10.33 %), *Pelmatojulus tectus* (overall contribution: 5.75 %), *Urodesmus cornutus* (overall contribution: 5.65 %), *Afolabina sanguinicornis* (overall contribution: 5.37 %) and *Coromus* sp. (overall contribution: 4.65 %) (Tab. 4).

Based on Bray-Curtis distance, the cluster analysis revealed that the millipede communities of secondary forest, primary forest, mangrove and agroforest formed a single cluster, suggesting a high degree of similarity among these habitat types, that are distinct from open meadows (Fig. 5). Moreover, the secondary forest, primary forest and mangrove also formed a cluster which is distinct from agroforest. However, both cluster groups are too close, suggesting very low dissimilarity between habitat types.

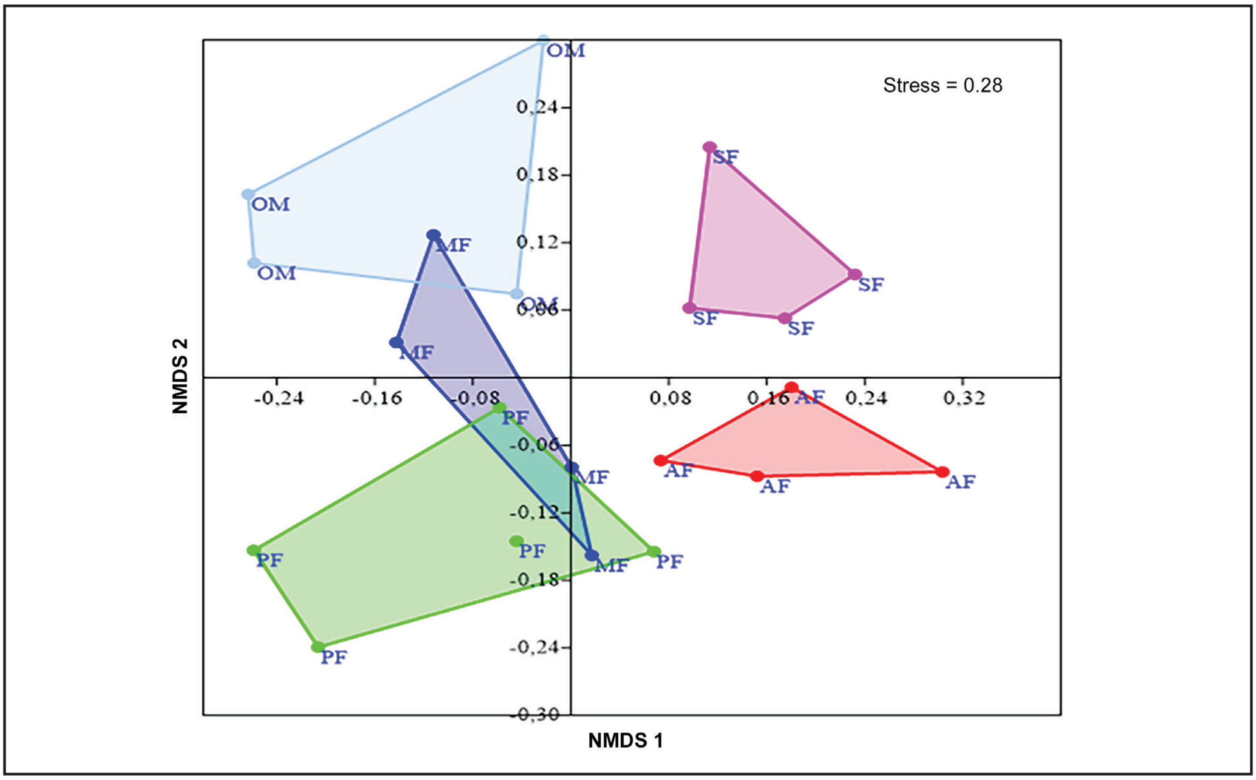

Figure 4. Non-Metric Multi-Dimensional Scaling (NMDS) of millipede faunal composition found in five Vegetation types in the Douala-Edéa National Park. Habitat codes are **AF** = Agroforest, **MF** = Mangrove forest, **PF** = Primary forest, **OM** = Open meadows, **SF** = Secondary forest.

Table 3. Results of the analysis of dissimilarity of millipede communities in habitat types (PERMANOVA) based on Bray-crurtis dissimilarity index.

	Mangrove	Primary forest	Secondary forest	Open meadow	Agroforest
	Bray-Curtis Index (global comparison: F = 3.485, p = 0.0001				
Mangrove	-	-			
Primary forest	0.2701				
Secondary forest	0.0272	0.0085	-		
Open meadow	0.0284	0.0084	0.0313	-	-
Agroforest	0.284	0.0082	0.0266	0.0287	-

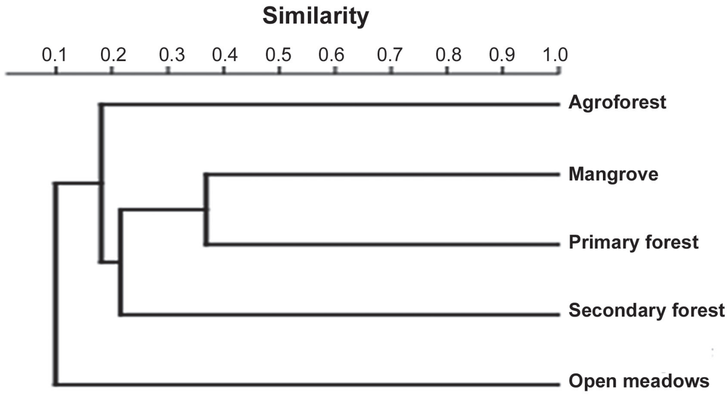

Figure 5. Cluster analysis based on Bray-Curtis index, showing dissimilarity between the different millipede communities in Douala-Edéa National Park.

Table 4. SIMPER analysis for millipede species contributing most to dissimilarities among the five environments (Magrove, Primary forest, Secondary forest, Open meadow and Agroforest) in the Douala-Edéa National Park.

Taxa	Average dissimilarity (a)	Contribution (b)	Cumulative % (c)	Mean abundance Mangrove	Mean abundance Primary forest	Mean abundance Secondary forest	Mean abundance Open meadow 4	Mean abundance Agroforest
Trichochalepuncus sp.	16.79	18.65	18.65		-	0.25	51	0.5
Kartinikus colonus	9.297	10.33	28.98	5	6.4	1.5	0.5	3.75
Pelmatojulus tectus	5.176	5.75	34.73	1.5	4.2	-	4.5	-
*Urodesmus cornutus**	5.091	5.655	40.39	2.25	2.2	-	10.3	0.5
*Afolabina sanguinicornis**	4.834	5.37	45.76	10.3	-	-	-	-
Coromus sp.	4.187	4.651	50.41	3.25	1.6	0.25	2.25	-
Laciniogonus sp.	3.171	3.522	53.93	-	3.6	0	-	-
Telodeiopus cananiculatus	3.085	3.427	57.35	1.75	-	2	-	-
*Paracordyloporus porati**	3.083	3.424	60.78	1.5	0.6	1.75	1.5	-
Aporodesmus gabonicus	2.989	3.32	64.1	4.25	1.8	-	-	-
Urotropis carinatus	2.442	2.712	66.81	-	2.8	-	-	-
Coenobothrus detruncatus	2.342	2.601	69.41	2.25	1	0.25	-	-
Pelmatojulus excisus	2.338	2.597	72.01	3	0.6	0.75	-	-
*Hemisphaeroparia mouanko**	2.322	2.579	74.59	1.25	0.2	-	2.5	0.75
Kyphopyge sp. 1	2.319	2.576	77.16	-	3.2	1.5	-	-
*Scytodesmus kribi**	2.278	2.531	79.7	-	-	2	-	-
Stemmiulus nigricolis	2.241	2.489	82.18	-	2.6	-	-	-
Coromus vitatus	1.781	1.978	84.16	1	1.2	0.5	-	-
*Coromus barombi**	1.713	1.903	86.07	3.5	-	-	-	-
Paracordyloporus sp.	1.506	1.672	87.74	-	3.4	0.25	-	-

Table 4 (continued).

Taxa	Average dissimilarity (a)	Contribution (b)	Cumulative % (c)	Mean abundance Mangrove	Mean abundance Primary forest	Mean abundance Secondary forest	Mean abundance Open meadow 4	Mean abundance Agroforest
Kyphopyge granulosa	1.172	1.302	89.04	0.75	1	0.5	-	-
*Diaphorodesmus dorcicornis**	1.114	1.237	90.28	-	-	0.75	-	-
Treptogonostreptus intricatus	1.103	1.225	91.5	1.25	-	-	-	-
Monachodesmus sp.1	1.094	1.215	92.72	-	-	-	5.25	-
*Coenobothrus bipartitus**	0.9745	1.082	93.8	-	1.2	-	-	-
Urotropis sp.	0.9559	1.062	94.86	-	1.4	-	-	-
*Scytodesmus valdaui**	0.9422	1.047	95.91	0.75	0.8	-	-	-
Stemmiullus sp.	0.6828	0.7584	96.67	-	1	-	-	-
*Heptadesmus granulosus**	0.6246	0.6939	97.36	-	1.6	-	-	-
Diaphorodesmoides sp.	0.6119	0.6796	98.04	1.25	-	-	-	-
Udodesmus sp.	0.4166	0.4628	98.5	-	-	-	2	-
Monachodesmus sp. 2	0.3645	0.4049	98.91	-	-	-	1.75	-
*Monachodesmus longicaudatus**	0.3645	0.4049	99.31	-	-	-	1.75	-
''Spirostreptus'' pancratius	0.3248	0.3608	99.67	-	0.4	-	-	-
*Paracordyloporus trissolabris**	0.158	0.1755	99.85	-	0.2	-	-	-
*Hemisphaeroparia integrata**	0.1366	0.1517	100	-	0.2	-	-	-
Overal dissimilarity		**90.3**						

[a] = Average species abundance in each environment, [b] = Species percentage contribution to dissimilarity, [c] = Cumulative dissimilarity among three environments, **" "** = represent species with uncertain generic position (Orphan species), ***** = represent local endemic species.

3.5 Millipede distributions and assemblage patterns

Kartinicus colonus was widely distributed as it was present in all habitat types (Tab. 5). The following 10 species were found exclusively in primary forest: *Paracordyloporus trisolabris*, *Coenobothrus bipartitus*, *Laciniogonus* sp., *Heptadesmus granulosus*, *Spirostreptus pancratius*, *Urotropis carinatus*, *Urotropis* sp., *Stemmiullus nigricollis*, *Stemmiullus* sp. and *Hemispheroparia integrates*. Similarly, *Diaphorodesmus dorcicornis* and *Systodesmus kribi* were restricted to secondary forest, while *Afolabina sanguinicornis, Diaphorodesmoides* sp., *Coromus barumbi* and *Treptogonostreptus intricatus* occurred exclusively in mangrove. *Monachodesmus longicaudatus, Monachodesmus* sp.1, *Monachodesmus* sp.2 and *Udodesmus* sp. were specific to open meadows (Tab. 5).

3.6 Millipede species abundance and dominance

A total of 799 individuals were collected during the study period. The highest abundance of millipedes was observed in open meadows with 334 individuals (41.80 % of all millipedes collected), followed by primary forest with 215 individuals (26.91 %) then mangrove with 179 individuals (22.40 %), secondary forest with 49 individuals (6.13 %) and agroforest with the lowest abundance having 22 individuals (2.75 %). Differences in millipede abundance among habitat types tested using the Kruskal-Wallis test were highly significant ($H = 20.05$, $p < 0.0001$). Three species, namely, *Trichochaleponcus* sp. (25.91 %), *Kartinikus colonus* (9.39 %) and *Urodesmus cornutus* (8.01 %) were the most abundant during the study period (Tab. 5).

3.7 Distribution of species abundance

The rank-abundance curves showed that species composition differed among vegetation types (Fig. 6). The millipede community at the sampled sites was characterized by a few abundant taxa. The Hill's number two index (N_2) showed that 14, 10, 8, 3 and 2 species were numerically dominant in primary forest, secondary forest, mangrove, open meadows, and in agroforest, respectively (Tab. 2).

Based on the Species Abundance Distributions (SAD) (Fig. 6A), the millipede community distribution fitted the Preston Lognormal's model in general (m = 1.073, v = 0.24, X^2 = 2.58, P = 0.46). The same trend was noted in open meadows (m = 1.096, v = 0.29, X^2 = 0.51, P = 0.48) (Fig. 6F). The Broken stick model of McAthur fitted the millipede community observed in primary forest (comparison to the McAthur's theoric values: X^2 = 2.49, P = 1.00) (Fig. 6B), as well as in mangrove forest (comparison to the McAthur's theoric values: X^2 = 5.99, P = 0.95) (Fig. 6D). The Species Abundance Distribution recorded in secondary forest (Fig. 6C), showed that millipede community distribution fitted the Motomura's model (Motomura's environment constant: m = 0.19, X^2 = 1.35, P = 0.97) and in agroforest (Fig. 6E), the millipede community distribution fitted the Fisher Log series model (α = 1.43, x = 0.94, X^2 = 1.25, P = 0.26)

4. Discussion

This study provided the first assessment of millipede communities in Douala-Edéa National Park. Our survey showed that the millipede fauna of this protected area is rich and diverse. The number of species (36) recorded is higher than that recorded by Mbenoun et al. (2017) who identified 27 millipede species in Campo Ma'an National Park in Cameroon. In a forest in Ivory Coast, Bourdanne (1997) reported the presence of 32 millipede species. Similarly, in a study conducted in a patch of rainforest in central Amazonia, Golovatch et al. (2011)

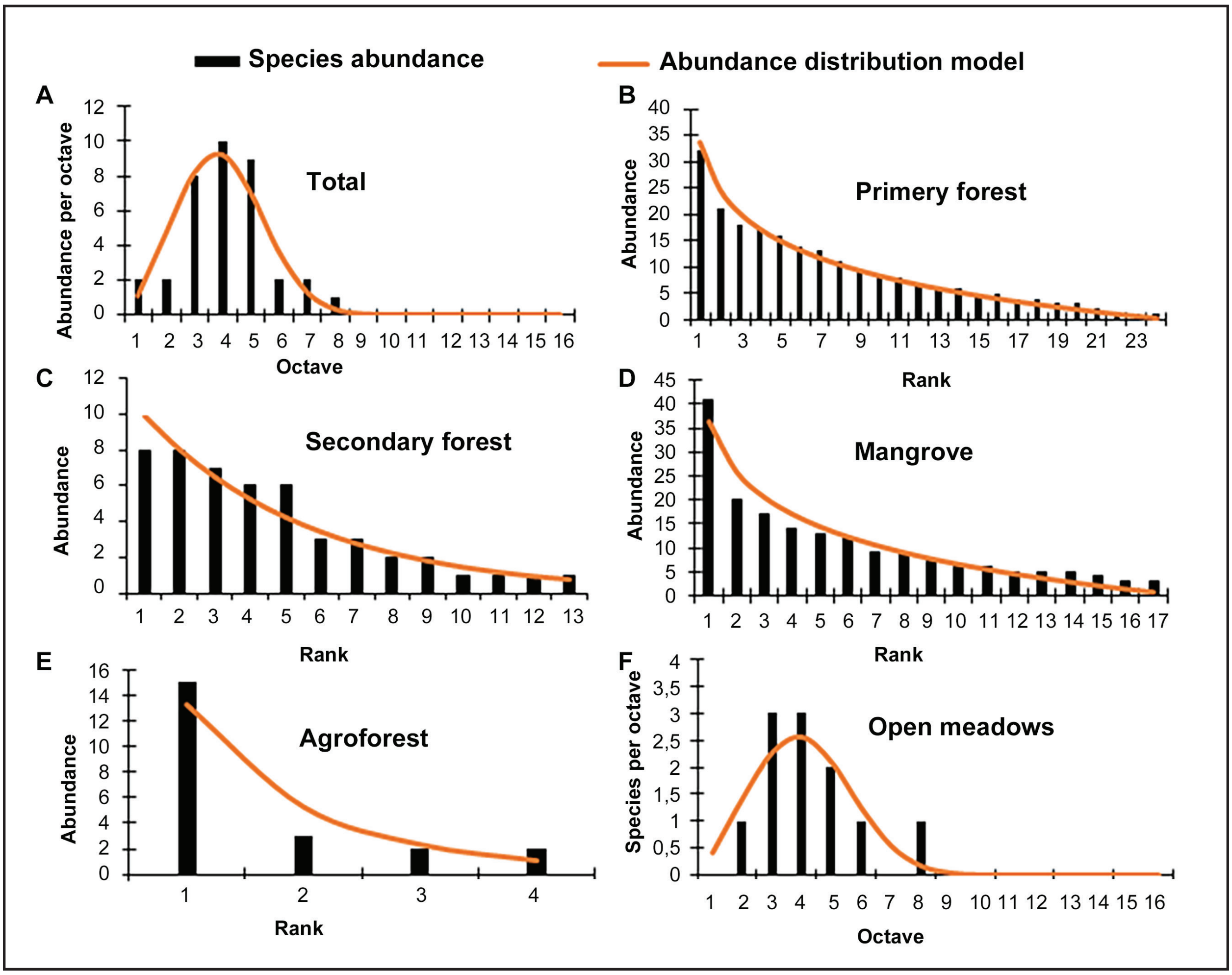

Figure 6. The abundance distribution model of millipede species in the different habitat in Douala-Edéa National Park.

Table 5. Absolute and relative abundance (in the brackets) of each millipede species by habitats types in Douala-Edéa National Park.

Families and Species	Open meadows	Agroforest	Mangrove	Secondary forest	Primary Forest	TOTAL
CHELODESMIDAE						
Afolabina sanguinicornis (Porat, 1892)	0	0	41 (5.13)	0	0	41 (5.13)
Diaphorodesmoides sp.	0	0	5 (0.63)	0	0	5 (0.63)
Diaphorodesmus dorcicornis (Porat, 1894)	0	0	0	3 (0.38)	0	3 (0.38)
Kyphopyge granulosa (Attems, 1931)	0	0	3 (0.38)	2 (0.25)	5 (0.63)	10 (1.25)
Kyphopyge sp.	0	0	0	6 (0.75)	16 (2.00)	22 (2.75)
Paracordyloporus porati (Verhoeff, 1938)	6 (0.75)	0	6 (0.75)	7 (0.88)	3 (0.38)	22 (2.75)
Paracordyloporus sp.	0	0	0	1 (0.13)	17 (2.13)	18 (2.25)
Paracordyloporus trissolabis Hoffman, 1963	0	0	0	0	1 (0.13)	1 (0.13)
CYPTODESMIDAE						
Aporodesmus gabonicus (Lucas, 1858)	0	0	17 (2.13)	0	9 (1.13)	26 (3.25)
Odontopygidae						
Coenobothrus bipartitus Porat, 1894	0	0	0	0	6 (0.75)	6 (0.75)
Coenobothrus detruncatus Carl, 1905	0	0	9 (1.13)	1 (0.13)	4 (0.50)	14 (1.75)
Laciniogonus sp.	0	0	0	0	18 (2.25)	18 (2.25)
Trichochalepuncus sp.	204 (25.53)	2 (0.25)	0	1 (0.13)	0	207(25.91)
OXYDESMIDAE						
Coromus barombi (Cook, 1896)	0	0	14 (1.75)	0	0	14 (1.75)
Coromus sp.	9 (1.13)	0	13 (1.63)	1 (0.13)	8 (1.00)	31 (3.88)
Coromus vitatus (Cook, 1896)	0	0	4 (0.50)	2 (0.25)	6 (0.75)	12 (1.50)
Heptadesmus granulatus (Verhoeff, 1938)	0	0	0	0	8 (1.00)	8 (1.00)
Scytodesmus kribi Cook, 1895	0	0	0	8 (1.00)	0	8 (1.00)
Scytodesmus valdaui (Porat, 1892)	0	0	3 (0.38)	0	4 (0.50)	7 (0.88)
PACHYBOLIDAE						
Pelmatojulus excisus (Cook, 1897)	0	0	12 (1.50)	3 (0.38)	3 (0.38)	18 (2.25)
Pelmatojulus tectus (Cook, 1897)	18 (2.25)	0	6 (0.75)	0	21 (2.63)	45 (5.63)
PYRGODESMIDAE						
Monachodesmus longicaudatus Golovatch, Nzoko Fiemapong & VandenSpiegel, 2015	7 (0.88)	0	0	0	0	7 (0.88)
Monachodesmus sp.1	21(2.63)	0	0	0	0	21 (2.63)
Monachodesmus sp.2	7 (0.88)	0	0	0	0	7 (0.88)
Udodesmus sp.	8 (1.00)	0	0	0	0	8 (1.00)
Urodesmus cornutus Golovatch, Nzoko Fiemapong & VandenSpiegel, 2015	42 (5.26)	2 (0.25)	9 (1.13)	0	11 (1.38)	64 (8.01)
SPIROSTREPTIDAE						
Kartinikus colonus Attems, 1914	2 (0.25)	15 (1.88)	20 (2.50)	6 (0.75)	32 (4.01)	75 (9.39)
''*Spirostreptus*'' *pancratius* Attems, 1914	0	0	0	0	2 (0.25)	2 (0.25)
Telodeiopus cananiculatus (Porat, 1894*)*	0	0	7 (0.88)	8 (1.00)	0	15 (1.88)
Treptogonostreptus intricatus (Voges, 1878)	0	0	5 (0.63)	0	0	5 (0.63)
Urotropis carinatus (Porat, 1892)	0	0	0	0	14 (1.75)	14 (1.75)
Urotropis sp.	0	0	0	0	7 (0.88)	7 (0.88)
STEMMIULIDAE						
Stemmiulus nigricollis Porat, 1894	0	0	0	0	13 (1.63)	13 (1.63)
Stemmiulus sp.	0	0	0	0	5 (0.63)	5 (0.63)
TRICHOPOLYDESMIDAE						
Hemisphaeroparia mouanko Golovatch, Nzoko Fiemapong, Tamesse, Mauriès & VandenSpiegel, 2018	10 (1.25)	3 (0.38)	5 (0.63)	0	1 (0.13)	19 (2.38)
Hemisphaeroparia integrata (Porat, 1894)	0	0	0	0	1 (0.13)	1 (0.13)
TOTAL	**334 (41.80)**	**22 (2.75)**	**179 (22.40)**	**49 (6.13)**	**215 (26.91)**	**799 (100)**

" " = represent species with uncertain generic position (Orphan species). The numbers in brackets represent the relative abundance generated from the formula n/N×100 where n represents the absolute abundance and N the total abundance at the end of the study period.

recorded 33 millipede species. Prior to our study, 36 millipede species were also recorded in a monsoon forest in southern Vietnam (Golovatch 1997, Golovatch & Kime 2009). This high species richness could be due to a variety of ecological conditions, rainfall regime, and temperature that are favorable to millipede in the Douala-Edéa National Park. In general, environments with high relative humidity and a high rainfall regime are favorable for millipedes. The littoral evergreen forest zone of Cameroon provided adequate ecological conditions for the propagation of millipede species. Bogyó et al. (2015) and Topp et al. (2006) reported that the occurrence of millipedes depends on the type of ecosystem, soil moisture and the presence of dead wood.

The present study showed that the Chelodesmidae was the most species-rich millipede family. In Campo Ma'an National Park, Mbenoun et al. (2017) instead found that the most dominant millipede family was the Odontopygidae, one of the largest millipede families endemic to the Afrotropical region (Enghoff, 2016). The difference observed between these two studies is likely because the studies were conducted in areas with different rainfall patterns and in different habitat types. The fact that the Campo Ma'an locality is at a higher altitude (500 - 600 m) than Douala-Edéa, which is closer to sea level, could also explain the difference. As the vast majority of Odontopygidae species are endemic to the mountains, they tend to prefer higher altitude environments (Enghoff, 2016). Indeed, we worked in 5 habitat types (primary forest, secondary forest, mangrove, agroforest, and open meadows), while Mbenoun et al. (2017) only considered two habitat types (primary and secondary forests). The Chelodesmidae, which dominated the millipede fauna of Douala-Edéa National Park, is one of the most important millipede families in the world in terms of the number of species (Minelli 2015).

The individual rarefaction curves suggest that additional sampling effort is needed in some of the studied vegetation types. The same result was obtained by Golovatch et al. (2011) in a patch of monsoon forest in southern Vietnam despite a 3-year sampling effort. The saturation plateau observed for mangrove and open meadows showed little change in species richness despite an increase in the number of specimens in both vegetation types. This result demonstrates that the sampling during this study was adequate (Longino & Colwell 2011).

Considering the species richness of millipedes in different habitat types, the results indicated that primary forest had the greatest number of species. In Campo Ma'an National Park in Cameroon, Mbenoun et al. (2017) made similar observations after collecting 23 millipede species in primary forest. This high species richness suggests a strong preference of millipedes for forest environments. Several authors among which David (2015), Hopkin & Read (1992), Rodrigues et al. (2017) and Sklodowski & Tracz (2018), reported that the large number of microhabitats and a high availability of food resources in forests are favorable for millipede species expansion. The difference observed between primary forest and secondary forest in terms of species richness can be explained by anthropogenic activities in the latter which can increase light regime and probably desiccation, two factors that are known to affect millipedes negatively (David & Handa, 2010). Also, the greater heterogeneity of primary forest compared to secondary forest due to more tree species of various ages trees in the former can justify our observation (Bogyó et al. 2015, Hopkin & Read 1992, Spelzhausen et al. 2020). Likewise, Hopkin & Read (1992) and Bogyó et al. (2015) reported that the occurrence of millipede species in forest habitats, is closely related to the high relative humidity and the availability of leaf litter. The difference in millipede species richness across the five land use systems recorded in Douala-Edéa National Park sufficiently demonstrates that habitat type and especially its composition are likely to influence millipede species composition. Closed habitats (primary forest, secondary forest, and mangrove) showed a higher species richness compared to open habitats (open meadows). As pointed out by Golovatch & Kime (2009) and Fournier et al. (2015), this could be related to the high floristic richness and the relative stability of these environments. The case of the agroforest where we obtained a lower species richness compared to other environments is probably because the homogenization of the vegetation through the implementation of a monoculture agroecosystem (such as palm oil plantation as in this study's case) might negatively impact millipede species richness. This suggestion is supported by Vuidot et al. (2011) and David & Handa (2010) who hypothesized that high species richness in Afrotropical forest region is correlated with high habitat heterogeneity. Generally, the low species richness of millipedes recorded in open meadows is the result of the adverse impact of habitat fragmentation and land use. Similar observations have been made by Bogyó et al. (2015) and Suárez et al. (2018). Furthermore, according to Previati et al. (2007), invertebrate species richness is reduced by disturbances associated with intensive field cultivation. In addition, Fahrig (2003) and Hornák et al. (2020) stated that species extinction or increased susceptibility to extinction is a result of habitat loss and fragmentation.

Kartinicus colonus was widely distributed, it was present in all habitat types. In a study conducted on ground dwelling macroinvertebrates in Kirimiri forest in Kenya, Omondi et al. (2020) obtained similar results. *Kartinicus colonus* could be as a generalist or cosmopolitan species, which is able to tolerate disturbances. Block & Brennen (1993) concurred to this by stating that the physiology of widely distributed species allows them to regulate or tolerate harsh and unfavourable habitat changes, some of which reduced food source, increased water loss and niche heterogeneity.

Overall, the millipede diversity indices showed significant variation among the vegetation types in the Douala-Edéa National Park. Primary forest, mangrove, and secondary forest, showed the highest values of Shannon-Weaver and Pielou indices, while agroforest and open meadows showed the lowest. The highest values of Shannon-Weaver and Pielou indices recorded in primary forest, mangrove, and secondary forest are a result of favorable conditions in these less disturbed habitats. Ayuke et al. (2009) also observed that undisturbed lands tend to have higher diversity of soil macrofauna than modified lands. Bogyó et al. (2015) also reported that the diversity of soil invertebrates and particularly millipedes decreased due to human-induced disturbances.

The highest values of diversity indices recorded in primary forest, mangrove and secondary forest compared to agroforest and open meadows indicate a greater uniformity in species abundance in the former group of habitat types. According to Lo Sardo & Lima (2019) this could represent greater community structural integrity as well as the presence of rare organisms.

We found a very low dissimilarity between habitat types. Nevertheless, the cluster analysis showed that primary forest, secondary forest, mangrove, and agroforest formed a distinct cluster from open meadows. Moreover, secondary forest, primary forest and mangrove also formed a separate cluster from agroforest. The clustered habitat types show that they share many species. Bogyó et al. (2015) reported similar observations.

This study also showed that differences in millipede abundance among habitat types were highly significant. The highest millipede abundance was in the open meadows because the most abundant species (*Trichochaleponcus* sp.) is a highly tolerant generalist species and potential pest that usually colonizes crop farms and open environments such as grasslands which are the least diversified environments in the tropical rainforest zone of Cameroon (Nzoko-Fiemapong 2020). As suggested by Mathieu et al. (2005) and Correia et al. (2018), there is a reduction in the total abundance of edaphic fauna in more diverse and mature vegetation such as primary forest. The most abundant species in the closed habitats (primary forest, mangrove, and secondary forest, according to the generated clusters, was *Kartinicus colonus*. Our result is similar to that of Mbenoun et al. (2017) who found that *Kartinicus colonus* is one of the most dominant species in primary forest in Campo Ma'an National Park. The high abundance of *Kartinicus colonus* could be the result of the influence of vegetation structure. Likewise, Woodcock et al. (2010) reported that terrestrial invertebrate communities and abundance can be influenced by vegetation structure, through niches or plant food provision. Warren and Zou (2002), Loranger-Merciris et al. (2008) and, Spelzhausen et al. (2020) supported this idea by stating that litter quality in various vegetation types is an important factor that influences species richness, assemblages, and abundance of millipedes. Suitable habitats for millipedes are characterized by large amounts of leaf litter and dead wood, closed canopy cover, and a humid microclimate (Bogyó et al. 2015). Therefore, as demonstrated by Sayer et al. (2010), a highly significant predictor of arthropod abundance can be the availability of suitable habitat, such as deep layer of plant litter. Tomlinson (2014) also indicated that plant litter quantity explains the abundance of most native species of soil arthropods in New Zealand.

4.1 Implications for conservation

The main threats to millipede species encountered during field investigations are bushfires, slash and burn agricultural practices, and overuse of chemical products. These practices negatively affect millipedes which are generally vulnerable to habitat disturbance. Similar findings were reported by Bourdanne (1997) in Ivory Coast, Bogyó et al. (2015) in Northeast Hungary, and Hornák et al. (2020) in Southern Moravia (Czech Republic). Due to these threats, there is an urgent need to consider several management strategies to conserve millipede fauna in Douala-Edéa National Park. As stated by Leyte-Manrique et al. (2019), one of the first steps in biodiversity conservation is to identify the vulnerability of species in order to seek the best strategies to mitigate that vulnerability. Our results highlight the importance of including different aspects of biodiversity to provide a comprehensive view of the impact of habitat disturbance on millipede communities, even in protected areas. In this case, conservation strategies should include the protection of forest remnants with the participation of local populations and local authorities in charge of wildlife conservation. These strategies should also include the promotion and support of agroforestry and agro-ecological systems based on conservation-compatible polycultures and the inclusion of restoration of degraded lands. In addition, long-term monitoring programs of species diversity in protected areas, including all disturbed habitats, are needed to assess population and community trends. Furthermore, the species conservation status and the effects of land-use policies and practices should be determined. Such actions could have a crucial and positive impact on the conservation of the millipede fauna in the Douala-Edéa National Park and Cameroon's Littoral forests as a whole.

5. Conclusion

This study demonstrated that the millipede fauna in the Douala-Edéa National Park is diverse and abundant. In the five habitats surveyed, 799 millipede individuals sorted into 36 species, 22 genera and 9 families were recorded. As we expected, the most species-rich habitat was the primary forest, while disturbed agroforest was the least species-rich habitat. The cluster analysis based on Bray-Curtis distance revealed that primary forest, secondary forest, mangrove and agroforest formed a single group distinct from open meadows. Some millipede species had a wide distribution while others were restricted to a specific habitat. The main threats to millipedes in the Douala-Edéa National Park are anthropogenic activities, such as bushfires, agricultural practices, clear-cutting, cocoa plantations, palm oil and use of chemicals. Conservation initiatives are therefore needed to ensure protection of these endemic millipedes in Cameroon.

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