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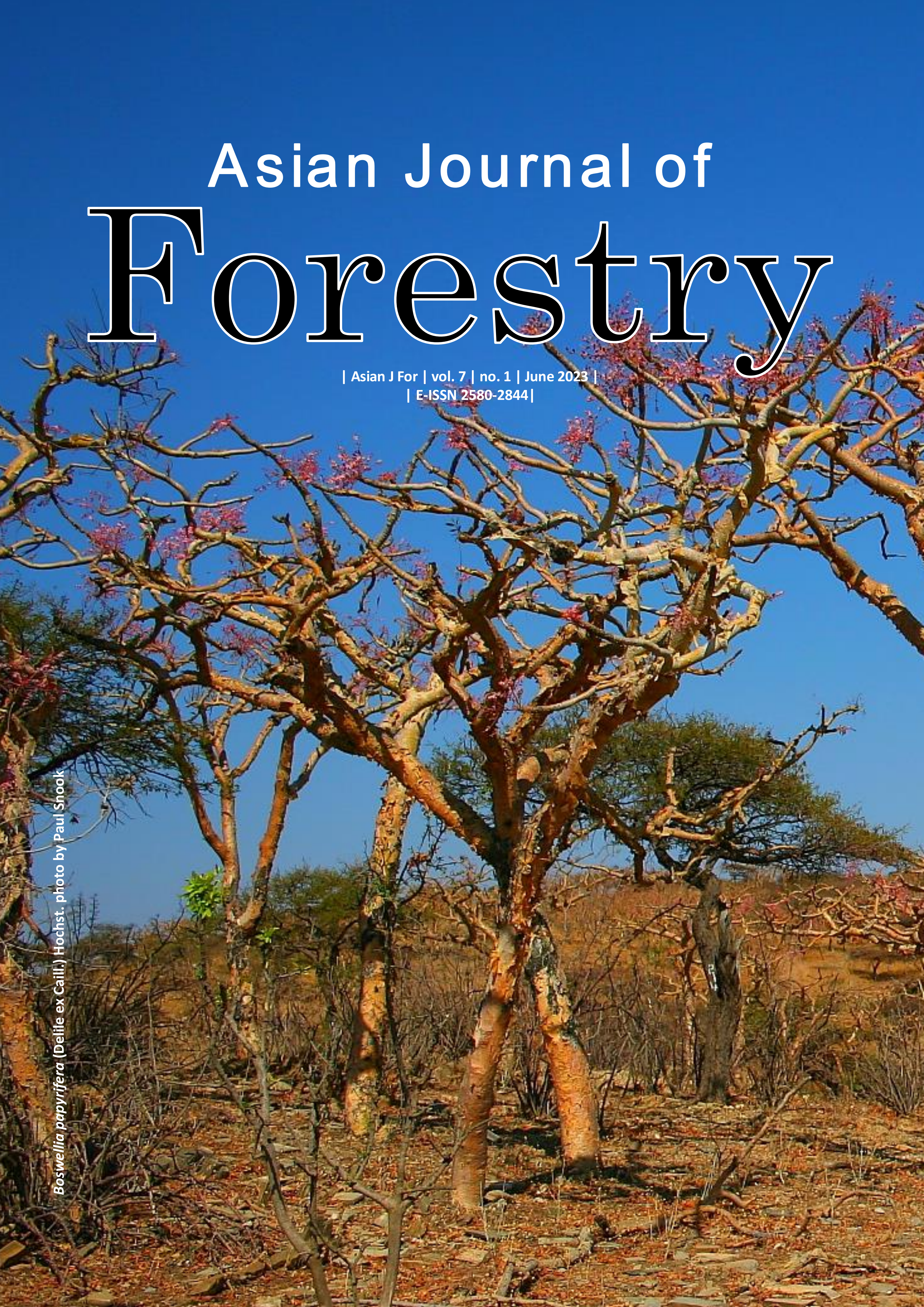

Boswellia papyrifera (Delile ex Caill.) Hochst. photo by Paul Snook

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Chapter in the book:

Webb CO, Cannon CH, Davies SJ. 2008. Ecological organization, biogeography, and the phylogenetic structure of rainforest tree communities. In: Carson W, Schnitzer S (eds.). Tropical Forest Community Ecology. Wiley-Blackwell, New York.

Abstract:

Assaeed AM. 2007. Seed production and dispersal of *Rhazya stricta*. 50th annual symposium of the International Association for Vegetation Science, Swansea, UK, 23-27 July 2007.

Proceeding:

Alikodra HS. 2000. Biodiversity for development of local autonomous government. In: Setyawan AD, Sutarno (eds.). Toward Mount Lawu National Park; Proceeding of National Seminary and Workshop on Biodiversity Conservation to Protect and Save Germplasm in Java Island. Universitas Sebelas Maret, Surakarta, 17-20 July 2000. [Indonesian]

Thesis, Dissertation:

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Recycling for deforestation reduction in Tanzania: Why are households not using waste charcoal?

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Abstract. *Ngowi NJ, Ngalawa AA. 2023. Recycling for deforestation reduction in Tanzania: Why are households not using waste charcoal?. Asian J For 7: 1-8.* Charcoal making and the subsequent waste charcoal produced contributes to deforestation and the production of greenhouse gases, the major drivers for climate change. Whereas climate change is increasingly becoming a public issue, Africa produces 63% of the total global wood charcoal. Since 2004, the demand for wood charcoal on the continent has risen by 30%, the highest in the world. The low efficiency of locally made earth kilns, between 8% and 36%, reported in Africa has contributed to increased waste charcoal production on the continent. However, more information is needed on the cycling use of waste charcoal in low-income countries. In order to improve forest resource use, we investigated the factors influencing the cycling use of wood waste charcoal in Kilosa District of East-Central Tanzania between 2020 and 2021. A total of 298 randomly selected households were involved in the survey. SPSS version 20 tools were used in the analysis through cross-tab descriptive statistics and the independent sample t-test. Results show that sex, age, expenditure on fuel energy, environment, and technology for cooking significantly affected household use of waste charcoal ($p < 0.05$). The availability of wastes ($v = 0.272$) was the most influential factor in the cycling use of waste charcoal. The paper shows that cycling the use of wood waste charcoal would reduce volumes of trees cleared for firewood and improve sanitation by removing rampantly disposed waste from the environment.

Keywords: Charcoal grates, deforestation, greenhouse, rampant, kiln

INTRODUCTION

The African continent ranks first in wood charcoal production, accounting for 30 million tonnes of global charcoal output (Steierer 2011). This trend has been increasing by 30% (https://www.grida.no/resources/7497). The increase in charcoal demand as the main energy source is attributed to many factors, including reliability, high calorific value, technology, urbanisation, and economic incentives (Steierer 2011), causing a significant increase in the production of the waste charcoal. Public policies, gender, cultural practices, behavior, technology, and cooking methods are among the factors that explain the choice of energy types in many developing countries (Masera et al. 2000; Wilson and Dowlatabadi 2007; Kowsari and Zerriffi 2011; Asada 2019). The changes in income between rural and urban households are associated with a shift in the use of energy type from the traditional to the modern one (Masera et al. 2000; Kowsari and Zerriffi 2011). Barnes and Halpern (2000) show that people maintain both the traditional and the modern energy types irrespective of changes in income or location. On the other hand, studies (Kowsari and Zerriffi 2011; Lewis and Pattanayak 2012; Sander et al. 2013; Tao et al. 2018; Wassie and Adaramola 2019, 2020; Doggart et al. 2020) assert that despite the introduction of various efforts including policy change, energy subsidies to electricity, and the use of alternative energy sources among others, still more than 90% of people in East Africa including Tanzania, prefer wood charcoal as the most favorite fuel energy for cooking, thus escalating pressure to forests and woodland ecosystems in the region (https://www.grida.no/resources/7497).

Studies (Antal and Grønli 2003; Ishengoma et al. 2016; Nweke 2017) report that waste charcoal such as tree barks, residues, and specks of dust are generated mainly due to the low efficiency of locally-made earth kilns during the carbonization process (Figure 1A), means used in the transportation, handling in the warehouses, and handling in the marketing or collection centers. Despite the potential of wood waste charcoal as recycled fuel energy, animal feed supplement Louis et al. (2018), and green manure, waste charcoal is considered useless in many low-income countries and thus disposed off with little or no consideration of its impact on the environment. The practice contravenes with research findings like the one by Eggleston et al. (2006), who report that the sector contributes to about 5% of the global greenhouse budget, with air pollution estimated at US$ 6.0 Billion in low-income countries in 2013 (World Bank and Institute for Health Metrics and Evaluation 2016). Thus, the rampant disposal of waste charcoal, as observed in the study area and illustrated in Figure 1B, may adversely contribute to greenhouse gases leading to increased global temperature and climate change.

Yu et al. (2021) show that the released amount of Carbon Dioxide gas (CO_2) through the energy from forest biomass is used by plants, thus making biomass fuel mainly renewable and carbon-neutral. Waste food, agricultural residues, and forests are among the biomass fuel sources of which the latter (forests) are the most vital biomass carbon producers (Gonçalves et al. 2018). However, Scheer and Moss (2012) report that cutting trees to obtain biomass fuel makes carbon neutrality out of the equation, leading to extra CO_2 in the environment, which contributes to the climate change phenomenon. Studies show that nearly 50% of renewable energy in the EU region is biomass, and the trend is slowly growing by 2030 (Ravilious 2020). In some EU countries, for instance, nearly 70% of renewable energy consumption in Denmark and around 25% in Sweden comes from forest biomass (Titus et al. 2021). In other places like Australia, Yu et al. (2021), biomass fuels alternatively, unlike fossil fuels, cut atmospheric CO_2 by 25 million tonnes per year.

Biomass, in general, and forest biomass, are significant energy sources in most developing and developed countries (Scheer and Moss 2012). However, approximately 15 billion trees are cut down annually, reducing the number of trees by 46% worldwide (Ehrenberg 2015). In Africa, where 93% of rural and 58% of urban households depends on wood biomass for fuel, estimates show that the extraction of wood fuels from trees contributes to 70% of deforestation on the continent (Wassie and Adaramola 2020, 2019). The scenario suggests that the utilization of wood biomass will remain fundamental and increase over time due to population growth and limited options for alternative energy sources in low-income populations (Subedi et al. 2014). It also suggests that more feedstock is required, resulting in bulk extraction of forest biomass for fuel (Titus et al. 2021). In Tanzania, deforestation has been accelerated by wood-fuel extraction, changing land use, and unsustainable extraction methods, among others (Scheer and Moss 2012; Manyanda et al. 2020). In some cases, the destruction is often not observed, as degraded forests often retain a closed canopy (Manyanda et al. 2020).

Different approaches have been implemented to reduce or curb deforestation and woodland degradation. They include: reusing biomass fuels, oil seeds for biodiesel, sugar plants for bioethanol, food waste, and pulp remains for biogas energy production, among others (C and E Advisory Ltd 2020, unpublished data). However, some countries like Djibouti, more prone to aridity than other countries in the region, have completely canceled any utilization of wood biomass at a large scale (C and E Advisory Ltd 2020, unpublished data). Subedi et al. (2014) show that biogas can lessen deforestation from wood fuel demand by 6% to 36% in 2010 and 4% to 26% by 2030. This is equivalent to 10-40% of absolute deforestation in 2010 and 9% to 35% by 2030.

Malimbwi and Zahabu (2008) and Wassie and Adaramola (2019) purport that the predominance use of wood fuel energy in domestic chores is the main cause of forest and woodland degradation in many low-income countries. For example, in Tanzania (https://www.climatelinks.org/resources/greenhouse-gas-emissions-factsheet-tanzania), report show that forests and changes in land use and agriculture are the largest sources of Greenhouse Gas Emissions (GHGs), contributing to 72.7% and 17.3%, respectively. On the other hand, energy and wastes contribute 7.8% and 1.6%, respectively. This situation invites the question of how countries like Tanzania can reduce the annual volume of trees cut, rampant disposal of waste charcoal, GHGs, and household budget on energy by formulating and adopting user-friendly strategies of recycling wood waste charcoal that eventually can contribute to improving community livelihoods and the environment. However, despite the potential of wood waste charcoal in recycled fuel energy, more is needed about the scope of utilization of these resources in Tanzania. Thus, unless the scope and determinants of wood waste charcoal utilization are well established, it is difficult to promote strategies for the use of recycled wood waste charcoal by the local community and integrate this into the local government plans for natural resources and livelihood security in the country.

Figure 1. Traditional earth kiln (A) and rampant disposal of waste charcoal at Maguha market (B)

This study aligns with Tanzania's Development Vision 2025 (www.mof.go.tz/mofdocs/overarch/vision2025.htm) and the National Environmental Policy (URT 2021), which emphasize the need for the reduction of waste for environmental sustainability by promoting environmentally - friendly technologies for sustainable development. The study on recycling waste charcoal in Tanzania aimed to understand the determinants of households' use of waste charcoal for reducing the cutting down of forests and woodlands for biomass fuel in east-central Tanzania.

MATERIALS AND METHODS

Description of the study area

This study was carried out in Kilosa District, Tanzania, located at latitude 5°55' and 7°53' South and longitude 36°30' and 37°30' East (Figure 2). The district occupies 14,918 km^2 of land (Ishengoma et al. 2016). Forests and woodlands occupy about 40% of the total land area. Village forest reserves occupy about 88,879 ha, national forests reserves 66,517 ha, district council forests reserves 8,168 ha, reserved village land forests 208,732 ha, and forest plantations are 1,692 ha (Ishengoma et al. 2016). The forest vegetation is characterized by miombo woodland mixed with shrubs and grasslands. The district experiences a semi-arid climate with an average annual rainfall of less than 1000 mm on low plains and a temperature of 25°C. The lowest temperature is 19°C in July, and the highest is 30°C in March. The district falls into three agro-ecological zones. The first is the mountains with an altitude of 2200 m extending along with the Eastern Arc system with pre-Cambrian metamorphic rocks covered with coarse soils. This area is suitable for wheat cultivation. The second zone is an upland plateau, also known as a cultivation steppe featuring an altitude of about 1100 m. Plains and a few hills with clay and loamy soils dominate this zone. The area is suitable for maize production and livestock keeping as the name depicts. The third zone comprises floodplains at a low altitude of 550 m, dominated by the Wami and Ruaha river basins. In 2012, the district had a population of 438,175 people (males 218,378 and females 219,797) (NBS 2013, unpublished data).

Research design, sampling process, and sample size

This study used both secondary and primary data sources. The secondary data were collected through a documentary review of various reports, journals, books, policy documents, and web pages. The sample size of 298 respondents shown in Table 1 was determined through a sampling process (Israel 1992).

$$\mathbf{n} = \frac{\mathbf{N}}{\mathbf{1} + \mathbf{N} \cdot (\mathbf{e})^{\mathbf{2}}}$$

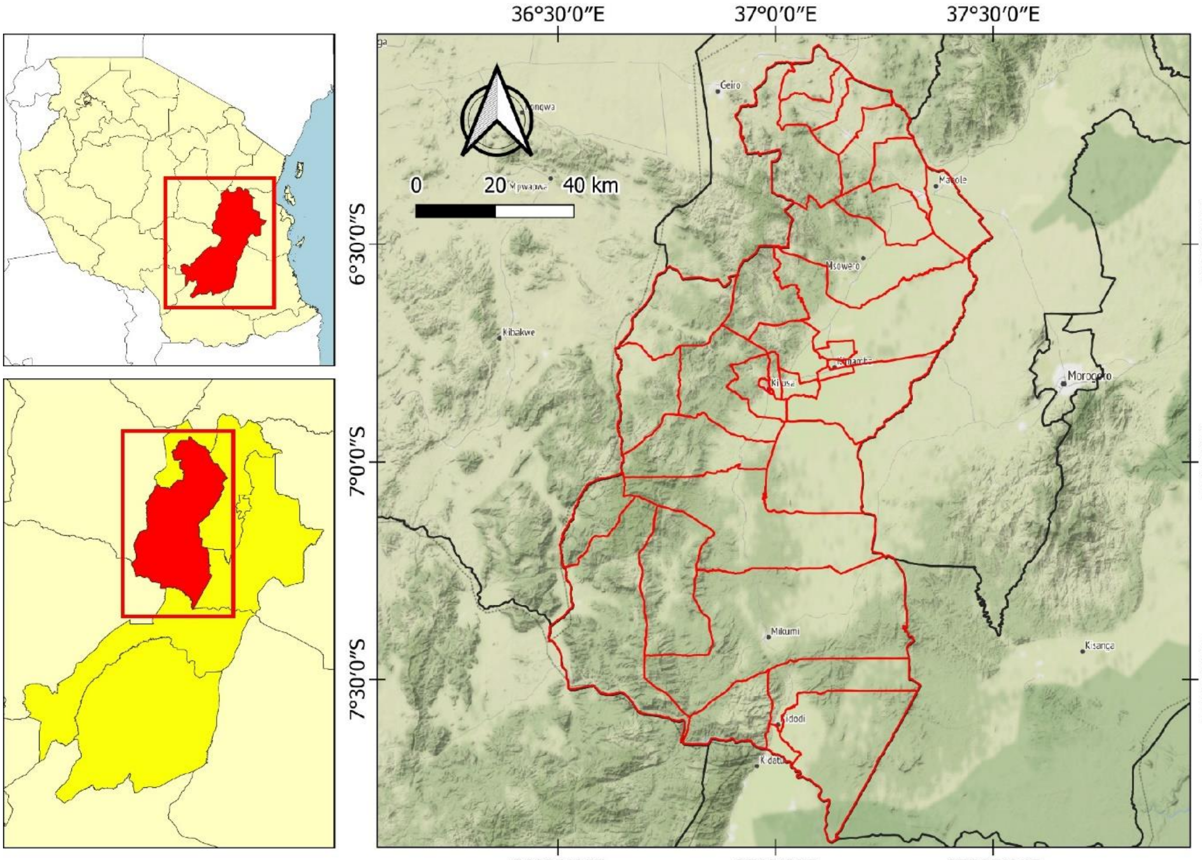

Figure 2. The location of the study area in the Kilosa District, Tanzania (Ishengoma et al. 2016)

In the formula, **n** is the Sample size, **N** is the Number of households, and **e** is the sampling error at 5%. A purposive sampling technique was used to select two study villages. The selection of the two villages was considered, among other factors: annual charcoal production (Ulaya Mbuyuni), marketing centers (Maguha), as well as people involved in the Sustainable Charcoal Project (both villages). This maximized the variability of data on waste charcoal (Ishengoma et al. 2016).

Data collection

Quantitative and qualitative approaches were used in gathering information. This is because a combination of both stories and statistics provided a better understanding of the problem being investigated than using either numbers or stories alone. Primary data were collected using a questionnaire administered to the heads of households. The questionnaire used to collect quantitative information included, among others, thirteen items on household cycling use of waste charcoal. The items were used to assess whether or not households had at least been using cycling waste charcoal a year before the survey period. In addition, qualitative information obtained through interviews with Focus Group Discussions (FGD) and personal observations were used to supplement quantitative information. People aged 18 years and above involved in the charcoal industry were interviewed to provide information on the charcoal-making process, use of waste charcoal, and factors a household considers influencing the cycling use of waste charcoal.

Determination of household cycling use of waste charcoal

The research framework in Figure 3 was used to determine the various features characterizing the cycling use of waste charcoal in Tanzania.

Independent variables: Socio-economic factors (gender, age, education, family size, income, house ownership), energy profile (asset holding, expenditure on fuel energy, routines and habits, and environment), and technology (calorific value, waste size, and grates). Together with the enablers (public policies and institutions) would influence household decisions on waste charcoal usage. In this study, cycling use of waste charcoal was applied as the dependent variable where the value of **“0”** was assigned as a non-user, i.e., a household that does not use waste charcoal, and **“1”** for a user. Households who had at least used waste charcoal within one year before the survey was used as the sampling frame because they could remember the information the study sought. Marie et al. (2021) have adopted a similar approach in Ethiopia.

Data analysis

The statistical significance relation between users and non-users of waste charcoal for quantitative data was assessed through IBM SPSS statistics with a p-value of 0.05 as a cut-off for statistical significance. Chi-square and independent sample t-test (Cramer’s v value) was used to determine any relationship between predictor variables (independent) and cycling use of waste charcoal (dependent variable) in the study area. A description method was used to present qualitative data collected through FGD and personal observation. To enhance clarity and more understanding of the subject, presentations have been supported by plates and direct statements from participants of the FGD.

The sequence presented in Figure 4 shows that the quantitative information collected first helped to inform the qualitative step and later assisted in the analysis and interpretation of the earlier.

Table 1. Sample size

Study area	Population	Number of households (N)	Respondents (n)
Ulaya Mbuyuni	3,257	581	237 (119)
Maguha	6,735	1,417	312 (179)
Total	9,992	1,998	549 (298)

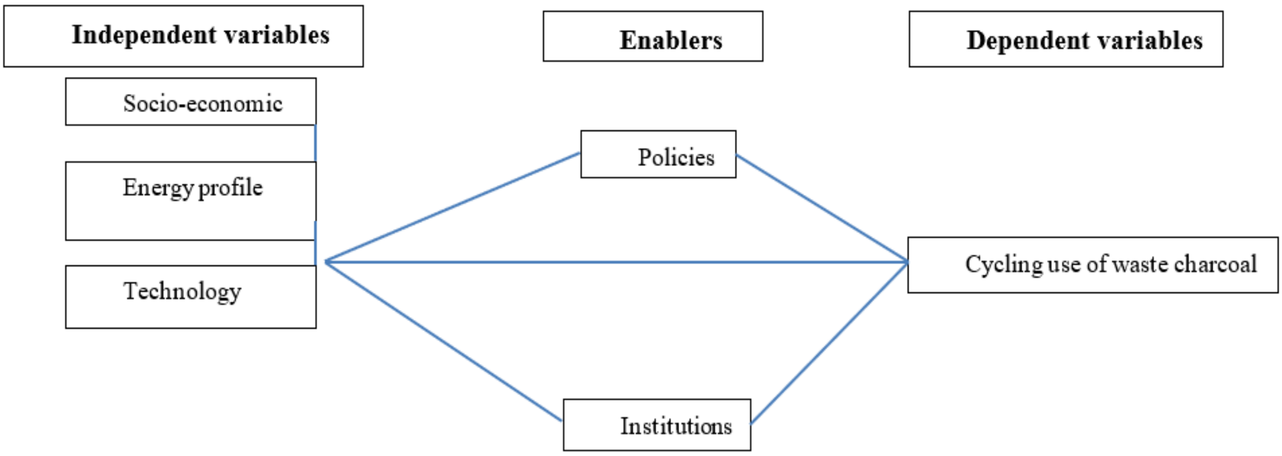

Figure 3. A research framework

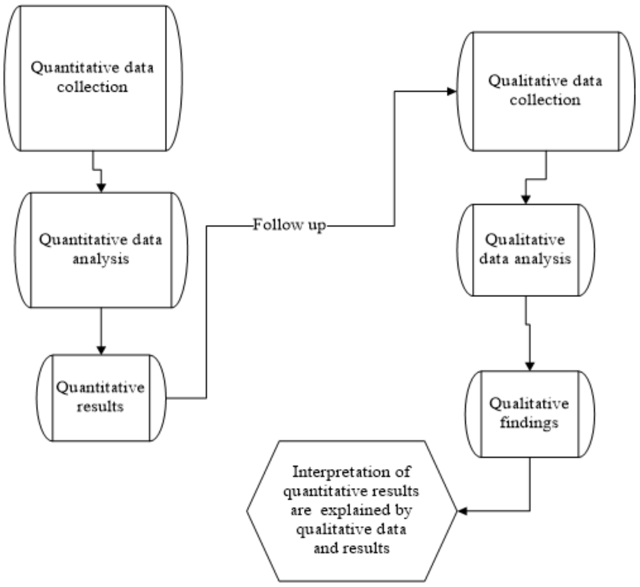

Figure 4. Processes adopted for quantitative and qualitative data collection and analysis

RESULTS AND DISCUSSION

Results

The results in Tables 2 and 3 show that a total of 298 respondents, determined by the sampling process, male-led households formed a large proportion of 164 respondents and 134 female respondents. Among waste-charcoal users, 129 (43.3%) females were leading in using waste charcoal by 23.5% to 19.8% for males (Table 3). The rest, 169 (56.7%), did not recycle waste charcoal. Results in Table 3 show that 30.2% of households used less than 50 kg of waste charcoal in a year. This was followed by 6.4% who used between 51 and 150 kg of waste charcoal. Only 3% were using 151 kg and above. The remaining 3.7% needed to learn how much waste charcoal they recycled.

The t-test results (Table 4) reveal that the mean age for non-waste charcoal users was higher (M = 3.20) than for waste users (M = 2.70). Age, sex, and expenditure of household head on fuel energy (see Table 4) show that age was the strongest factor determining the use of waste charcoal among household-socioeconomic factors (v = 0.257) followed by sex (v = 0.163) and expenditure on fuel energy (v = 0.123) and the relationship are significant ($p < 0.05$). Further, the results revealed that the mean age, education, family size, and house ownership for waste charcoal users were smaller than non-waste users' by 0.47 from 0.66 for waste users. Results show that the availability of waste charcoal (p = 0.68) was the most influential factor impacting the use of waste charcoal (v = 0.272), although it was not significant ($p > 0.05$) when compared to the environment and technology for cooking. Independent t-test results show that the difference in means for public policies and institutions-related factors was higher for waste charcoal users (M = 1.52) than non-user households (M = 1.43).

Table 2. Household cycling use of waste charcoal

Variable	Category		Total	χ2	df	p-value	Cramer's V
	Non-user	user					
Male	105	59	164	7.945	1	0.005**	0.163
Female	64	70	134				
Age	169	129	298	19.704	1	0.000***	0.257
Expenditure	86	212	298	4.508	1	0.034**	0.123
Policies and institutions	169	129	298	0.993	2	0.321	0.058
Wastes availability	169	129	298	22.114	3	0.68	0.272
Technology	169	129	298	7.034	1	0.008**	0.154
Education	169	129	298	1.164	3	0.818	0.062
Family size	169	129	298	3.782	3	0.189	0.113
House ownership	169	129	298	0.112	1	0.739	0.019
Income	169	129	298	11.613	3	0.922	0.197
Environment	169	129	298	19.916	4	0.020**	0.259
Access to recyclables	169	129	298	4.285	2	0.953	0.120
Asset holding	169	129	298	10.964	3	0.701	0.192

Note: **, *** are significant at 0.05 and 0.01 levels of significance, respectively

Table 3. Quantity of waste charcoal used per household in a year

Response			Quantity of waste used per household per year				Total
			Did not know	50 kg and below	51 - 150 kg	151 kg and above	
	No	Count (%)	26(8.7%)	84(28.2%)	30(10.1%)	29(9.7%)	169(56.7%)
	Yes	Count (%)	11(3.7%)	90(30.2%)	19(6.4%)	9(3.0%)	129(43.3%)
Total		Count (%)	37(12.4%)	174(58.4%)	49(16.4%)	38(12.8%)	298100.0%)

Table 4. Determinants of use of waste charcoal

Variables	Non-user (n = 169)		User (n = 129)		t-value	p-value
	Mean	SD	Mean	SD		
Sex	0.38	0.487	0.54	0.500	- 2.847	0.005**
Age	3.20	0.917	2.70	1.115	4.275	0.000***
Education	1.02	0.607	1.01	0.566	0.231	0.818
Family size	1.56	0.596	1.47	0.586	1.316	0.189
House ownership	0.80	0.402	0.78	0.414	0.333	0.739
Income	1.54	1.029	1.55	1.046	- 0.098	0.922
Expenditure on energy	0.66	0.474	0.78	0.419	- 2.132	0.034**
Asset holding	2.87	0.791	2.91	0.870	- 0.385	0.701
Environment	2.05	1.017	2.33	1.085	- 2.337	0.020**
Policies and Institutions	1.43	0.730	1.52	0.782	- 0.994	0.321
Technology for cooking	0.73	0.446	0.58	0.495	2.675	0.008**
Access to recycles	0.03	0.23	0.03	0.174	- 0.059	0.953
Availability of wastes	1.60	0.701	1.46	0.586	1.834	0.68

Note: **, *** are significant at 0.05 and 0.01 levels of significance, respectively

Discussion

The findings suggest that people do not use waste charcoal as a source of energy at the household level because they need more knowledge about how to use it. This study found that fewer households used waste charcoal than those that did not. This was supported by qualitative information from participants of the FGD, who noted that "waste charcoal is thrown away." They have no use". This was evidenced through observation in transect walk where waste piles were poured at different places (see Figure 1 B). When asked why waste charcoal was not used, they gave three reasons: (i) the incompatibility of the sizes of particles that constitutes the waste with the sizes of the grates of the cooking stoves, thus passing easily through grates within the cooking stoves, (ii) people do not have appropriate knowledge and skills of making the small particles big enough to fit in their cooking stoves, and (iii) waste charcoal they produce little energy, therefore, a limiting factor to its use. This study shows that people do not use it because their knowledge is restricted to using the waste in the form it exists without any modification. For example, waste charcoal was found as small particles used to light charcoal stoves in some households. This suggests that the people are aware of the possibilities of using the waste should they be trained on how to modify the small particles and make it big enough and thus compatible with the grates of their cooking stoves along with the capacity of the modified sizes of the wastes in generating enough energy for household use.

These results are supported by a study by Ma et al. (2013) in 28 OECD countries, which show that one factor might have a positive effect in one case and a negative one in others. Furthermore, these results are consistent with findings in Uganda by Asada (2019) and in Thailand by Dunn et al. (1982) that low calorific value for waste charcoal was suitable in rice cooking and steaming banana, respectively. Participants of the focus group discussion explained that: Waste charcoal was not used because they did not generate enough energy for household use. In their own words, they said, "Heat produced from waste charcoal is of low quality, and they burn for a short while. Who is to use such a source of energy, particularly when you come from farm "shamba" work, tired and hungry?". (Ulaya Mbuyuni, December 2020).

This quotation implies that people cannot modify the waste charcoal particles and make them feel like normal charcoal particles. Further, it suggests that the dominant economic activity of the people (agriculture) needs to give them more time to research how to modify the waste charcoal. The hand-to-mouth scenario that characterizes their agriculture has made them think that charcoal for household use should always be in a form that is ready for consumption.

Although household behavior was not among the 13 factors investigated, this study revealed that nearly 1% of respondents were unaware that fuel energy could be tapped by cycling waste charcoal. This has impacted the overall results from other factors investigated by showing that households that discovered the usefulness of charcoal wastes reused them. This study found that 3.7% of households using waste charcoal did not know the quantity of waste charcoal recycled because the very little waste generated by households was used by mixing with other charcoal of larger particle sizes and not cumulatively added to the total amount of waste produced.

The findings also suggest that the youth are more innovative than the elders. The youth had innovations of increasing their income by making use of the waste charcoal, as evidenced by the results showing that households that used waste charcoal were of relatively younger mean age (M = 2.7) compared to (M = 3.4) for non-user - households. The explanation for this is that households of younger age were more involved in charcoal making and transportation processes, so they had greater chances of accessing waste left in kins in the wild, during packing and transportation of charcoal or selling centers, and bringing them home for use compared to the elderly one. This study's results corroborate previous findings from Peša (2017) study, which found that behavior rendered large quantities of waste from sawdust in wood processing industries to remain unutilized in the Copper belt of Zambia. In addition, Brown et al. (2017), in a study conducted in South Africa, found that behavior was one of the limiting factors to adopting electric cooking from traditional energy sources.

Figure 5. Charcoal sacks found in the study sites, each weighing beyond 50 kg

Concerning the environment and health, findings suggest that people were aware of environmental and health hazards caused by mishandling and poor disposal of waste charcoal. Most charcoal makers and dealers did not use masks in undertaking their respective activities. This research shows the negative impacts of the increased waste from wood charcoal on the environment. According to Nyundo, one of the participants in the FGD said: “Waste charcoal contributes to human diseases by increasing health risks. When you inhale dust from waste charcoal, it can destroy the functioning of your lungs. However, if you work in the charcoal industry and stay without a shower, you will likely suffer from skin diseases. This is because the skin will be rough and dry. This dryness is likely to cause skin diseases as charcoal dust can penetrate your body through the small openings on the skin”. (Maguha, December 2020). This quotation reflects that people know the health hazards associated with poor handling of waste charcoal.

The environment variable shows a p-value of 0.020 and technology for a cooking p-value of 0.008, indicating a significant effect on waste charcoal cycling. For instance, the rain was found to have influenced the use of waste charcoal and household choice of energy type. Households with outdoor kitchens preferred using firewood over any other type of energy because it tended to cook faster, saving time for other activities at home, including farming. The results of this study are similar to the findings of the study conducted by Brown et al. (2017) in South Africa, which show that the environment has both direct and indirect influence on the use of types of energy sources. This research shows that waste availability ranked first, followed by environmental variables in influencing the cycling use of wood waste charcoal.

This research revealed that public policies and institutions did not impact waste charcoal utilization ($p = 0.321$) and Cramer’s v value ($v = 0.058$. These findings contradict those from Ma et al. (2013), Malimbwi and Zahabu (2008), and Doggart et al. (2020), which showed that government policies were the most important factor in improving the utilization of energy in low-income countries. The contradiction is mainly due to the enforcement of rules and guidelines according to the Government Notice (Forest Royalty Collection). The rules set a maximum of 50 kilograms (kg) per sack of charcoal as the basis for calculating royalty of Tanzanian Shillings 250/= per Kg of charcoal (equivalent to US $ 0.114/kg. Calculated at an exchange rate of 1.00 US $ = 2200/=Tanzanian Shillings). However, most sacks weigh beyond 50 kg, contrary to the guidelines. This uncontrolled weight is based on various factors, including the area where charcoal is being processed and tree species used. For instance, in *Brachystegia*, *Commiphora*, *Combretum*, and *Albizia* species-dominated woodlands, charcoal makers were found parking the charcoal bags beyond the recommended weight in order to ease the exercise of transporting the charcoal bags from the point of production, but also the practice provided an opportunity for charcoal traders to avoid payment of appropriate royalty to the respective authorities during transportation of the bags to the final destination where charcoal was sold (Figure 5).

In conclusion, this research shows that the waste availability variable ranked first, followed by environment, and age, in influencing the use of wood waste charcoal. Most households’ waste charcoal users used at least one bag of 50 kg or less in a year. This paper shows that cycling the use of waste charcoal would reduce volumes of trees cleared for firewood and rampant disposed of wastes from the environment. This study recommends that people be trained in appropriate knowledge and skills to make the small waste particles big enough to fit in their cooking stoves.

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Boswellia papyrifera a tree of economic importance in dry land Sudan: Dendrometric parameters and tapping characteristics

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Abstract. *Dafa-Alla M, Abdelkari HA, Abuelbashar AI, Hassan MH. 2022. Boswellia papyrifera a tree of economic importance in dry land Sudan: Dendrometric parameters and tapping characteristics. Asian J For 7: 10-16.* We conducted this research in Abugadaf Natural Forest Reserve (ANFR) located within Blue Nile state, Sudan. The objectives were to examine dendrometric parameters and to analyze tapping characteristics for frankincense production. We randomly distributed 97 sample plots; in each, we counted *B. papyrifera* trees, measured their diameter at breast height (dbh), total height, bole height, and crown diameter, and examined the correlation between crown diameter and dbh. We examined 116 tapping spots made on 13 randomly selected trees. In each tree, we measured upper and lower tapping heights, calculated potentially untapped bole height, counted the number of tapping spots, measured their dimensions, recorded their directions, and compared mean numbers of tapping spots at four directions using one-way ANOVA (α=0.05). Results revealed that the distributions of dbh and the total height of tapped *B. papyrifera* followed a bell shape, and that of bole height and crown diameter followed a bell-shaped skewed to the right. Results revealed a significant (P=0.000) correlation between crown diameter and dbh (N=499) that trees were tapped at the four directions without significant difference (P=0.427) between mean numbers of tapping spots at the four directions. The study concludes that current tapping practice doesn't strictly adhere to recommended tapping guides, particularly with minimum lower tapping height and preference of concentration of tapping in east-west directions.

Keywords: *Boswellia papyrifera*, dendrometry**,** frankincense, tapping spots

INTRODUCTION

The genus Boswellia's economic importance stems from the production of frankincense (Sommerlatte and Wyk 2022), which expresses the characteristic aromatic natural oleo-gum-resins that occur in all its parts (Gebrehiwot et al. 2003) and used as incense. Frankincense production differs widely among the five chief producing species, with *B. papyrifera* being the main producing species at present (Bongers et al. 2019**)**. Ecologically the species is important since it grows in harsh environments, where most tree species may not grow (Abiyu et al. 2010), are generally resource-poor regions with few livelihood alternatives, so harvesting frankincense in the dry season can contribute significantly to limited household income.

Frankincense is a natural resin obtained through incisions made into the trunk of trees of the genus *Boswellia*, enhancing the yield of resins. Usually, tapping involves deliberate shaving of a very thin, i.e., 2 mm deep and 4-8 mm wide, external circular layer of the bark along the stem (Figure 1), starting at 0.5m height using a hand tool known as Mengaf (Gessmalla et al. 2015) which is a hand tool with a metal end, about 2cm wide and a wooden handle (Figure 2) (Tadesse et al. 2004). Upon incision, the bark of *B. papyrifera* exudes a white milky liquid, which later hardens on exposure to air into globular or club-shaped droplets or tears (Figure 3) called frankincense (Lemenih and Kassa 2009).

Tapping process involves making repeated incisions into the bark over a series of months (Worku and Bantihun 2018) at a distance of about 30cm (Al-Aamri 2015) to 50cm (Tadesse et al. 2004). The particular details of the tapping, such as its duration, the time of year it is undertaken, and the interval between individual tappings, vary according to the species and the customs in the production area (Greenhalgh 2019). It is repeated four times, where the second and subsequent tapings are usually done at 15 - 30 days intervals. For small-size trees, tapping is done on the eastern and western sides (Gessmalla et al. 2015) and in four directions for larger trees.

The collection of resin starts after 30-35 days and extends up to the end of June, and generally, there are about 4-9 pickings per season, depending on the weather conditions (Ali et al. 2009). The hardened resin is removed at each tapping round, and the tapping wound is re-opened and enlarged (Tolera et al. 2013). Renewal of the wound at the correct interval of days is critical; otherwise, the old wounds may heal completely (Lemenih and Kassa 2011). During tapping, the cut surface gradually moves upward, and each year a fresh cut is started at a different site (Woldie 2011) immediately above the old wounds (Ali et al. 2009). Tree tapping is ceased 2-3 weeks before the rainy season (Gessmalla et al. 2015).

Figure 1. New (upper) and old (lower) tapping spots on the stem of *B. papyrifera*. Source: Abuelbashar (2020)

Figure 2. Mengaf is the traditional tool for tapping the *Boswellia* tree. Source. Ali et al. (2009)

Figure 3. Frankincense on the stem of *B. papyrifera* ready for harvest. Source. Abuelbashar (2020)

The number of tapping spots on each tree depends on the tree's diameter (Gebrehiwot et al. 2003) and height and foliage cover (Al Aamri 2015). The rules for forest use practices must be established to keep forests in good condition to sustain the provision of ecosystem services (Sist et al. 2014). Despite the economic and ecological benefits of the species, very little effort has been undertaken to develop the resource. Lack of reliable information is one of the major constraints that hamper species' efficient and economical development and use (Gessmalla et al. 2015). The underlying assumptions in this research were that current tapping intensity and frequency are appropriate while the tapping technique is traditional, wasteful, and needs technical improvements. The main objective of this study is to contribute to filling the information gap of the *B. papyrifera* resource base and tapping characteristics for frankincense production. Specific objectives are to analyze dendrometric parameters, explore diameter at breast height (dbh) and height class distribution, investigate current tapping practice characteristics, and explore statistical relationships between some measurable parameters of *B. papyrifera* in the Abugadaf natural forest reserve.

MATERIAL AND METHODS

Study area

We conducted this study in ANFR East of the Blue Nile River within Elrosairis locality, Blue Nile state, Sudan. It lies between longitudes 34° 50 45′ & 34° 54 45′ and altitudes 11° 25 10′ & 11° 30 10′ (Figure 4). With an estimated area of about 4,624.4 ha, it is one of a few dryland natural forests remaining within an extensive agricultural landscape in the state. ANFR is composed of thirteen tree species, including *Boswellia papyrifera* (Delile) Hochst., *Combretum aculeatum* Schweinf., *Lannea fruticose* (Hochst. Ex A. Rich.) Engl., *Sterculia setigera* (Delile), *Acacia Senegal* (L) Wild., *Anogeissus leiocarpa* (DC.) Guill. & Perr., *Balanites aegybtiaca* (L.) Delile, *Dichrostachys cinerea* (L.) Wight et Arn., *Acacia seyal* var *seyal* Delile, *Ziziphus spina-christi* (L.) Desf., *Hyphaene thebaica* (L.) Mart., *Adansonia digitata* L. and *Cordia sinensis* Lam. The most abundant tree species is *B. papyrifera,* with a maximum mean density of 51(±27) trees/ha (17.6%) relative to a mean tree density of 290 (±49) trees/ha at ANFR. *B. papyrifera* is one of the species in the forest that has been most utilized for economic benefits from frankincense production.

Dendrometric characteristics of *B. papyrifera*

We initially conducted a reconnaissance survey to explore the range of dbh of *B. papyrifera* trees and to estimate the mean minimum dbh used for tapping *B. papyrifera* for frankincense production. We randomly distributed 97 circular sample plots (0.10 ha) across the forest; in each tree, we counted *B. papyrifera* trees, measured their dbh (1.3m above ground level) to the nearest 0.1cm using a caliper, height to the nearest 0.1m using a Suunto clinometer (Suunto Corp., Finland) and crown diameter to the nearest 0.1m using a measuring tape. Next, we described the structure of the species using frequency distributions of dbh, total height, bole height, and crown diameter. Finally, we calculated the arithmetic mean diameters (D) of the trees using equation (1) (Mengich et al. 2020).

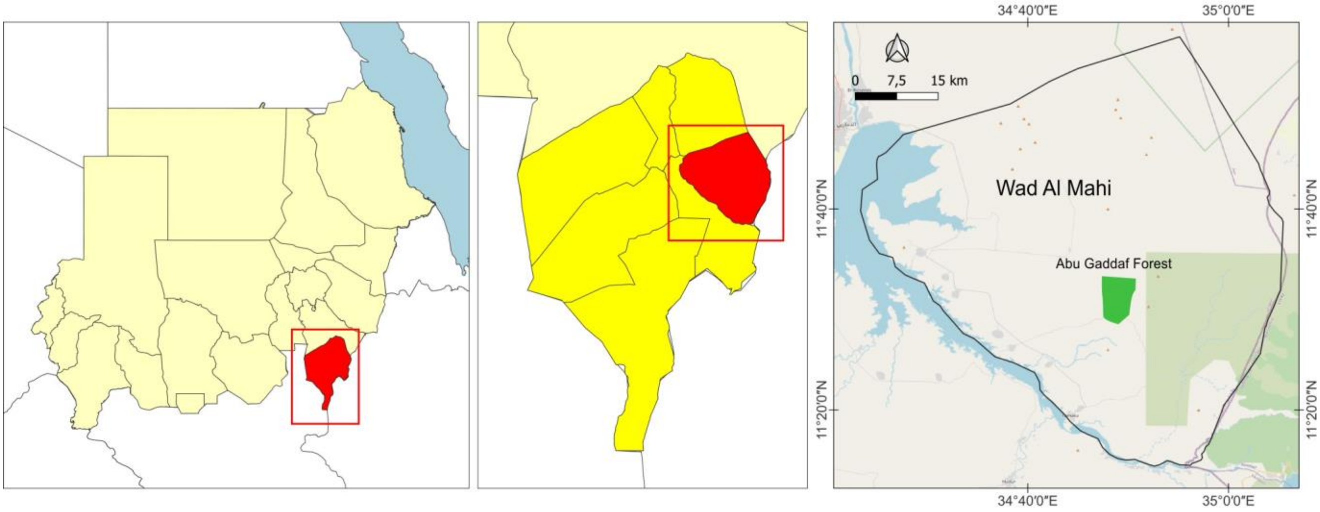

Figure 4. Study area in Elrosairis, Blue Nile state, Sudan

$$D = \frac{1}{n}\sum_{i=1}^{n} d_i$$

Where
d_i = diameter of the individual (ith) tree
n = total number of trees in the plot

The dbh range was divided into ten dbh classes with a dbh class interval of 5cm. Then, we assigned each tree to an appropriate dbh class. Next, a clinometer will be used to measure the total height and bole (stem) of standing *B. papyrifera* trees.. then, the trees will be grouped into 5 total-height classes that begin at 2m and end at 14.5m with a class interval of 2.5m and seven bole-height classes that start at 1.0m and end at 7.9m with a class interval of 1.0m. Finally, it's calculated using the equation below (2).

$$H = \frac{1}{N}\sum_{n=1}^{N} h_i$$

The determination of the crown diameter of each tree was used as the average of two perpendicular widths through the crown vertical projection; the first measurement was along the direction of the longest branch of a tree. The second was along the direction perpendicular to the first measurement (Kimondo et al. 2014) using a tape measure to the nearest 0.1m. Next, the distributed crown diameters are into seven classes that start at 1.0m and end at 18.4m with a class interval of 2.5m. Finally, the Pearson correlation between crown diameter (cm) and dbh (cm) was examined.

In each sample plot, the total and pole heights are estimated to the nearest 0.1m and upper and lower tapping heights to the nearest 1cm. Then, a total of 116 tapping spots were examined by randomly selecting thirteen *B. papyrifera* trees. Then counted, the mean number of tapping spots per tree, measured their length, width, and depth dimensions, and determined their geographical directions was done. Finally, a one-way ANOVA was used to compare the mean number of tapping spots (α=0.05) made in the four tapping directions.

RESULTS AND DISCUSSION

Dendrometric parameters of *B. papyrifera*:

The mean dbh of *B. papyrifera* trees was 28.9cm (±7.4) (range 9.0-51.0cm) which is comparable to (Groenendijk et al. 2012) and is different from Abuelbasher (2020), who reported complete absence of trees with dbh ≥ 35cm southeastern Sudan and Gelaye (2012) who reported diameters of harvested trees from 11 to 30.5 cm. Figure 5 displays that the abundance of stems of tapped trees was very high in the middle, where 51.4% of the total individuals lie within two central dbh classes, 25-29cm, and 30-34cm. Trees with larger and smaller dbh were less abundant. The dbh distribution of tapped *B. papyrifera* was bell-shaped, displaying a higher number of individuals in the middle diameter classes and progressive reduction towards the lower and higher diameter classes. The same distribution pattern is reported by Gelaye (2012) in Ethiopia. The pattern of diameter class distribution indicates the general trends of population dynamics and recruitment processes of a given species (Abyot et al. 2014), with a bell-shaped diameter distribution indicating a hampered regeneration (Hido et al. 2020). The considerable number of individuals of the species in the middle diameter classes could be managed sustainably to improve their regeneration and produce resins. The diameter distribution of many large top canopy species does not follow a reversed-J tendency as they have relatively few individuals in the smaller sizes, suggesting little regeneration and recruitment of species in more recent years (Feyera et al. 2007). A short left-end tail of the dbh distribution curve is attributed to excluding untapped seedlings and saplings from the survey.

The results showed that the mean total height (N= 499) of the species was 7.9m (±2.2m) (range 2.0-13.0m) which is comparable to Gelaye (2012), who reported tree height from 4.57 to 13m. The distribution of *B. papyrifera* by total height demonstrates a higher number of medium-height individuals than short and tall individuals. The total height distribution (Figure 6) follows a bell-shaped pattern, with a maximum number of trees (16.2%) within a central bole height class 7.5-8.4m and decreasing numbers towards low and high total height classes. The short left-end tail of the total height distribution curve is attributed to exclusion of <10cm dbh seedlings and saplings from the survey; that at the right-hand tail demonstrates that some 15% of trees surpass the maximum total height of up to 12 m due to more favorable growing environments. The latter is slightly lower than Ethiopia's 14-16 m (Groenendijk et al. 2012).

The mean bole height was 3.1m (±0.3) (range 1.2-7.0 m). The distribution of bole height was bell-shaped (Figure 7) and skewed to the right, indicating that it was generally clustered around a low bole height class. 80.2% of total individuals lie within the lower three bole-height classes with fewer frequencies in higher ones. The short left-end tail of the bole height distribution curve is attributed to the presence of a few very large dbh resin-producing trees in the survey. Therefore, the species can be categorized with progressively declining numbers of trees with increasing bole height.

The results revealed that the mean crown diameter of *B. papyrifera* was 7.4m (±3.2) (range 1.0 - 16.0m) which was relatively higher than the 4.5 m reported by Gelaye (2012). The distribution of the crown diameter took a bell shape (Figure 8), skewed to the right. 85% of trees lie within the first four crown classes, and only about 15% of tree crowns frequencies lie within the three larger crown classes towards the right end. Crown diameter indicates that most *B. papyrifera* individuals are medium size trees. Analysis of Pearson's two-tailed correlation between crown diameter and dbh (N=499) revealed a significant (α=0.01) positive correlation (P=0.000, r= 0.758). The result supports the earlier findings of Schrender et al. (1993) that dbh is correlated to crown diameter.

Characteristics of current tapping practice

Results unveiled that tapping of frankincense tree at ANFR was made at a minimum dbh of 9.0 cm and up to 51.0 cm, comparable to 9.0-45.0cm reported by Eshete et al. (2005). However, this result contradicts the commonly applied tapping guide of a minimum tapping dbh of 10.0cm. The cessation of tapping on small, younger trees (10-15 cm) is recommended, as small trees have a low frankincense yield. Tapping causes more physiological stress to smaller trees than larger trees (Eshete et al. 2012), and they may not be able to recover their wounds after tapping (Abiyu et al. 2010). Prohibiting tapping trees with small dbh allows them to grow to a larger, more productive size and reproduce without interference (Gonzalez 2020).

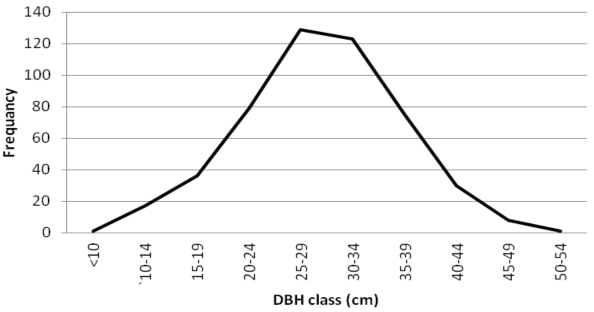

Figure 5. Distribution of *B. papyrifera* in Abugadaf Natural Forest Reserve by DBH classes

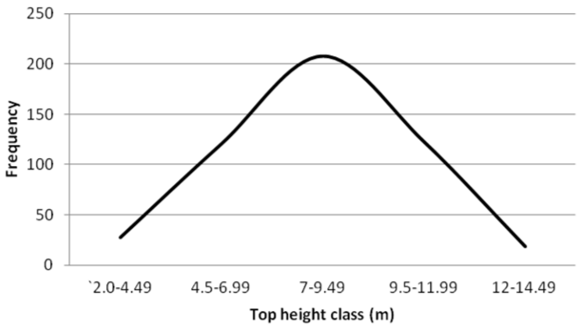

Figure 6. Distribution of *B. papyrifera* in Abugadaf Natural Forest Reserve by total height classes

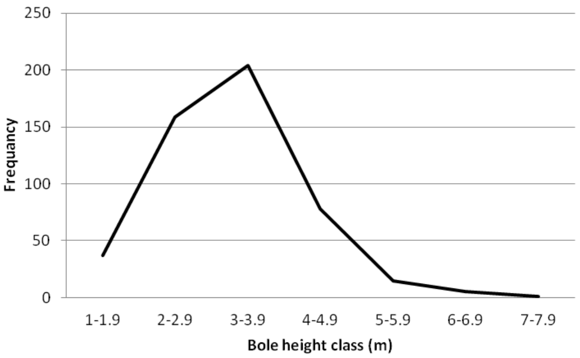

Figure 7. Distribution of *B. papyrifera* in Abugadaf Natural Forest Reserve by bole height classes

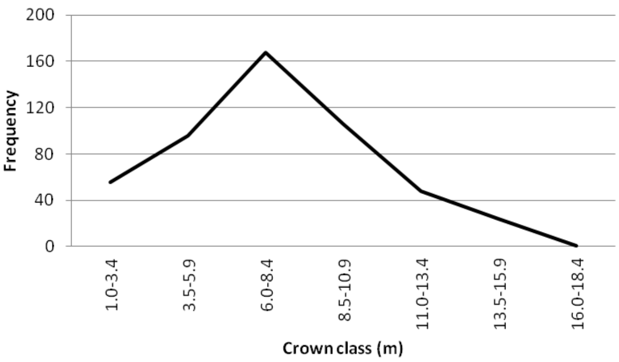

Figure 8. Distribution of *B. papyrifera* at Abugadaf Natural Forest Reserve by crown diameter

The results of this study revealed that mean upper and lower tapping heights (N=492) were 1.76m (±0.13) (range 1.05-2.20m) and 0.64m (±0.15) (range 0.3-1.0m) (Figures 9 and 10), respectively. Generally, the estimates of mean tapping heights fall within the range of the tapping heights of 0.50-2.0m applied in the local practice (Mengistu et al. 2013). With a mean bole height of 3.1m, the potential tapping height (over 0.64m mean lower tapping height) under current practice was 2.46m. Therefore, the mean actually tapped part of the bole height was 1.12m (±0.2) (range 0.5-1.6m), equivalent to 45.5%. The result implies that more than half of the potential tapping height was untapped as it was out of reach of tappers, and it remains a potential for tapping, which may help in better distribution of tapping spots for a better frankincense yield. That is of particular significance as close to 20% of the trees fall within bole height classes bigger than the mean bole height of the forest. In the current tapping practice in Sudan, the tree's height is not a determinant factor for the number of wounds since the laborers tap into the reachable parts of the trees (Nour 2008). The traditional practice of tapping requires innovative techniques to reach higher parts to maximize the use of bole height. The resin yield of *B. papyrifera* can be improved by modifying the tapping techniques (Ali and Gebauer 2007), tools, and skills.

While minimum and maximum numbers of tapping spots per tree were 1 and 12, results revealed that the mean number of tapping incisions per tree was 9.0(±3.0). The result generally follows the recommended tapping intensity per tree of 6-16 in Ethiopia (Abiyu et al. 2010, Lemenih and Kassa 2011), 6-10 in southeastern Sudan (Nour 2008 and 6-15 in West Sudan (Abtew et al. 2012) The result may be well understood provided that only 10.8% of the trees fall within small dbh class of <20 cm which is recommended to be tapped at two spots, one in each of east and west directions, 41.7% and 47.4% of trees fell within dbh class of 20-29 cm and >30cm, respectively, where the recommended tapping intensities are 3 and 4 tapping spots per a direction per tree (Gebrehiwot et al. 2003).

Results disclosed that trees were tapped on average at nine spots in the four directions with a mean of 3 spots in the East and 2 in each of the other three directions. The intensity of tapping is predominantly in the East, equally followed by North and South, and lower in the West direction (Figure 11). All-direction tapping of trees illustrates the dominance of trees with larger dbh (55%). In addition, resin yield per tree increases significantly with several tapping spots (Gelaye 2012). Table 1 displays a one-way ANOVA of the mean numbers of tapping spots (N=116) between four directions (α=0.05). One-one ANOVA revealed no significant difference (p=0.427).

The mean length, width, and depth of end-of-season tapping incisions were 6.6cm(±2.5) (range 3.0-17.0), 4.6cm(±1.3) (range 2.0-9.0), and 4.3mm(±1.6) (range 1.0-8.0), respectively. Tapping has recently intensified since the number of tapping wounds, and their dimensions have increased, leading to severe damage of many B. papyrifera trees and reduced production of viable seeds (Ogbazghi 2006).

Currently, due to the high demand for frankincense, up to 27 tapping spots are made per tree in some commercial sites (Kebede 2010). Recently, more tapping spots per tree and more tapping rounds per season are becoming common (Eshete et al. 2012) may be due to increasing global and domestic demand for frankincense**.**

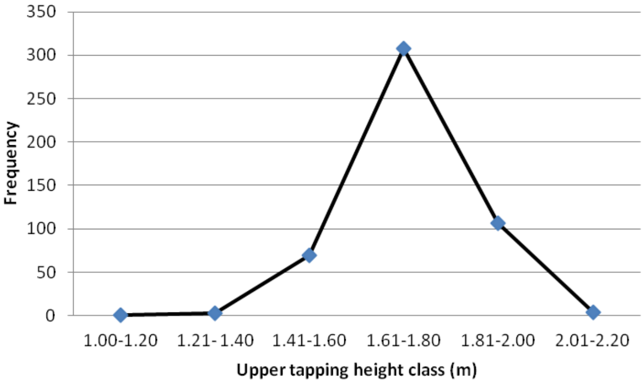

Figure 9. Distribution of *B. papyrifera* trees at Abugadaf Natural Forest Reserve by tapping upper-height

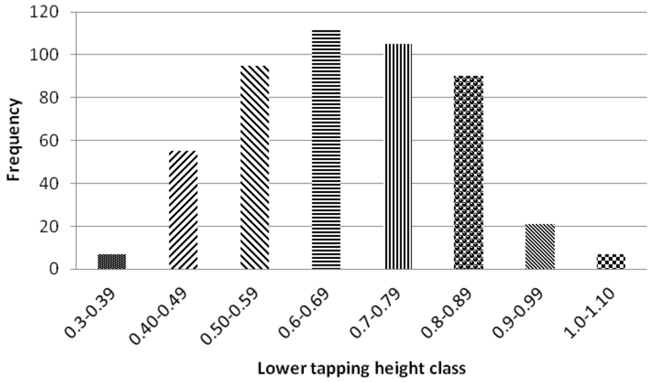

Figure 10. Distribution of *B. papyrifera* trees at Abugadaf Natural Forest Reserve by tapping lower-height

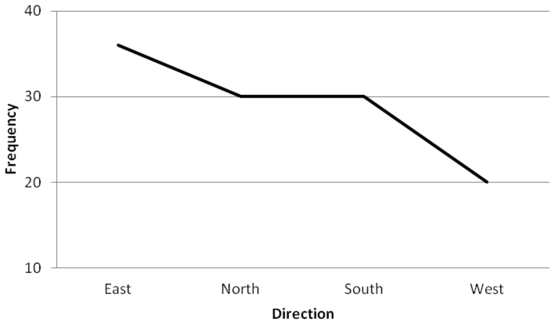

Figure 11. Frequency of tapping spots in the direction at Abugadaf Natural Forest Reserve

Table 1. One-way ANOVA of the direction of tapping and number of tapping spots

Sources of variation	Sum of Squares	df	Mean Square	F	Sig.
Between the direction of tapping	41.775	3	13.925	0.934	0.427
Within tapping directions	1669.217	112	14.904		
Total	1710.991	115			

The research concludes that the dbh distribution of tapped *B. papyrifera* is dominated by trees in the middle dbh classes that could be managed sustainably to improve their regeneration and production of frankincense. There is a significant and positive correlation between crown diameter and dbh, which calls for a better understanding of the influence of tapping on physiology and biomass production, particularly for small frankincense-producing trees. The current tapping practice of *B. papyrifera* for frankincense production doesn't firmly follow recommended tapping guides. Considerable bole length is currently unutilized as it is out of laborers' reach. The study recommends the development of a tapping protocol that maintains the resource base and sustains frankincense production. The protocol should emphasize the improvement of tapping techniques, tools, and skills.

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Floristic composition and structure of closed and open forests in the Banco National Park, Abidjan, Côte d’Ivoire

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Abstract. *Gnahore E, Douffi KG-C, N’guessan YJ, Amba AJG, Ibrahima D, Kone M, Bakayoko A. 2023. Floristic composition and structure of closed and open forests in the Banco National Park, Abidjan, Côte d’Ivoire. Asian J For 7: 17-26.* Natural ecosystems are pressured to degradation caused by human activities. To develop conservation guidelines for these ecosystems, it was necessary to have enough information on their biodiversity. This study aimed to investigate the species richness, diversity, structure, and composition of vegetation in the Banco National Park (BNP), Abidjan, Côte d’Ivoire, in two forest types, namely closed and open forests. Vegetational data was collected using a sampling plot and itinerant methods. The results showed that 345 and 283 species were recorded in the closed and open forest, respectively, suggesting that the closed forest appears richer than the open forest. Fabaceae, Rubiaceae, and Apocynaceae were the most dominant families in both forest types. Both forest types had a high Shannon-Weaver diversity index and high Pielou equality index. The most important species in the open forest included *Chrysophyllum subnudum* Baker (Sapotaceae), *Allanblackia floribunda* Oliv. (Clusiaceae) and *Funtumia africana* (Benth.) Stapf (Apocynaceae), while in the closed forest were *Strombosia pustulata* Oliv. (Olacaceae), *Turraeanthus africanus* (Welw. ex C.DC.) Pellegr. (Meliaceae), and *Monodora myristica* (Gaertn.) Dunal (Annonaceae). The diametric structure showed the greatest proportion of individuals with low diameter classes, indicating regeneration potential. However, there were great differences in stand structure between the open and the closed forest, especially in high-diameter classes, indicating that the open forest has been pressured by timber cutting. The results of this study can serve as baseline information to develop conservation and rehabilitation strategies in BNP to sustain its biodiversity components.

Keywords: Banco National Park, Côte d’Ivoire, closed forests, conservation, forest structure, open forests

INTRODUCTION

Forest ecosystems provide many ecosystem services necessary for life maintenance, including water supply, carbon sequestration, flood regulation and protection against desertification and soil erosion. In addition, forest vegetation contributes to climate change mitigation and biodiversity conservation (Koubouana et al. 2016). Forests, particularly tropical forests, contain high biological diversity, including at the genetic level (Slik et al. 2015). Nonetheless, many forests in the world are pressured by deforestation and forest degradation, although such pressures are not uniform and vary according to the density of the population, the accessibility from urban areas, the quality of basic infrastructure, and the economic potential of the forest including if it is converted to other land use (e.g., agriculture, plantation). The destruction of forests due to human activities leads to species’ disappearance and ecosystem services deterioration (Toko et al. 2012; McDonald et al. 2013). The pressures on forests are expected to increase due to population and economic growth since there would be an increase in the use of forest products (e.g., wood) and forest services (Gnahore et al. 2022).

Situations related to managing the environment and natural resources are major challenges humans face today. In African countries, the trend toward degradation of forest resources due to human activities was worrying. This is due to, among other things, the high dependence of populations on natural resources. West Africa has a significant annual deforestation rate, with an estimated more than 3.94 million hectares of forest lost from 2010 to 2020 (FAO and PNUE 2020). In Côte d’Ivoire, the richest and most diverse forests are now among the most threatened ecosystems on the planet (Achard et al. 2002). The forests in the country are heavily affected by agriculture, which is the main driver of deforestation (Kouakou et al. 2019). This loss of natural forests results in the destruction of some natural habitats and the reduction of forest cover, which reduces biodiversity (Flores et al. 2018). The destruction of Ivorian forests has increased considerably, urging the government to implement conservation strategies by creating national parks and nature reserves. These practices have been recognized worldwide as the most effective strategy for conserving natural resources (Wondie 2015). However, despite creating national parks and nature reserves in Côte d’Ivoire, such conservation areas are infiltrated by populations to fulfill their daily needs. This is the case of the Banco National Park (BNP) under various human pressures (Gnahore et al. 2022).

Most African states have based their conservation strategy on creating and extending protected areas to secure their plant and animal resources better. Nonetheless, two decades after creating some protected areas such as the BNP, conservation is still a major challenge. Today, BNP has become the preferred target of the local population in search of firewood, food, and medicinal plants (Gnahore et al. 2022). In addition to these deplorable conditions for the protection of the BNP, there is the development of human settlements and neighborhoods near the park (Oura 2012). Several authors have reported these human activities as the dominant forms of disturbance of vegetation structure and physiognomy within protected areas (Tankoano et al. 2016). This situation has led to disturbances in the vegetation in the park and in the current state of knowledge, changes in land use, floristic richness and vegetation structure are unknown. Despite the degradation of forest resources and the modification of landscapes in the BNP, the precise and available information on vegetation dynamics and floristic diversity in this protected area was insufficient. Knowledge of forest area, floristic composition and its dynamics provides information essential for establishing and monitoring environmental and economic policies.

Concerning the increasingly strong anthropogenic pressures, an ecological problem today is understanding how tropical plant communities' dynamics are affected by human disturbances. The long-term conservation of biodiversity depends on knowledge of the structure, species richness and ecological characteristics of vegetation (Okende 2021). It is, therefore, more than urgent to assess the remaining forest in protected areas in Côte d'Ivoire to ensure better protection. It was necessary to assess floristic diversity to know the state of plant resources, hence motivating this study. Knowledge of the current vegetation dynamics, floristic composition and structure of BNP is important for sustainable management of the park. The main objective of this study is to contribute to a better knowledge of the flora in BNP by investigating the floristic parameters (i.e., richness, composition and floristic diversity) and structural parameters (i.e., density, diameter class, basal area) of closed and open forests in the park.

MATERIALS AND METHODS

Study area

The Banco National Park (BNP), Côte d'Ivoire was created on 31 October 1953. It covers an area of 3834.34 ha and is located between latitude 5°23' N and longitude 4°03' W. It is located within the city of Abidjan between the communes of Abobo, Adjamé, Attécoubé and Yopougon (Figure 1). The river network is essentially composed of the Banco River. This river is fed by groundwater related to rainwater infiltration under the forest canopy. The city of Abidjan, where the BNP is located, has a sub-equatorial (Atitean climate) with four humid and cool winter seasons. The average annual rainfall recorded by the SODEXAM meteorological station from 2010 to 2021 was 1733 mm. The average temperature was 27.2°C with an amplitude of 4.3°C (Tiébré et al. 2014). The soil of the BNP has the same composition as all the soils of the Abidjan region. The type of soil, wet all year round, was said to be psammohygrophile.

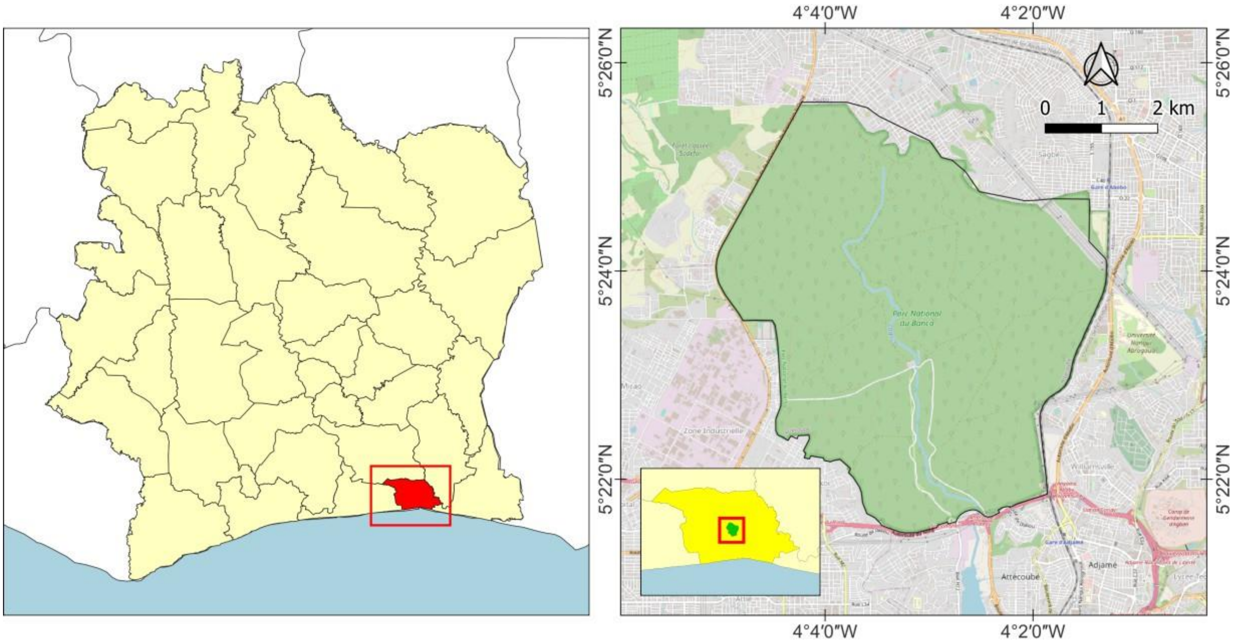

Figure 1. Map of the study area in Banco National Park, Côte d'Ivoire

The vegetation of the BNP consists mainly of dense evergreen humid forest. The dominant trees are *Turraeanthus africanus* (Welw. ex C.DC.) Pellegr (Meliaceae), *Synsepalum afzelii* (Engl.) T.D.Penn. (Sapotaceae), *Berlinia confusa* Hoyle (Fabaceae), *Blighia welwitschii* (Hook.f.) Brenan (Fabaceae), *Coula edulis* Baill. (Olacaceae), *Dacryodes klaineana* (Pierre) H.J.Lam (Burseraceae), *Lophira alata* Banks ex C.F.Gaertn. (Ochnaceae), *Petersianthus macrocarpus* (P.Beauv.) Liben (Lecythidaceae), and *Piptadeniastrum africanum* (Hook.f.) Brenan (Fabaceae). It was specific to an association vegetal, typical of the southeast of the country: the psammohygrophile forest with *T. africanus* and *Heisteria parviflora* Sm. (Olacaceae). In addition, the park is subject to strong anthropogenic pressure on the periphery due to illegal activities such as poaching, harvesting of non-timber forest products, etc.

Vegetation data collection

Inventories were carried out using two sampling methods: plot and itinerant methods. The plot method is a classic method already used by various researchers (e.g., Missa et al. 2018; Tesfay et al. 2019; Misganaw et al. 2021). For this study, we established 130 square plots measuring 25 m x 25 m (625 m²) arranged regularly on each side with the help of stakes. In these squares, all woody species (trees, shrubs, and lianas) were inventoried, taking into account diameters greater than or equal to 10 centimeters at the breast height of the ground. Initially, the floristic inventory concerned all woody species (lianas, shrubs, and trees) with a DBH (diameter at breast height) ≥ 10 cm. The second step was to delimit within these plots a nested plot with a size of 5 m x 5 m (25 m²), called regeneration plots. Here, all plant species (shrubs, lianescents, or herbaceous) encountered were identified without considering their DBH. These circumference measurements were made using a two-meter-long metric ribbon. We used the itinerant method to increase the chances of encountering the maximum number of species (Gnahore et al. 2018; Missa et al. 2018). This second method involved going through each study plot in all directions, noting all newly encountered plant species. All species that had not been documented in the initial surveys were recorded to complete the floristic list of the area.

Samples of unknown species were collected for the preparation of a herbarium. These herbarium samples were identified at the Swiss Center for Scientific Research in Côte d'Ivoire. Two forest types, open and closed forests, were considered to determine differences in structural and floristic parameters (Figure 2). The open forests present a state of deterioration, more or less advanced due to human pressures, while the closed forests are intact and never affected by human pressures.

Data analysis

Species Importance Value Index (SIV) and Family Importance Value (FIV)

The information collected on the field sheets was used to describe diversity, floristic composition, and plant structure. This collected data was entered, classified, and processed using Word (2013 version) and Excel (2013 version).

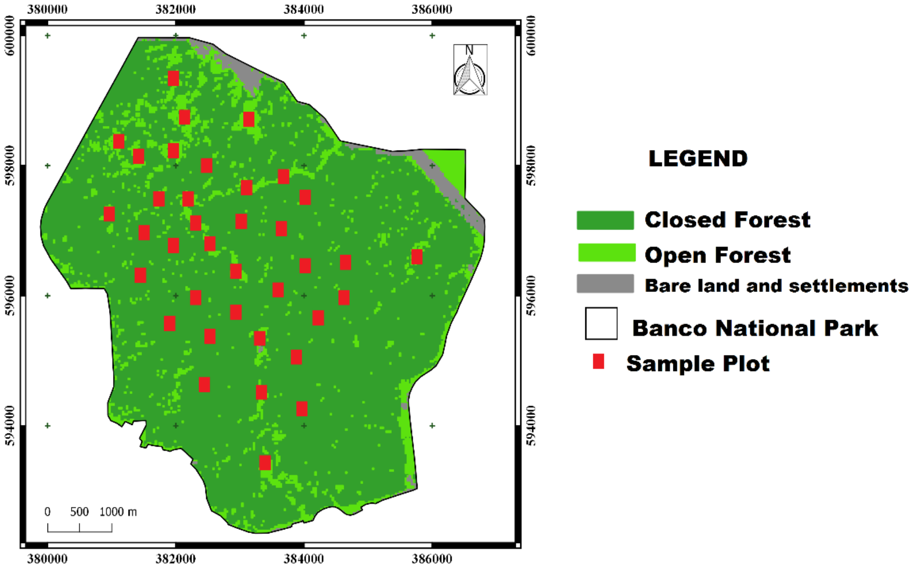

Figure 2. Arrangement of transects and observation plots in Banco National Park, Côte d'Ivoire

Concerning floristic parameters, floristic richness and composition were determined through the frequencies of occurrence in each type of forest. Structural parameters were assessed through stem frequency, density, basal area and species dominance. The importance of species or families in a given plant community was assessed from the Species Importance Value Index (SIV) and the Family Importance Value (FIV). The Importance values ranged from 0 to 300. When the value is zero, there is no dominance. Based on the different parameters, dominant species or families were defined as those with a value ≥ 10 (Missa et al. 2018; Thammanu et al. 2021). Those different parameters have been calculated according to the following formulas:

SIV = Relative density + Relative frequency + Relative dominance

FIV = Relative density + Relative dominance + Relative diversity

Where:

$$\text{Relative Density} = \frac{\text{Number of stems of species}}{\text{Total number of stems}} \times 100$$

$$\text{Relative frequency} = \frac{\text{Frequency of a species}}{\text{Sum of all species}} \times 100$$

$$\text{Relative dominance} = \frac{\text{Total basal area of a species}}{\text{Total basal area o all species}} \times 100$$

$$\text{Relative diversity} = \frac{\text{Number of species in a family}}{\text{Total number of species}}$$

The basal area (BA) of all trees in the sample plots was calculated using the formula:

$$BA = \sum \pi \left(\frac{d}{2}\right)^2$$

Where BA = Basal Area (m^2 / ha), d = diameter at breast height (cm) and π = Pie (3.142).

Diversity indices

To compare the floristic diversity between the two forest types, the Shannon-Weaver (H’) and Pielou equitability (E) indices were used (Pielou 1966) and calculated using the PAST 2.16 software. The indices were calculated from the specific contribution of each species. These indices provide a better understanding of an ecosystem’s biological diversity. The Shannon-Weaver Diversity Index (H’) quantified the floristic diversity of each area. It is often used to express the diversity of different sites. The following formula calculated it:

$$H' = -\sum_{i=1}^{s} pi\ln pi$$

Where H’ is the Shannon-Weaver index, s is the total number of species, pi is the proportion of individuals in the ith species, and ln is the natural logarithm.

From the Shannon-Weaver Diversity Index, the Pielou equitability index (E) was deducted. It provides information on the distribution of numbers of each species. It varies from 0 when a single species strongly dominates to 1 when every species has equal numbers. This index was calculated from the formula below:

$$E = \frac{H'}{\ln S}$$

With S: total number of species (species richness), H’ was the Shannon-Weiner index, and ln was the natural logarithm.

Average stocking density of major species

The average density of stems per hectare assessed the average stocking density of the main species of the BNP. The average density is the quotient of the total number of individuals inventoried in the biotope by the total area sampled in the environment in hectares (Hosny et al. 2018). The formula calculated it:

$$N = n/S$$

Where: N the average density, expressed in individuals per hectare; n: the total number of individuals inventoried and S the total area sampled in the environment in hectares.

Diameter classes

Diameter classes are used to characterize the diameter structure of vegetation. The distribution of stems of tree species in different diametric classes evaluated the diameter classes in this study. In this study, the number of stems of all species was assessed by diameter class, which was used to construct histograms. In this study, ten diameter classes were used (Hosny et al. 2018; Misganaw et al. 2021): [10-20 cm], [20-30 cm], [30-40 cm], [40-50 cm], [50-60 cm], [60-70 cm], [70-80 cm], [80-90 cm], [90-100 cm] and [100 cm; + ∞].

Sorensen similarity index (Cs)

The Sorensen similarity index was used to assess the similarity between the two forest types using the formula:

$$Cs = \frac{2S}{Y + Z} \times 100$$

Where: Y = number of species in the area y; Z = number of species in the area z; S = number of common species in both forest types. The index of 0 indicates that both forest types have totally different species and the index of 10 implies that both forest types have exactly similar species

Statistical analysis

For the comparison of the means of the structural and floristic parameters between the two forest types, Student’s parametric t-test was used. The two forest types were considered as independent samples. This test was valid only for populations with normal distribution and equal variances (Tiébré et al. 2014). The significance test was performed at $p = 0.05$ (Tesfay et al. 2019; Thammanu et al. 2021). Normality was tested using the Shapiro-Wilk test. When the calculated probability was significant, the Tukey test compares the means two by two and assesses the significant differences between them. The XLSTAT 2014 software was used to carry out these statistical tests.

RESULTS AND DISCUSSION

Species richness and floristic composition

The floristic inventory identified 556 species combined in open and closed forests. A total of 345 plant species belonging to 211 genera and 73 families were identified in the closed forest against 283 species belonging to 229 generas and 80 families in the open forest. In addition, the number of species presents exclusively in the open forest was greater than that encountered in the closed forest. Indeed, 260 species were observed only in the open forest, while 156 species were only in the closed forest. Among these exclusive species, we noted, for example, *Bombax buonopozense* P.Beauv. (Malvaceae), *Capparis tomentosa* Lam. (Capparaceae), *Croton hirtus* L'Hér. (Euphorbiaceae), *Eulophia gracilis* Lindl. (Orchidaceae) and *Majidea fosteri* Sprague Radlk. (Sapindaceae) in the open forest. In the closed forest, we found *Trilepisium madagascariense* DC. (Moraceae), *Clerodendrum polycephalum* Baker (Lamiaceae), *Decorsella paradoxa* A.Chev (Violaceae), *Panda oleasa* Pierre (Pandaceae), and *Phyllocosmus africanus* Hook.f. Klotzsch (Ixonanthaceae). The most significant families in the open forest in decreasing order were Fabaceae (40 species), Apocynaceae (23 species), Rubiaceae (18 species), Annonaceae (15 species), and Euphorbiaceae (15 species). At the specific level, the most commonly encountered species were *Baphia nitida* Lodd (Fabaceae), *Cola heterophylla* (P.Beauv.) Schott & Endl (Malvaceae), *Strombosia pustulata* Oliv. (Olacaceae) and *Dichapetalum pallidum* (Oliv.) Engl. (Dichapetalaceae).

In the closed forest, the most dominant families with at least 10 species were Fabaceae (35 species), Rubiaceae (32 species), Apocynaceae (20 species), Annonaceae (13 species), Sapindaceae (13 species), Malvaceae (12 species), Celastraceae (11 species) and Meliaceae (10 species). The families Fabaceae, Rubiaceae and Apocynaceae alone represented 87 species or 25.21% of all species. The most abundant species were *Albertisia cordifolia* (Mangenot & Miege) Forman (Menispermaceae), *Angylocalyx oligophyllus* (Baker) Baker f. (Fabaceae), *Cola caricifolia* (G.Don) K.Schum (Malvaceae), *C. edulis*, *T. africanus*, *Isolona campanulata* Engl. & Diels (Annonaceae), *Microdesmis keayana* J.Léonard (Pandaceae), *S. pustulata*, *B. welwitschii*, *P. africanum* and *Rhaphiostylis cordifolia* Hutch. & Dalziel (Icacinaceae). The most abundant families of the two forest were Rubiaceae, Fabaceae and Apocynaceae. Student t Test statistical analysis showed a significant difference between the two forest types (P < 0.0001). Comparison of the means of species-specific richness in the dominant families reveals a significant difference (P < 0.05) between the families Fabaceae and Apocynaceae. However, there was no significant difference (P = 0.057) within the Rubiaceae family (Table 1).

Diversity indices of the different forests types

The floristic diversity was estimated using the Shannon-Weaver (H') and Pielou (E) indices. In the open forest, the diversity is categorized as high with the Shannon-Weaver (H') index of 3.36, while the equitability of the Pielou index of 0.84 corresponds to high equitability. On the other hand, in the closed forest, the diversity index was high, with a value of 3.26, while equitability was high, with a value of 0.83.

Species and Families Importance Value Indices

The Importance Value Index reveals that the most important species in the open forest with a value greater than or equal to 10 % in decreasing order were *C. subnudum, Allanblackia floribunda* Oliv. (Clusiaceae), *Funtumia africana* (Benth.) Stapf (Apocynaceae), *Elaeis guineensis* Jacq. (Arecaceae), *Pentaclethra macrophylla* Benth. (Fabaceae) and *Bridelia grandis* Pierre ex Hutch. (Phyllanthaceae) (Table 2). The most ecologically important species in closed forests were *S. pustulata, T. africanus*, and *Monodora myristica* (Gaertn.) Dunal (Annonaceae), *F. africana, Antiaris toxicaria* Loes. *var. africana* Scott-Elliot ex A.Chev (Moraceae), *B. confusa, Buchholzia coriacea* Engl. (Capparaceae), *C. edulis* and *L. alata* (Table 2). Similarly, the predominant families in the open forest with important value index greater than 10% in decreasing order were Fabaceae, Apocynaceae, Ebenaceae, Clusiaceae, Arecaceae, Phyllanthaceae, Meliaceae, and Sapotaceae. In the closed forest, the most significant families were Fabaceae, Meliaceae, Apocynaceae, Annonaceae, and Olacaceae (Table 3).

Table 1. Floristic parameters in two different forest types of Banco National Park, Côte d'Ivoire

Floristic parameters		Forest types		Statistical parameters	
		Open forest	Closed forest	T	P
Species	Total	283	345	21.9	< 0.0001*
	Average number	31.39 ± 1.02^b	40.19 ± 0.19^a		
Genera	Total	229	211	17.9	< 0.0001*
	Average number	28.72 ± 0.9^b	38.56 ± 0.17^a		
Families	Total	80	73	5.7	0.018*
	Average number	21.62 ± 0.48^b	25.51 ± 0.09^a		
Fabaceae	Total	35	40	8.511	0.01*
	Average number	11.375 ± 0.46^a	9.3 ± 0.48^b		
Apocynaceae	Total	28	20	8.647	0.01*
	Average number	6.125 ± 0.56^a	4.2 ± 0.29^b		
Rubiaceae	Total	23	32	4.2	0.057
	Average number	2.25 ± 0.66^a	3.7 ± 0.34^a		

Note: Average = (mean ± standard error). *Significant overall means effects, p < 0.05. Means with different letters are significantly different based on Student t test

Density, basal area and species representativeness

In the open forest, 19 stems with a diameter at breast height greater than or equal to 10 centimeters over an area of 625 m² corresponding to 304 stems per hectare, were identified. *Millettia zechiana* Harms (Fabaceae) (22.82%) was the most represented species, followed by *F. africana* (18.47%) and *Macaranga beillei* Prain (Euphorbiaceae) with 17.39%. In the closed forest, 496 stems were inventoried per hectare, or 28 stems on an area of 625 m². *F. africana* and *A. floribunda* were the most abundant, representing 59.36% and 25.55%, respectively. These two species alone represented 84.91% of this forest's total number of individuals. The basal area of the closed forest was 36.22 m² per hectare. The species *B. confusa* had the highest basal area and occupied 1.62 m² per hectare, followed by *T. africanus* with 1.07 m² per hectare and *F. africana* with 0.89 m² per hectare. For the open forest, the basal area was 17.33 m² per hectare. *Pycnanthus angolensis* (Welw.) Warb. (Myristicacaeae) occupied 2.02 m² per hectare, followed by *M. zechiana* with 0.13 m² per hectare and *M. beillei* with 0.14 m² per hectare. The three species contributed 22.62% to the total basal area. Despite this variation, the difference between average densities was insignificant (Table 4). The closed forest had an average basal area of 1.791 ± 0.003 m² per hectare and had the highest value for the square plots measuring 25 m x 25 m (625 m²). As for the open forest, the average basal area was 0.36 ± 0.013 m² per hectare. The mean comparison test reveals that the difference in basal areas between the two forest types was statistically significant (Table 4).

Diameter distribution

The diameter class distribution of all woody species of the open forest showed that the classes 10 to 20 cm contained the highest number of stems, with 3308 stems per hectare, followed by the class between 20 and 30 cm, with 761 stems per hectare. For classes between 30 and 40 cm; 40 to 50 cm; 50 to 60 cm, and 60 to 70 cm contained 69, 14, 10, and 5 stems per hectare. The dominance of individuals of small diameters between 10 and 20 cm is clearly noticeable, followed by the class between 20 and 30 cm. In general, a steady decrease in the number of stems was observed when the diameter classes increased. An absence of stems from the class between 70 and 80 centimeters in the whole was noticed. In the closed forest, the class of 10 to 20 cm in diameter had the highest density (3308 stems per hectare) against 896 stems per hectare in the open forest (Figure 3).

Table 2. The Importance Value Index (IVI) of the species in two differents forest types in Banco National Park, Côte d'Ivoire

Species	RDo (%)	RD (%)	RF (%)	IVI (%)
Closed forest				
Strombosia pustulata Oliv.	28.00	4.70	10.15	42.85
Turraeanthus africanus (Welw. ex C.DC.) Pellegr.	13.71	4.93	8.47	27.11
Monodora myristica (Gaertn.) Dunal	5.95	3.04	9.91	18.90
Funtumia africana (Benth.) Stapf	1.81	6.59	10.15	18.55
Antiaris toxicaria Loes. var. *africana* Scott-Elliot ex A.Chev	9.73	2.00	5.81	17.54
Berlinia confusa Hoyle	0.48	5.26	11.19	16.93
Buchholzia coriacea Engl.	1.35	3.78	10.33	15.46
Coula edulis Baill.	1.83	3.04	9.09	13.96
Lophira alata Banks ex C.F.Gaertn.	2.35	3.59	6.81	12.75
Pycnanthus angolensis (Welw.) Warb.	1.83	1.98	5.57	9.38
Vitex grandifolia Gürke	5.79	1.29	2.06	9.14
Allanblackia floribunda Oliv.	6.27	0.78	2.06	9.11
Myrianthus libericus Rendle	1.60	1.15	4.33	7.08
Other species (332)	19.30	57.87	4.07	81.24
Total (%)	100	100	100	300
Open forest				
Chrysophyllum subnudum Baker	27.83	33.33	55.64	116.80
Allanblackia floribunda Oliv.	20.48	5.03	8.57	34.08
Funtumia africana (Benth.) Stapf	0.47	12.58	10.25	23.30
Elaeis guineensis Jacq.	6.03	3.93	9.41	19.37
Pentaclethra macrophylla Benth.	10.03	2.47	4.60	17.05
Bridelia grandis Pierre ex Hutch.	1.47	12.00	0.21	13.68
Diospyros sanza-minika A.Chev.	3.99	0.45	1.88	6.32
Guarea cedrata (A.Chev.) Pellegr.	0.47	5.33	0.41	6.21
Drypetes chevalieri Beille ex Hutch. & Dalziel	3.98	0.45	1.67	6.10
Antiaris toxicaria Loes. var. *africana* Scott-Elliot ex A.Chev	2.06	0.45	3.55	6.06
Coula edulis Baill.	3.67	0.45	1.04	5.16
Microdesmis keayana J.Léonard	0.22	0.65	0.21	1.08
Other species (271)	19.03	22.93	2.56	44.79
Total (%)	100	100	100	300

Note: RDo: Relative dominance, RD: Relative density, RF: Relative frequency, IVI: Importance Value Index

Species that had small diameters between 10-20 cm in the closed forest were *S. pustulata*, *M. keayana*, *B. nitida* and *C. edulis* while *T. africanus*, *Ceiba pentandra* (L.) Gaertn. (Malvaceae) and *P. africanum* were recorded in the largest diameter class. Species such as *M. myristica*, *A. toxicaria* Loes. var. *africana* Scott-Elliot ex A.Chev, *B. confusa* Hoyle had been mainly found in the intermediate class (30 to 60 cm) while *Vitex grandifolia* Gϋrke (Lamiaceae), *Myrianthus libericus* Rendle (Urticaceae), *Hymenostegia afzelii* (Oliv.) Harms (Fabaceae) et *Omphalocarpum ahia* A.Chev. (Sapotaceae) were recorded in large classes (70 to 90 cm) in diameter. *L. alata*, *B. coriacea* and *A. floribunda* were found in the 90-120 cm diameter class.

Acacia mangium Willd. (Fabaceae), *Senna siamea* (Lam.) H.S.Irwin & Barneby (Fabaceae), *Anthonotha macrophylla* P.Beauv. (Fabaceae) and *B. nitida* were the majority in the smallest diameter class (10 to 20 cm) in the open forest. *Taberneamontana crassa* Benth. (Apocynaceae), *V. grandifolia* and *Gilletiodendron kisantuense* (De Wild.) J.Leonard (Fabaceae) were the species with the largest diameters (90 to 120 cm), while the two largest trees (*C. pentandra*, *P. africanum* and *A. floribunda*) were over 100 cm in diameter. In general, the appearance of the histograms showing the diameter class distribution resembles the reverse J-shape relationship (Figure 3).

Discussion

A total of 556 species were counted across both forest types. The results showed that 345 and 283 species were recorded in the closed and open forest, respectively suggesting that the closed forest appears richer than the open forest. The higher number of species in the closed forest could be explained by the protection and monitoring of this forest by the Ivorian Office Parks and Reserves officers. This result agrees with the findings reported by Soro et al. (2019) in the Taï national park (Côte d'Ivoire). The disturbances could also explain this lower richness in the open forest suffered in this area that would have favored the proliferation of pioneer species (Prévost 1981). Kouakou et al. (2019) have showed through their work that human disturbances are estimated at 95% and that only 5% are of natural origin in the world. Indeed, local communities infiltrated the BNP to collect plants (Gnahore et al. 2022). This situation may cause a reduction in the number of species in the open forest. The low number of species recorded in the open forest is similar to the number of species reported by Yadav et al. (2020) in the Terai region of Nepal.

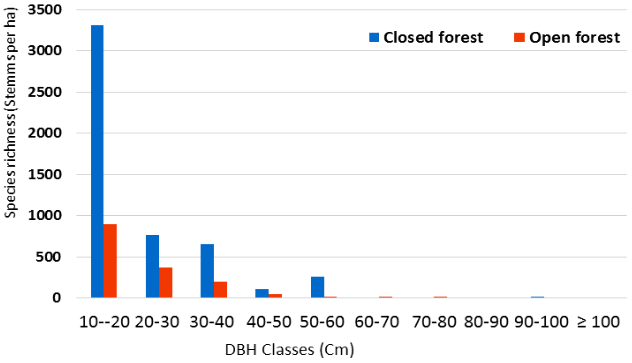

Figure 3. The diameter class distribution of stems in two different forest types of Banco National Park, Côte d'Ivoire

Table 3. The Importance Value Index of Families (VIF) of two different forest types in the Banco National Park, Côte d'Ivoire

Families	RDo (%)	RD (%)	RF (%)	VIF (%)
Closed forest				
Fabaceae	7.48	10.14	23.21	40.83
Meliaceae	22.72	2.89	8.86	34.47
Apocynaceae	10.23	5.79	17.25	33.27
Annonaceae	2.92	3.76	6.73	13.41
Olacaceae	5.05	1.44	4.84	11.33
Lecythidaceae	3.78	1.15	4.93	9.86
Moraceae	4.94	2.02	2.03	8.99
Ochnaceae	2.99	1.44	3.59	8.02
Myristicaceae	4.21	0.57	1.98	6.76
Capparaceae	1.64	0.86	3.78	6.28
Clusiaceae	3.73	1.73	0.78	6.24
Lamiaceae	1.17	1.73	1.84	4.74
Urticaceae	0.66	1.44	1.15	3.25
Other families (60)	28.48	65.04	19.03	112.55
Total (%)	100	100	100	300
Open forest				
Fabaceae	7.45	24.31	17.77	49.53
Apocynaceae	17.38	14.61	4.44	36.43
Ebenaceae	14.18	1.37	4.44	19.99
Clusiaceae	7.61	5.37	2.22	15.2
Arecaceae	10.08	3.93	1.11	15.12
Phyllanthaceae	7.98	2.62	3.33	13.93
Meliaceae	5.44	1.51	5.55	12.5
Sapotaceae	7.11	2.16	3.06	12.33
Olacaceae	3.96	1.5	2.22	7.68
Moraceae	2.04	0.72	3.33	6.09
Putranjivaceae	2.83	0.45	2.22	5.5
Pandaceae	2.2	0.65	1.11	3.96
Other families (68)	11.74	40.8	49.2	101.74
Total (%)	100	100	100	300

Note: RDo: Relative dominance, RD: Relative density, RF: Relative frequency, VIF: Importance Value Index of Families

Table 4. Structural parameters in two different forest types of Banco National Park, Côte d'Ivoire

Structural parameters	Forest type		Statistical parameters	
	Closed forest	Open forest		
Density (stems / ha)	496	432	-	
Average number of density	288 ± 5.093a	146.73 ± 23.053a	t = 2.344	P = 0.099
Basal area (m² / ha)	36.22	17.33	-	-
Average number of basal area	1.791 ± 0.003a	0.36 ± 0.013b	t = 3.305	P < 0.0001*

Note: Average = (mean ± standard error). *Significant overall means effects, p < 0.05. Means with different letters are significantly different based on Student t test. (-) indicates data absent

Fabaceae and Apocynaceae were the dominant families in the studied area. These families are noted as predominant in species in other forest ecological studies. Fabaceae's dominance is also characteristic of old-growth forests. The dominance of the Fabaceae family could be explained by the fact that all the plots studied in the park still retain their forest aspects. This family is also reported as the most diverse at the species level in the Gra-Kahsu natural vegetation in southern Tigray of Ethiopia (Tesfay et al. 2019). The most important families shown by the index of importance in the closed forest were Fabaceae, Rubiaceae, Apocynaceae and Ebenaceae, while those in the open forest were Fabaceae, Rubiaceae and Apocynaceae, suggesting that Fabaceae and the Rubiaceae are the most important families in Banco National Park.

Several studies agree with our observations in Ivorian forests (Tuo et al. 2017). By comparison, the Rubiaceae family found in different forest types had a similar species richness to the study done by Senbeta in the Maji and the Berhane-Kontir forests (Senbeta et al. 2014). The Importance Value Index was used to show the importance of species for good conservation (Tadele et al. 2013; Berhanu et al. 2017) and to understand the ecological significance of the vegetation species in community structure (Premavani et al. 2014). The high proportion at the species level of *S. pustulata*, *F. africana*, *T. africanus* and *M. myristica* in the closed forest indicates a good recovery of stands and this could be explained by the high amount of litter that enriches the soil and the absence of anthropogenic pressures. These results are in line with other studies by Gnahore et al. (2018) in the unburned savannah in the Lamto Scientific Reserve (Côte d'Ivoire) and Zin and Mitlöhner (2020) in the primary and secondary moist evergreen forests in the Tanintharyi Nature Reserve (TNR) Buffer Zone, Myanmar. Similarly, several studies have shown that many of the most prominent species are used for conserving protected areas (Tadele et al. 2013; Berhanu et al. 2017). The species with a lower Importante Value Index may indicate a threat and must be considered in conservation measures (Getie and Getahun 2020).

Based on the number of common species, the Sorensen similarity index was 47.6%. This index, ranging from 0 to 100% and less than 50%, indicates a low similarity between the two forest types. The low percentage of similarity index could be explained by the fact that the two forest types belong to different plant communities, which vary in floristic composition. A similar study was reported by Tesfay et al. (2019) in Gra-Kahsu natural vegetation, southern Tigray of Ethiopia.

The Shannon-Weaver diversity index and the Pielou equitability index were high in both forest types, although the open forest had a slightly higher Shannon-Weaver diversity index. The difference in the value of the Shannon-Weaver index could be explained by the presence of anthropized formations in the open forest due to the impact of human activities (Kouakou et al. 2019). According to Bouko et al. (2007), species diversity is affected by the degree of degradation. The results agree with Naidu and Kumar (2016), which stated that high diversity corresponds to a high species richness. Concerning the Pielou equitability index, the values obtained showed a similarity between the two forest types. This similarity indicates a homogeneous species distribution throughout the forest (Misganaw et al. 2018; Kouakou et al. 2019). The open forest had more heterogeneous environmental conditions because it was near the local communities and consequently highly prone to illegal infiltration (Gnahore et al. 2022). The open forest has been heavily infiltrated for the collection of most species, which has disturbed the canopy through openings that promote the proliferation of new species and increased species diversity. The results obtained in this study corroborate those of Zin and Mitlöhner (2020) in the primary and secondary moist evergreen forests in the Tanintharyi Nature Reserve (TNR) Buffer Zone, Myanmar.

A comparative analysis of the diametric structures of the two forest types indicates a clear difference between them. The diameter class distribution showed a reverted J-shape in both forest types, withstands dominated by individuals of small diameters. This same reverted J-shape has been obtained by Tesfay et al. (2019) in Gra-Kahsu natural vegetation, southern Tigray of Ethiopia, which reflects a decrease in plant individuals as the diameter class increases. In the open forest, there was a complete absence of individuals with a diameter greater than 80 centimeters. This showed that the reconstituted vegetation was still in the juvenile stage. The dominance of small-diameter individuals replacement of the loss of individuals after the anthropization of flora and vegetation (Tra et al. 2021). The absence of individuals with a large diameter in the open forest could be explained by the cutting of woody species by the surrounding communities, which did not occur in the closed forest (Yohannes et al. 2017). The reverted J-shape signifies a regeneration of species individuals after the anthropization of an environment (Goncalves et al. 2018), however young individuals are more vulnerable to anthropogenic disturbance (Bharathi and Devi-Prasad 2017).

On the other hand, a basal area of 17.33 m² per hectare is within the range proposed by Sokpon and Biaou (2002), which characterizes open forests as a basal area between 12 and 25 m² per hectare. The value of the basal area obtained could be explained by the fact that the open forest was in a state of degradation. Similarly, in his study of the Gra-Kahsu natural vegetation in southern Tigray of Ethiopia, Tesfay et al. (2019) found a basal area of up to 35 m² per hectare. The value obtained during this study in the closed forest was higher than that found by Tesfay. The higher value could be explained by the fact that the closed forest is not sufficiently subject to human activities. Also, the results showed that basal area values between 25 and 50 m² per hectare characterize dense humid forests. These results are similar to this study. Differences in basal area values between the two forest types may be due to differences in floristic composition and human activities. Structural parameters are essential for understanding forest ecology (Naidu and Kumar 2016). We also noted a structural variability between the forest types.

In conclusion, anthropogenic pressure and climate change are the leading causes of ecosystem degradation. To guide conservation, it was, therefore, necessary to have a range of information on the current state of biodiversity. The overall objective of this study was to contribute to the characterization of the vegetal biodiversity of Banco National Park (BNP). The study in the park contributed to a better understanding of BNP's floristic composition and plant structure. This study revealed 345 plant species belonging to 211 genera and 73 families in the open forest, while the closed forest possessed 283 species belonging to 229 genera and 80 families. In both forests, the most important families were Rubiaceae, Fabaceae and Apocynaceae. The Shannon-Weaver diversity indices were high in the closed and open forests. The most important species were *C. subnudum*, *T. africanus* and *P. africanum*. The diametric structure showed the greatest proportion of individuals with a low diameter class, representing regeneration potential. However, great differences were observed in the open and the closed forest vegetation structure. These results demonstrate that this forest has great ecological value despite the anthropogenic activities influencing its integrity. The different characteristics of BNP revealed by this study are knowledge necessary to biodiversity management and rehabilitation in all of its components. All these assets militate in favor of the reinforcement of the conservation and rehabilitation strategies of biodiversity in all its components.

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Vegetation diversity, structure, composition and carbon stock of community managed forests of Mid-hills, Nepal

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Abstract. *Joshi P, Joshi R, Sapkota RP, Panta M, Chand P. 2023. Vegetation diversity, structure, composition and carbon stock of community managed forests of Mid-hills Nepal. Asian J For 7: 29-36.* Depending on management practices, forests can serve as both carbon sinks and sources. The goal of reducing carbon emissions and increasing the carbon sink is thought to be feasible if carbon reservoirs in current forests are protected and conserved. This study was objectively conducted to assess the vegetation diversity, structure, and carbon stock of the Mid-hills of Nepal. The study was undertaken in the Lanta Community Forest, Jajarkot District, Nepal, which has an extent of 38.65 hectares. Data for vegetation analysis and carbon stock assessment were collected using systematic random sampling using quadrats of 10×10 m with a total number of 35 quadrats. Within each quadrat, individual trees and bamboo were identified in the sites, and their height (m) and DBH (cm) were measured. Density, frequency, basal area, and Important Value Index (IVI) were calculated as structural parameters of vegetation. The Above-Ground Tree Biomass (AGTB) and Below-Ground Tree Biomass (BGTB) were calculated using an allometric equation based on tree diameter, height, and wood-specific gravity. The species diversity, species richness, and evenness were found to be 2.2, 2.35, and 0.83, respectively. A total of 14 tree species, with 723 individuals and one bamboo species were recorded. *Rhododendron arboreum* Sm. had the highest tree density with 211 trees/ha, while *Tsuga dumosa* (D.Don) Eichler had the highest IVI. Total wood volume, biomass, and total carbon stock were estimated at 15.37 m^3 ha^{-1}, 31.99 t ha^{-1}, and 15.03 t ha^{-1}, respectively. There was a strong negative correlation (r = -0.59) between *R. arboreum* and *R. campanulatum* D. Don and a strong positive correlation (r = +0.65) between *Malus sikkimensis* and *Machilus* species. For the preservation and sustainable management of community forests, information regarding the structure, composition, and dominance of tree species is provided by the study. The establishment of community forests is thus demonstrated in this article as a means of promoting the protection and preservation of regional biodiversity.

Keywords: Basal area, community forestry, DBH, Gramineae, IVI

INTRODUCTION

Forests comprise over 31% of the earth's land surface and are estimated to contain 289 Gt of carbon in biomass, greater than the total quantity of carbon in the atmosphere (FAO 2010). Depending on management practices, forests can serve as both carbon sinks and sources. The goal of reducing carbon emissions and increasing the carbon sink is thought to be feasible if carbon reservoirs in current forests are protected and conserved (Brown et al. 1999). Besides their role in climate change mitigation by storing carbon, forests are also the main home of biodiversity. Forests have canopy layers and various structures that serve as flora and fauna habitats and provide various ecosystem services (Dronova 2017).

Nepal occupies 0.03% of the world's land area (Government of Nepal 2019), but it is home to almost 3.2% of the world's known flora (MoFSC 2014). There are 6,073 species of angiosperms, 26 gymnosperms, 534 pteridophytes, 1,150 bryophytes, 365 lichens, 1,822 fungi, and 1,001 algae species recorded in Nepal (MoFSC 2014), with 284 blooming plant species are native to the country (MoFSC 2014). Nepal's forest covers 44.74 percent of the country's total land, in which 40.36 percent of it is covered by forest, while Other Wooded Land (OWL) covers the remaining 4.38 percent (DFRS 2018). Based on the Forest Act, 2019, national forests in Nepal are divided into six type namely government-managed, protected, community, leasehold, religious, and collaborative forests (Poudel 2019). The community forestry program in Nepal is a government effort to reduce forest degradation, promote sustainable forestry practices, and improve the community's livelihood (Ojha et al. 2009).

The capacity of a forest as a carbon sink can be referred to from the Above-Ground biomass (AGB) stored by forest vegetation (Ketterings et al. 2001; Nur et al. 2022). Carbon storage estimation helps gather data relevant to Greenhouse Gas (GHG) reduction (Adame et al. 2020). Data and information regarding above-ground biomass are also required to undertake climate change-related initiatives and programs, such as REDD+ (Reducing Emissions from Deforestation and Forest Degradation). The state and dynamic of above-ground biomass can also provide essential information on forest management to inform the necessary interventions (Brown et al. 1999). For instance, carbon (C) sequestration in existing forests can be increased by reducing timber harvesting or lengthening rotations, but it must be compensated by the reduced financial benefits, the risks of catastrophic wildfires, and the costs of C emissions from the manufacture of materials to replace wood products. Another example is the afforestation of marginal farmland, which is already happening in many regions of the eastern US, facilitated through federal incentive programs like the Conservation Reserve Program (CRP) and the Wetlands Reserve Program (WRP) (Brown et al. 1999).

Most of the community-based forest management systems are being managed by local people. However, there are continuous debates about whether forest management practices have improved or worsened after the local communities take control of forest resources. Therefore, this study aimed to investigate vegetation diversity, structure, composition, and carbon stock of community managed forests in Lanta Community Forest, Kushe Rural Municipality, Jajarkot District, Nepal. We expect the results of this study can enrich the understanding of the effectiveness.

MATERIALS AND METHODS

Study area

The study was conducted in Lanta Community Forest, a core area of Red Panda in Kushe Rural Municipality, Jajarkot District, Nepal. Jajarkot District is situated in the Bheri Zone of Karnali Province with a total of 2230 km^2 (Figure 1). Lanta Community Forest has an extent of 38.65 ha. The geographical coordinates of the area are 28° 50′ 24″ N and 82° 10′ 12″ E. It has an altitude ranging from 3000-5000 masl with an average annual rainfall of 183.5 mm and an average temperature of 21℃ and humidity of 56.14%, respectively. The study area has a loam type of soil.

Data collection

Field survey

Field observation was conducted to determine the observation sites that were identified in a Focus Group Discussion (FGD). A systematic random sampling strategy was used to collect vegetation and environmental data. The first quadrat, measuring 10×10m, was carried then the subsequent quadrats were laid at the interval of 100 m. Within each plot, habitat factors, including altitude, slope, aspect, canopy cover, primary vegetation, and distance to road, water supply, and settlements, were documented at each plot. In addition, individual plant species within the plot were identified, and the height (cm), seedling density (individuals/m^2), litter layer cover (%), and dead bamboo density (stems/m^2) were also measured.

Canopy, slope, and elevation

The densiometer was used to determine the canopy cover of the vegetation within the quadrat (Lemmon 1957), which was classified into low (<25%), medium (25-50%), high (50-75%), and extremely high (>75%). In addition, the slope was categorized as extremely low (<10°), low (10°-20°), moderate (20°-30°), high (30°-40°), and extremely high (>50°). Similarly, the elevation range was also categorized from 2000m to 3000m.

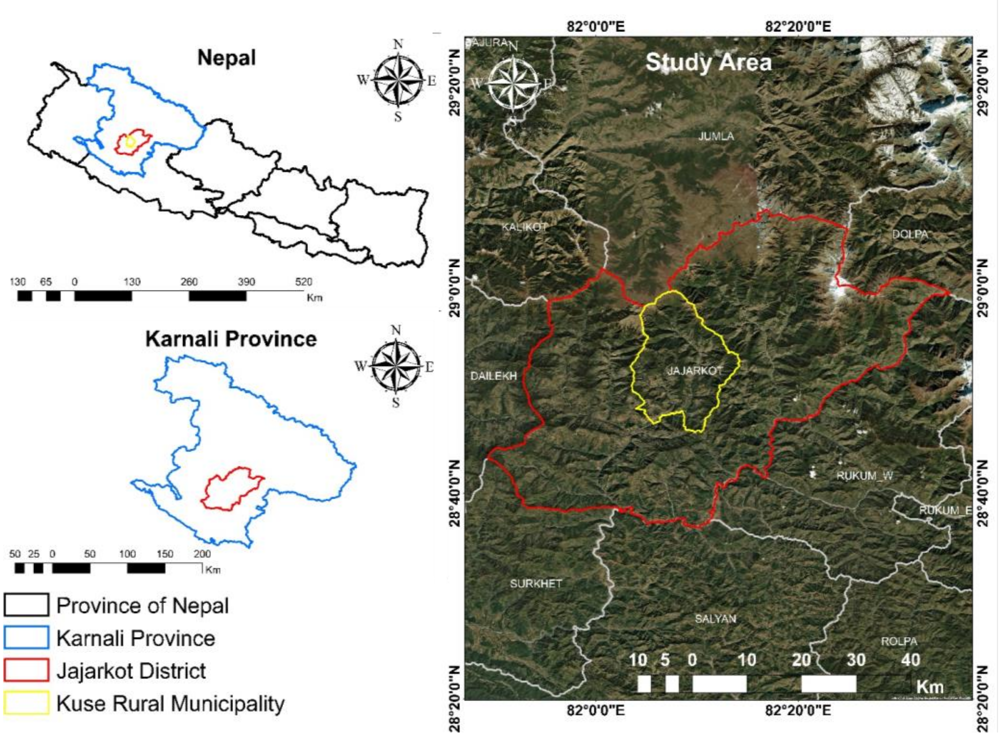

Figure 1. Map of the study area in Lanta Community Forest, Kushe Rural Municipality, Jajarkot District, Nepal

Secondary data

The documents and information about bamboo from DFO, Sector Forest Office, and Community Forest Office were reviewed and documented. Furthermore, pertinent references to various publications, the thesis and other publicly available information were examined.

Biodiversity parameters

Shannon-Wiener Diversity Index (H)

The species diversity was estimated to get a better quantitative picture of the forest community (Joshi et al. 2019). The Shannon-Wiener Diversity Index (H') was used to calculate the species diversity using the formula below:

$H' = -\sum (ni/N)\ln(ni/N) = -\sum Pi \ln pi$ (Shannon and Weaver 1949)

Where; N is the total number of species, and ni is the number of individuals in each species; Pi = ni/N.

Evenness Index (e)

Evenness Index indicates the individual distribution/dispersion of a species whether a species is clumped or evenly distributed. The following formula was used to calculate it:

$e = H'/\ln S$ (Odum 1971)

Where: H' is the Shannon-Wiener Diversity Index and S is the total number of species.

Species richness (D)

The Margalef Index was used to calculate the species richness using the following formula

$D = (S-1)/\ln N$ (Margalef 1958)

Where: S represents the total number of species and N represents the total number of individuals.

Floristic structure and composition

The floristic composition of the vegetation community in the studied area was analyzed using the Important Value Index (IVI). It is the sum of Relative Density (RD), Relative Frequency (RF), and Relative Basal Area (RBA) (Zobel et al. 1987). The following formulas were used to calculate such parameters.

Density and Relative Density (RD)

Density is the number of individual trees per unit area and was calculated as follows:

$$\text{Density (ind/ha)} = \frac{\text{Number of individuals of a species}}{\text{Total number of plots studied} * \text{area of plot}} \times 10000$$

Relative density is the density of a species about the overall density of all species in an area and was calculated as follows:

$$\text{Relative density (\%)} = \frac{\text{Density of species A}}{\text{Total densities of all species}} \times 100$$

Frequency and Relative Frequency (RF)

Frequency is the dispersion of species in a community and was calculated as the proportion of sampling units in which a specific species is found as follows:

$$\text{Frequency (\%)} = \frac{\text{No of plots in which species A occured}}{\text{Total number of plots sampled}} \times 100$$

The relative frequency of a species is the frequency of a species about the total frequency of all species in the community.

$$\text{Relative frequency (\%)} = \frac{\text{Frequency of a species A}}{\text{Total frequency of all species}} \times 100$$

Basal area and relative basal area (RBA)

The basal area is the horizontal area occupied by the stem of a species and indicates the dominance of a species (Hanson and Churchill 1961). It was calculated as follows:

Basal area (sq.cm) = $A = \pi d^2/4$

Similarly,

$$\text{Relative basal area (\%)} = \frac{\text{Basal area of species A}}{\text{Total basal area of all species}} \times 100$$

Important Value Index (IVI)

The Importance Value Index shows each species' dominance and ecological success within a community. It was calculated as follows:

IVI = Relative Density (RD) + Relative Frequency (RF) + Relative Basal Area (RBA)

Biomass estimation

Above-Ground Tree Biomass (AGTB)

Using diameter tape and clinometers, the diameter at breast height (DBH = 1.3 m) and height of individual trees more than or equal to 5 cm DBH were measured in each square plot within a 100 m^2 area. Each tree was numbered and labeled as accurately as possible with its species name. Trees on the edge of the plots were included if more than 50% of their base area fell within the plots, and were omitted if less than 50% of their basal area fell outside the plot. Trees with their trunks inside the sampling plots and branches out were omitted, whereas those with their trunks inside the plots and branches out were included. The diameter tape was wrapped around the stem precisely at the measurement place with great care.

The formula devised by Chave et al. (2005) was used to calculate AGTB:

$AGTB = 0.0509 \times § \times D^2 \times h$

Where, § = Specific gravity of wood (g/cm^3)
D = Tree diameter at breast height (cm)
h = Height of the tree (m)

The tree species wood specific gravity was measured and computed. Then, the biomass stock in kg/m^2 was computed by adding the sum of all the individual biomass weights (kg) of sampling plots and dividing by the area of sampling plots (100 m^2). Finally, the value was multiplied by 10 to get the ton/ha value. After multiplication with a carbon proportion of 0.47, the biomass stock was turned into carbon stock.

Below-Ground Tree Biomass (BGTB)

The root-to-shoot ratio, which has become the standard method for separating root biomass from the more easily measured shoot biomass, is one of the most popular descriptors of the relationship between roots (belowground) and shoot (aboveground) biomass. We used the root-to-shoot ratio of 1:5 to estimate BGTB, meaning that BGTB was equal to 20% of AGTB (MacDicken 1997).

Total carbon stock

By combining the carbon stocks of AGBT and BGTB, the total carbon stock of the forest was estimated.

Statistical analysis

Microsoft Excel 2007 was used to tabulate and analyze the data, which was then presented in graphs and tables.

RESULTS AND DISCUSSION

Biodiversity parameters

For the vegetation analysis of Lanta Community Forest, various biodiversity parameters were examined. In total, there were 14 tree species recorded with a total of 253 individuals from 35 sample plots of community forest in this study. The species were *Tsuga dumosa, Rhododendron arboreum, Rhododendron campanulatum, Abies spectabilis, Quercus semicarpifolia, Myrica esculenta, Daphne papyracea, Machilus* species, *Prunus cornuta, Pyrus pashia, Malus sikkimensis, Helixanthera ligustrina, Rosa sericea* and *Lyonia ovalifolia*. In addition, one species of bamboo identified, namely *Thamnocalamus* sp.

The Shannon-Wiener diversity index in the studied area was 2.20 (Table 1), which is greater than the range (0.9-1.0) reported by (Sharma et al, 2018) in Resunga Sacred Grove, Gulmi, and (1.80-1.91) reported by (Shrestha et al. 2020) in sacred groves in Kathmandu Valley, and Ganesh Community Forest and Ramnagar Community Forest in Nepal tropical (Joshi et al. 2019). However, our result is significantly lower than the range (2.69-4.53) reported by (Sahu et al. 2012) in a tropical holy forest environment in the Niyamgiri Hill Range in India Eastern Ghats. The species evenness index of our study area was 0.83, which is higher than that in Ganesh CF and Ramnagar CF (Joshi et al. 2019), indicating that the forest is more diverse and species are distributed more uniformly.

Floristic structure and composition

The average plant density in Lanta CF was 52 individuals per hectare (Table 2). The *R. arboreum* (211 ind ha^{-1}) was the most densest species, while *A. spectabilis* and *H. ligustrina* had less than 4 individuals per hectare (Table 2). The *T. dumosa*, *Q. semicarpifolia, D. papyracea,* and *R. campanulatum* were found to have 80 individuals per hectare, 71 individuals per hectare, 86 individuals per hectare, and 77 individuals per hectare, respectively. Similarly, the relative density of *R. arboreum* (29.25%) was highest and *A. spectabilis* (0.4%) had less than 1%. Similarly, the bamboo species *Thamnocalamus* was strongly growing and had a density of 3683 individuals per hectare.

Forest structure, functional diversity, ecological processes, and other ecosystem services are all influenced by tree density. The density of *R. arboreum* with 211 ind ha^{-1} is higher than Ganesh CF, which has the most densest tree species *Mallotus philippensis* (85 ind ha^{-1}), but lower than Ramnagar CF, which has the most densest tree species *Shorea robusta* (499 ind ha^{-1}) (Joshi et al. 2019). As a result, when compared to similar types of community forests in Nepal, the observed tree density in the current study might be deemed modest.

The frequency reflects how often the species appears in the forest. The *R. arboreum* (85.7%) had the highest percentage frequency, while *A. spectabilis* had the lowest (2.85%). In Lanta CF, the average basal area of trees was 79 m^2 ha^{-1} with *T. dumosa* was the most dominant species. Although it had a lower density (80 trees per hectare), it has the largest basal area (706.57 m^2 ha^{-1}) among the species due to its great girth size. Similarly, *A. spectabilis* was the recessive species with the smallest basal area (0.02 m^2 ha^{-1}).

Table 1. Biodiversity parameters in Lanta Community Forest, Jajarkot District, Nepal

Parameters	Value
Numbers of species	14
Number of individuals	253
Shannon-Wiener Index (H’)	2.20
Margalef Richness Index (D)	2.35
Evenness Index (e)	0.83

Table 2. The density, frequency and basal area of species in Lanta Community Forest, Jajarkot, Nepal

Species	Density (trees/ha)	Freq. (%)	BA (m^2 ha^{-1})
Tsuga dumosa	80	51.42	706.57
Rhododendron arboreum	211.42	85.71	99.51
Quercus semicarpifolia	71.42	22.85	218.72
Prunus cornuta	34.28	11.42	9.32
Machilus spp.	22.85	11.42	2.72
Pyrus pashia	25.71	22.85	7.85
Malus sikkimensis	11.42	5.71	1.89
Myrica esculenta	51.42	22.85	22.72
Rhododendron campanulatum	77.14	28.57	15.09
Helixanthera ligustrina	2.85	5.71	9.24
Daphne papyracea	85.71	40	1.29
Abies spectabilis	2.85	2.85	0.02
Rosa sericea	34.28	20	1.52
Lyonia ovalifolia	11.42	8.57	2.88

Note: BA: Basal area (m^2 ha^{-1})

The average basal area was found to be 78.52 m^2 ha^{-1} in present study, which was higher than the basal area of 37.28 m^2/ha reported by (Bhuju 2000) for Churia Forest in eastern Nepal and 34.20 m^2 ha^{-1} reported by (Marasini 2003) for disturbed Churia forest in Rupandehi District. In comparison to the forest of Churia, the Lanta Community Forest appears to be less disturbed. This could be related to the fact that the Lanta CF is located distant from human settlements and is not suitable for leisure visits.

Similarly, *R. arboreum* has the highest relative density (29.25%) and relative frequency (25.21%), whereas *A. spectabilis* has the lowest relative density (0.4%) and relative frequency (0.84%) (Table 3). The *T. dumosa*, the dominating species, had the highest relative dominance (64.26%), while *A. spectabilis* had the lowest (0.002 %) (Table 3). The IVI was found highest for *T. dumosa* (90.46) followed by *R. arboreum* (63.51), *Q. semicarpifolia* (36.5), *D. papyracea* (23.74), *Rhododendron campunulatum* (20.45), *M. esculenta* (15.9), *P. pashia* (10.99), *R. sericea* (10.76), *P. cornuta* (8.95), *Machilus* sp. (6.77), *L. ovalifolia*(4.36), *M. sikkimensis* (3.43), *H. ligustrina* (2.92) and the lowest IVI was obtained for *A. spectabilis* (1.24) as shown in Table 3.

Important Value Index (IVI) values express any species' dominance and ecological succession. The *T. dumosa* had the highest IVI (90.46), indicating that it takes up most space and resources. Conversely, with the lowest IVI index value (1.24), *A. spectabilis* is the most recessive species, which only consumes the least amount of area and resources.

Biomass estimation

The Lanta CF is located in temperate region of Nepal, where gymnosperms with enormous trunks present. As the result, the volume of wood would be enormous. The average wood volume in the Lanta CF is 1.090 m^3 ha^{-1}, with *T. dumosa* (10.5 m^3 ha^{-1}) having the largest volume, followed by *Q. semicarpifolia* (3.57 m^3 ha^{-1}), *R. arboreum* (0.84 m^3 ha^{-1}) and least *A. spectabilis* (0.0001 m^3 ha^{-1}) as shown in Table 4.

As shown in Table 4, the average biomass in Lanta Community Forest was found to be 2.285 t ha^{-1} , with *T. dumosa* (15.28 t ha^{-1}) having the highest biomass followed by *Q. semicarpifolia* (7.89 t ha^{-1}), *R. arboreum* (5.08 t ha^{-1}) and the least was *A. spectabilis* (0.003 t ha^{-1}). Forest biomass estimation is influenced by forest structure, such as tree density, diameter, basal area, tree height, and age. Furthermore, because biomass is closely tied to species selection and scheduled operations, such as planting, harvesting, and collecting other forest products, biomass in the forest reveals possibilities for improvements in forest management (Lal, 2005).

The present studies' mean biomass (2.285 t ha^{-1}) was lower than that of Dahal (2007) in the mid-hills of Central Nepal (233.0 t ha^{-1}) and *Eucalyptus* forest (96 t ha^{-1}) in Northeast Australia (Zerihun et al. 2006), community managed Hill Sal forest (120 Mg ha^{-1}) in central Nepal reported by Shrestha et al. (2020).

The amount of carbon sequestered in the forest varies depending on the forest type and tree density. The average carbon stock of Lanta CF was found to be 1.07 t ha^{-1}. *T. dumosa* (7.18 t ha^{-1}) was found to have the highest biomass, followed by *Q. semicarpifolia* (3.71 t ha^{-1}), *R. arboreum* (2.39 t ha^{-1}), least *A. spectabilis* (0.0014 t ha^{-1}). However, the results are not as comparable to those published by (Pandey and Bhusal 2016) from Sal forest in two different ecological areas (Hill and Terai) of Nepal, and by (Karki et al. 2016) from ICIMOD Knowledge Park, Godavari, which ranged from 234.54 to 479.29 t ha^{-1} in 2012. Forests, which store 20-100 times more carbon per unit area than croplands, play a critical role in lowering CO_2 levels in the atmosphere (Brown and Pearce 1994).

Table 3. The relative density, relative frequency, relative dominance and IVI of species in Lanta Community Forest, Jajarkot, Nepal

Species	RD (%)	RF (%)	RDo (%)	IVI
Tsuga dumosa	11.07	15.13	64.26	90.46
Rhododendron arboreum	29.25	25.21	9.06	63.51
Quercus semicarpifolia	9.88	6.72	19.9	36.5
Prunus cornuta	4.74	3.36	0.85	8.95
Machilus sp.	3.16	3.36	0.25	6.77
Pyrus pashia	3.56	6.72	0.71	10.99
Malus sikkimensis	1.58	1.68	0.17	3.43
Myrica esculenta	7.11	6.72	2.06	15.9
Rhododendron campanulatum	10.67	8.4	1.37	20.45
Helixanthera ligustrina	0.4	1.68	0.84	2.92
Daphne papyracea	11.86	11.76	0.11	23.74
Abies spectabilis	0.4	0.84	0.002	1.24
Rosa sericea	4.74	5.88	0.14	10.76
Lyonia ovalifolia	1.58	2.52	0.26	4.36

Note: RD: Relative density (%), RF: Relative frequency (%), RDo: Relative dominance (%), IVI: Importance value index

Table 4. Wood volume, living biomass and carbon stock of species in Lanta Community Forest, Jajarkot, Nepal

Species	Wood volume (m^3 ha^{-1})	Biomass (t ha^{-1})	Carbon stock (t ha^{-1})
Tsuga dumosa	10.50	15.28	7.18
Rhododendron arboreum	0.85	5.08	2.39
Quercus semicarpifolia	3.58	7.9	3.71
Prunus cornuta	0.06	0.55	0.26
Machilus sp.	0.02	0.15	0.07
Pyrus pashia	0.02	0.19	0.09
Malus sikkimensis	0.01	0.15	0.07
Myrica esculenta	0.18	1.07	0.50
Rhododendron campanulatum	0.09	1.13	0.53
Helixanthera ligustrina	0.05	0.17	0.08
Daphne papyracea	0.003	0.12	0.056
Abies spectabilis	0.0001	0.003	0.001
Rosa sericea	0.006	0.11	0.05
Lyonia ovalifolia	0.014	0.11	0.05

Table 5. Correlation matrix on the abundance between species in Lanta Community Forest, Jajarkot, Nepal

Species	TD	RA	QS	PC	MS	PP	MS	ME	RC	HL	DP	AS	RS	LO	TS
TD	1														
RA	-0.18	1													
QS	-0.32	0.00	1												
PC	-0.23	0.03	-0.14	1.00											
MS	0.34	0.04	-0.16	-0.06	1										
PP	-0.01	-0.05	-0.25	0.25	-0.17	1.00									
MS	-0.01	-0.02	-0.11	0.03	0.65	0.00	1.00								
ME	-0.28	-0.06	0.04	-0.14	-0.15	-0.18	-0.10	1							
RC	0.28	-0.59	-0.26	-0.06	-0.06	0.00	-0.12	-0.24	1.00						
HL	0.04	-0.02	-0.08	-0.05	-0.06	-0.09	-0.04	-0.08	0.15	1					
DP	0.02	-0.14	-0.16	-0.03	-0.10	-0.36	-0.15	0.23	0.19	0.16	1.00				
AS	0.22	-0.30	-0.08	-0.05	-0.06	-0.09	-0.04	-0.08	0.39	-0.03	0.02	1			
RS	-0.25	-0.01	-0.12	-0.11	-0.12	-0.19	-0.08	0.49	-0.16	0.12	0.37	-0.06	1.00		
LO	-0.02	0.27	-0.09	-0.09	-0.10	-0.15	-0.06	0.00	-0.05	-0.05	0.38	-0.05	-0.03	1.00	
TS	0.03	-0.12	-0.09	0.20	0.17	0.43	0.25	-0.19	-0.02	-0.03	-0.52	0.00	-0.10	-0.39	1

Note: TD: *Tsuga dumosa, RA*: *Rhododendron arboretum,* QS: *Quercus semicarpifolia,* PC: *Prunus cornuta,* MS: *Machilus* sp., PP: *Pyrus pashia,* MS: *Malus sikkimensis,* ME: *Myrica esculenta,* RC: *Rhododendron campanulatum,* HL: *Helixanthera ligustrina,* DP: *Daphne papyracea,* AS: *Abies spectabilis,* RS: *Rosa sericea,* LO: *Lyonia ovalifolia,* and TS: *Thamnocalamus* spp.

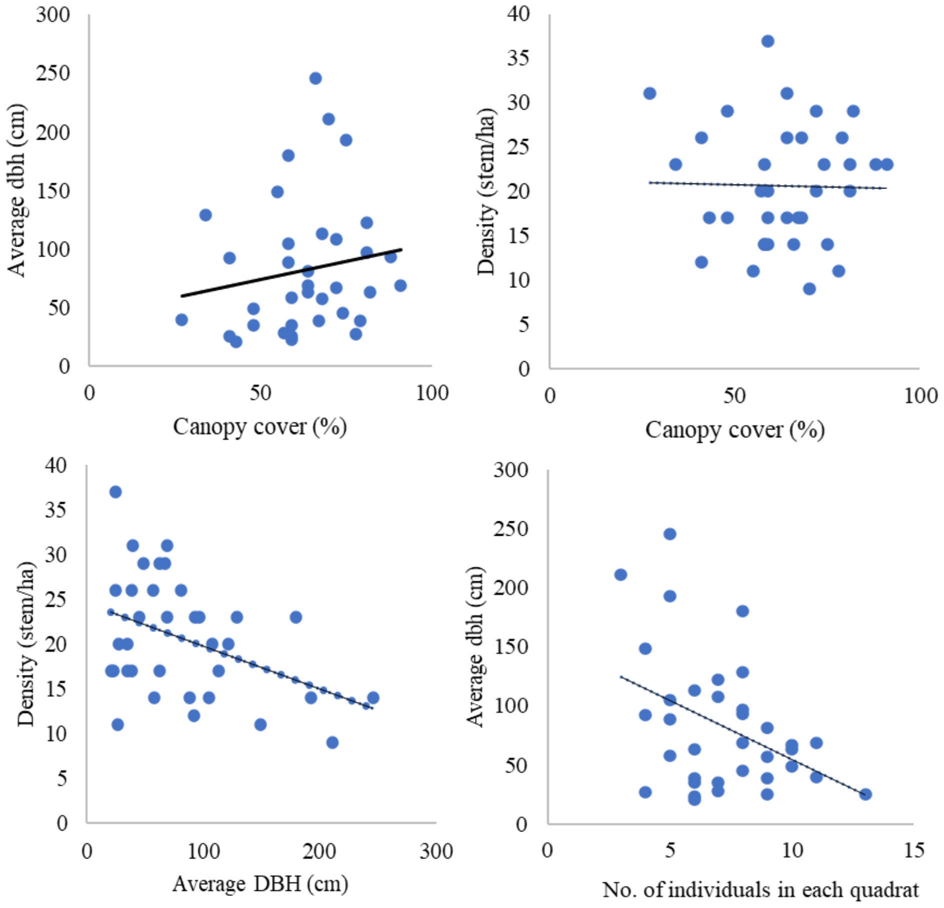

Figure 2. Correlation between structural parameters: A. Average dbh (cm) and canopy cover (%); B. density (stem/ha) and canopy cover (%); C. density (stem/ha) and average dbh (cm); and D. between average dbh (cm) and no. of individuals in a quadrat

Correlation between species

Most of the species in Lanta CF had a slightly negative relationship, while some species had a slightly positive one. However, there was a strong positive correlation between *Machilus* spp. and *M. sikkimensis* with a Pearson correlation coefficient of +0.65. Similarly, there was a strong negative correlation between *R. arboreum* and *R. campanulatum*, with a Pearson's correlation coefficient of -0.59.

The abundance of eight species in Lanta CF had a slight negative correlation with *Thamnocalamus* species, whereas the abundance of five species had a slightly positive correlation. The *D. papyracea* had a moderate negative correlation with a coefficient of -0.52 among the recorded species, whereas *P. pashia* had a moderate positive correlation with a Pearson correlation coefficient of +0.43. However, *A. spectabilis* was one of the few species with neither a positive nor a negative relationship with the *Thamnocalamus* species (r = 0).

In Lanta CF, the canopy cover was found to be positively correlated (r = 0.16) with average dbh (cm), suggesting that the present study is by the universally accepted concept, as shown in Figure 2A. The positive correlation between the average dbh of both canopy covers is because as the tree gets older, its girth size increases and also increases the branches and leaves. Conversely, there was a negative correlation between density (stem/ha) and canopy cover (r =-0.02), as shown in Figure 2B. The negative correlation between canopy cover and density might be due to the presence of large trees with widely dispersed branches which increased the canopy cover without increasing the tree number. Likewise, the average dbh (cm) was negatively correlated with density (stem/ha) (r = -0.4) and several individuals in each quadrat (r = -0.4), as shown in Figure 2C and 2D, respectively.

In conclusion, the vegetative characteristics and the biomass and carbon stock of the Lanta community forest were investigated. The observation plots in the Lanta CF recorded 14 tree species with 253 individuals and one bamboo species with 1289 individuals. The CF had a Shannon-Wiener diversity index of 2.20, a Richness Index of 2.35, and a species evenness index of 0.83. Based on the Important Value Index (IVI), *T. dumosa* was the dominant tree species in Lanta Community Forest. The total tree density was 723 trees per hectare, with the *R. arboreum* (211 individuals per hectare) being the densest. The forest's total wood volume, biomass, and carbon stock were also estimated to be 15.375 m^3 ha^{-1}, 31.99 t ha^{-1}, and 15.03 tons/ha, respectively. The amount of carbon stock differed among tree species due to tree structural factors such as density, basal area, height, and wood-specific gravity. The *R. arboreum* and *R. campanulatum* were found to have a strong negative correlation among all species, while *M. sikkimensis* and *Machilus* sp. had a high positive correlation. Similarly, the bamboo species had a negative association with *D. papyracea* and a positive association with *P. pashia*. Based on the results of the study, we recommend the following suggestions: (i) to increase the tree species richness in the forest, equal emphasis should have given to each tree species; (ii) to maintain the tree species diversity emphasis should have given to the proportionate distribution of all tree species throughout the forest; (iii) to normalize the species composition in the forest equal emphasis should have given to the less valuable species; (iv) to maintain the mean stem volume/ha, the balanced distribution of pole and trees over the entire forest area should be retained; and (v) to control the effect of limiting factor in the forest, regular and proper care should be employed during the management of the forest.

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Ecotourism development as a community-based conservation effort in Ayah Mangrove Forest, Kebumen, Central Java, Indonesia

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Abstract. *Afifah RN, Putri A, Hartanti AN, Negari SIT, Pratama MSR, Zuaini PAK, Al Madani AR, Muryanto BS, Muhammad F, Astikasari L, Indriyani S, Kurniawati I, Sunarto, Kusumaningrum L, Budiharta S, Flores AB, Setyawan AD. 2023. Ecotourism development as a community-based conservation effort in Ayah Mangrove Forest, Kebumen, Central Java, Indonesia. Asian J For 7: 37-44.* The mangrove ecosystem is a transition ecosystem between terrestrial and marine zones affected by tidal waves, water inundation, and high saline water and soil. Mangrove forest plays an important role in physical, ecological and socio-economic aspects. One potential function of mangrove forests from the socio-economic aspect is ecotourism. Ayah Mangrove Forest in Ayah Village, Kebumen District, Central Java Province, Indonesia, has the potential to be developed as a mangrove-based ecotourism area that combines the objective of income generation and mangrove conservation. This research was conducted to examine the conservation and ecotourism efforts by the people of Ayah Village and to assess the local community's perception of the development of mangrove-based ecotourism. The research was conducted in November 2022. Primary data were collected using field observation and interviews with 100 respondents aged 17-60 years using the snowball sampling method, while secondary data were collected using a literature review. The results showed that efforts to protect the Ayah Mangrove Forest area included establishing zones for protection, rehabilitation, and utilization of the mangrove forest. There are also mangrove rehabilitation efforts that involve the community in the form of the establishment of nurseries, mangrove planting, and maintenance of planted seedlings. The local community also carry out garbage cleaning program and public outreach. Besides the conservation aspect, Ayah Mangrove Forest is also utilized for ecotourism objects that generate income for the local community by selling tickets, boats, foods, beverages and souvenirs, tour guides, and parking attendants. Most of the local community has good perceptions and agreed on the development of Ayah Mangrove Forest as an ecotourism area. They are willing to be involved in ecotourism planning and development activities. The community agreed to participate in tour guide activities, selling souvenirs, providing homestays, and participating in ecotourism counseling. Most of the local community has also agreed to form a mangrove conservation community to maintain the sustainability of mangrove forest ecotourism. Nonetheless, support from the government is required, especially in the promotion and rehabilitation programs in the ecotourism development of Ayah Mangrove Forest.

Keywords: Ayah Mangrove, conservation, ecotourism, forest, Kebumen

INTRODUCTION

The mangrove ecosystem is located in the transition zone between terrestrial and marine zones, which is affected by intertidal waves, brackish water, riverine system, and high saline water and soils (Martuti et al. 2018). These unique environmental conditions give the mangrove ecosystem a high level of biodiversity in the form of flora and fauna. Mangrove forest refers to vegetation in the mangrove ecosystem that is tolerant to regular inundation of seawater and sea tides. Plant species that can grow in mangrove forests are generally very limited because the water is brackish, a mixture of seawater and freshwater, making it difficult for other plants to adapt to this area. Mangrove vegetation can be found on estuaries, coral beaches, inland coral reefs with thin sand, or alluvial soil types. Mangrove forest is recognized as the most productive ecosystem and has various functions for the environment and the surrounding community (Sari et al. 2018).

Mangrove forest plays many physical, ecological and socio-economic functions. In the physical aspect, mangrove forests act as barriers to soil erosion, beach abrasion, strong waves and winds, storms, and even tsunamis (Saenger 2002). Ecologically, mangrove forest is the habitat of various fauna as a place to find food (feeding grounds), breeding and spawning grounds of aquatic biotas, such as fishes, crustaceans, and mollusks (Putriningtias et al. 2019; Irwansyah et al. 2021; Wiraatmaja et al. 2022). Also, it is the important habitat of many species of mammals, birds, amphibians, and reptiles. Meanwhile, mangrove forest delivers numerous socio-economic benefits for the surrounding communities, including the sources of timber for building materials, firewood and charcoal, paper, dyes, and so on. Furthermore, with the increasing issue of climate change, more recent studies have revealed that mangrove forest is among the largest carbon sink in the world compared to other ecosystem types, including tropical lowland forest (Dinilhuda et al. 2018; Ely et al. 2021). Nowadays, mangrove forest is also gaining popularity to be developed as an ecotourism object promoted as a win-win solution of mangrove management that integrates conservation and economic objectives.

Ayah Mangrove Forest is located in Ayah Village, Kebumen District, Central Java Province, Indonesia. This mangrove forest is close to the popular Logending Beach, giving Ayah Mangrove Forest the potential to be developed as an ecotourism attraction. A previous study documented various species in Ayah Mangrove Forest, including *Rhizophora mucronata*, *Sonneratia caseolaris*, *Avicenna marina*, *Rhizophora apiculata*, *Acanthus ebracteatus*, *Acrostichum aureum*, *Bruguilera gymnorrhiza*, and *Nypa fruticans* (Murniasih et al. 2022). Among such species, *R. mucronata* dominated the vegetation community due to its rapid regeneration, giving the species a large forest cover which is essential in protecting the coastal area in Ayah Beach. Furthermore, the large coverage of mangrove forests in Ayah Beach also creates a natural beauty with stunning views and a pleasant environment which is a good aspect of developing ecotourism.

Currently, Ayah Mangrove Forest is managed as a mangrove conservation area for disaster mitigation and as a tourism object. Nonetheless, the management carried out by the community focuses on mangrove conservation which is overshadowed by the large contribution of the local community group (i.e., KTH Pansela) and financial support through CSR schemes from various companies. The community group created a management body in the form of BumDes (village business unit) to optimize the ecological and economic benefits of Ayah Mangrove Forest. Ecologically, the community gets indirect benefits such as carbon sinks, oxygen suppliers, and disaster mitigation from abrasion to waves and tsunamis. Economically, the community gets a direct impact from ecotourism through the creation of economic opportunities to support tourism activities such as selling food for souvenirs and providing lodging to tour guides. The economic activities generated from the tourism sector in Ayah Mangrove Forest play an important role in driving the economy in Ayah Village. Therefore, this research was conducted to examine the conservation and ecotourism efforts by the people of Ayah Village and to assess the local community's perception of the development of mangrove-based ecotourism.

MATERIALS AND METHODS

Study period and area

The research was conducted in November 2022 in the Ayah Mangrove Forest area, which is located in Ayah Village, Ayah Sub-district, Kebumen District, Central Java, Indonesia (Figure 1). The Ayah Mangrove Forest has an extent of 18.5 hectares. Ayah Mangrove Forest is divided into five zones: protection, utilization, plant collection, rehabilitation, and special use.

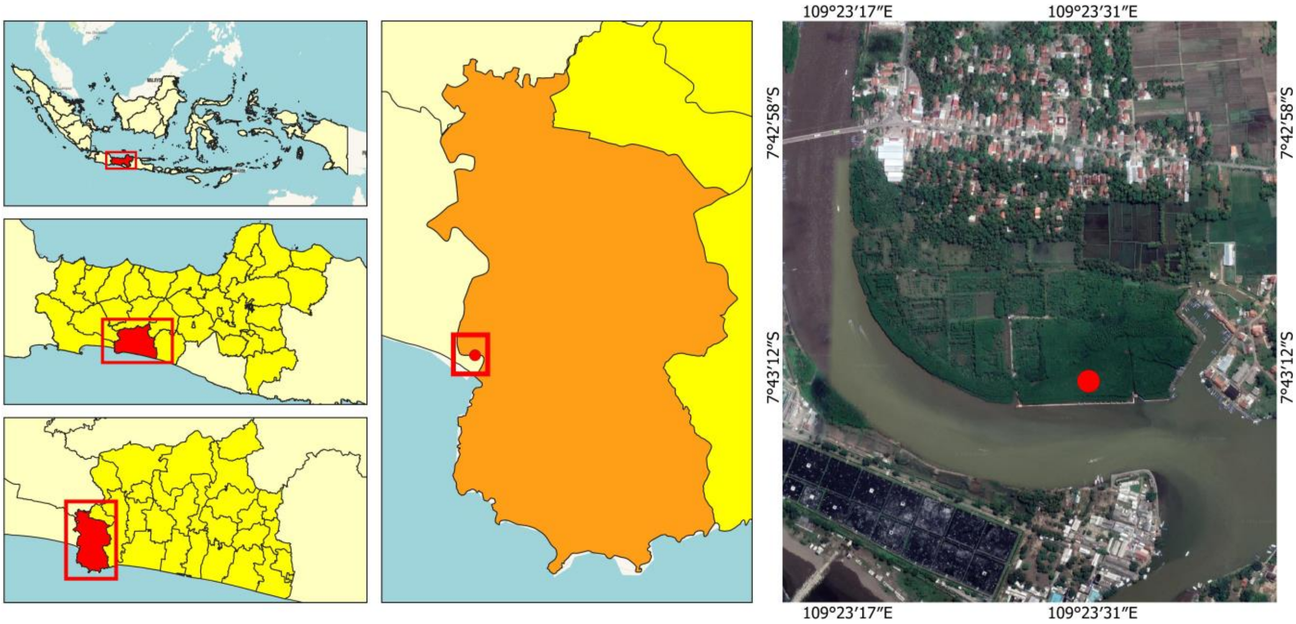

Figure 1. Map of study location in Ayah Mangrove Forest, Ayah Village, Ayah Sub-district, Kebumen District, Central Java, Indonesia

Data collection procedure

This study used primary and secondary data. The primary data were collected using field surveys, interviews, questionnaires, and documentation. Direct observations were carried out during the field survey to determine the actual conditions in the surrounding environment around the Ayah Mangrove Forest area. The interviews aimed at obtaining opinions and perceptions through dialogue and asking questions about the conditions of the research location (Katz-Buonincontro and Anderson 2018). The interviews were conducted using the snowball sampling technique. The target of the interview was the Ayah Mangrove Forest manager as the key informant. The questionnaires were distributed to 100 respondents consisting of residents and tourists who visited Ayah Mangrove Forest aged 17 years and above. Secondary data were collected from literature studies to complement the primary data to be analyzed.

Data analysis

The data were analyzed using an inductive qualitative approach with a descriptive method because it requires an explanation through investigation and direct responses to be able to describe the actual situation and what happened on the ground. The inductive approach was conducted by analyzing the data obtained, then specific relationships were developed to generate hypotheses or theories (Fitrah and Luthfiyah 2018). The community's perceptions included participation, ecotourism community, government intervention, the economic impact of ecotourism, visitors' influence on the environment, and the desire for sustainable ecotourism development. The results of public perceptions were analyzed using the scoring method. The obtained data were analyzed descriptively to relate it with the aspect of conservation, utilization, and community perceptions of the Ayah Mangrove Forest.

RESULTS AND DISCUSSION

Conservation and preservation of Ayah Mangrove Forest

Mangrove conservation includes all efforts to protect various natural resources contained in the mangrove ecosystem, including flora, fauna, water, and soil. The conservation of mangroves would not only benefit the natural resources contained but also could support coastal protection in, which it could prevent 70% risk of beach abrasion (Maulidia et al. 2022). Local-level policies are developed to protect mangrove ecosystems, including Ayah Mangrove Forest. For example, at the district level, the Forestry and Plantation Office of Kebumen District has enacted regulations to prevent disturbances to a mangrove forest and develop a particular zoning system. Some regulations prohibit visitors from bringing flammable items or those that can damage and cause disturbances to the mangrove ecosystem.

In 2008, the district office also legalized the community group, namely *Kelompok Tani Hutan Pantai Selatan* (KTH Pansela; South Coast Forest Farmers Group), as the management body of the mangrove conservation area as the fundamental basis so that conservation would not be interrupted. Further, the district government also provided capacity building through various training and extension programs to improve the management of Ayah Mangrove Forest.

According to Rahmayani (2015), protecting mangrove ecosystems has benefits, including: (i) protecting ecological processes and preserving biodiversity, (ii) supporting research and development activities, (iii) mechanisms for financial and ecotourism management. Some efforts to protect Ayah Mangrove Forest have been implemented by forming zones to facilitate monitoring. For example, based on Table 1, Ayah Mangrove Forest is divided into several zones, one of which is a protection zone that may not be used for any extractive and production activities. Based on interviews with managers and observations at the Ayah Mangrove Forest, the total protection zone is 5.90 Ha or 32%.

Management of mangrove areas is essential to maintain the stability of coastal ecosystems that are useful and sustainable. Community involvement in the development and management of mangrove forest ecosystems could be a strategic and appropriate step by developing community-based management of mangrove ecosystems while at the same time raising public awareness of the importance of protecting natural resources for human lives today and future generations (Amdani 2022). Poor mangrove management will have an impact on the existence of mangroves themselves. For example, untreated garbage in the mangrove area can disturb the pneumatophores of mangrove roots, affecting the breathing process, which can lead to mangrove death (Ruhama et al. 2018).

The preservation of mangrove forest resources is critical for the long-term viability of coastal and marine ecosystems. The existence of mangrove forests also plays an important role in preventing disasters from tidal waves. Unfortunately, as time goes by, more and more people are irresponsible and deliberately damage the existence of mangrove forests, affecting the existence of mangrove forests.

Table 1. Zonation in Ayah Mangrove Forest, Kebumen, Central Java, Indonesia

Zone	Extent (Ha)	Percentage
Protection	5.90	32%
Utilization	6.40	35%
Plant collection	4.40	24%
Rehabilitation	1.30	7%
Special use	0.50	3%
Total	18.50	100%

Therefore, preserving the resources in the mangrove forest is necessary to avoid a much bigger impact. There are several efforts to preserve mangrove forests that can be carried out, including: (i) establishing a nursery to produce mangrove seedlings; (ii) planting mature seedlings in degraded areas; (iii) maintaining the planted seedling to grow and increase the forest cover; (iv) carrying out environmental improvements around the mangrove forest; (v) carrying out spatial management by segmenting the protection area (including rehabilitation zone) and utilization area (including ecotourism); (vi) providing public awareness and education about the importance of mangrove forests for human life and how to preserve them.

Mangrove forest conservation implemented in the Ayah Mangrove Forest area is carried out through reforestation, enrichment planting, and environmental rehabilitation around the coast, which KTH Pansela mainly conducts. In addition, this organization encourages the community's participation in mangrove conservation through public outreach and involvement. Several programs developed by this organization in mangrove conservation include conducting seedlings production, planting mangroves, maintaining mangroves vegetation, tourism, outreach, and outdoor studies. Some of the mangrove seedlings planted by the KTH Pansela were obtained from their nurseries, and some were bought from outside. Mangrove seedlings are also obtained from the Forestry Service of Central Java Province. The mangrove planting is divided into two stages, the intensive stage is carried out in the first six months after planting, followed by routine maintenance for two years, and regular maintenance and monitoring after two years.

According to research by Yuliani et al. (2018), besides the mangrove reforestation, the KTH Pansela has also made several efforts related to mangrove conservation, including: (i) Protection of life support systems. This effort is based on the regulations enacted by the Kebumen District Forest Management Office in the mangrove ecosystem area of Ayah Village. (ii) Conservation of flora and fauna. This effort is carried out by facilitating the mangrove area with trash cans so visitors do not litter in the mangrove area and imposing a ban on the construction of permanent buildings in the mangrove forest area. (iii) Sustainable utilization of the mangrove ecosystem. This is done by establishing fishing businesses such as shell crabs and shrimp in mangrove habitats and developing mangrove areas as educational tourism destinations.

Utilization of Ayah Mangrove Forest

Mangrove forest provides various ecosystem goods and services. Mangrove plants have strong roots, so they can act as a barrier from waves, winds, abrasion and tsunamis, retain mud and trap sediment transported by water, and neutralize water pollution from heavy metals to certain limits (Zakiyah et al. 2020). Mangrove forests also maintain the balance of ecosystems and biodiversity in coastal areas. Mangrove ecosystems serve as spawning and nursery ground, feeding grounds for marine organisms, such as fish, shrimp, crabs, clams, snails, and other animals (Rahmila and Halim 2018). In addition, mangrove forests are habitats for several wild animals, such as monkeys, snakes, otters, monitor lizards, and birds (Ali et al. 2020). Economically, the goods generated from mangrove forests can be used by the community as food ingredients and fuel wood to improve the surrounding community's economic conditions (Afonso et al. 2022). In addition, fishermen and farmers on the coast depend on natural resources from mangrove forests as a source of livelihood and sustenance for the community. Communities obtain marine products such as fish, shrimp, crabs, clams, snails, and other types of animals (Ismail et al. 2018). Mangrove forests can also be developed as an ecotourism attraction. Tourism activities in mangrove forest areas could impact the surrounding community's economy by providing jobs and business opportunities (Razak et al. 2018).

In Ayah Mangrove Forest, communities obtain economic benefits from fishing and farming activities conducted around the mangrove area. The local people create ponds for fish farming around the mangrove forest since it can maintain water salinity and regulate tides (Ahmed et al. 2017). From fishing activities, the community obtains various marine products such as mullet, tilapia, milkfish, white snapper, clam shells, full-blooded clams, crabs, and snails that they get in the Ayah Mangrove Forest area and are sold to fish auctions located not far from this area (Figure 2).

Figure 2. Fishermen collecting marine products around the Ayah Mangrove Forest area, Kebumen, Central Java, Indonesia

Besides the marine products, the economic benefits are also derived from wood products of mangroves forest, such as firewood for self-use or sale. Some residents take sap from the nipa plant (*Nypa fruticans*), a raw material for brown sugar, alcohol, vinegar, sweets, and syrup (Figure 3). To collect the sap in the nipa plant, the community traditionally uses a container made of bamboo or a plastic bottle placed under the nipa bunch. The community sells nipa sugar for IDR 16,000 per kilogram. Some people use mangrove leaves and stem as natural dyes for fabrics. Mangrove plants contain many tannins and natural brown dyes (Pringgenies et al. 2018). The local community has also utilized mangrove plants for traditional medicinal purposes, but it is only used by the community and is not produced commercially. For example, the local community has widely used extracts and raw materials from mangroves to cure ulcers and smallpox.

Ayah Mangrove Forest is also developed as an ecotourism area since it has the advantage of being close distance to the popular Logending Beach. The utilization of mangrove forests as ecotourism is in line with the shifting in the tourist industry from old tourism to new tourism, which emphasizes natural ecotourism destinations with elements of education and conservation (Murtini et al. 2018). Many facilities and utilities are provided in Ayah Mangrove Forest, such as boats that can be rented to explore the area (Figure 4). Also, there are bamboo bridge facilities to facilitate tourists' mobility in enjoying the mangrove forest. The ecotourism development in Ayah Mangrove Forest aims to improve the local people's economy by creating jobs and business opportunities around the area. They can set up shops, stalls, restaurants, transportation, and other activities around the Ayah Mangrove Forest ecotourism area.

Community perceptions of ecotourism development

Developing an area for ecotourism can be conducted through many strategies, such as involving the surrounding community in tourism activities (Hidayat 2016). The community perceptions of ecotourism development in this study were obtained from interviews with 100 respondents, with the results are presented in Table 2. The interviews consisted of several aspects, including participation in ecotourism development, responses to ecotourism groups, government intervention, the economic impacts of ecotourism, the impact of visitors on the environment, and the desire to develop ecotourism sustainably. The respondents had age of more than 16 years old, with the majority being over 35 years old or in the adult category, suggesting that the decision-making is more mature (Putra et al. 2018). The majority of the respondents were sellers, fishermen, and homemakers. Other jobs included pond labor, parking staff, self-employed, private employees, etc.

Figure 3. Utilization of nipa plant (*Nypa fruticans*) for its sap to be processed for various food products in the Ayah Mangrove Forest, Kebumen, Central Java, Indonesia

Figure 4. Utilities and facilities in Ayah Mangrove Forest in the form of boats and bridges

When developing an ecotourism area, it is important to provide opportunities for the local community to participate in all aspects, including planning, development, and management (Tiga et al. 2019). Community perceptions can indicate community participation in developing ecotourism. There were five questions to represent community participation in the development of ecotourism, namely planning activities, tour guide activities, selling souvenirs, providing homestays, and counseling about ecotourism. The interview result showed that the community's perception of participating in developing ecotourism in Ayah Mangrove Forest was very high, with around 97% of the respondents agreeing to participate. The community hoped there would be education and coaching activities to drive the community's economic activities. These include training and involvement in ecotourism concepts, creating small businesses, tour guides, and accommodation management. Community empowerment can also be conducted by providing credit so they can start businesses such as souvenir shops, food stalls, boat rentals, etc. However, several respondents did not agree to provide homestays around the mangrove forest because there is no proper infrastructure. Moreover, according to them, the ecotourism in Ayah Mangrove Forest does not need homestays because visitors rarely spend a long time there.

The local community manages Ayah Mangrove Forest ecotourism under the group of KTH Pansela. The management of Ayah Mangrove Forest is considered well-organized. Some efforts made by the management body to protect the mangrove ecosystem include monitoring activities, planting mangroves, installing signage boards, and establishing protection zone. The Ayah Mangrove Forest is divided into a zoning system to maintain the mangrove forest area. The community is very supportive of those activities carried out by the manager, however only about 57% of the respondents were willing to participate in ecotourism management because they are already tied up with daily work.

The respondents perceived that Ayah mangrove ecotourism needs intervention from the government to support the development of the area, for example, by promoting Ayah Mangrove Forest so that many visitors come to enjoy the beauty of the ecotourism area. In addition, government intervention is required in the form of the provision of mangrove seedlings and conducting socialization.

However, most respondents agreed that the government's involvement is only partial since they perceived the main actor of ecotourism in Ayah Mangrove Forest as the local community. The government, along with the private sector, NGOs, and the community, have an important role in developing ecotourism, including mangrove-based ecotourism in Ayah Village. Mangrove ecosystems will be well-maintained if all stakeholders are involved (Handayani and Suruwaky 2020).

Ecotourism can support the local community's economy so they can live prosperously. Ecotourism is hoped to generate economic benefits for the surrounding community in the form of a source of income from ecotourism activities, such as selling food and drinks, being a parking attendant, and renting boats. About 56% of the respondents agreed that Ayah mangrove ecotourism could improve the economy in the area. However, about 71% of the respondents agreed that the economic benefits are only for some people.

Table 2. The community's perception on ecotourism development in the Ayah Mangrove Forest, Kebumen, Central Java, Indonesia

Aspect	Agree	Less agree	Not agree
Participation in ecotourism development			
Planning activities	97	3	0
Tour guide activities	92	3	5
Selling souvenirs	91	3	6
Providing homestays	84	10	6
Counseling about ecotourism	96	3	1
Community response to ecotourism groups			
Group formation	97	3	0
Willingness to join the group	57	18	25
Government intervention in Ayah mangrove ecotourism			
Partial intervention	77	19	4
Full intervention	20	32	48
Economic benefits of ecotourism			
Economic benefits for the whole community in Ayah Village	56	37	7
Economic benefits only for some participants	71	17	12
Environmental impacts of ecotourism			
Garbage in the mangrove forest area	38	5	57
Garbage in the village	35	5	60
Destruction of mangrove trees	2	6	92
Destruction of ecotourism facilities	1	6	93
The desire to develop sustainable mangrove ecotourism	96	2	2

The development of Ayah Mangrove Forest as an ecotourism object will likely impact the environment. In particular, the behavior of visitors who litter and destroy trees and existing facilities can endanger the sustainability of ecotourism. Indiscriminate waste disposal will cause water and soil pollution and cause plants to be unable to live properly. The most common waste in mangrove ecotourism is all kinds of plastic. The same condition was also reported in research in the Mangrove ecosystem area of Lesah Village (Loliwu et al. 2021). The types of marine debris found at the study site were plastic, rubber, metal, and glass waste, with plastic being the most dominant type of waste. The solutions that can be taken to reduce this impact include limiting the number of visitors, monitoring, encouraging visitors not to litter, and installing signage boards.

Most respondents agreed on the sustainability of ecotourism development in Ayah Mangrove Forest. They attributed this as an opportunity to improve the economy of the Ayah Village. In addition, managing the mangrove forest as an ecotourism object can also maintain, protect and rehabilitate the function of mangroves. Ayah Village has KTH Pansela, which promotes mangrove forest conservation and encourages community participation in protecting and conserving mangrove forests. This organization has several mangrove conservation programs, including seedlings provision, mangrove planting and maintenance, ecotourism management, public outreach, and field studies.

In conclusion, Ayah Mangrove Forest has the potential to be developed as one of the ecotourism objects in the Kebumen District, Central Java, Indonesia. There have been several conservation, protection, and utilization efforts in Ayah Mangrove Forest, and mangrove-based ecotourism can be aligned with conservation programs. Most of the local community has good perceptions and agreed on the development of Ayah Mangrove Forest as an ecotourism area. They are willing to be involved in ecotourism planning and development activities. The community agreed to participate in tour guide activities, selling souvenirs, providing homestays, and participating in ecotourism counseling. Most of the local community has also agreed to form a mangrove conservation community to maintain the sustainability of mangrove forest ecotourism. Nonetheless, support from the government is required, especially in the promotion and rehabilitation programs in the ecotourism development of Ayah Mangrove Forest.

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Volume and aboveground biomass models for a dry evergreen montane forest in Tanzania

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Abstract. *Mwaluseke ML, Mwakalukwa EE, Maliondo SMS. 2023. Volume and aboveground biomass models for a dry evergreen montane forest in Tanzania. Asian J For 7: 45-53*. Models available for accurately estimating trees and shrubs' volume and aboveground biomass from dry evergreen montane forests in Tanzania are largely lacking. Therefore, this study was conducted to develop volume, and aboveground biomass models for a dry evergreen montane forest of Lendikinya Forest Reserve found in Northern Tanzania. A total of thirty sample trees and shrubs with a diameter range of 5-58.5 cm were destructively harvested and used in this study. Specifically, the study developed (i) the height-diameter model and (ii) the total volume and aboveground biomass models. The following height, volume, and biomass models appear to be suitable for estimating tree height, volume, and biomass of tree and shrub species found in the study site: Height (m) =2.3249 + 6.6101/DBH + 0.2847DBH (R2 = 0.78, RMSE = 1.79, AIC = 164.37), ln(Volume, m^3) = -9.845 + 1.915 ln(DBH) + 1.089ln(Ht) (R2 = 0.97, RMSE = 0.296, AIC = -144.18) and ln(Biomass, kg) = -1.666 + 0.853ln(WD x DBH2 x Ht) (R2 = 0.95, RMSE = 0.324, AIC = 224.13). Both models yielded low bias, hence indicating an excellent fit. These models will be useful in understanding the condition of the forest and the potential of this forest in storing carbon hence, the possibility of benefiting from the ongoing negotiations of REDD+ schemes for payment for avoided deforestation and degradation through sustainable management of the reserve.

Keywords: Allometric, biomass, carbon, evergreen, Lendikinya Forest Reserve, models

INTRODUCTION

There is increasing interest in understanding the contribution of forest ecosystems in mitigating climate change effect (Lorenz and Lal 2010; Njana et al. 2018; Mauya et al. 2019; Leley et al. 2022; Mauya et al. 2022; Nugroho et al. 2022). That can be achieved by quantifying the carbon currently locked up in these ecosystems (Njana et al. 2018; Mauya et al. 2019; Leley et al. 2022). The estimation or quantification of carbon stocks is based on allometric equations developed in the same forests or similar vegetation types in cases where there are no existing models for such a specific forest (Chave et al. 2005; Henry et al. 2011; Ngomanda et al. 2014; Mwakalukwa et al. 2014). However, it is generally accepted that indirect methods using allometric volume equations, form factor and biomass expansion factors and/or with basic wood density (Chave et al. 2014; Njana et al. 2017) and models developed from other vegetation types to some extent, do not provide reliable estimates of carbon stocks in the studied vegetation (Henry et al. 2011; Chave et al. 2014; Djomo et al. 2016; Njana et al. 2018; Mauya et al. 2019). Specific models for each vegetation type are more precise in estimating biomass hence carbon stocks of the particular forest compared to the generalized models developed from other or similar vegetation types (Brown 2002; Mugasha et al. 2013; Chave et al. 2014; Daba and Soromessa 2019; Mauya et al. 2019; Asrat et al. 2020). Therefore, it is encouraged that locally specific models be developed to improve the accuracy and predictive capacity of the models using samples obtained from the same vegetation types (Mugasha et al. 2013; Mauya et al. 2014; Mwakalukwa et al. 2014; Feyisa et al. 2018; Njana et al. 2018; Asrat et al. 2020). That is especially very important when the country (project) is expecting to benefit from the ongoing initiative of payment for Reducing Emission from Deforestation and Degradation (REDD+) (Njana et al. 2018; Mauya et al. 2019; Mauya et al. 2022).

According to Mauya et al. (2019), countries are required to develop four key components if they aim to undertake REDD+ activities and to be eligible for financial compensation (FAO 2014): (i) a national strategy or action plan; (ii) a national forest reference emission level (FREL) and/or forest reference level (FRL); (iii) a robust and transparent national forest monitoring system for Measurement, Reporting and Verification (MRV) of the REDD+ activities; and (iv) a system for providing information on how the safeguards are addressed or respected. The Tanzania mainland has up-to-date national forest inventory data collected through a National Forest Resources Monitoring and Assessment (NAFORMA) Project between 2009 and 2014 (MNRT 2015). Those data could serve to address the above requirements. In addition, the availability of as many up to date allometric models to assist in quantifying existing carbon stocks and estimating potential emissions is necessary.

In Tanzania, most of these models have been developed for different vegetation types (Malimbwi et al. 2016). Specifically, models have been developed for Miombo forests (Mugasha et al. 2013; Mauya et al. 2014; Mwakalukwa et al. 2014; Manyanda et al. 2019), mangrove forests (Njana et al. 2016a,b), lowland forests (Mugasha et al. 2016a), mountain humid/rain forest (Masota et al. 2014; Masota et al. 2015), thicket and associated trees (Makero et al. 2016), *Acacia*-*Commiphora* woodlands (Mugasha et al. 2016b) and plantation forests (Mugasha et al. 2016c; Zahabu et al. 2016). However, no volume and biomass models have been developed for dry evergreen montane forests in Tanzania, unlike other areas such as Ethiopia that have been studied (Tetemke et al. 2019; Asrat et al. 2020).

This study intended to provide robust stand volume and biomass models for dry evergreen montane forests of Lendikinya Forest Reserve (LFR) in Northern Tanzania to assist in better planning and management. Stand-level models can also help to understand the contribution of the forest in mitigating climate change based on future REDD+ initiatives in Tanzania, particularly for effective monitoring, reporting, and verification of Greenhouse Gas (GHG) emissions. Therefore, the specific objective of the study was to develop volume and aboveground biomass models of trees and shrubs with a diameter ≥ 5 cm found in the LFR. The models would be used for dry evergreen montane forests in that region.

MATERIALS AND METHODS

Study area description

Lendikinya Forest Reserve (LFR), with a total area of 3,689 ha and gazetted in 1969 (JB No. 1854), is a dry evergreen montane forest located in the eastern part of Monduli District (latitudes 2° and 4° S and longitude 36° and 37° E) in Arusha Region, Tanzania (Meindertsma and Kessler 1997) (Figure 1). Monduli Local Government Authority manages the LFR. Generally, the district's climate is arid to semi-arid, with average rainfall between 400 to 900 mm per annum and wide variations in relief and soil types (UNDP 2003). The weather range from as low as 11.5°C in July to a maximum temperature of up to 29°C in December. For the lower altitudes in May, humidity during the night reaches 100%. LFR is surrounded by four villages: Lashaine, Monduli Juu, Alkatani, and Lendikinya. Moreover, the economic activities of the people of the area depend on livestock, agro-pastoralism, and tourism. LFR forms part of the rift valley characterized by depressions. The woodland harbors large wild animal species such as *Loxodonta africana* (Elephants), *Giraffa camelopardalis* (Giraffe), *Syncerus caffer* (Buffaloes), and a variety of birds and insects (Meindertsma and Kessler 1997; UNDP 2003).

Field sampling

The field survey was conducted in May-June 2014. A total of 30 sample trees and shrubs (Table 1) with a diameter range of 5-58.5 cm were selected based on species composition and diameter classes of species available in the forest (Mwaluseke 2015, unpublished data). The selection ratio was five trees to 1 shrub (Chaturvedi and Raghubanshi 2012; Mwakalukwa et al. 2014; Malimbwi et al. 2016; Asrat et al. 2020; Teshome et al. 2022). Few shrubs were selected because they possessed lower diameter size classes and are the least contributor to total volume and biomass than the large-sized trees (Asrat et al. 2020; Teshome et al. 2022). The selected trees and shrubs species were first identified before they were felled (Table 2) (Mwakalukwa et al. 2014; Malimbwi et al. 2016). After identification, the trees were measured for diameter at breast height (DBH at 1.3 m) using diameter tape/caliper to the nearest 0.1 cm (Teshome et al. 2022). The trees were also measured for their height using a Suunto hypsometer. In addition, height measurements were also taken for the three selected stems (small, medium, and largest in terms of diameter) using the same instrument (a Suunto hypsometer). Next, the tree was felled 10 cm from the ground level using a chainsaw. Then, using a tape measure, the total height of the felled tree was measured before it was segregated into different components: stems, branches, and leaves (Mugasha et al. 2013; 16a). The upper diameter limit selection depends on the wood's utilization (Mwakalukwa et al. 2014). In the study area, the important use of the wood was mainly for poles and timber production, with few tree species used for charcoal production, mainly utilizing the top diameter of the 5 cm portion. Therefore, both volume and biomass models were developed based on DBH ≥ 5 cm.

For each felled sample tree, twigs and leaves were removed from branches and tied into bundles (piles) (Mandal et al. 2013; Mwakalukwa et al. 2014; Mugasha et al. 2016a; Asrat et al. 2020). Depending on their weights, stem and branch billets were tied in bundles and weighed separately to obtain green weights, and eventually, the green weight of a whole tree was obtained by summating the weight of individual tree sections (Mwakalukwa et al. 2014; Mugasha et al. 2016a). The subsamples were then weighed using a Portable Digital scale, while large and heavy billets or piles were weighed using a conventional scale (Asrat et al. 2020). For dry weight estimation, a subsample from each pile of twigs and leaves and two stem discs (one from the stem at 1.3 cm aboveground and one from a branch at approximately 30 cm from the point of branching) were obtained and weighed for green weight in the field and taken to the laboratory for oven-dry weight determination (Mandal et al. 2013; Asrat et al. 2020). All samples were carefully marked for respective species names, tree numbers, and DBH. The lightweight data was packed in envelopes for laboratory analysis. Furthermore, to facilitate biomass weighing and the construction of volume models, each of the stem and branch sections of the felled trees was trimmed into billets of length 1-2.5 m and measured at mid-diameters (dm) (Malimbwi et al. 1994; Mugasha et al. 2013; Asrat et al. 2020).

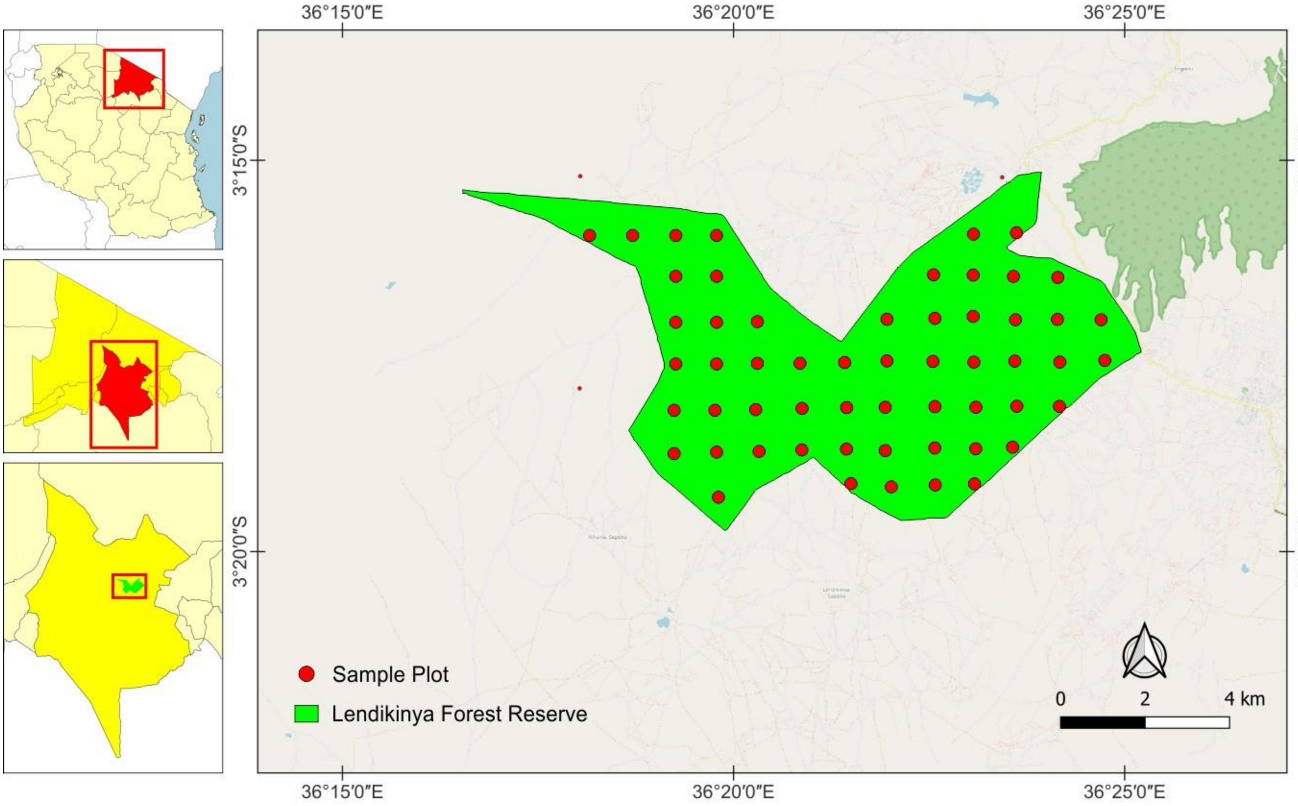

Figure 1. Location of Lendikinya Forest Reserve in Monduli District, Arusha, Tanzania

Laboratory analyses

In the laboratory, all large disc samples were split into small blocks and marked. Then, all block subsamples were soaked in water for seven days until they were saturated (Malimbwi et al. 1994; Mugasha et al. 2013). After that, the water displacement method was used to determine the green volume of each subsample (i.e., small discs and blocks) (Chaturvedi and Raghubanshi 2012). Eventually, the subsamples were oven-dried at a standard temperature of 103 ± 2°C for four and half days to a constant weight, and their oven-dry weight was recorded (Mandal et al. 2013; Mwakalukwa et al. 2014; Asrat et al. 2020). Wood basic density (g cm^{-3}) was determined as the ratio of oven-dry weight to fresh volume, while dry biomass was computed by multiplying the fresh weight with oven-dry weight to green weight ratio (Chaturvedi and Raghubanshi 2012; Mwakalukwa et al. 2014; Njana et al. 2016a,b; Asrat et al. 2020).

Data preparation

Before the model development, data were assessed for additivity (Mugasha et al. 2016a). The additivity concept was attained by summing the biomass and volume components (stem wood, branches, and leaves). Thus, ensuring that the total biomass and volume obtained equaled the total biomass and volume predicted by allometric equations (Asrat et al. 2020; Teshome et al. 2022). Next, for each tree, the volume of individual stems and branch billets was calculated using Huber's formula ($V=\pi\ Ld^2_m/4$; where d_m= Billet's mid-diameter, L=Billet length, π = Pai, V= billet volume). Then, the total volume of each sample tree/shrub with a diameter ≥ 5 cm was calculated by adding the individual volume of stem and branch billet sections. This dataset was used to develop mixed-species volume models (Mwakalukwa et al. 2014; Mauya et al. 2014).

Dry biomass (kg) was estimated as the product of estimated biomass ratios (oven-dry to fresh weights) for each subsample (i.e., stem and branch discs and leaves) and their corresponding fresh weights (kg) measured in the field. The total aboveground biomass of each tree was the sum of individual oven-dry biomass of stem, branch, and leaves (Mugasha et al. 2013; Mugasha et al. 2016a; Asrat et al. 2020). The resulting dataset was used to develop aboveground mixed-species-biomass models.

Statistical analysis

Before volume and biomass models were developed, height models were developed to determine the heights of unmeasured live trees and shrubs (Mwakalukwa et al. 2014; Mugasha et al. 2016a). A total of 139 of both felled and unfelled sample trees and shrubs were mixed to capture their variations and accuracy in the estimates. The height dataset was then fitted to the formulated models, as shown in Table 3. DBH was the only predictor variable in the height models, while height (H) was the dependent variable.

For the volume and biomass models development, logarithmic linear models formulated and used elsewhere (Malimbwi et al. 1994; Mwakalukwa et al. 2014) in other forests for the prediction of either volume or biomass or both were also adopted in this study (Table 4). In these models, both dependent and independent (i.e., DBH and height) variables were natural logarithmic transformed to attain linearity. The transformation was also intended to reduce he heteroscedasticity's effect and obtain models that fit the dataset well.

Model 5 in (Table 4) partly resembled model 2, but model 5 further included average wood basic density, intended to capture its contribution to biomass prediction by the model (Chaturvedi and Raghubanshi 2012; Njana et al. 2016a,b). In transformed regression equations, the analysis of the models was preceded by plotting dependent variables (volume and biomass) against each of the independent variables (DBH and height) to determine the range and likely shape of the functional relationship and the heteroscedasticity assessment (Mwakalukwa et al. 2014). Since logarithmic transformation leads to another problem of geometric mean instead of arithmetic mean in the predicted output, a correction factor "CF" following Baskerville (1972) was used. Those equations ((CF = exp(MSE/2) where MSE = Mean Square Error of the regression, CF = A Correction Factor, Exp = Exponential function)) were used to solve the problem by multiplying it with the predicted output after back-transformed.

The statistical tests employed in evaluating the models regarding the goodness of fit criteria follow Parresol (1999) and Mandal et al. (2013). According to FAO (2012), the most frequently used statistical tests in determining the performance of a model were: coefficient of determination (R^2), Root Mean Square Error (RMSE), and Akaike's Information Criterion (AIC), all of which were computed by the following equations (R^2 = 1-RSS/TSS, where; R^2 = coefficient of determination, RSS = Residual sum of squares, TSS = Total Sum of Squares; RMSE = $\sqrt{}$RSS/n-Ψ, Where; n = Number of observations in a data model, (Ψ) = Number of parameters present in a model, RSS = Residual sum of squares, RMSE = Root Mean Square Error, and AIC = n(lnRSS/n)+2k, Where; n = Number of data points (observations) in a given equation, ln = Natural logarithm, RSS = Residual Sum of Squares, k = Number of parameters in a given equation, AIC = Akaike's Information Criterion) respectively. For each category available, the best model selected had higher R^2 but lower RMSE and AIC values (Parresol 1999; Mugasha et al. 2013; Asrat et al. 2020).

Table 1. Distribution of felled sample trees

Diameter class (cm)	**5-10**	**10.1-20**	**20.1-30**	**30.1-40**	**40.1-50**	**> 50**
Number felled	3	10	8	4	2	3

Table 2. Felled sample trees showing their DBH, height, basic density, volume, and biomass content

Botanical Name	**Dbh (cm)**	**Height (m)**	**Stem (ρ) (gcm^{-3})**	**Branch(ρ) (gcm^{-3})**	**Volume (m^3)**	**Biomass (kg)**
Euclea natalensis A.DC.	5.0	6.2	0.614	0.583	0.0158	13.0283
Turraea holstii Gürke	7.5	4.5	0.509	0.448	0.0055	11.2261
Capparis tomentosa Lam.	10.0	4.6	0.735	0.733	0.0273	32.224
Clausena anisata Willd. Hook. f. ex Benth	10.9	8.0	0.704	0.559	0.0383	30.9263
Indigofera sp.	12.3	7.3	0.506	0.481	0.0527	37.7481
Vangueria madagascariensis J.F.Gmel.	13.8	6.3	0.514	0.528	0.0541	39.5458
Carissa edulis (Forssk.) Vahl	15.0	9.6	0.680	0.606	0.0927	130.867
Elaeodendron buchananii (Loes.) Loes	16.1	5.0	0.624	0.603	0.0722	46.3013
Hibiscus sp.	17.0	4.7	0.612	0.600	0.0793	94.080
Ozoroa insignis Delile	18.5	8.4	0.510	0.426	0.1679	105.683
Teclea simplicifolia (Engl.) I. Verd	19.0	7.7	0.749	0.685	0.1611	175.140
Albizia schimperiana Oliv.	19.9	15.6	0.487	0.512	0.2699	156.608
Vangueria madagascariensis J.F.Gmel.	20.0	3.8	0.521	0.508	0.0577	26.570
Ozoroa insignis Delile subsp. latifolia	20.5	11.9	0.529	0.531	0.5090	246.516
Calodendrum capense (L.f.) Thunb.	20.6	15.0	0.484	0.440	0.3553	196.553
Acacia xanthophloea Hochst. Ex Benth.	21.0	11.7	0.740	0.671	0.2438	255.754
Celtis africana Burm.f.	22.0	14.8	0.562	0.643	0.2824	206.441
Olea sp.	24.0	9.5	0.707	0.703	0.2748	328.615
Maytenus senegalensis (Lam.) Exell	26.0	6.5	0.685	0.640	0.2579	159.633
Maytenus senegalensis (Lam.) Exell	27.4	7.8	0.673	0.608	0.2815	173.052
Acacia nilotica (L.) Willd. ex Delile	28.4	11.2	0.797	0.735	0.4143	358.659
Calodendrum capense (L.f.) Thunb.	30.1	10.0	0.583	0.480	0.4537	273.221
Acacia xanthophloea Hochst. Ex Benth.	32.3	12.8	0.766	0.654	0.7446	514.147
Acacia nilotica (L.) Willd. Ex Delile	33.1	11.3	0.746	0.708	0.6321	400.086
Albizia schimperiana Oliv.	38.7	16.2	0.485	0.490	0.8717	497.065
Celtis africana Burm.f.	41.0	18.8	0.667	0.662	1.1654	671.814
Acacia gerradii Benth.	43.0	14.6	0.816	0.786	1.2700	770.793
Diospyros abyssinica subsp. abyssinica	50.1	21.3	0.628	0.644	2.5103	1339.99
Vangueria madagascariensis J.F.Gmel.	54.0	20.6	0.561	0.473	3.8998	1748.28
Vangueria madagascariensis J.F.Gmel.	58.5	23.8	0.552	0.539	4.4837	2062.34

Table 3. Height model forms tested

Model No.	Height model form
1	$H = b_0 + b_1 DBH$
2	$H = 1.3 + b_1 DBH + b_2 DBH^2$
3	$H = b_0 + \frac{b_1}{DBH} + b_2 DBH$
4	$H = \left(\frac{b_0 DBH}{b_1 + DBH}\right) - b_2$

Note: H=Tree height, b_o,= Intercept of DBH, b_1, and b_2 are constant parameters while DBH is as defined previously

Table 4. Volume and biomass model forms tested

Model No.	Model form
1	$Ln\ (Y) = a + b \times Ln\ (DBH) + c \times Ln\ (DBH^2) + d \times Ln\ (Ht) + e \times Ln(Ht^2)$
2	$Ln\ (Y) = a + b \times Ln\ (DBH^2 \times Ht)$
3	$Ln\ (Y) = a + b \times Ln\ (DBH) + c \times Ln\ (Ht)$
4	$Ln\ (Y) = a + b \times Ln\ (DBH)$
5	$Ln\ (Y) = a + b \times Ln\ (WD \times DBH^2 \times Ht)$

Note: Y=Volume (m^3) or Biomass (kg); DBH (cm) and Ht=Tree height (m) and a, b, c, d, and e are constant parameters to be analyzed, Ln = natural logarithm, WD = Wood basic density

The accuracy of the model was checked by percentage bias (PBIAS = $\sum(X_{obs}-X_{pre})/\sum X_{obs})x100$, where; PBIAS = Percentage Bias, X_{obs} = Observed value derived from the equation, X_{pre} = Predicted value derived from the equation). According to Mandal et al. (2013), the Percentage Bias (PBIAS) was used to compare and evaluate the predicted and observed values for accuracy assessment. The lowest value of PBIAS indicated by a candidate model gives a better-fit result (Mandal et al. 2013). Regarding Osman et al. (2013), a graphical plot on residuals versus predicted values was important in visualizing the performance of prediction models. The best-selected models were then used to predict corresponding height, volume, and dry biomass quantities (Mugasha et al. 2013; Mwakalukwa et al. 2014; Asrat et al. 2020). All analyses were carried out in Microsoft Excel Spreadsheet, PAST, and Minitab 15 software.

RESULTS AND DISCUSSION

Height models

Four model forms were formulated to predict height as a dependent variable, with diameter as the independent variable (Table 5). First, model 2 had the lowest R^2 value but the highest values of RMSE and AIC. That indicates the poorest performance of all other models. Next, models 1, 2, and 4 had similar RMSE values (1.79). Then Models 1 and 4 had similar R^2 values (0.77). Finally, Model 3 had the highest R^2 and the lowest AIC value of all other models. The goodness of fit shown by model 3 implied that the model fitted the data well and was considered the best model for the height prediction for the unmeasured tree heights.

When standardized residuals for model 3 were plotted against predicted values, it showed that most of the residuals were evenly distributed on both sides but were mostly closer to the horizontal line (zero), indicating the model fitted the data well (Figure 2). In addition, the model also had a very small bias of 0.23%, indicating a reduced error in the height prediction.

Volume models

Four models were parameterized for volume prediction. Table 6 shows that all four models had higher R^2 values ranging from 0.91 to 0.97. Model 1 had higher R^2 and AIC values than all other models except for parameter "a." All other parameters were insignificant at $p < 0.05$, indicating poor performance than other models. Like model 1, model 2 had higher $R^{2,}$ but all other parameters were significant at $p < 0.05$, and their AIC value was much lower than model 1, indicating it is a better model. Model 3 had a similar R^2 value as models 1 and 2 but was not comparable with model 4, which had the lowest R^2 and highest RMSE and PBIAS indicating poor performance than all other models. Since model 3 had the lowest AIC and percentage bias, it was considered the best model for volume prediction.

When standardized residuals of model 3 were plotted against predicted values (Figure 3), the scatter plot did not show any noticeable pattern. Most standardized residuals were distributed close but along the horizontal line (zero), implying that a model fitted well the data.

Biomass models

Five model forms were parameterized for biomass prediction (Table 7). Results showed high values of R^2 with a range of 0.89 to 0.96. Since model 1 had the highest R^2 and smallest values of RMSE and AIC, it was expected to be the best model, but almost all its parameters were not significant at ($p < 0.05$). Apart from models 1 and 4, all other models had equal R^2, but higher values of RMSE and AIC and all their parameters were significant at ($p < 0.05$). Among the significant models, that is, models 2 up to 5, it was observed that model 5 had the lowest RMSE and AIC values than the other three significant models indicating it to be the best model.

When standardized residuals for model 5 were plotted against predicted values (Figure 4), the scatter plot did not show any noticeable pattern. Most of the standardized residuals were distributed along but close to the horizontal line (zero), indicating the attainment of homoscedasticity and the model fitted well the data.

Table 5. Height models showing parameter values and performance

Model No.	Parameter estimate and standard error			Goodness of fit		
	b_0	b_1	b_2	R^2	RMSE	AIC
1	3.3266 (0.3466)	0.1306 (693576.1)	-	0.77	1.79	165.09
2	1.3 (0)	0.3923 (0.0241)	-0.00171 (0.00063)	0.72	2.03	200.20
3	2.3249 (0.8977)	6.6101 (5.4716)	0.2847 (0.0241)	0.78	1.79	164.37
4	2.61e+08 (0)	1.0e+09 (0)	-3.3266 (0)	0.77	1.79	165.06

Note: b_0: constant; b_1: intercept for independent variable DBH, and b_2: constant parameters of models, and the numbers in blackest are their standard errors

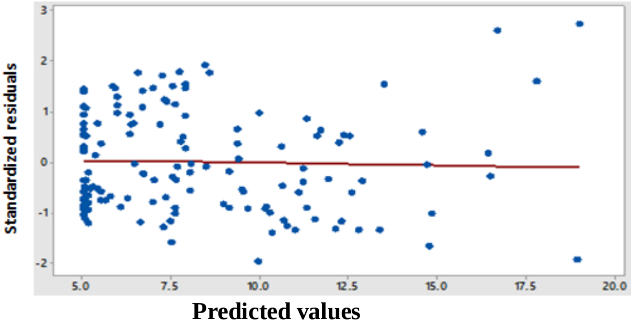

Figure 2. Distribution of residuals from the horizontal line for height model 3

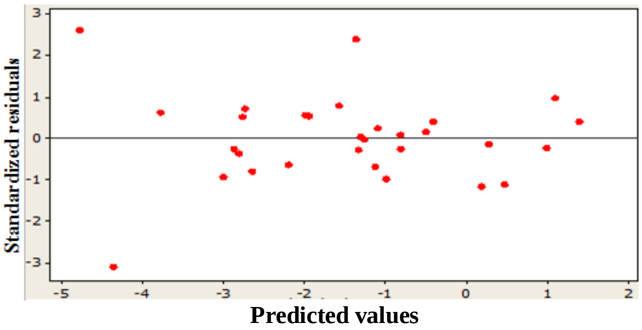

Figure 3. The distribution of residuals from line zero for the volume model 3

Table 6. Volume models showing parameter values and performance

Model No.	Parameter estimates						Goodness of fit			Accuracy
	a	b	c	d	e	CF	R^2	RMSE	AIC	%BIAS
1	-10.02 -1.28^{s} 0.000^{p}	0.7 0.8 0.383^{p}	0.2171 0.158^{s} 0.179^{p}	2.741 1.12^{s} 0.022^{p}	-0.402 0.26^{s} 0.14^{p}	1.043	0.97	0.292	61.2	0.
2	-9.86 0.30^{s} 0.000^{p}	0.99 0.04 0.000^{p}				1.043	0.97	0.292	-111.18	-1.14
3	-9.845 0.30^{s} 0.000^{p}	1.9 0.1 0.000^{p}	1.089 0.16^{s} 0.000^{p}			1.044	0.97	0.296	-144.18	-2.27
4	-9.583 0.48^{s} 0.000^{p}	2.63 0.15 0.000^{p}				1.118	0.91		-138	2.

Note: a is intercept, b, c, d, and e are constant parameters, and the superscript "s" and "p" are the standard error and probability of a parameter, respectively

Discussion

The height and diameter relationship found in this study was non-linear. Non-linear regression model 3 had a higher R^2 than the other models tested. Marshall et al. (2012) argued that individual tree height is not simply correlated with diameter; instead, the ratio is related to species and the condition of the area. Differences in the structure "architecture" of the woody plants, especially shrubs, might have affected the performance of the height models tested.

The developed volume and biomass models were important for assessing volume and carbon stock in LFR. However, the models tested performed differently. For both volume and biomass, model 3, which included height (Ht) in addition to (DBH) as independent variables, improved the fit more than when (DBH) alone (Model 4). Marshall et al. (2012) reported an overestimation of 55 t ha^{-1} or 31.5% biomass when height was excluded in the biomass prediction model. That is in agreement with other studies (Malimbwi et al. 1994; Chave et al. 2005, Marshall et al. 2012; Mugasha et al. 2013; Mwakalukwa et al. 2014) that argued regression equations incorporating height are most likely to be accurate as they incorporate more information on the size of stems than those which utilized diameter alone. It was further shown that the inclusion of wood basic density in model 5 significantly improved the fit as supported by higher R^2 and low RMSE.

Table 7. Biomass models showing parameter values and performance

Model No.	Parameter estimates a	b	c	d	e	CF	Goodness of fit R^2	RMSE	AIC	Accuracy %BIAS
1	-3.029	0.445	0.2217	3.384	-0.589	1.051	0.96	0.315	180.43	3.21
	1.37^s	0.93^s	0.16^s	1.21^s	0.28^s					
	0.037^p	0.638^p	0.201^p	0.01^p	0.056^p					
2	-2.074	0.853				1.054	0.95	0.325	257.28	-3.36
	0.33^s	0.039^s								
	0.000^p	0.000^p								
3	-2.05	1.585	1.01			1.054	0.95	0.326	236.81	-2.31
	0.33^s	0.16^s	0.18^s							
	0.000^p	0.000^p	0.000^p							
4	-1.807	2.251				1.117	0.89	0.47	201.54	1.33
	0.48^s	0.15^s								
	0.001^p	0.000^p								
5	-1.666	0.853				1.054	0.95	0.324	224.13	-1.93
	0.31^s	0.04^s								
	0.000^p	0.000^p								

Note: The superscript "s" and "p" are the standard error and probability of a parameter, respectively

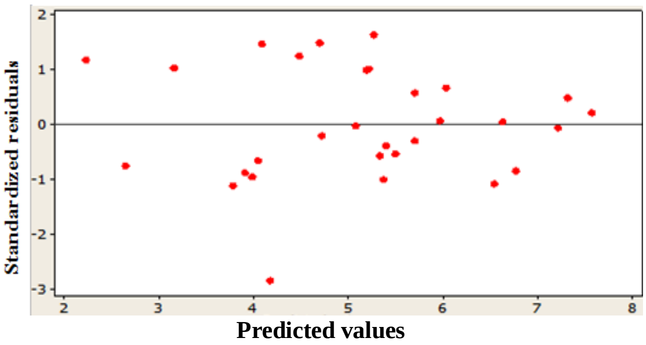

Figure 4. The distribution of residuals about the horizontal line for biomass model 5

The higher diversity and species composition experienced in the tropical forests, particularly the dry evergreen montane forest of LFR, support the argument for developing site-specific equations for mixed species (Djomo et al. 2016; Mokria et al. 2018; Asrat et al. 2020). Furthermore, Mwakalukwa et al. (2014) argued that due to the variation in species composition from site to site and the impact of site conditions on the shape of trees, the use of mixed-species regression models calibrated using data from sites with similar conditions and species composition is a logical choice. However, using biomass-generalized allometric equations available for tropical forests (Brown et al. 1989; Chave et al. 2005; Djomo et al. 2016) gave lower results and tended to give higher errors (Teshome et al. 2022). For instance, Brown et al. (1989) biomass model gave an error of 5.5%, while that of Chave et al. (2005) had a much higher error of 15.1%, implying that site-specific mixed species biomass models developed in this study are more accurate by having a much lower error of 1.93% in the prediction.

The differences in the estimates might also be attributed to differences in diameter size classes used to construct these generalized models. For instance, Brown et al. (1989) used a maximum of 40 cm. In contrast, this study developed the volume and biomass models with a maximum diameter of 58.5 cm. Moreover, locally abundant species are not represented in the databases used to develop the generalized models, thus failing to accurately predict the true biomass estimates in a particular forest (Mugasha et al. 2013; Mwakalukwa et al. 2014; Mugasha et al. 2016a). Therefore, caution should be taken when using generalized models where local site-specific mixed-species models are unavailable (Djomo et al. 2016; Teshome et al. 2022). The use of site-specific models is recommended to ensure that high precision in the quantification of woodland resources is achieved (Mwakalukwa et al. 2014; Njana 2017; Mauya et al. 2019).

In the allometric models selected for height prediction, the volume and biomass (carbon) quantification had high R^2 and lower RMSE, AIC, and percentage bias than the existing generalized equations developed for vegetation in dry tropical forests. This study showed that height model 3, volume model 3, and biomass model 5 were the best-predicting models in the study area. Despite DBH being the common predictor variable in most developed allometric models, the inclusion of height in the volume equation, and the use of wood density in biomass model 5 increased the goodness of fit (Henry et al. 2010; Mugasha et al. 2013; Mwakalukwa et al. 2014). The developed models provide important managerial tools that will assist managers, planners, and policymakers manage the LFR more sustainably, especially for future REDD+ project implementation phases in Tanzania.

In conclusion, this study, for the first time, reports volume and biomass models for dry evergreen montane forests found in Tanzania. The reported site-specific models developed based on destructively sampled trees data from dry evergreen montane forests in Northern Tanzania yielded low bias, indicating an excellent fit. These models may also be considered for application in other dry evergreen montane forests lacking site-specific models after carefully evaluating the required conditions (i.e., tree-size distribution, species composition, and site characters). These developed models add to the knowledge about volume and biomass models developed from various vegetation types found in Tanzania.

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Identification and characterization of honeybee flora calendar in Southwest Jimma Zone, Ethiopia

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Abstract. *Mossie T, Worku H. 2023. Identification and characterization of honeybee flora calendar in Southwest Jimma Zone, Ethiopia. Asian J For 7: 54-66.* The study was conducted to identify, characterize and document major bee forages, develop an appropriate flora calendar, their phenology, and pollen potential in the various agro-ecological conditions of the Jimma Zone, Ethiopia. A total of 90 beekeepers were purposefully selected from three districts and interviewed to get primary data. The density and abundance of flowering plants were determined using quadrat sampling techniques. In addition, pollen specimens were collected using pollen traps at seven-day intervals and were also traced back to plant species level under a light microscope. The study has revealed the presence of 141 pollen and/or nectar-source honeybee plant species belonging to sixty-two families in the study area. Herbs were the most dominant bee flora growth forms, accounting for 62 (44%) of a total of 141 honeybee plant species, followed by trees at 48 (34%) and shrubs at 31 (22%), respectively. The herbaceous plant had a greater density value of plant species per plot than did trees and shrubs. One hundred fifteen (81.6%) were both sources of pollen and nectar, whereas fifteen (10.6%) were pollen sources and the remaining eleven (7.8%) were nectar-source plant species. The Shannon diversity index and evenness were found to be 2.8 and 0.6, respectively. This indicated that the study area has a rich bee floral plant species and is suitable for beekeeping. Two main flowering periods of honeybee plants were followed by two honey flow seasons. Therefore, beekeepers should follow the floral calendar of honeybee plants to exploit the potential of the area for honey production.

Keywords: Bee flora species, flowering period, honeybees, trapped pollen

INTRODUCTION

Apiculture is a livestock industry that contributes significantly to a country's national and international economies, mainly in Africa. Ethiopia is still one of the top ten natural honey producers worldwide due to its diverse floral resources and favorable ecosystem conditions (Fichtl and Admassu 1994; Gidey and Mekonen 2010; Bareke and Addi 2020). It, directly and indirectly, contributes to household income and the national economy (Fenet and Alemayehu 2016). The production of honey, beeswax, pollen, royal jelly, and other by-products is the direct income source for the users, while contributing to plant pollination and the conservation of the natural environment, is the indirect role of honey bee production. Honeybees and plants have had a strong relationship for over 50 million years (FAO 1986). Beekeeping conserves natural resources and protects the global environment. It can be integrated with agricultural practices like crop production, horticulture crops, and the conservation of natural resources (Gezahegn 2001; Bareke and Addi 2020). Honey bees require feed for their production and reproduction, like other livestock species. It depends on flowering plants for their nutrition and protection. About 40, 000 plant species are used as honeybee forage across the world (Crane 1990). Among the flowering plants found in Ethiopia, 500 species are rich in nectar and pollen (Fichtl and Admasu 1994). Plants are classified as nectar or pollen-source plants based on the honeybee's activity of extending their proboscis and hind legs into flowers, respectively (Wubie et al. 2014; Jenberie et al. 2016; Pande and Gi 2018). Honeybees' main food sources are pollen and nectar. Nectar is a major component in the production of honey, whereas pollen is used as larval food, which is important in colony reproduction (Façade and Paul 2006).

Ethiopia has abundant natural and cultivated flora, as well as diverse agro-ecosystem and climatic conditions ideal for beekeeping. The presence of numerous honey plants is important for the country's honeybee colonies, production, and productivity. The botanical composition of natural vegetation differs depending on the agro-ecosystem , climate, and soil type (Gebretsadik 2016). The type and quantity of flora present determine the productivity and reproduction performance of honeybees (Amssalu 2007). Oromia is one of the Federal Republic of Ethiopia's regional states rich in natural resources and has favorable climatic conditions for improved beekeeping development. The region has virgin forests with high biodiversity, such as Harena, Yayu, Dindin, Anfarara, Munessa, Jibat, Chilimo, and Menagesha-Suba that are ideal for beekeeping. The region also contains cultivated crops such as oil and horticultural crops, as well as pulses, all of which can help to further the development of beekeeping. These make the region one of the potential areas for honeybee production.

Despite the region's diverse agro-ecosystem and climatic conditions, abundance of natural and cultivated flora, beekeepers lack a floral calendar for honeybee foraging and honey production. Flora calendar is a timetable that indicates the approximate duration of the flowering period, abundance, distribution, and honey potential of honeybee forages in various agro-ecosystem zones of the country (Amssalu 2004; Admasu et al. 2004). Identification and documentation of bee forage and their flowering calendar is critical for the sub-sectors development endeavors since the flowering periods of honeybee plants differ depending on the diversity of plant habits and environmental conditions (Tilahun 2003). Therefore, establishing a floral calendar is a critical tool for planning various beekeeping management operations, such as hive super adding, and predicting the frequency and period of honey flow in a given area. The length of the flowering period, nectar and pollen production, and honeybee plant availability in a specific area are all determined by agro-ecosystem and season. Therefore, assessing the different agro-ecosystem zones for determining the availability of bee forage, their life forms and establishing a flowering calendar of honey plants that enable effective seasonal colony management is paramount important. Furthermore, for optimal honey production, beekeepers should be aware of the flowering seasons of both main and minor nectar and pollen sources of plants in the vicinity of their apiary site (Francis 1990; Pearson and Braiden 1990). The study was conducted with the objective to characterize and document major bee forages, develop an appropriate flora calendar, and identify major mono- and multi-floral honey sources for effective bee management in various agro-ecosystem conditions of the Jimma Zone.

MATERIALS AND METHODS

Study area profile

The study was conducted in beekeeping potential areas of the Jimma Zone of the Oromia Regional State, Ethiopia, which geographically lies at a latitude of about 7013’-8 056’N and a longitude of about 35052’-37037’E (Figure 1). The area has high humidity and is rich in fauna and flora biodiversity. Three study districts (Goma, Gera, and Shebe Sombo) were selected based on ecosystem differences and beekeeping potential (Bareke and Addi 2019).

Honeybee flora inventory

The study was carried out in three beekeeping potential districts of Gera, Goma, and Shebe, representing highland, midland, and lowland agro-ecosystem. Agro-ecosystem representation was used to exploit bee flora species in different ecosystems of the study area. Three kebeles were selected from each district depending on their agro-ecosystem variation and potential for beekeeping activities. Household beekeepers were selected based on their experience in beekeeping and after discussion with district experts. A total of ninety (90) beekeepers, thirty (30) from each district, were also purposefully selected to get sound information on honey source plant lists, flowering periods, duration, beekeeping experience, number of colonies, number of harvests per year, and presence of poisonous plants. Well-structured questionnaires or checklists were created to collect both primary and secondary data from respondents, depending on their beekeeping experience in potential districts. A group discussion with experts, community groups, development agents, and farmer beekeepers was held to generate relevant information. Necessary and supportive data on plant nature and habitats, feeding resources, and plant phenology were collected following field observation. The types of honeybee forage, honey flow season, plants with adverse effects, swarming seasons, and management practices were considered during data collection.

Pollen sample collection and laboratory analysis

In total, 18 honey bee colonies were established in nine different locations across three districts of the study area in different agro-ecosystems.

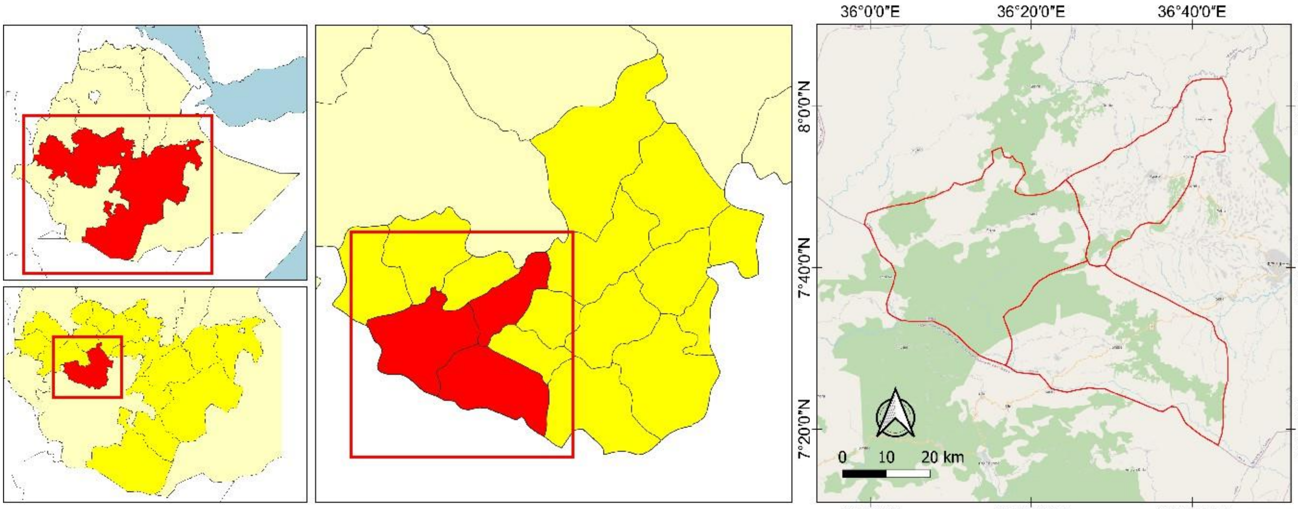

Figure 1. Study area map where samples were collected in the Jimma Zone, Oromia State, Ethiopia

At each site, two honeybee colonies were established for pollen trapping and pollen sample collection. Honeybee colonies were fitted with pollen traps and loads to collect and dislodge pollen pellet samples at seven-day intervals. In one year, a total of 66 pollen specimens were collected and used to determine the botanical origin of honeybee pollen. The fresh and dry weight of pollen pellets was recorded. The collected pollen was dried at room temperature overnight, and the fresh and dry pollen pellets were weighed and sorted by color. Each sorted pollen sample was identified at generic and species levels under the light microscope following diluting with ether solution. Using reference materials, plant species were identified from each type of pollen by comparing the shape, size, and apertures of the pollen. Pollen that we couldn't identify botanically using either analysis technique has been labeled as "unidentified."

Honeybee flora species composition and diversity

Assessment of plant species composition and diversity was performed in purposively selected districts based on beekeeping potential. For vegetation analysis using quadrat sampling techniques, two representative kebeles were chosen from each district based on vegetation coverage and ecosystem difference (highland, midland, and lowland). The diversity and composition of honeybee plants were determined according to Tesfaye et al. (2013) plant density determination method. The quadrat/plot size varied depending on vegetation types. Honeybee plants were classified as trees, shrubs, and herbaceous. Tree and shrub sampling plots were 20 m × 20 m in size, whereas herbaceous plant sampling plots were 2 m x 2 m in a two-kilometer radius every 0.1 km from the hive to estimate the frequency and density of bee plants. The main plots were laid out systematically considering the availability of vegetation coverage, and then small quadrats of 2 m x 2 m plots were laid out at different sites of the main plots to understand the forgeable area of honey plants. A total of 60 plots were taken for the districts, representing different agro-ecosystems. Then, plant species within quadrats were counted for the assessment of plant density and frequency in specific sampling sites. Honeybee flora species abundance was defined and computed in all quadrats, and density was calculated in hectares. Plant specimens were collected during flowering seasons with necessary botanical features like leaves, flowers, and a portion of the stem. The collected specimens were pressed, identified, and then compared to the published report at the Holeta Bee Research Center.

Richness and diversity of bee forage plants

The Shannon-Wiener diversity index, species richness, and Shannon's evenness were used to determine the diversity of bee forage plant species. The Shannon-Wiener diversity index is the most widely used non-sample-size-dependent measure of species diversity (Ramirez-Arriaga et al. 2011).

Shannon index (H') =$\sum$ (pi*lnpi), where H' = Shannon index, pi= proportion of individual species, and Ln= log base n.

Evenness (J) = H'/ H'max = H'/INS, where H' =Shannon diversity index, H'max = lns where S is the number of species, Ln = logbase. The value of evenness is found between zero to one (Kent and Coker 1992).

Statistical analysis

Data on bee flora species, abundance, frequency, diversity, and pollen count were summarized using descriptive statistics. The data were thoroughly examined using Microsoft Excel and the Statistical Package for Social Sciences (SPSS). And the results were presented in a table format.

RESULTS AND DISCUSSION

Survey result

A total of 39 pollen and/or nectar source plant species belonging to 23 families were identified during the survey work (Table 1). Bee floral plant species were classified as herbs, shrubs, and trees and wild and cultivated based on growth forms and source of bee plants. According to the survey results, trees (62.5%) were the most important source of bee forages, followed by herbs (25%) and shrubs (12.5%). The foremost sources of honeybee forages were wild 116 (82.3%) and cultivated 25 (17.7%). Honeybee plant species indicated by beekeepers during the survey were categorized as very good, good, and poor based on their abundance in the study area And most of the bee floral species identified through the survey were categorized as high in their abundance.

Coffea arabica, Croton macrostachyus, Vernonia spp., *Guizotia scabra, Eucalyptus camaldulensis, Cordia africana, Mangifera indica,* and *Combretum molle* were the most common honeybee plant species identified by beekeepers in different agro-ecosystems (Table 1). The dominant honeybee plant species in the highland were *Vernonia* spp., *Schefflera abyssinica, C. macrostachyus, C. arabica,* and *Bidens* spp., while the most frequently visited bee floral species in lowland ecology were *C. africana, G. scabra, C. molle, E. camaldulensis, Bidens* spp., and *C. arabica.* On the other hand, *G. scabra, Vernonia* spp., *C. arabica, C. macrostachyus,* and *Bidens* spp., were the most abundant floral plant species in midland agro-ecology based on survey results. *Vernonia* spp., *C. africana,* and *G. scabra* were the most abundant plant species in the highland, midland, and lowland, respectively. Frequently indicated bee floral species by beekeepers were *Vernonia* spp, *C. arabica, C. macrostachyus,* and *G. scabra* with 90 (100%), 83 (92.2%), 77 (85.6%) and 73(81%) rate, respectively. The most widely distributed bee flora species in all agro-ecosystem were *Vernonia* spp.. *S. abyssinica* and *C. molle*, two bee flora species, were only found in highland and lowland agro-ecosystem, respectively.

Honey yields were harvested twice a year by 68.9% of beekeepers and three times by 17.8% of beekeepers in different agro-ecologies of the study area. The average honey yields for highland, lowland, and midland were 25.3, 23.3, and 30.2 kg from frame hives, respectively. The major honey flow seasons in the study area across different agro-ecosystem are October to December, February to April, and May to June (Figure 2). The flowering period and duration of flowering time of honeybee plants differ significantly across the study area's agro-ecosystems ($p<$ 0.05). The major flowering months of the study area are September, October, February, March, April, May and June (Table 1). On the other hand, according to beekeepers who participated in the survey, the study area's dearth periods were August, July, and January. The maximum and minimum flowering duration of bee plant species were ninety and seven days, respectively. *C. macrostachyus*, *Vernonia* spp., *E. camaldulensis*, *G. scabra*, and *Trifolium* spp. had the longest flowering periods and offered a steady supply of nectar and pollen to honeybees on the hunt.

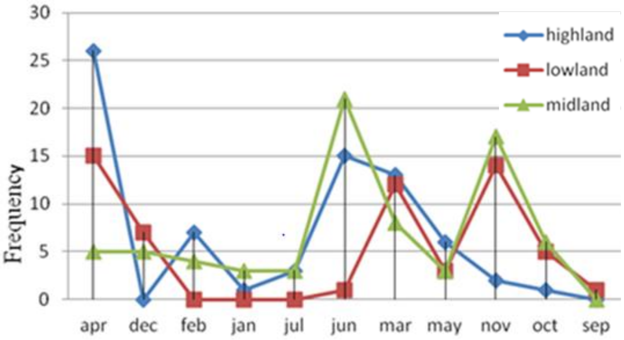

Figure 2. Honey flow month in different agro-ecosystems

Table 1. Major honeybee plants identified by beekeepers

Local name	Scientific name	Family name	Species abundance	Life forms	Food source	Flowering month	Duration (days)
Buna	*Coffea arabica*	Rubiaceae	2	Shrub	Pol/Nec	Feb, Mar	60
Bisana	*Croton macrostachyus*	Euphorbiaceae	2	Tree	Pol/nec	Jun, May, Aug, Apr	90
Girawa	*Vernonia* spp.	Asteraceae	1	Tree	Nectar	Dec-Mar	90
Tufo	*Guizotia scabra*	Asteraceae	2	herb	Pol/nec	Aug-Dec	90
Bahrzaf	*Eucalyptus camaldulensis*	Myrtaceae	3	Tree	Pol/nec	Year round	90
Wanza	*Cordia africana*	Boraginaceae	2	Tree	Poll/nec	Aug, Jul, Sep	60
Abalo	*Brucea antidysenterica*	Simaroubaceae	3	Tree	Pol/Nec	Apr,Mar	15
Avocado	*Persea americana*	Lauraceae	2	Tree	Pollen	Sep, Oct, Jan, Feb, Apr	90
Mango	*Mangifera indica*	Anacardiaceae	3	Tree	Pol/Nec	Mar, Dec, Feb	60
Tensa	*Combretum molle*	Combretaceae	1	Tree	Nectar	Mar, Apr, Feb	60
Boqolo	*Zea mays*	Poaceae	2	herb	Pollen	Jul, Jun, May	60
Adeye ababa	*Bidens* spp.	Asteraceae	1	herb	Pollen	Sep, Oct, Nov, Dec	60
Girar	*Acacia* spp.	Fabaceae	2	Tree	Pollen/nectar	Apr, May, Dec, Jan, Feb	60
Geteme	*Schefflera abyssinica*	Araliaceae	1	Tree	Pollen/nectar	Apr, Mar, May	60
Kerero	*Aningeria altissima*	Sapotaceae	2	Tree	Nectar	Apr, Jun, May, Jul	60
Turba abeba	*Brugmansia suaveolens*	Solanaceae	3	Shrub	Pollen/nectar	Almost year round	60
Siddessa	*Trifolium* spp.	Fabaceae	3	Herb	Pollen/nectar	Sep, Oct, Jan, Feb, Mar	90
Rejii	*Vernonia rueppellii*	Asteraceae	2	Shrub	Pollen/ nectar	Dec, Jan, Feb, Mar	60
Sesbania	*Esbania sesban*	Fabaceae	2	Shrub	Pollen	Year round	15
Sesa	*Albizia gummifera*	Fabaceae	2	Tree	Pollen /nectar	Feb, Mar	30
Wandabiyo	*Apodytes dimidiate*	Icacinaceae	2	Tree	Pollen/nectar	Feb, Oct, Mar	30
Bayya	*Olea welwitschi*	Oleaceae	2	Tree	Pollen/nectar	Dec,Feb, Jan	60
Keryo	*Polyscias fulva*	Araliaceae	2	Tree	Pollen/nectar	Apr, Mar, May, Jun	30
Mashila	*Sorghum bicolor*	Poaceae	2	Herb	Pollen	Sep, Oct, Mar	30
Kenchib	*Euphorbia tirucalli*	Euphorbiaceae	3	Herb	Nectar	Sep, Oct, Mar	60
Nuge	*Guizotia abyssinica*	Asteraceae	3	Herb	Pollen/Nectar	Sep, Oct	30
Sio	*Rhus* spp.	Anacardiaceae	2	Tree	Pollen/nectar	Aug, Jul, Sep, Oct	60
Sombo	*Ekebergia capensis (E. rueppeliana*	Maliaceae	3	Tree	Pollen /nectar	Dec, Jan, Mar, Oct, Nov	30
Zytune	*Psidium guajava*	Myrtaceae	3	Tree	Pollen/nectar	Jan	7
Giravilla	*Grevillea robusta*	Proteaceae	2	Tree	Pollen /nectar	Mar	20
Ruze	*Oryza sativa*	Poaceae	3	Herb	Pollen	Sep, Oct	30
Sesame	*Sesamum indicum*	Pedaliaceae	2	Herb	Pollen /nectar	Mar,May, Jun,Ssep	60
Sole	*Olinia rochetiana*	Penaeaceae	2	Tree	Pollen/nectar	Sep, Mar	60
Bedesa	*Syzygium guineense*	Myrtaceae	2	Tree	Pollen /Nectar	Jan, Feb, Mar, Sep, Aug	
Maget	*Trifolium* spp.	Papilionaceae	2	Tree	Nectar	Mar	
Korch	*Erythrina abyssinica*	Fabaceae	3	Shrub	Pollen /nectar	Jan, Mar	30
Zembaba	*Phoenix reclinata*	Arecaceae	3	Tree	Pollen	Mar	15
Derbata	*Terminalia laxiflora*	Combretaceae	3	Tree	Nectar	Sep, Mar	30
Seho	*Allophylus abyssinicus*	Sapindaceae	3	Tree	Pollen/nectar	Aug	60

Poisonous honeybee plant species were also identified in the study areas, along with their flowering times and durations. About 65 (77.2%) of respondents were aware of the presence of poisonous plants for honeybees. The remaining 25 (27.8%) had no awareness of the availability of poisonous honeybee plants in their surrounding areas. Honeybee poisonous plants found in the study area were key abeba, tikur enchet, and nime tree. *Euphorbia cotinifolia* is a shrub that belongs to the family Euphorbiaceae, which is the most frequently, identified poisonous plant species. *E. cotinifolia* plant species mainly found in highland and midland agro-ecosystems. The major flowering months of *E. cotinifolia* species are September to November, February to April, and May to June in the study areas. The maximum and minimum flowering durations of plants were 90 days and 7 days, respectively.

Bee pollen analysis

Twenty-four honey bee plant species belonging to ten families were identified from a total of sixty-six (66) pollen samples collected in different districts (Table 2). *Guizotia abyssinica, Vernonia* spp., *C. arabica*, and *Eucalyptus* spp. were the major pollen-source honeybee plant species identified in the study areas (Figure 3). On the other hand, *Bersama abyssinica, Olea africana, Syzygium guineense* and *Syzyjium* spp. were the minor pollen sources of honey bee plant species, as the present findings indicated. The current study found that the highest proportion of pollen grains was collected in October (46.3%), November (14.6%), February (12.2%), and December (11.0%). The lowest pollen grains were collected in July and August.

Honeybee flora species abundance and density

Ninety eight honeybee plant species belonging to 47 families were identified from 60 main plots and subplots (Table 3). These honeybee plant species were classified as herbs, shrubs, and trees depending on growth forms of plants. Herbs were the most frequently visited plant growth form, accounting for 49 (50%) of all visits, followed by trees at 26 (26.5%) and shrubs at 23 (23.5%) in sample plots. The Fabaceae (31.9%), Asteraceae (19.1%) and Poaceae (14.8%) families had the most honey bee plant species encountered in quadrat samples.

The most common or top ten floral honeybee plant species in highland sample plots/quadrats were *Cynoglossum lanceolatum*, *C. arabica*, *Isoglosa* species, *Snowdenia polystachya*, *Pennisetum glaucum*, *Desmodium* species, *Tinospora cordifolia*, *Acanthus eminens*, *E. camaldulensis* and *Cyclamen purpurascens* (Table 4). The *C. lanceolatum*, *S. polystachya*, *Bidens* spp., *Sorghum bicolor*, *Isoglosa* spp., *Kalanchoe pinnata*, *Erythrina abyssinica*, *Euphorbia tirucalli*, *Vernonia auriculifera* and *Lippia adoensis* were the most frequent honeybee plant species in lowland plots, whereas *C. lanceolatum*, *E. camaldulensis*, *Colocasia esculenta*, *S. polystachya*, *K. pinnata*, *Erica* spp., *Psidium guajava*, *Arum maculatum* and *G. scabra* were the dominant honeybee plant species in midland sample plots (Table 4).

Diversity and composition of honeybee forages

A total of one hundred forty-one pollen and/or nectar-source bee plant species belonging to sixty-two families were discovered based on the survey, pollen load collection, and plant inventory data results. Herbs were the most dominant bee flora, accounting for 62 (44%) of a total of 141 honeybee plant species, followed by trees at 48 (34%) and shrubs at 31 (22%), respectively (Figure 4). The families with the highest number of species were Fabaceae 18(12.8%), Asteraceae 11 (7.8%), Poaceae 9 (6.4%), Solanaceae 6 (4.3%), Acanthaceae 4 (2.8%) and Euphorbaceae 4 (2.8%) in the study area (Figure 5). Among a total of one hundred forty-one honeybee plant species, one hundred fifteen (81.6%) were both sources of pollen and nectar, whereas fifteen (10.6%) were pollen sources and the remaining eleven (7.8%) were nectar source plant species.

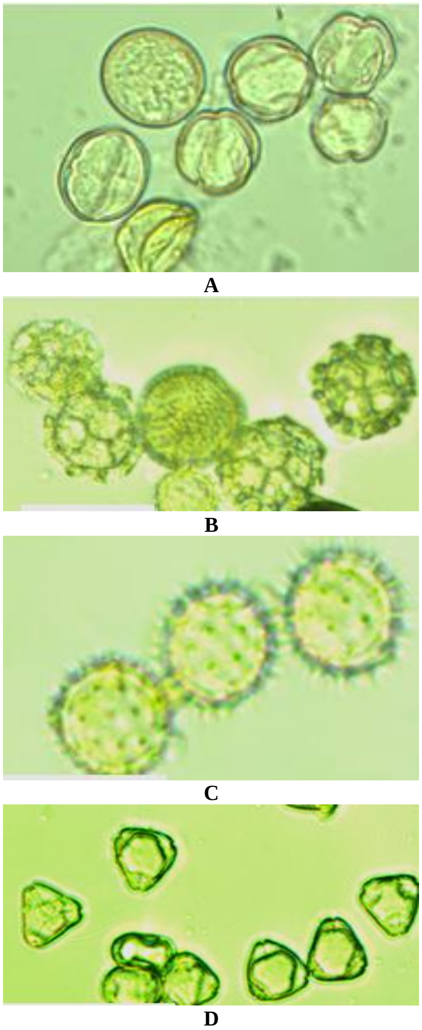

Figure 3. Major honeybee flora species identified through pollen analysis. A. *Coffea arabica*, B. *Vernonia* spp., C. *Guizotia* spp., D. *Eucalyptus* spp.

Table 2. Bee plant species identified from pollen analysis and harvesting period

Scientific/species name	Family name	Life forms	Source of food	Harvesting period
Guizotia abyssinica	Asteraceae	Herb	Pollen/nectar	Sep-Feb
Vernonia amygdalina	Asteraceae	Tree	Pollen/nectar	Jan-Feb, Apr
Coffea arabica	Rubiaceae	Shrub	Pollen/nectar	Jan-Apr, Oct-Nov
Eucalyptus spp.	Myrtaceae	Tree	Pollen/nectar	De-Jan, Oct-Nov
Bidens spp.	Asteraceae	Herb	Pollen/nectar	Oct-Jan
Trifoluma spp.	Fabaceae	Herb	Pollen/nectar	Oct, Dec-Jan
Parkinsonia aculeata	Fabaceae	Tree	Pollen/nectar	Oct, Nov
Rubus spp.	Rosaceae	Herb	Pollen/nectar	Oct, Dec-Jan
Schefflera abyssinica	Araliaceae	Herb	Pollen/nectar	Oct, Jan
Grass spp.	not id	Herb	Pollen/nectar	Oct
Plantago lanceolata	Plantaginaceae	Herb	Pollen	Mar
Olea africana	Oleaceae	Tree	Pollen/nectar	Feb
Bersama abyssinica	Francoaceae	Tree	Pollen/nectar	Oct
Brassica spp.	Brassicaceae	Herb	Pollen/nectar	Oct
Caesalpinia sp.	Fabaceae	Herb	Pollen/nectar	Dec
Caesalpinia decapitala	Fabaceae	Shrub	Nectar	Dec
Combretum molle	Combretaceae	Tree	Nectar	Oct
Datura arborea	Solanaceae	Shrub	Pollen	Dec
Syzygium guineense (Willd.) DC	Myrtaceae	Tree	Pollen/nectar	Oct
Echoriopis spp.	Cactaceae	-	Pollen	Dec
Ejursaw spp.	not id	-	Pollen	Feb
Rubuytmaeso lanceolata	not id	-	Pollen	Mar
Syzygium spp.	Myrtacea	Tree	Pollen	Oct

Table 3. Honeybee plant species density and frequency in sample quadrats

Local name	Scientific name	Family	Plant type	Plant count	Plant density/ha	Plot obs.
Ambebesa/sesa	*Albizia gummifera*	Fabaceae	Tree	44	267	9
Arenchi	*Pavonia urens*	Malvaceae	Herb	89	8663	8
Bahrzaf	*Eucalyptus camaldulensis*	Myrtaceae	Tree	704	4689	8
Banana	*Musa acuminata*	Musaceae	Tree	82	2050	3
Bisana	*Croton macrostachyus*	Euphorbiaceae	Tree	34	174	12
Buna	*Coffea arabica*	Rubiaceae	Shrub	1739	4329	30
Castor/gulo	*Ricinus communis*	Euphorbiaceae	Shrub	47	279	10
Demekese	*Ocimum lamiifolium* Hochst	Labiatae	Herb	16	4782	4
Dergu	*Isoglosa* species	Acanthaceae	Herb	464	69074	50
Emo	*Colocasia esculenta*	Araceae	Herb	352	1996	10
Girawa	*Vernonia* spp.	Asteraceae	Tree	77	261	22
Girnche/chifrig	*Sida schimperiana*	Malvaceae	Herb	222	2500	19
Kello/adey abeba	*Bidens* spp.	Asteraceae	Herb	497	37499	30
Metene	*Cynoglossum lanceolatum*	Boraginaceae	Herb	2514	68281	110
Muja	*Snowdenia polystachya*	Poaceae	Herb	944	141250	31
Rejii	*Vernonia auriculifera*	Asteraceae	Shrub	133	762	13
Susbania	*Esbania sesban*	Fabaceae	Shrub	72	679	9
Tufo	*Guizotia scabra*	Asteraceae	Herb	60	67944	25
Ulmaye/limich	*Clausena anisata*	Rutaceae	Shrub	22	244	7
Abayi/qalawa	*Maesa lanceolata*	Myrsinaceae	Tree	14	200	4
Arebe duberti	*Carduus schimperi*	Asteraceae	Herb	58	42916	7
Birbira	*Millettia ferruginea*	Fabaceae	Tree	7	113	3
Bosoke	*Kalanchoe* sp.	Crassulaceae	Herb	95	52000	9
Chat	*Kalanchoe pinnata*	Crassulaceae	Shrub	320	4001	4
Cheda dima	*Euphorbia tirucalli*	Euphorbiaceae	Shrub	142	1354	5
Desmodium	*Desmodium* species	Fabaceae	Herb	209	29583	20
Endod	*Phytolacca dodecandra*	Phytolaccaceae	Herb	29	46250	5
Gomenzer	*Brassica carinata*	Brassicaceae	Herb	15	37500	2
Haallaal	*Urera hypselodenron*	Urticaceae	Shrub	44	83750	4
Kontir	*Caesalpinia decapetala*	Fabaceae	Shrub	65	1116	4
Korch	*Erythrina abyssinica*	Fabaceae	Tree	143	1131	6
Mango	*Mangifera indica*	Anacardiaceae	Tree	2	50	2
Qortobi	*Plantago lanceolata*	Plantaginaceae	Herb	42	31875	6
Sanaa maki	*Senna didymobotrya*	Fabaceae	Shrub	8	200	2

Sindedo	*Pennisetum thunbergii*	Poaceae	Herb	18	45000	2
Sokoro	*Acanthus eminens*	Acanthaceae	Herb	97	681	5
Ulaga	*Ehretia cymosa*	Boraginaceae	Tree	5	58	4
Wanza	*Cordia africana*	Boraginaceae	Tree	17	148	9
Zeytuna	*Psidium guajava*	Myrtaceae	Tree	145	925	7
Adenguare	*Phaseolus vulgaris*	Fabaceae	Herb	3	7500	1
Agam	*Carissa spinarum*	Apocynaceae	Shrub	4	100	1
Alenge	*Arum maculatum*	Araliaceae	Herb	121	302500	1
Allala	*Allamanda* spp.	Apocynaceae.	Herb	13	16250	2
Ananno	*Periploca linearifolia*	Asclepiadaceae	Shrub	30	375	2
Apple	*Malus pumila*	Rosaceae	Shrub	2	50	1
Asangira	*Datura stramonium*	Solanaceae	Herb	9	11250	2
Askira	*Millettia ferruginea*	Fabaceae	Tree	15	63	6
Avocado	*Persea americana*	Lauraceae	Tree	5	25	5
Baddessa/Dokima	*Syzygium guineense*	Myrtaceae	Tree	2	50	1
Besobila /kefo	*Salvia nilotica/Ocimum basilicum*	Lamiaceae	Herb	6	15000	1
Boqqo	*Bersama abyssinica*	Melianthaceae	Tree	1	25	1
Bosoka	*Eriobotrya japonica*	Rosaceae	Tree	1	25	1
Butte	*Ammocharis tinneana*	Amaryllidaceae	Herb	8	10000	2
Cassava	*Manihot esculenta*	Euphorbiaceae	Shrub	124	1550	2
Cheka	*Calpurnia aurea*	Fabaceae	Tree	4	100	1
Chibo	*Vernonia leopoldi*	Asteraceae	Shrub	18	450	1
Damisa	*Centella asiatica*	Apiaceae	Herb	7	17500	1
Dhumuga	*Justica schimperiana*	Acanthaceae	Shrub	16	400	1
Diniche	*Solanum tuberosum*	Solanaceae	Herb	20	16666	3
Dobbi/sama	*Urtica simensis*	Urticaceae	Herb	34	42 500	2
Enselal	*Foeniculum vulgare*	Apiaceae	Herb	2	5000	1
Enset	*Ensete ventricosum*	Musaceae	Shrub	33	413	2
Girar	*Acacia* spp.	Fabaceae	Tree	3	75	1
Gomera	*Capparis tomentosa*	Capparidaceae	Shrub	1	25	1
Guriyo	*Tinospora cordifolia*	Menispermaceae	Herb	141	88125	4
Harbu/shola	*Ficus sur*	Moraceae	Tree	2	50	1
Hidda bofa	*Momordica foetida*	Cucurbitaceae	Herb	1	25	1
Hiddaa lafaa	*Dregea schimperi*	Asclepiadaceae	Herb	8	6666	3
Hiddi	*Solanum incanum*	Solanaceae	Shrub	5	4166	3
Jajjab	*Setaria megaphylla*	Poaceae	Herb	26	32500	2
Karaba	*Sida rhombifolia*	Malvaceae	Herb	26	7222	9
Kase	*Lippia adoensis*	Verbenacae	Herb	86	107,500	2
Kishkishe	*Senna septemtrionalis*	Fabaceae	Herb	7	17500	1
Kunche	*Chenopodium album*	Amaranthaceae	Herb	21	17500	1
Kusaye	*Lantana trifolia*	Verbenaceae	Shrub	2	50	1
Lochisa	*Bersama abyssinica*	Melianthaceae	Herb	35	875	1
Mixoo/dido/didu	*Galiniera saxifrage*	Rubiaceae	Shrub	2	50	1
Mulberry	*Morus alba*	Moraceae	Shrub	3	75	1
Nanaye	*Pennisetum glaucum*	Poaceae	Herb	196	61 250	8
Pepper /berberi	*Capsicum annuum*	Solanaceae	Herb	18	4500	1
Qalawa/qaawaa	*Grewia mollis*	Tiliaceae	Tree	1	25	1
Qumudu	*Nymphoides indica*	Menyanthaceae	Herb	41	34166	2
Raafu	*Kleinia grantii*	Asteraceae	Herb	17	21250	3
ret/alovera	*Aloe debrana*	Xanthorrhoeaceae	Herb	17	42500	1
Rhodus	*Chloris gayana*	Poaceae	Herb	1	2500	1
Shajara	*Cyclamen purpurascens*	Primulaceae	Herb	80	33333	6
Shenkora	*Saccharum officinarum*	Poaceae	Shrub	6	150	1
Shultee	*Rumex nepalensis*	Polygonaceae	Herb	6	15000	1
Siddisa/wazma	*Trifolium rueppellianum*	Fabaceae	Herb	39	97500	1
Siglu	*Fagaropsis angolensis*	Rutaceae	Tree	6	150	1
Sorghum	*Sorghum bicolor*	Poaceae	Herb	135	1688	2
Suufi/suff	*Carthamus tinctorius*	Asteraceae	Herb	56	46666	3
Togo	*Dieliptera* spp.	Acanthaceae	Tree	10	25000	1
Tsid	*Juniperus procera*	Cupressaceae	Tree	40	1000	1
Uregessa	*Clausena anisata*	Rutaceae	Tree	11	138	2
Vetch	*Vicia sativa*	Fabaceae	Herb	6	15000	1
Welensu	*Erythrina brucei*	Fabaceae	Tree	41	513	2
Yeriwo garo	*Solanecio* spp.	Asteraceae	Herb	1	25	1

Table 4. Honeybee plant species density and their frequency of occurence in ecosystems

Scientific name	**Family name**	**Highland**			**Lowland**			**Midland**			
		Plant count	**Plant den./ha**	**Plot obs.**	**Plant count**	**Plant den./ha**	**Plot obs.**	**Plant count**	**Plant dens./ha**	**Plot obs.**	**Type of plants**
Vernonia spp.	Asteraceae	48	100	12	10	42	6	19	119	4	Tree
Coffea arabica	Rubiaceae	425	1181	9	604	1373	11	710	1775	10	Tree
Croton macrostachyus	Euphorbiaceae	20	84	6	1	25	1	13	65	5	Tree
Vernonia rueppellii	Asteraceae	27	135	5	92		5	14	167	3	Shrub
Albizia gummifera	Fabaceae	8	67	3	1	25	1	35	175	5	Tree
Eucalyptus camaldulensis	Myrtaceae	112	933	3	3	75	1	589	3681	4	Tree
Malus pumila	Rosaceae	2	50	1	-	-	-	-	-	-	Shrub
Acacia spp.	Fabaceae	3	75	1	-	-	-	-	-	-	Tree
Acanthus eminens	Acanthaceae	93	581	4	-	-	-	4	100	1	Herb
Allamanda spp.	Apocynaceae.	-	-	-	13	16250	2	-	-	-	Herb
Aloe debrana	Xanthorrhoeaceae	-	-	-	-	-	-	17	42500	1	Herb
Ammocharis tinneana	Amaryllidaceae	8	10000	2	-	-	-	-	-	-	Herb
Arum maculatum	Araliaceae	-	-	-	-	-	-	121	30250	1	Herb
Bersama abyssinica	Melianthaceae		36	900	2	-	-	35	875	1	Tree
Bidens spp.	Asteraceae	32	13333	6	407	56 527	18	58	24166	6	Herb
Brassica carinata	Brassicaceae	11	27500	1	-	-	-	4	10000	1	Herb
Caesalpinia decapetala	Fabaceae	23	192	3	-	-	-	42	924	1	Shrub
Calpurnia aurea	Fabaceae	-	-	-	-	-	-	4	100	1	Tree
Capparis tomentosa	Capparidaceae	-	-	-	1	25	1	-	-	-	Shrub
Capsicum annuum	Solanaceae	-	-	-	-	-	-	18	4500	1	Herb
Carduus schimperi	Asteraceae	26	16250	4	-	-	-	32	26666	3	Herb
Carissa spinarum	Apocynaceae	-	-	-	-	-	-	4	100	1	Shrub
Carthamus tinctorius	Asteraceae	-	-	-	56	46666	3	-	-	-	Herb
Centella asiatica	Apiaceae	7	17500	1	-	-	-	-	-	-	Herb
Chenopodium album	Amaranthaceae	21	17500	3	-	-	-	-	-	-	Herb
Chloris gayana	Poaceae	1	2500	1	-	-	-	-	-	-	Herb
Clausena anisata	Rutaceae	2	50	1	9	56	4	22	276	4	Shrub
Colocasia esculenta	Araceae	-	-	-	54	675	2	286	1021	7	Herb
Cordia africana	Boraginaceae	-	-	-	8	67	3	5	31	4	Tree
Cyclamen purpurascens	Primulaceae	80	33333	6	-	-	-	-	-	-	Herb
Cynoglossum lanceolatum	Boraginaceae	997	51, 927	48	643	53 583	30	874	68281	32	Herb
Datura stramonium	Solanaceae	-	-	-	9	11250	2	-	-	-	Herb
Desmodium species	Fabaceae	142	29583	12	-	-	-	104	21862	9	Herb
Dieliptera spp.	Acanthaceae	-	-	-	10	25000	1	-	-	-	Tree
Dracaena afromontana	Dracaenaceae	12	300	1	-	-	-	-	-	-	Tree
Dregea schimperi	Asclepiadaceae	8	6666	3	-	-	-	-	-	-	Herb
Ensete ventricosum	Musaceae	-	-	-	-	-	-	33	413	2	Shrub
Ehretia cymosa	Boraginaceae	4	33	3	1	25	1	-	-	-	Tree

Eluesine folicofolia	Poaceae	4	10000	1							Herb
Erica spp.	Ericaceae	1	2500	1	-	-	-	166	69166	6	Herb
Eriobotrya japonica	Rosaceae	-	-	-	1	25	1	-	-	-	Tree
Erythrina abyssinica	Fabaceae	38	475	2	105	656	4	-	-	-	Tree
Erythrina brucei	Fabaceae	-	-	-	-	-	-	41	513	2	Tree
Sesbania sesban	Fabaceae	28	116	6	1	25	1	43	538	2	Shrub
Euphorbia tirucalli	Euphorbiaceae	41	512	2	101	842	3	-	-	-	Shrub
Manihot esculenta	Euphorbiaceae	-	-	-	-	-	-	124	1550	2	Shrub
Fagaropsis angolensis	Rutaceae	-	-	-	-	-		6	150	1	Tree
Ficus sur	Moraceae	2	50	1	-	-	-	-	-	-	Tree
Foeniculum vulgare /Anethum graveolens	Apiaceae	-	-	-	-	-	-	2	5000	1	Herb
Galiniera saxifrage	Rubiaceae	2	50	1	-	-	-	-	-	-	Shrub
Grewia mollis	Tiliaceae	1	25	1	-	-	-	-	-	-	Tree
Guizotia scabra	Asteraceae	4	10000	1	56	28000	5	10 2	29944	19	Herb
Isoglosa species	Acanthaceae	266	24629	27	120	30000	10	78	15000	13	Herb
Juniperus procera	Cupressaceae	-	-	-	-	-	-	40	1000	1	Tree
Justica schimperiana	Acanthaceae	-	-	-	-	-	-	16	400	1	Shrub
Kalanchoe pinnata	Crassulaceae	-	-	-	109	1363	2	211	2638	2	Shrub
Kalanchoe sp.	Crassulaceae	54	27000	5	41	25000	4	-	-	-	Herb
Kleinia grantii	Asteraceae	-	-	-	-	-	-	17	21250	2	Herb
Lantana trifolia	Verbenaceae	-	-	-	-	-	-	2	50	1	Shrub
Lippia adoensis	Verbenacae	-	-	-	86	107500	2	-	-	-	Herb
Maesa lanceolata	Myrsinaceae	9	75	3	-	-	-	5	125	1	Tree
Mangifera indica	Anacardiaceae	-	-	-	1	25	1	1	25	1	Tree
Millettia ferruginea	Fabaceae	5	63	2	15	63	6	2	50	1	Tree
Momordica foetida	Cucurbitaceae	-	-	-	1	25	1	-	-	-	Tree
Morus alba	Moraceae	-	-	-	-	-	-	3	75	1	Shrub
Musa acuminata	Musaceae	32	800	1	13	325	1	-	-	-	Tree
Nymphoides indica	Menyanthaceae	-	-	-	-	-	-	41	34166	3	Herb
Ocimum lamiifolium	Labiatae	-	-	-	12	11325	3	4	100	1	Herb
Pavonia urens	Malvaceae	41	5625	4	1	2500	1	47	538	3	Herb
Pennisetum glaucum	Poaceae	196	61 250	8	-	-	-	-	-	-	Herb
Pennisetum thunbergii	Poaceae	-	-	-	-	-	-	14	35000	1	Herb
Periploca linearifolia	Asclepiadaceae	-	-	-	-	-	-	30	37875	2	Shrub
Persea americana	Lauraceae	-	-	-	-	-	-	5	25	5	Tree
Phaseolus vulgaris	Fabaceae	-	-	-	-	-	-	3	7500	1	Herb
Phytolacca dodecandra	Phytolaccaceae	15	37500	1	14	8750	4	-	-	-	Herb
Plantago lanceolata	Plantaginaceae	-	-	-	9	11250	2	33	20625	4	Herb
Psidium guajava	Myrtaceae	-	-	-	9	75	3	136	850	4	Tree
Ricinus communis	Euphorbiaceae	4	50	2	6	75	2	37	154	6	Shrub
Rumex nepalensis	Polygonaceae	6	15000	1	-	-	-	-	-	-	Herb
Saccharum officinarum	Poaceae	-	-	-	-	-	-	6	150	1	Shrub
Salvia nilotica/Ocimum basilicum	Lamiaceae	-	-	-	-	-		6	15000	1	Herb
Senna septemtrionalis	Fabaceae	-	-	-	7	17500	1	-	-	-	Herb
Senna didymobotrya	Fabaceae	6	150	1	2	50	1	-	-	-	Shrub

Setaria megaphylla	Poaceae	-	-	-	26	32500	2	-	-	-	Herb
Sida schimperiana	Malvaceae	-	-	-	55	11 458	12	-	-	-	Herb
Sida rhombifolia	Malvaceae	26	7222	9	-	-	-	-	-	-	Herb
Snowdenia polystachya	Poaceae	200	5000	10	526	77 352	17	218	136250	4	Herb
Solanecio sp.	Asteraceae	-	-	-	-	-	-	1	25	1	Herb
Solanum incanum	Solanaceae	-	-	-	5	4166	3	-	-	-	Herb
Solanum tuberosum	Solanaceae	-	-	-	-	-	-	20	16666	3	Herb
Sorghum bicolor	Poaceae	-	-	-	135	1688	2	-	-	-	Herb
Syzygium guineense	Myrtaceae	-	-	-	-	-	-	2	50	1	Tree
Tinospora cordifolia	Menispermaceae	141	88125	4	-	-	-	-	-	-	Herb
Trifolium rueppellianum	Fabaceae	-	-	-	39	97500	1	-	-	-	Herb
Urera hypselodenron	Urticaceae	1	25	1	11	138	2		-	-	Shrub
Urtica simensis	Urticaceae	-	-	-	-	-	-	34	42500	2	Herb
Vernonia auriculifera	Asteraceae	-	-	-	92	460	5	-	-	-	Shrub
Vernonia leopoldi	Asteraceae	18	450	1	-	-	-	-	-	-	Shrub
Vicia sativa	Fabaceae	6	15000	1	-	-	-	-	-	-	Herb

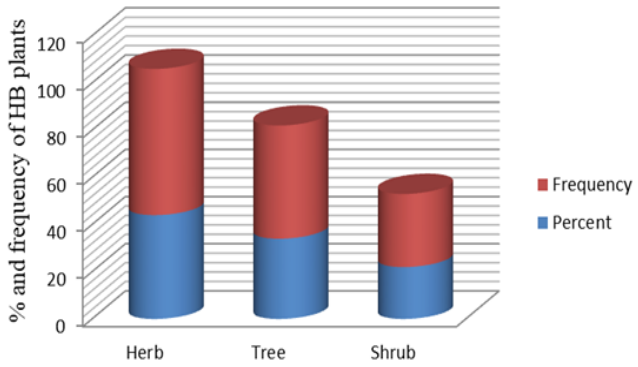

Figure 4. Growth forms of bee plant species

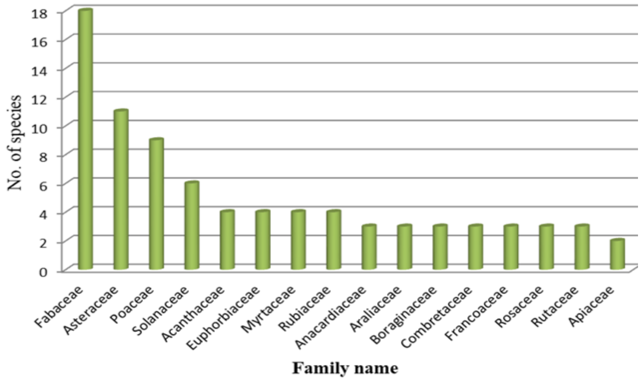

Figure 5. Number of species within each family

Table 5. Shannon diversity indices of honeybee plant species in ecosystems

Agro-ecosystem	Richness	Shannon	H'max (lns)	Shannon Evenness =H'/lnS
Highland	50	2.7	3.9	0.6
Lowland	47	2.6	3.8	0.6
Midland	59	3	4.1	0.7

Note: S: Number of species, H'max: Maximum diversity, H': Shannon index

Species diversity, richness and evenness of honeybee plant species

The Shannon diversity index analysis revealed that the midland agro-ecosystem had the most species diversity in sample plots compared to the highland and lowland agro-ecosystem. The species diversity in highland and lowland ecosystems was the same. The species richness varied by ecosystem. Midland ecosystem had comparatively the most species (59 species in 34 families), followed by highland (50 species in 29 families) and lowland ecosystem (47 species in 26 families) (Table 5). In the study areas, the midland ecosystem had the highest evenness, followed by the highland and lowland ecosystem. The Shannon diversity index of the study area was found to be 2.8, which fell in the range between 1.5 and 3.5 (Kent and Coker 1992). This indicated the area possessed a good plant density. The Shannon evenness value (0.6) showed that the honeybee plant species counted in the quadrats were evenly distributed in the sample plots and sites.

Discussion

The species of bee plants indicated by beekeepers through survey were more or less comparable to those found by plant inventory and pollen analysis. This has demonstrated that beekeepers' indigenous knowledge is significant for bee plant inventory results. *Vernonia* spp., *C. africana*, and *G. scabra* were the most abundant plant species in the highland, midland, and lowland, respectively. The most frequently indicated bee floral species by beekeeprers were *Vernonia* spp, *C. arabica*, *C. macrostachyus*, and *G. scabra* with 90 (100%), 83 (92.2%), 77 (85.6%) and 73 (81%) rate, respectively. The highest bee floral plant species has been known as the best indicators of adaptation to the area and climatic condition (Wubie et al. 2014). However, no single bee floral plant species has been identified by beekeeper respondents in the midland. This demonstrated that the midland has an overlapping agro-ecosystem in terms of bee flora plant species or vegetation distribution. According to the survey results, trees (62.5%) were the most important source of bee forages, followed by herbs (25%) and shrubs (12.5%). This finding is consistent with the findings of Kebede and Gebrechirstos (2016) and Haftom et al. (2013) in Tigray, who found trees to be a major source of feed for honeybees. The current survey findings, however, contradict the findings of Teklay (2011), who reported that herbs were the most common floral plant species. This finding variation might link with the changes in geographical location, soil type and climatic situation.

Honey yields were mainly harvested twice a year by 68.9% of beekeepers in different agro-ecosystems of the study area. This result is consistent with the study done by Shegaw and Giorgis (2021), which found that there were two main harvesting seasons. Honeybee plants are present at different periods of the year because plant flowering times vary depending on species, topography, climate, and farming practices (Rijal et al. 2018). The major flowering months of the study area were September, October, February, March, April, May and June. Bareke and Addi (2019) and Zeleke et al. (2019) conducted comparable studies in the Gera forests and selected parts of South Nations Nationalities and Peoples of Ethiopia, respectively. On the other hand, according to beekeepers who participated in the survey, the study area's dearth periods were August, July, and January. Another scholar, Shegaw and Giorgis (2021) carried out an analogous study in selected areas of the South Nations Nationalities and Peoples of Ethiopia. A drought period can cause the depletion of stored food inside the hive, which has a negative impact on honeybee productivity. Therefore, beekeepers should know the dynamics of honeybee colonies in accordance with bee floras, flowering periods, and duration of flowering times in different agro-ecosystem. Almost all beekeepers in the study area were familiar with the honeybee colony dynamics conditions. This finding agrees with the results of the study conducted by Fichtl and Admassu (1994), Lemessa (2006) and Teklay (2011). Beekeepers identified bee flora depending on the intensity of flowers visited by honeybees. The knowledge gained in identifying bee flora assists beekeepers in recognizing the honey harvesting season and managing the beehives. The identification of flora calendar assists beekeepers in planning various beekeeping activities (Genet 2002). In fact, not all honey bee plants are equally important in the lives and honey production of different bee species. The most frequently identified poisonous plant species in the study area, according to current findings, was *E. cotinifolia* (key abeba). In the Kaffa zone of southwest Ethiopia, analogous findings were reported by Addi (2018). *E. cotinifolia* is a shrub that belongs to the family Euphorbiaceae, which bears flowers at different months of the year. This plant is easy to adapt and propagate by cutting, and it also acts as a living fence in the study area.

The highest pollen grains were collected in October and November because the majority of plant species bloom following the long rainy season (June to August). The lowest pollen loads, on the other hand, were recorded in July and August because rain impairs honeybees' ability to fly, which in turn lowers their ability to collect pollen. Low temperatures may also impede the growth and flowering of bee plant species, which would reduce pollen production and nectar secretion. The findings are consistent with those of studies carried out in the Kaffa zone, southeast Oromia zone, and central Ethiopia by Lemessa and Addi (2009) and Bareke and Addi (2020), respectively. Contrary to the current findings, Wubie et al. (2014) reported that most pollen grains were collected during the main rainy season. This might occur since the flowering period differs with different agro-ecosystem areas.

In general, a total of one hundred forty-one pollen and/or nectar source honeybee plant species belonging to sixty-two families were discovered based on the survey, pollen load collection and plant inventory data results. Herbs, trees, shrubs, and grass were among the plant growth forms that honeybees use as forages. Herbs were the most dominant bee flora, accounting for 62 (44%) of a total of 141 honeybee plant species, followed by trees at 48 (34%) and shrubs at 31 (22%), respectively. This finding is consistent with previous findings, as herbs are the most dominant bee flora plants in different parts of Ethiopia (Teklay 2011; Abebe and Temam 2016; Addi 2018; Bareke and Addi 2019; Bareke and Addi 2020). The predominant of herbs are due to disturbance and existence of gaps in the forest (Bareke and Addi 2019). The Fabaceae and Asteraceae families have the highest number of species, with eighteen and eleven species, respectively. The study carried out in the Gera forests also revealed that the Fabaceae family had the dominant species composition, followed by Asteracea, which is consistent with current findings (Mulugeta et al. 2015). This study was focused on the overall floristic composition of Gera forest rather than identifying specific species of honeybee flora.The present findings, on the other hand, contradict previous reports, as Asteraceae family has the highest species composition in Kaffa Zone and Gera forests (Addi 2018; Bareke and Addi 2019; Bareke and Addi 2020). Not all Fabaceae species are plants that attract honeybees. As a result, it is not a dominating honeybee plant family in different study sites. However, the Asteraceae family is the most common bee foraging family in many forest areas (Bareke and Addi 2020). The Asteraceae family's dominance may be ascribed to the ability of certain of its species to produce honey (Bareke and Addi 2019). The main sources of honeybee forages were wild 116 (82.3%) and cultivated 25 (17.7%). These findings indicated that majority of bee floral plant species were found in wild sources, since beekeepers had no practices to cultivate bee floral plant species. The best predictor of adaptation to the area and local conditions is thought to be the highest frequency of bee plant species. Due to their climate preferences for growth, Boraginaceae, Rubiaceae, Poaceae, and Myrtaceae were the most prevalent families in sample plots. Herbaceous plant species had a greater density value of plant species per plot than did trees and shrubs. This result is consistent with Wubie et al. (2014) findings. This is a result of lower seed sizes taking up a significant portion of the plots. The trees and shrubs density was lower due to desertification.

The majority of honeybee plant species were considered as the basic sources of both pollen and nectar in the study area. Forage sources (pollen/nectar) were confirmed with published and pollen specimen accounts. The present study revealed that bee plant species were the main source of both pollen and nectar, rather than a single source of nectar or pollen. The findings also demonstrated that species of pollen-producing plants are more numerous than nectar-producing ones. This finding is aligned with those reported by Bareke and Addi (2020). Nectar and pollen are used for honey production and colony multiplication, respectively. Not all honeybee plants are similarly significant to bees and honey production. Only 16% of flowering plants are the origins of the majority of the honey in the world (Crane 1990). This shows that there are only a handful of significant honey source plants in each geographical area.

The analysis of bee forage diversity, richness, and evenness was estimated in different agro-ecosystem systems using the Shannon-Wiener diversity index. The midland agro-ecosytem moderately had the most species diversity, richness, and evenness compared to the highland and lowland agro-ecosystems. These findings, however, contradicted the findings of Wubie et al. (2014), who indicated that highland agro-ecology had more species diversity and richness than midland and lowland ecological systems. This variation could be attributed to differences in the geographical location, soil type, and climatic conditions of the study areas. Nevertheless, this doesn't mean that areas with a higher quantity of plant diversity are good for honey production, since the productivity of the beekeeping sector is reliant on the abundance and density of plants. The Shannon diversity index and evenness were found to be 2.8 and 0.6, respectively. This finding indicated that the study area possessed good plant density evenly distributed in the sample plots and sites. The higher the evenness and Shannon index values, the more even the species and diversity in the ecosystem or plots. The current finding further supported the notion that the species diversity and evenness in sample plots fell within acceptable bounds.

According to current findings, the study area has a diverse range of floral species, which may aid in the production of honey for national and international markets. A total of one hundred forty-one pollen and/or nectar source honeybee plant species belonging to sixty-two families were identified in the study area. The Fabaceae and Asteraceae families have the highest number of species. Herbaceous plant species had a greater density value of plant species per plot than did trees and shrubs. And honeybees prefer herbaceous floral plant species such as weeds and a few trees and shrubs to produce honey. Two main flowering periods of honeybee plants were followed by two honey harvesting periods indicated in the study area. The identification of bee plant species, as well as their floral calendar, aids beekeepers in planning various beekeeping activities. Therefore, beekeepers should follow floral calendar of honeybee plants to exploit the potential of the area for honey production.

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Impacts of different salinity levels on seedling growth and survival of black mangrove (*Avicennia germinans*)

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Abstract. *Matto AA, Jaikishun S, Ram M. 2023. Impacts of different salinity levels on seedling growth and survival of black mangrove (*Avicennia germinans*). Asian J For 7: 67-73.* Mangroves are important for coastal protection and provide numerous ecosystem services such as breeding, feeding, and nursery grounds for commercial fishes. This research examined the survivability of black mangrove (*Avicennia germinans* (L.) L.) seedlings in seven salinity concentrations. Growth parameters were collected every ten days, and a post-harvest analysis was done at the end of the data collection period. The results indicated that as the salinity level increased, the growth of the seedlings was reduced. Salinity can affect the different parameters collected in the growth of young seedlings. In the control, which was given zero salinity (0 ppt), it was noted that the growth parameters were not significantly affected. The lower treatments, such as T2 (15 ppt), 3 (30 ppt), and 4 (45 ppt), indicated that the seedlings strive better in these conditions. In treatment 7, which was the highest salinity level (90 ppt), the growth parameters of the seedlings were reduced. The findings of this research indicated that black mangrove seedlings need a low salinity level to have a good growth establishment. Given the adaptability of *A. germinans*, these mangrove species are anticipated to be flexible to cope with a wide range of future climatic circumstances.

Keywords: *Avicennia germinans*, climate change, mangroves, salinity, seedlings

INTRODUCTION

Mangroves are shrubs or forests that grow along tidal estuaries, classified into true and mangrove associates (Numbere 2018a,b). They are placed in these categories based on their morphological and physiological specializations (Sandilyan and Kathiresan 2012). Mangroves are integral to maintaining our marine ecosystem (Sandilyan and Kathiresan 2012; Carugati et al. 2018). Mangroves are home to a wide variety of flora and fauna. Mangrove forests play a significant role in maintaining the productivity of inshore and offshore fisheries (Ball 2002; Nagelkerken 2009; Kodikara et al. 2017). They provide nurseries for fish, help support extensive aquatic species, aid in supplying nutrients to the ecosystem, and provide shelter (Baustian et al. 2017). In mangrove areas and mudflats, are a high abundance of diatom, blue-green algae, and dinoflagellates. From the decomposition of mangroves, forest litter is organic material that can be used as food sources for organisms in mangrove-dominated estuaries (Noor et al. 2015; Baustian et al. 2017).

In Guyana, mangroves are mostly found along the coast. The dominant species of mangroves is the *Avicennia germinans* (L.) L. (black), which fringes along the coast, *Rhizophora mangle* L. (red) are less abundant, and *Laguncularia racemosa* (L.) C.F.Gaertn. (white) is distributed partially along Guyana's coast (Bovell 2010). The *A. germinans* and *L. racemosa* are found as pioneering species in mudflats and sandbanks on the coast. The *R. mangle* is more abundant in estuaries and sheltered river banks. This zonation pattern of mangroves is common from Corentyne to the Essequibo Coast (Bovell 2010).

Globally, mangrove cover has declined significantly, and the degradation continues at an alarming rate. Sandilyan and Kathiresan (2012) reported a depletion rate of 0.66-1.0% per year, three to five times higher than the average loss rate for land forests. Globally, an estimated 20% (3.6 million hectares) of mangroves were lost between 1980 and 2005 (Maiti and Chowdhury 2013). The impact of carbon emissions and mangrove degradation has been a global issue. Although carbon-rich mangrove forests are deforested and degraded due to land use and land-cover change, other threats include releasing heavy metal toxic substances and industrial waste (Maiti and Chowdhury 2013; Kodikara et al. 2017).

Climate change is also a significant contributor to mangrove depletion, particularly sea-level rise, which is considered a threat to mangrove habitat and functionality (Sandilyan and Kathiresan 2012; Jaikishun et al. 2017; Carugati et al. 2018). In the Caribbean, mangroves have declined by 24% over the last quarter-century. One of the reasons for this is anthropogenic activities and the increase in the effects of climate change. In addition, there has been an increase in salinity on the mudflats, ocean levels, and more intense rainfalls (Wilson 2017).

In Guyana, mangroves are important since they're an income earner for many coastal communities (NAREI 2015). Hollowell (2005) reported that in Guyana, a sea-level rise in the 0.09 to 0.88 m range had been predicted by the year 2100. Sea level rise could result in drastic changes in the extant mangrove ecosystems. In addition, this prolonged flooding resulting from increased sea-level rise can lead to plant death.

Several ecological factors drive mangrove health; ecological requirements for salinity, appropriate hydrology, topography, and suitable species composition are major causes of mangrove restoration failure (Kodikara et al. 2017). The health, distribution, growth, and productivity of mangroves are affected by salinity. The high spatial and temporal fluctuations, driven by inputs of freshwater and seawater, inundation, groundwater seepage, and evaporation are often observed (Baustian et al. 2017; Kodikara et al. 2017; Carugati et al. 2018). Hence, species-specific studies of mangroves are vital for the replanting process. Salinity affects mangrove species differently (Bovell 2010; Baustian et al. 2017; Kodikara et al. 2017). This study was designed to assess the impacts of different salinity levels on seedling growth and survival of black mangroves (*A. germinans*).

MATERIALS AND METHODS

Study site

This experiment was conducted in Kitty, Georgetown, on Guyana's Atlantic Coast. This area is in region four, Demerara-Mahaica.

Experimental design

The Randomize Block Design (RBD) was ideal for this experiment because it allows the seedlings to be grouped into replicates and does not limit the number of replicates and treatments used (Figure 1). It was done in three replicates; three plants were treated with 0, 15, 30, 45,60, 75, and 90 ppt salt (NaCl) solution, with 0 ppt being the control. The natural mangrove seedlings were obtained from the Grove area, with permission from NAREI.

Data collection

Data was collected for three months, and the following parameters were recorded: the height of the plants was measured and determined using a measuring tape, the stem diameter was measured using a Vernier calliper, the total number of leaves was counted at ten days intervals, the number of surviving mangrove species was counted every ten days, the mortality of the plants was recorded.

Post-harvest analysis

The wet and dry masses of the plant parts were collected. Three replicates were harvested and washed thoroughly, blotted dry, and separated plant components (roots, stem, and leaves). The mass of the wet parts was recorded. The dry part was wrapped in paper and oven-dried at 80°C for 24 hr. The weights were recorded until a constant weight was reached (Kodikara et al. 2017).

Total leaf area

Three random leaves from each treatment were chosen, and the total leaf area was quantified using graph paper (Kodikara et al. 2017).

Figure 1. Experimental plot

Chlorophyll content

Leaf samples were harvested, and 1 g was weighed and immersed in 10 mL of dimethyl sulfoxide (DMSO). It was then placed in 15 mL test tubes and wrapped in aluminium foil. The samples were then incubated for four hours at 65°C. After the expiration of the period, the samples were removed and left to cool. Samples were then individually poured into the test tubes with DSMO being the blank (Wellburn 1994; Tait and Hik 2003). The mixture was then placed in a spectrophotometer to obtain the readings at 670 nm, 645 nm, and 663 nm (Sandilyan and Kathiresan 2012; Kodikara et al. 2017).

Soil analysis

A soil analysis test was conducted on the two soils at John's Lab, University of Guyana. The following nutrients were tested for; nitrogen, phosphorus, potassium, calcium, magnesium, iron, and copper.

Data analyses

Analysis of the data collected was done using Microsoft Excel 2010. The statistical test, one-way fixed-factor ANOVA, was used to find the p-values of parameters (height, stem diameter, leaf thickness, number of leaves, wet and dry mass) and hence the significant differences.

RESULTS AND DISCUSSIONS

Black mangroves are Guyana's most dominant species and can tolerate high salinity levels. However, in the seedling stage, it is important to have a moderate saline condition. This study used two types of soil to cultivate mangroves. The EC and pH of the soil were noted in Table 1. In the natural soil analysis, the EC indicated that it contained a high salinity level, and the pH indicated that the soil was mostly acidic. However, in the loam soil, the EC showed that the soil had moderate salinity, and the pH was close to neutral (Table 1).

Table 1. EC and pH of the natural and loam soils

Parameters	Natural soil	Loam soil
Electrical conductivity (Sm^{-i})	1281	569
pH	5.20	6.60

Mangrove seedlings can be subjected to ion toxicity, osmotic stress, nutritional shortage (N, Ca, K, P, Fe, Cu, and Mg), and oxidative stress resulting from soil salinity, which inhibits water intake from the soil. One element, especially sodium, harms plants in high concentrations (Reef et al. 2010; Shrivastava and Kumar 2015). While NaCl is a necessary plant micronutrient that influences enzyme activity in the cytoplasm, it is a co-factor in photosynthesis and critical for turgor pressure creation and cell expansion (Shrivastava and Kumar 2015). Therefore, high accumulation might result in ion toxicity and plant damage. Furthermore, excessive sodium accumulation in cell walls can quickly cause osmotic stress and cell death. It also makes the soil less permeable, hence, the reason for stunted growth and even death. This also causes an imbalance in leaf tissues, affecting the leaf thickness (Reef et al. 2010; Shrivastava and Kumar 2015).

From the soil analysis results collected, it was shown that nitrogen is an essential nutrient for plant growth. As depicted in the graphs, the control and T7 did not contain a significant amount of nutrients because these mangroves were grown in pots and because of the uptake of nutrients and lack of supply through the nutrient cycle and competition (Reef et al. 2010; Shrivastava and Kumar 2015). Despite nitrification levels being normally low, the top layer of soil and a thin layer of aerobic soil around the mangrove roots maintain populations of nitrifying bacteria, which can transform ammonium into nitrate for the plant. High salinity affects photosynthesis and the demand for nitrogen in mangroves (Reef et al. 2010). Mangroves cultivated in pots/bags appear to prefer nitrate to ammonium and exhibit a significant drop in plant nitrogen uptake. The reduction in nitrogen level is also shown to affect stem diameter, dry mass, and plant growth (Reef et al. 2010; Campos et al. 2014). A saline soil usually contains large amounts of Mg and Ca, both compete with the soil particles, in particular, Na, for space, so in the results, it was seen that Ca dominated in the control; however, in the natural soil, Mg was less than loam soil because the natural soil already was a high saline soil and Na would have the largest nutrient in the soil (Figure 2) (Reef et al. 2010; Chen and Ye 2014; Noor et al. 2015; Shrivastava and Kumar 2015).

Phosphorus is another important nutrient for plant growth. Phosphate accessibility is limited in saline soils due to ionic strengthening effects, which diminish phosphate activity. Phosphorous activity cannot be replenished through biological fixation (Reef et al. 2010; Chen and Ye 2014). Because phosphate ions precipitate with Ca^{2+}, soil salinity inhibits plant phosphorus absorption (Reef et al. 2010; Chen and Ye 2014).

Potassium content is a factor that affects plant growth at a high salinity level (Niste et al. 2014). In Figure 2, the control in both soils shows a higher level of K than T7. Potassium is essential in protein synthesis in mangroves. The soil solution's high salinity and Na+ concentration can impact plant K^+ absorption. The loss in chlorophyll and photosynthetic function in the mangrove seedlings was shown because of K^+ deficiencies (Reef et al. 2010; Shrivastava and Kumar 2015). Several types of research have demonstrated that as the NaCl salinity or the Na^+/Ca^{2+} ratio in the root media increases, the K^+ concentration in plant tissue decreases (Shrivastava and Kumar 2015).

Magnesium showed a high concentration in the natural soil control and a low level in loam soil. Copper and iron have significantly low levels in both control and T7. Cu stress appears to substantially influence mangrove plants by reducing root permeability (Figure 2). Moreover, black mangrove seedlings were discovered to have larger outer cell layers and greater lignification within the exodermis (Reef et al. 2010; Chen and Ye 2014; Kodikara et al. 2017). This could have indicated that reduced permeability could directly prevent excessive Cu from entering the roots.

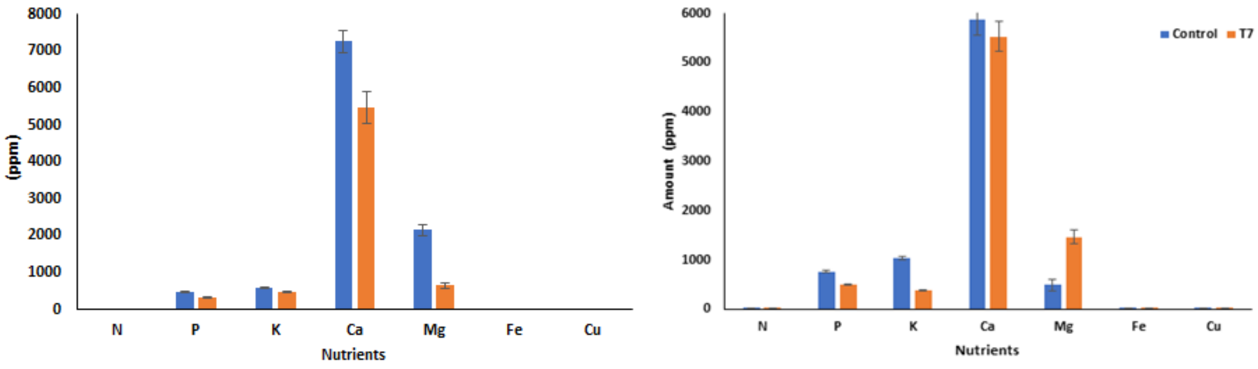

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Figure 2. Average nutrient content in (A) natural soil and (B) loam soil. Bars represent Mean±SD

In mangroves, the metal binding inside the cell wall, either in ionic form or combined with cell wall structural components like lignin, has long been recognized as an important detoxification method (Chen and Ye 2014). Natural cations (Cu, Fe, Mn), including vital plant nutrients, are displaced by high Na^+ concentrations. The findings revealed that the combined impacts of iron shortage have a substantially greater impact on plant development and chlorophyll concentration due to the increased saline conditions of the soil; iron, a micronutrient, is suppressed as salinity increases. Iron deficiency hurts the chlorophyll concentration of juvenile leaves. Mineral toxicity, nutrient insufficiency, and nutritional imbalance are frequently associated with soil salinity and acidity (Reef et al. 2010; Chen and Ye 2014).

The experiment yielded three mortalities. The treatments that have morality were one in T5 (60 ppt) and two in T7 (90 ppt). From the results obtained, these specific treatments were not very productive for the seedlings to grow properly. Further, T7 showed that the seedlings could not have good growth establishment. This is due to the increased stress on the seedlings caused by the high salinity level; the earlier stage of seedling growth requires low salinity for good growth. Hence, the lower treatment levels have half of no mortality. The necessary nutrients responsible for growth, such as nitrogen, phosphorus, and potassium, were limited in the soil due to increased saline conditions. This could have been a factor responsible for mortality.

The growth parameters percentage was calculated using the average by subtracting the final from the initial data and finding the percentage. The control treatment is significantly different from the salinity treatments. Growth was observed at 43%; in the natural soil, the plants grew better in T3, which showed a 65.8% increase in height; in the loam soil, the control showed a 56.7% growth rate. Growth was observed better in T4, with a 61% increase. This treatment replicates the seawater salinity, which was tested to be 45 ppt.

In comparison to the higher levels of salinity, it was noted that growth rates were lower as the salinity levels increased. When compared to the control T1, there was a 43% growth in natural soil and 56.7% in loam soil (Figure 3). In the control, the seedlings have depicted a slow growth rate as compared to T3 in natural and T4 in loam soils. When comparing the control to the treated seedlings, there was slower growth, but the higher treatments proved to have had the slowest growth rates (Liang et al. 2008; Nagelkerken 2009; Kodikara et al. 2017). The one-way ANOVA showed significant differences in the growth rate of plants in the natural and mixed soils, respectively (p=0.000353 and p=0.000293). Production is also influenced by salinity, and at the stage of development, an increase in salinity reduces height and growth, particularly in *Avicennia* spp. (Whigham et al. 2009; Nguyen et al. 2015; Budiadi et al. 2022).

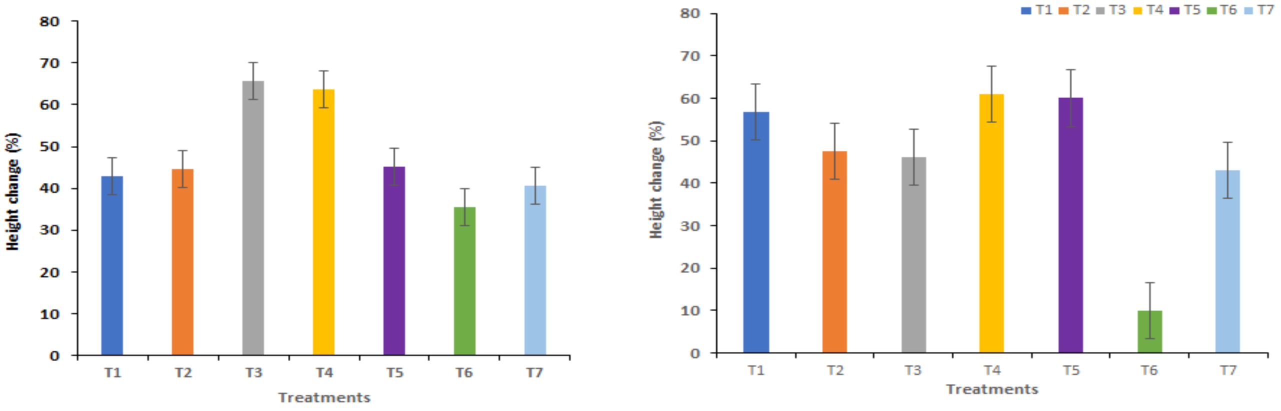
Despite the ability of black mangroves to tolerate high levels of salinity, seedlings need moderate salinity to survive and have a good root establishment. They are age specific as they grow older and tend to tolerate a higher salinity. An extremely high salinity condition can increase the stress level in mangrove seedlings (Kodikara et al. 2017). An increase in temperature can result in reduced precipitation and increased urban and agricultural demand for groundwater is projected to raise estuary salinity, increasing the risk of succumbing to salt stress. According to some models, temperature changes, salinity, and sea level will cause mangroves to spread into higher latitudes (Madrid et al. 2014).

In the stem diameter, it was seen that the control in natural soil was not significantly affected even though there was a drop of 93% (Figure 4A). In the loam soil, the control was not affected at all at 93%; this can be due to the soil being less saline than the natural soil (Figure 4B). T2 in nature gave the highest percentage of 95%, and T1, T4, and T5 had the highest and the same 93% recorded. T7 in both soils showed a decline in stem diameter (p~ 0.0232 and 0.0379), showing a difference in the stem diameter; however, it was not a large significant difference.

Increases in water salinity influence critical nutrient uptake and salt accumulation, resulting in osmolality increases in tissue sap. In NaCl media, many researchers found that when salinity increases, the formation of the xylem, the width of vascular bundles, and the diameters of mangrove stems can be reduced. Although in the soil, nitrogen plays an important role in stem diameter in the soil due to the minimal amounts indicated in the soil analysis; this could have hurt this parameter.

It was noted that leaf emergence in the control (T1) was not affected because there was 68.9% leaf emergence but showed a decline in T6 for the natural soil and T7 for the loam soil (Figure 5A). In the loam soil, the control showed a high increase and a steady fluctuation (Figure 5B). There was no significant difference in this parameter in the natural and loam soil, respectively (p= 0.0672 and 0.0732). The percentage of leaf emergence of seedlings on natural soil tends to be lower than that of seedlings on loam soil. Seedlings in the salinity treatment could grow new leaves but showed a lower percentage than the control. The rate of leaf creation and death and the length of leaf life have significant physiological implications for the entire plant. It enables the plant to maintain a good growth rate (Suárez and Medina 2005).

The combination of a considerable fall in leaf production, leaf expansion rates, and a reduction in leaf life span resulted in a lower plant leaf under hypersaline conditions. The capacity to sequester ions in leaves is projected to decline with time due to continuous exposure to excessive salinity (Suárez and Medina 2005; Nguyen et al. 2015). Although there were differences noted, it was not to a large extent. This could be because there was grasshopper infestation and caterpillar, which could have affected the results as the leaves of the plants were eaten away.



A B

Figure 3. Percentage of seedling height increase of seedlings in (A) natural soil and (B) loam soil. Bars represent Mean±SD

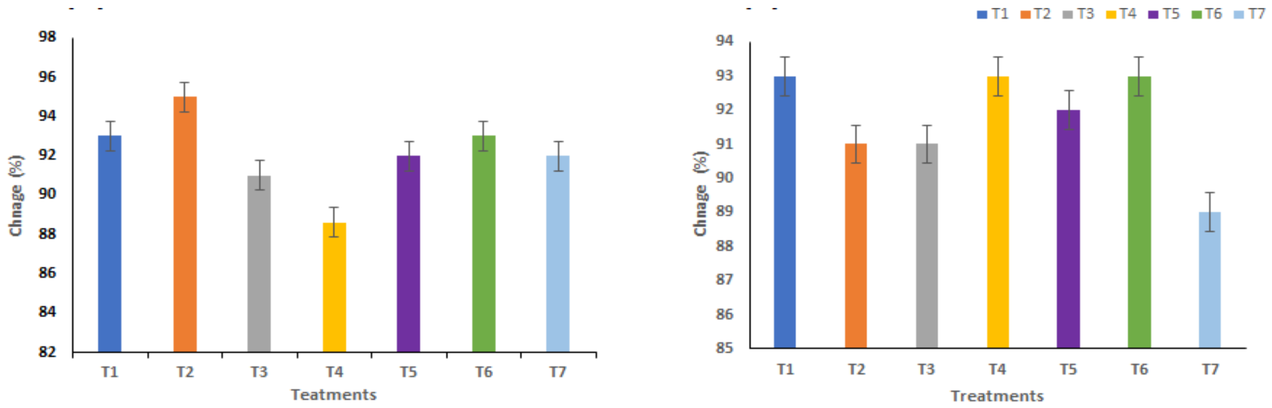


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Figure 4. Percentage stem diameter percentage in (A) natural soil and (B) loan soil. Bars represent Mean±SD

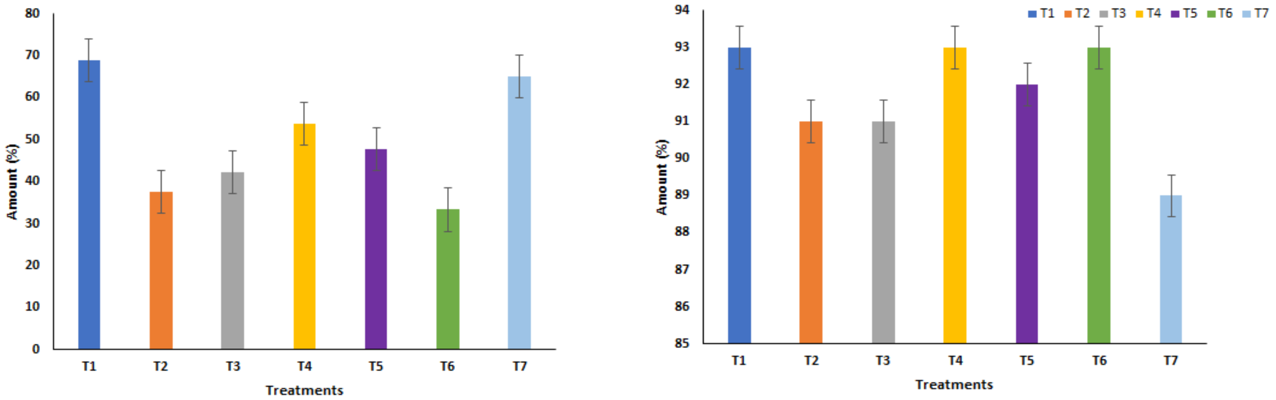

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Figure 5. The percentage of leaf emergence in (A) natural and (B) loam soil was obtained from different treatments. Bars represent Mean±SD

The percentage leaf thickness ranges showed T2 and T3 in the natural, having the highest rise at 82% in leaf thickness, and the loam soil was noted to have 84% in T4 in leaf thickness. The control T1 in natural was shown to have 79.4% and the loam 78%, respectively. The T7 was noted to have the lowest leaf thickness percentage at 71%. The natural and loam soil, respectively (*p*= 0.0634 and 0.0868), indicated no significant difference in leaf thickness in this parameter. However, the higher salinity range showed a decline in leaf thickness. This could be because, at higher salinity ranges, the leaves were unable to photosynthesize as compared to lower levels; they were also unable to absorb large amounts of water as salinity levels increased. As salinity increases, the ability of the seedlings to absorb water is reduced due to an increase in the Na ions, which affect the other nutrients present in the soil, such as potassium and phosphorus (Kodikara et al. 2017). Various factors could have also affected the leaf thickness, such as inclement weather conditions and prolonged rainfall.

Post-harvest

Leaf surface area (*p*= 0.054421 and *p*=0.005111) showed a significant difference in the leaf surface area. As salinity levels increase, leaf demography is affected, especially in the size of the leaf. Higher saline conditions indicated a smaller leaf surface area (Table 2). Leaves are very useful in determining the stress level in mangrove plants in their survival in saline conditions (Sobrado 2000; Suárez and Medina 2005; Kodikara et al. 2017). Salt levels cause a reduction in both the length and width of the leaf in black mangrove species in high salinity, altering the leaf's overall shape. Further reductions in leaf life span and the incapacity to sustain fresh leaf production result in a significant drop in leaf area per plant at very high salinity. Reduced leaf area can be caused by salinity and nitrogen levels (Sobrado 2000; Suárez and Medina 2005; Budiadi et al. 2022).

The natural soil's wet and dry mass (*p*=0.021808, *p*=0.005137) differed significantly. The wet and dry masses for the loam soil, respectively, were significant (*p*=0.047641 and *p*=0.034195). The values indicated were not large but indicated a relationship between salinity and wet and dry masses of mangroves—the percentage water content for natural and mixed soil, respectively (*p*= 0.011446 and *p*=0.021808). Salinity can be seen as a factor in water absorption. The average percent water content of plants cultivated in lower-saline settings was considerably higher than that of plants produced in high-saline conditions, suggesting that mangroves store more water in low-saline environments. Low osmotic potential under hypersaline could be the reason for increased moisture absorption (Basak et al. 2004).

A factor that would account for the plants' modest growth rates and shrubby size at hypersaline sites is that a linear association was found between diurnal carbon gain and maximum rate of CO_2 uptake in a variety of rainforest canopy species. Light saturated photosynthesis rate shows the maximum achievable benefits from a given investment in photosynthetic gear.

Managing a saline environment, on the other hand, presents unique problems in terms of maintaining favourable water and ion balances (Whigham et al. 2009). Salinity can influence the ability of roots to absorb moisture while preventing most ions from entering the transpiration stream in mangrove ecosystems, regardless of the quantity of water. As a result, the average percentage water content of black mangroves was lower as the salinity level increased. Furthermore, the enhanced salinity stress resulted in a considerable drop in the fresh and dry weights; hence, the overall ability to absorb water and reduce grew in lockstep with the increase in salinity augmentation (Whigham et al. 2009; Kodikara et al. 2017).

Chlorophyll is the primary colour pigment involved in photosynthesis. Furthermore, using the absorption rate and the results obtained, it was shown that in the control (T0), for natural and loam, it was showed the chlorophyll level was not severely affected. There was a rise from T2, T3 was constant, a rise in the natural soil, and a drop in the loam soil. As the salinity level continues to rise to T5, T6, and T7, the total chlorophyll decreases (Figure 6). The chlorophyll concentration is a reliable measure of the photosynthetic function under harsh conditions. Because of salt toxicity, which causes the burning of leaves or other succulent portions and degradation of other pigments, chlorophyll content reduction owing to salinity stress is quite prevalent in salt-sensitive plant species (Whigham et al. 2009; Kodikara et al. 2017). This is because they are saline-tolerant species that can withstand such worsening salinity stress. However, it has been discovered that when trees are subjected to increased salt stress, their chlorophyll levels dropped due to enzymatic chlorophyll breakdown (Basak et al. 2004).

Table 2. Leaf surface area in natural and mixed soil obtained from different treatments

Treatments	Total Leaf Area (cm^2)		Water content (%)	
	Natural soil	Loam soil	Natural soil	Loam soil
1 (control)	5.3±4.55	16.8±6.32	70.6±10.4	83.5±12.5
2	8.5±5.7	7.6±4.7	81.8±7.1	88.7±1.28
3	6.6±4.56	9.1±5.88	78.6±4.2	83.5±9.11
4	7.1±2.02	18.6±7.92	79.6±6.59	79.7±13.11
5	5.1±3.38	22.8±8.33	82.3±9.84	81.3±8.45
6	5.6±3.21	10±6.23	54.6±5.7	75.4±9.05
7	4.5±1.76	5±4.66	74.6±6.3	80.3±5.60

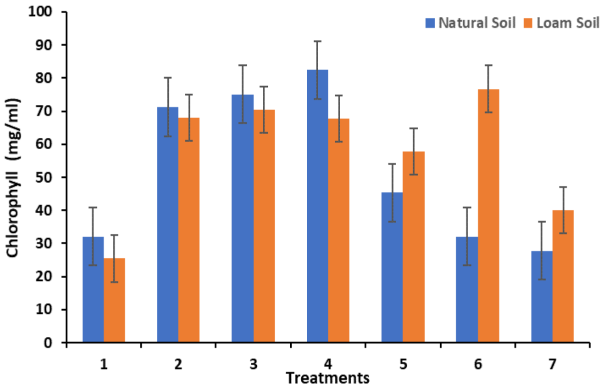

Figure 6. Total chlorophyll in natural and loam soil was obtained from the seven treatments

Salinity therapy increased salt production at moderate salinity. This allowed plants to modestly lower water absorption while maintaining a high level of carbon assimilation. However, salt secretion tended to be reduced under hypersaline conditions, which could be the result of a saturated mechanism (Basak et al. 2004). Salt secretion is a very active system that comprises several metabolically regulated phases. As a result, declines in the solutes transported to leaves were more relevant in hypersaline circumstances (Ball 2002; Basak et al. 2004; Budiadi et al. 2022).

In conclusion, the growth parameters indicated that as the salinity level increased, the mangrove growth was reduced. The lower treatments, such as T2, T3, and T4, proved to be ideal saline conditions for the growth of black mangrove seedlings. As the seedlings grow older, they can be exposed to higher salinity ranges. Seedlings must get the right conditions to have a good growth establishment. The post-harvest analysis indicated that high salinity ranges affect the wet and dry masses and water content. The chlorophyll content was reduced, indicating that the rate of photosynthesis declined. This study indicated that mangrove seedlings are at a greater risk of surviving as salinity increases.

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