BIODIVERSITAS
Volume 26, Number 9, September 2025
Pages: 4793-4804

ISSN: 1412-033X
E-ISSN: 2085-4722
DOI: 10.13057/biodiv/d260948

Stand composition and structure of *Rubroshorea curtisii* in a lowland Dipterocarp remnant forest of Bukit Tiban, Riau Islands, Indonesia

ARIDA SUSILOWATI[1,♥], HENTI HENDALASTUTI RACHMAT[2], NELLY ANNA[1], DENI ELFIATI[1], FIFI GUS DWIYANTI[3], KOICHI KAMIYA[4], FAUZAN ALAWI[5], KESUMA WIJAYA[6], IDA MALLIA GINTING[1]

[1]Faculty of Forestry, Universitas Sumatera Utara. Jl. Lingkar USU, Pancur Batu, Deli Serdang 20353, North Sumatera, Indonesia. Tel./fax.: +62-61-8220605, ♥email: arida.susilowati@usu.ac.id

[2]Research Centre for Ecology and Ethnobiology, National Research and Innovation Agency. Jl. Raya Bogor No. 970, Cibinong, Bogor 16915, West Java, Indonesia

[3]Departement of Silviculture, Faculty of Forestry and Environment, Institut Pertanian Bogor. Jl. Lingkar Akademik Kampus Dramaga, Babakan, Bogor 16680, West Java, Indonesia

[4]Graduate School of Agriculture, Ehime University. 1553 Shimoidaimachi, Matsuyama, Ehime 791-0112, Japan

[5]Serindit Philosophy Center Batam. Jl. Tiban I No. 20, Patam Lestari, Sekupang, Batam 29424, Riau Islands, Indonesia

[6]Conservation Forest Management Unit Batam. Jl. Pramuka No. 7b, Sungai Harapan, Sekupang, Batam 29425, Riau Islands, Indonesia

Manuscript received: 23 April 2025. Revision accepted: 30 September 2025.

Abstract. *Susilowati A, Rachmat HH, Anna N, Elfiati D, Dwiyanti FG, Kamiya K, Alawi F, Wijaya K, Ginting IM. 2025. Stand composition and structure of* Rubroshorea curtisii *in a lowland Dipterocarp remnant forest of Bukit Tiban, Riau Islands, Indonesia. Biodiversitas 26: 4793-4804. Rubroshorea curtisii* is an ecologically and commercially valuable Dipterocarpaceae species native to the Malay Peninsula, Peninsular Thailand, and northern Borneo, but in Indonesia its distribution is restricted to the Riau Islands. Habitat disturbances in this area directly threaten the species' persistence. This study assessed the stand composition, structure, and floristic diversity of *R. curtisii* in Bukit Tiban Protected Forest, Batam Island, Riau Islands. Vegetation data were collected from 25 nested plots (20×20 m for trees, 10×10 m for poles, 5×5 m for saplings, and 2×2 m for seedlings) along five transects purposefully located in areas where *R. curtisii* occurs. All individuals were identified and measured to calculate relative density, frequency, basal area, and Importance Value Index (IVI), along with Shannon–Wiener diversity (H′), evenness (E), and species richness (R). A total of 44 species from 13 families were recorded, including seven Dipterocarpaceae species. *Rubroshorea curtisii* dominated all strata with the highest IVI values in seedlings (21.38%), poles (30.83%), and trees (34.00%), indicating its keystone ecological role. The stand exhibited an inverted J-shaped diameter distribution, characterized by a predominance of small-diameter trees (10-19.9 cm) and a scarcity of large individuals, reflecting regeneration imbalance. Diversity was highest in tree strata (H′ = 3.10) and lowest in saplings (H′ = 1.70), while evenness remained high (E = 0.91-0.97). Tree strata showed the highest richness (R = 7.05), whereas saplings had the lowest (R = 1.25). Despite ongoing regeneration, reduced mature individuals and disrupted recruitment suggest vulnerability to fragmentation, logging, and limited reproductive events. Conservation priorities should include protection of mature trees, enrichment planting, canopy restoration, and long-term monitoring of reproductive ecology. Integrating community participation and habitat rehabilitation is essential to ensure sustainable management of *R. curtisii* and the ecological resilience of remnant dipterocarp forests in the Riau Islands.

Keywords: Dipterocarpaceae, diversity, remnant forest, *Rubroshorea curtisii*

INTRODUCTION

Tropical forests are essential for maintaining a diversity of species, which contribute to biodiversity via the web of life (Bodo et al. 2021; Awoke et al. 2022). Increasing human populations, and land conversion into settlements, agricultural activities contribute significantly to the decline of tropical forests (Trozzo et al. 2019; Nath et al. 2020; Tilahun et al. 2022). Remnant forests are frequently viewed as the final refuge for numerous species (Wintle et al. 2019; Faria et al. 2021), a seed source for forest regeneration (Cadavid-Florez et al. 2020), conservation area of native species (Kowarik and von der Lippe 2017), and an essential support system for ecosystem recovery after disturbances (Wu et al. 2022). The remaining forests are acknowledged for their beneficial connection to plant diversity (Malkinson et al. 2018). Unfortunately, remnant forests exhibit higher extinction rates (Anjos et al. 2021). Evaluating remnant forest ecosystems is essential for the maintenance and conservation of species (Susilowati et al. 2024; Yang et al. 2021; Wu et al. 2022).

Rubroshorea curtisii (synonymous with *Shorea curtisii*) is distributed in the Malay Peninsula, Thai Peninsula, and North Borneo and can be found in coastal or inland ridge forests (Symington 1943) with well-drained, low-nutrient soils in northern Borneo (Ashton 1982). This species' wood is used for paneling, furniture, flooring, and veneer. Molecular phylogenetic evidence suggested Rubroshorea should be separated from Shorea (Ashton and Heckenhauer 2022). *Rubroshorea curtisii,* known as a canopy-to-emergent tree over 50 meters in height, regulates forest microclimate, stratifies the canopy, sustains arboreal species, sequesters carbon, cycles nutrients, stabilizes stand structure and preserving vertical stratification (Appanah and Turnbull 1998; De Frenne et al. 2021; Suhaimi et al. 2023).

Habitat fragmentation, timber exploitation, seed predation, and the reduction of natural pollinators have been reported as factors inhibiting natural regeneration of *R. curtisii* (Ng et al. 2024; Tani et al. 2025). This threat is exacerbated by climate change which has affect regeneration patterns and spatial distribution (Ganivet et al. 2020). Based on IUCN Red List this species classified as "Least Concern (LC)," but local populations in several regions, including Indonesia, are threatened with decline due to its narrow distribution and habitat degradation (Barstow 2018). Although the species is categorized as LC on a global scale, its status within Indonesia requires immediate focus. Native to the Riau Islands, this species faces local extinctions in Bintan and Singkep, alongside declining populations attributed to habitat loss. It qualifies as a threatened species under national criteria, underscoring the necessity for targeted conservation strategies.

The remnant forests in the Riau Islands, Indonesia contain a variety of potential Dipterocarpaceae (Susilowati et al. (2023). As one of the remnant forests in the Riau Islands, the Bukit Tiban protected forest is home for various important species, including *R. curtisii*. This 1770-hectare area proximity to settlements has led to significant anthropogenic pressures as local communities regularly use the species for construction such as housing, fences, animal cages (Yuliastrin 2020; Firdamayanti et al. 2024). Biodiversity loss, reduced natural regeneration, and a shift in vegetation structure are all triggered by this condition (Both et al. 2019).

Studies on the composition and structure of dipterocarp forest stands have been extensively conducted, particularly in the Malay Peninsula and Borneo (Ediriweera et al. 2020; Lepun and Heng 2020; Suhaimi et al. 2023). Most research has focused on the dipterocarp community, but few have investigated specific species' native habitats, such as *R. curtisii* in Indonesia. Specific population analyses could reveal the species' ecological function in stand dynamics and disturbance vulnerability (O'Connor et al. 2017; Malik 2025). Floristic evaluation of juvenile and mature trees is crucial for forest regeneration and population structure (Sosef et al. 2017; Paul et al. 2019; Shu et al. 2021). Unfortunately, information related to the abundance of *R. curtisii* and its floristic composition in its natural habitat is still limited to several remnant forests. Therefore, this study aims to assess the stand composition, structural profile, and floristic diversity of the natural habitat of *R. curtisii* in the Bukit Tiban Protection Forest, Riau Islands. Understanding the stand composition, structural profile, and floristic diversity patterns of *R. curtisii*'s natural habitat is crucial for formulating ecosystem-based conservation strategies.

MATERIALS AND METHODS

Research location

This research was conducted in the Bukit Tiban Protected Forest (BTPF) located on Batam Island, Riau Islands Province, Indonesia (Figure 1). Administratively, BTPF is located at 1°05′N, 104°00′E in two different sub-districts, namely Sekupang and Batu Aji.

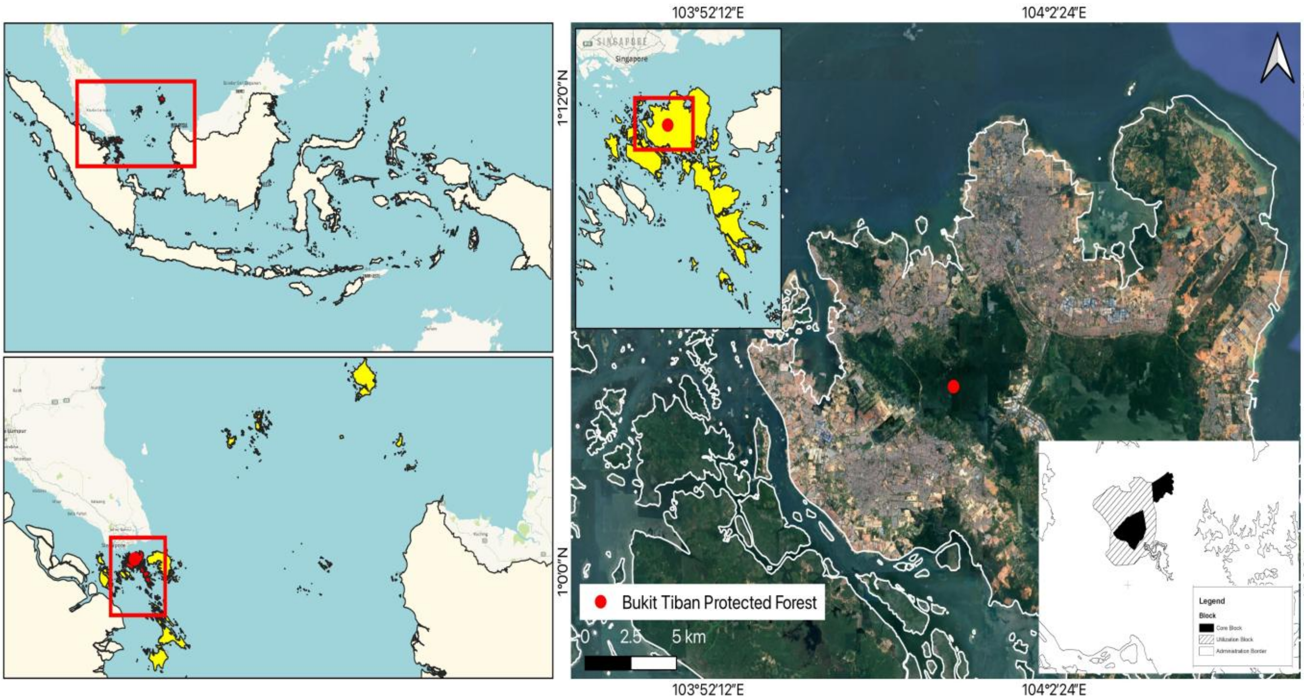

Figure 1. Research location in Bukit Tiban Protected Forest, Batam Island, Riau Islands Province, Indonesia

The BTPF is located in the middle of a community settlement that directly borders the Muka Kuning Nature Tourism Park and the Mata Kucing Tourism Forest. The BTPF is a lowland tropical forest area situated at an altitude of 8-150 meters above sea level, with temperatures ranging from 21°C to 34.8°C and high humidity ranging from 45% to 100%. The land has the characteristics of sloping, with forest cover varied from slightly barren to a good forest cover. Like other forest areas, the BTPF is one of the remaining forest areas in the Riau Islands Province. This location managed by the Forest Management Unit II Batam, which is part of the Riau Islands Province Forestry Service. Anthropogenic disturbances are among the major concerns as in some locations, there have been converted into several nonlegal agricultural activities. Inappropriate land use accompanied by the characteristics of sloping land, is thought to increase the potential for disasters such as landslides (Firdamayanti et al. 2024), so that reforestation in this area is urgently needed.

Data collection and plot design

A total 5 transect consist with 25 plots were established for vegetation analysis. Data collection used nested plots with different sizes according to growth class: 20×20 m for trees (DBH ≥10 cm), 10×10 m for poles (DBH 5-9.9 cm), 5×5 m for saplings (DBH 1-4.9 cm), and 2×2 m for seedlings (height <1.5 m). Plots were placed purposively in locations representative of vegetation conditions especially, the presence of *R. curtisi*. Each individual tree within the plot was recorded for its species, its Diameter at Breast Height (DBH) was measured using a diameter tape, and its height was measured using a clinometer. The coordinates of each plot were recorded using a GPS. The target species and all vegetation in the sample plots were recorded based on their growth stage. Vegetation data obtained at the pole and tree stages comprised the species name, height, and diameter of each tree. The data obtained at the seedling and sapling stages comprised the species name and the number of individuals for each species.

Species identification

Species names were established during data collecting. Identification was conducted in the field using identification keys and supported by authoritative literature such as Flora Malesiana (van Steenis 1983), Tree Flora of Sabah and Sarawak (Soepadmo and Wong 2004) and Pedoman Indentifikasi Pohon Pohon Dipterocarpaceae Sumatera (Newman et al. 1999). Unidentified specimens were immediately preserved in herbarium specimens and verified in the laboratory in consultation with taxonomists from KPHL II Batam For taxonomically ambiguous specimens, voucher samples also verified through morphological analysis at Center for Forest Development herbarium in Bogor, West Java, Indonesia. This dual approach ensured accurate species documentation while building local capacity in forest biodiversity monitoring. Current threats were identified through a focused discussion with key stakeholders, including area managers, local leaders, and environmental activists. Threats are defined as activities or events that caused ecological impact on forest cover change or activities disrupting forest structure, composition or regeneration. Its level can be measured based on ecological impact, frequency (occurrence rate), and irreversibility (recovery potential).

Vegetation analysis

Species abundance was determined based on the number of individuals analyzed using the Importance Value Index (IVI). According to Curtis and Mclntosh (1950), the IVI is calculated by accumulating the values of relative density, relative frequency, and relative dominance (basal area). The IVI for the seedling and sapling is the sum of the relative density and the relative frequency, hence the highest IVI for these stages is 200%. For pole and tree, the IVI is calculated as the sum of Relative Density (RD), Relative Frequency (RF), and Relative Basal Area (RBA), with a maximum of 300% as follow:

IVI = Relative Density + Relative Frequency (for seedling and sapling)

IVI = Relative Density + Relative Frequency + Relative Basal Area (for pole and tree)

Relative density is obtained by quantifying the comparison of the number of individuals of each species to the area of observation. In detail, relative density can be calculated using the following formula:

$$\text{Relative Density} = \frac{\textit{Total number of individual species}}{\textit{Total number of all individual of species}} \times 100\%$$

In contrast to relative density, relative frequency is calculated based on the number of observation plots where a species is found. In other words, relative frequency describes how often a species appears in an observation plot. Relative frequency also shows the ratio of the occurrence of a species to all species found at the study site. In more detail, relative frequency can be calculated using the following formula:

$$\text{Relative Frequency} = \frac{\textit{Frequency of respective species}}{\textit{Frequency of all species}} \times 100\%$$

At the pole and tree level, the DBH of all individuals found in the research plot was quantified to obtain the basal area. The DBH value was calculated by the proportion of the tree basal area to the area of the observation plot. Seedlings and saplings were excluded from the basal area calculation because they have not reached the standard breast height (1.3 m) required for DBH measurement. Their stem diameters are too small to be comparable with mature trees, and their contribution to the total stand basal area is negligible. These growth stages are instead analyzed separately as part of forest regeneration dynamics. Although the number of individuals greatly determines the Relative Basal Area (RBA) value obtained, the size of the trunk of the individuals found also has a significant effect on the RBA value. In more detail, the RBA was be calculated using the following formula:

$$\textit{Relative Basal Area} = \frac{\textit{Basal area of respective species}}{\textit{Basal area of all species}} \times 100\%$$

In addition to calculating the abundance of vegetation based on the IVI, in this study, several ecological indices, such as the Shannon-Wiener diversity index (H'), species evenness index (E), and Margalef richness index (R), were also analyzed. According to Turkis and Elmas (2018), the diversity index is an index that can estimate the proportion of coverage of a particular species in the total sample of an ecosystem. The value of the diversity index can be obtained by quantifying the logarithm of the proportion of the number of individuals of a species to the number of individuals of all species found.

The species diversity of the examined community can be calculated using the Shannon-Wiener diversity index (H') formula referring to Magurran (2004). The greater the species diversity index of a community, the more stable the community is. The diversity index will produce a maximum value if there are the same number of individuals in all species, but if the number of species found is very small, the resulting species diversity value will be very low (approaching 0).

$$H' = -\sum_{i=1}^{S} Pi \, ln \, Pi$$

Where, H' represents the diversity index, Pi is the comparative value between the number of individuals of a species (ni) to the number of individuals of all species (S).

To determine whether individuals are distributed more evenly among the species present at a growth stage, the species Evenness index (E) referring to Magurran (2004) was calculated. A higher evenness index of a growth stages indicates that the distribution of individuals to species will be even.

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

Where, H' represents the Shannon Wiener diversity value and S represents the number of species found in the research location. The species evenness index is also known as the Shannon equivalence index, where this index produces values in the range of 0 to 1. Values approaching 1 assume that the evenness in the ecosystem is complete. Conversely, values approaching 0 describe that the evenness of species is low.

To estimate species richness in the research location, the Margalef index (R) referring to Magurran (2004) is calculated by calculating the proportion of the number of species found to the logarithm of the number of individuals of all species. The Margalef species richness index (R) depicts the species richness of a community during study. This index can be used to determine the level of richness of particular type of vegetation compared to others in the community. The formula used to calculate species richness is as follows:

$$R = \frac{(S-1)}{Ln\ (N)}$$

Where, R represents the Margalef richness index, S represents the total species found, and N is the total number of individuals at the study site. The Margalef index (R) will produce high values in communities that include many species, where the number of individuals of each species decreases more slowly when moving from abundant species to relatively fewer species. Wardhana et al. (2022) classify the Margalef index value into three categories, namely low if R is less than 2.5, moderate if the R value is in the range of 2.5 to 4, and high if the R value obtained is more than 4.

For threat analysis, discussion results among stakeholders then thematically analyzed to identify recurrent threats and validate field observations. This participatory approach ensured interdisciplinary integration of ecological data with socio-economic perspectives.

Data analysis

Numerical data were analyzed using Microsoft Excel 2019 for basic calculations, while diversity analysis and stand structure comparison were performed using PAST v4.03 (Hammer et al. 2001) and R v4.3.0 (R Core Team 2023). The significance threshold was set at $p<0.05$.

RESULTS AND DISCUSSION

Floristic composition

The research findings indicate that 44 tree species from 13 families have been identified in the observation plots. The Dipterocarpacea family is predominant in this study area, as evidenced by the presence of seven species: *R. curtisii*, *R. parvifolia*, *Hopea mengarawan*, *H. beccariana*, *Dipterocarpus coriaceus*, *D. fagineus*, and *D. rigidus*. The findings of the Importance Value Index (IVI) indicate a change in species dominance throughout various growth stages. *Rubroshorea curtisii* (Dipterocarpaceae) exhibited the highest seedling viability (IVI: 21.38%), with *Calophyllum pulcherrimum* and *Gironniera parvifolia* following closely behind (Table 1). The occurrence of pioneer species such as *G. parvifolia* and *Eurycoma longifolia* indicates that fast-growing species play a crucial role in the initial stages of regeneration. Nonetheless, the prevalence of dipterocarp species during the initial phases indicates that the climax community may demonstrate sustainability (Sadeghi et al. 2014; Rüger et al. 2023; Yatar et al. 2023).

In the early stages of growth, there was a notable shift in dominance, with *C. pulcherrimum* and *H. beccariana* exhibiting the highest Importance Value Index, exceeding 48% (Table 1). This highlights the significance of shade-tolerant non-dipterocarp species in competitive regeneration within the middle strata. *Santiria laevigata* (IVI: 49.10%) was the most prevalent species during the pole stage, with *G. parvifolia* and *R. curtisii* closely trailing it. During the tree stage, dominance returned to significant canopy species, particularly *Koompassia malaccensis* (IVI: 59.42%) and *R. curtisii* (34%). This finding highlighting the crucial role of shade-tolerant climax species in shaping mature stand structure, consistent with the Southeast Asian rainforest succession model Prohaska et al. 2023). The visualization of IVI trends reveals a consistent pattern of changing species dominance throughout various growth stages (Figure 2). Pioneer species, such as *E. longifolia* and *G. parvifolia*, play a crucial role in the initial strata, yet their significance decreases as one move through the tree layers. Conversely, shade-tolerant species (*R. curtisii* and *K. malaccensis*) established dominance within the tree strata. This trend aligns with recent findings that suggest tropical forest regeneration dynamics are influenced by a mix of environmental factors (light, moisture, and soil nutrients) and species-specific competitive interactions (Both et al. 2019; Hammond et al. 2021).

The process of succession follows an established successional pattern, with pioneers playing an important role in the early stages but gradually reducing as shade-tolerant and emerging species take over in the latter phase. This pattern is consistent with general ecological theory, which posits that pioneer species help with early phases of regeneration but are eventually replaced by climax species, which define mature forest communities (Romadini et al. 2022). The presence of *R. curtisii* in all strata illustrates its importance as a keystone species, adjusting from the start of regeneration to establishing dominance in mature stands. IVI trends (Figure 2) show a constant shift in species dominance across development layers. Pioneer species (such as *E. longifolia* and *G. parvifolia*) are important in the early strata but dwindle as they progress through the tree layers. Shade-tolerant species (*R. curtisii* and *K. malaccensis*) grow in dominance within the tree layers. This trend is consistent with recent findings indicating that tropical forest regeneration dynamics are driven by a combination of environmental factors (light, moisture, and soil nutrients) and species-specific competitive interactions.

The Kruskal-Wallis analysis revealed a significant difference in species IVI across growth strata (H: 9.01, p: 0.029) (Conover 1999). This suggests that fluctuations in species dominance are not random, but are controlled by real ecological processes in the dynamics of dipterocarp forest renewal. In other words, shifts in vegetation composition and structure from seedlings to trees indicate the presence of a progressive ecological filtering mechanism. A Dunn post-hoc test with Holm correction revealed significant differences at the early-late successional transition across multiple strata combinations (Table 2).

Table 1. Top ten species with high Important Value Index (IVI) in each growth stage

Growth stage	Rank	Species	Family	RD (%)	RF (%)	RBA (%)	IVI (%)	Ecological Guild
Seedling	1	*Rubroshorea curtisii*	Dipterocarpaceae	13.04	8.33		21.38	Shade tolerant
	2	*Calophyllum pulcherrimum*	Calophyllaceae	11.96	8.33		20.29	Shade tolerant
	3	*Gironniera parvifolia*	Cannabaceae	8.70	6.25		14.95	Pioneer
	4	*Koompassia malaccensis*	Fabaceae	6.52	8.33		14.86	Shade tolerant
	5	*Eurycoma longifolia*	Simaroubaceae	6.52	6.25		12.77	Pioneer
	6	*Dipterocarpus rigidus*	Dipterocarpaceae	4.35	6.25		10.60	Shade tolerant
	7	*Hopea beccariana*	Dipterocarpaceae	4.35	6.25		10.60	Shade tolerant
	8	*Gironniera nervosa*	Cannabaceae	4.35	4.17		8.51	Shade tolerant
	9	*Syzygium densiflorum*	Myrtaceae	4.35	4.17		8.51	Shade tolerant
	10	*Ochanostachys amentacea*	Olacaceae	3.26	4.17		7.43	Shade tolerant
Sapling	1	*Calophyllum pulcherrimum*	Calophyllaceae	25.45	25.00		50.45	Shade tolerant
	2	*Hopea beccariana*	Dipterocarpaceae	23.64	25.00		48.64	Shade tolerant
	3	*Dipterocarpus coriaceus*	Dipterocarpaceae	21.82	12.50		34.32	Shade tolerant
	4	*Rubroshorea curtisii*	Dipterocarpaceae	10.91	12.50		23.41	Shade tolerant
	5	*Eurycoma longifolia*	Simaroubaceae	9.09	12.50		21.59	Pioneer
	6	*Polyalthia hypoleuca*	Annonaceae	9.09	12.50		21.59	Shade tolerant
Pole	1	*Santiria laevigata*	Burseraceae	15	16.67	17.44	49.10	Shade tolerant
	2	*Gironniera parvifolia*	Cannabaceae	10	11.11	13.18	34.29	Pioneer
	3	*Rubroshorea curtisii*	Dipterocarpaceae	10	11.11	9.72	30.83	Shade tolerant
	4	*Eurycoma longifolia*	Simaroubaceae	10	11.11	6.54	27.65	Pioneer
	5	*Hopea beccariana*	Dipterocarpaceae	10	5.56	9.33	24.89	Shade tolerant
	6	*Calophyllum pulcherrimum*	Calophyllaceae	10	5.56	8.56	24.12	Shade tolerant
	7	*Dillenia grandifolia*	Dilleniaceae	5	5.56	6.24	16.79	Pioneer
	8	*Mesua nuda*	Calophyllaceae	5	5.56	6.24	16.79	Shade tolerant
	9	*Sandoricum beccarianum*	Melliaceae	5	5.56	6.24	16.79	Pioneer
	10	*Syzygium* sp.	Myrtaceae	5	5.56	6.24	16.79	Shade tolerant
Tree	1	*Koompassia malaccensis*	Fabaceae	15.22	10.00	34.20	59.42	Shade tolerant
	2	*Rubroshorea curtisii*	Dipterocarpaceae	8.70	7.50	17.81	34.00	Shade tolerant
	3	*Shorea parvifolia*	Dipterocarpaceae	8.70	7.50	5.91	22.11	Shade tolerant
	4	*Santiria laevigata*	Burseraceae	6.52	7.50	1.85	15.87	Shade tolerant
	5	*Calophyllum pulcherrimum*	Calophyllaceae	4.35	5.00	4.05	13.40	Shade tolerant
	6	*Syzygium* sp.	Myrtaceae	4.35	5.00	1.58	10.92	Shade tolerant
	7	*Bouea oppositifolia*	Anacardiaceae	4.35	5.00	1.26	10.60	Shade tolerant
	8	*Ochanostachys amentacea*	Olacaceae	4.35	2.50	2.34	9.18	Shade tolerant
	9	*Endospermum diadenum*	Euphorbiaceae	2.17	2.50	4.37	9.04	Pioneer
	10	*Vitex cofassus*	Lamiaceae	2.17	250	3.86	8.53	Pioneer

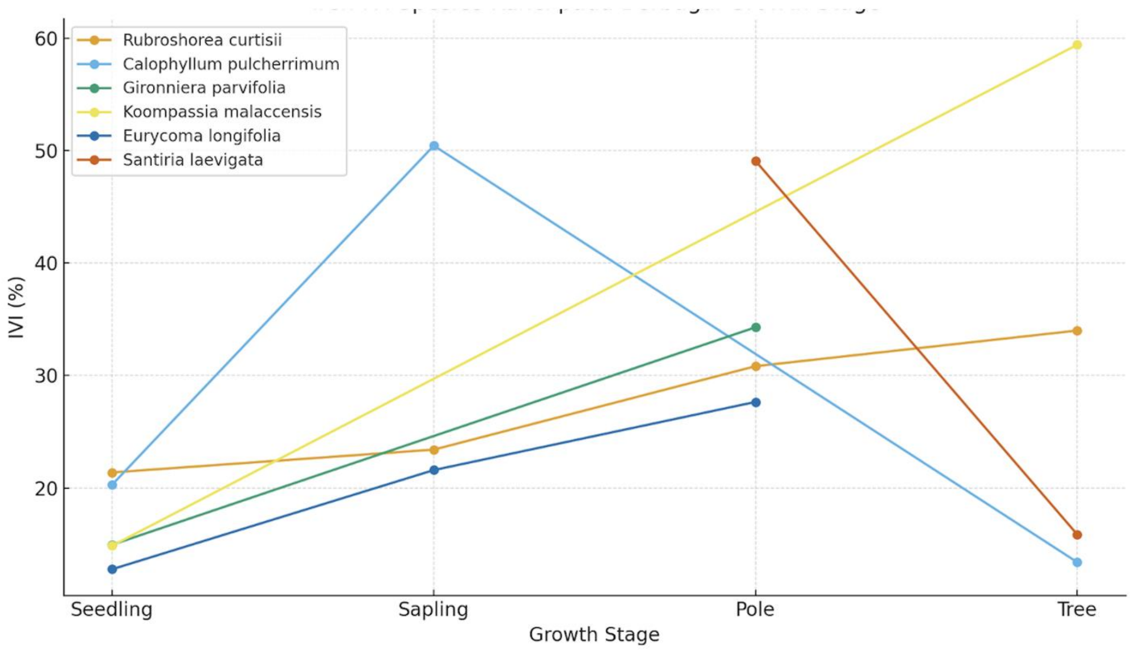

Figure 2. Important Value Index (IVI) Shifting of dominant tree trend based on growth stages

Table 2. Results of the Dunn post-hoc test with Holm adjustment for comparing species importance values (IVI) across growth strata

Comparison	p-value (Holm)	Significance
Seedling vs sapling	0.021	Significant (p<0.05)
Seedling vs pole	0.064	Not Significant
Seedling vs tree	0.008	Very Significant (p<0.01)
Sapling vs pole	0.327	Not Significant
Sapling vs tree	0.041	Significant (p<0.05)
Pole vs tree	0.033	Significant (p<0.05)

These findings confirm that dipterocarp forest vegetation communities go through two major filtering phases those were (i) during the early regeneration transition, when many pioneer species are replaced by shade-tolerant species, and (ii) during the transition to mature stands, when dominance shifts to large canopy species with high basal area. Meanwhile, in the middle growth strata, community differences were rather minor, indicating some compositional stability prior to large structural shifts. From the perspective of ecology, these statistical data demonstrate that the IVI is an effective indicator for detecting ecosystem transition stages. The Kruskal-Wallis test found considerable overall variance, whereas the post-hoc Dunn test revealed that differences were predominantly concentrated between early and late successional stages. This is consistent with long-term studies in dipterocarp forests, which indicated that tree community composition varies drastically during the early regeneration and mature canopy stages but remains rather constant in the interim (Prohaska et al. 2023).

These findings are consistent with the pattern of tropical forest regeneration, which is typically characterized by an initial phase of intense competition for light, followed by a relatively stable intermediate phase, and finally a shift toward climax species dominance (Both et al. 2019; Do et al. 2020). Statistical analysis results further support the hypothesis that succession is not a linear process, but rather a series of "tipping points" indicated by large variations in species composition (Di Biase et al. 2021). In the Bukit Tiban Forest, logging, fragmentation, and land use change have the potential to interrupt these critical stages, resulting in a regeneration trap in which climax species fail to replace pioneers. This state hastens the loss of forest structure and affects ecosystem functions like carbon storage and biodiversity protection (Prohaska et al. 2023; Yatar et al. 2024). To ensure successional continuity in Bukit Tiban, conservation activities must focus on protecting important regeneration phases by assisted natural regeneration and enrichment planting of key species such as *R. curtisii* and *K. malaccensis.*

Forest structure

Forest stand structure is a spatial representation of the composition, size, and distribution of trees within a forest community, both vertically (height or canopy strata) and horizontally (stem diameter). This structure reflects the stages of succession, regeneration dynamics, and the stability of the forest ecosystem. A healthy stand structure is characterized by the presence of trees of various diameter and height classes, from seedlings, saplings, poles, to mature trees, forming a tiered canopy layer. This pattern allows for a stable microclimate, supports biodiversity, and maintains ecological functions such as carbon storage, water absorption, and wildlife habitat.

The tree horizontal structure in the Bukit Tiban Forest has an inverted J-shape pattern (Figure 3), with a large number of individuals in small diameter classes and a dramatic fall as diameter class size increases. This pattern occurs often in tropical rainforests and implies that natural regeneration continues, with individuals recruited in small diameter classes and only a small proportion surviving to large diameter classes. This is in line with the findings of the species importance value (IVI) analysis, which revealed that shade-tolerant dipterocarp species played the major role in maintaining populations in the early strata before cementing dominance in greater diameter classes.

The Kruskal-Wallis test for the distribution of individuals across diameter classes showed a significant difference (H: 11.24, p: 0.041). This demonstrates that tree distribution across diameter classes is not uniform, but rather follows the ecological stratification typical of dipterocarp forests. A post-hoc Dunn's test with Holm's correction revealed that the most significant difference was between the small diameter classes (5-15 cm) and the large diameter classes (>50 cm; $p<0.01$). This difference confirms the existence of a growth bottleneck, where only a few individuals successfully survive the intense competition to reach the canopy stage. In contrast, the comparison between the middle classes (15-30 cm and 30-50 cm) was not significant ($p>0.05$), indicating relative stability in the middle growth phase before strong selection occurs in the final phase.

These findings align with research in the dipterocarp forests of Sabah and Kalimantan, which also reported uneven diameter distribution patterns, with the loss of large trees due to anthropogenic pressure being the main differentiating factor between primary and disturbed forests (Matsuo et al. 2021; Prohaska et al. 2023). Similarly, Ediriweera et al. (2020) emphasized that the presence of large-diameter trees is crucial for ecosystem function, particularly in carbon storage. In the context of Bukit Tiban Forest degradation, the dominance of small-diameter classes implies continued regeneration, whereas the scarcity of large trees shows selective logging and fragmentation. Zhou et al. (2023) study found that forest disturbance causes the loss of large-diameter trees and uneven spatial distribution, leading to a decline in forest structural and functional diversity. Similarly, Hayward et al. (2021) emphasized that the dominance of pioneer species after disturbance can slow vertical recovery and prolong the early successional phase, meaning key forest ecological functions are not immediately restored. This has the potential to produce a regeneration trap in which climax species like *R. curtisii* and *K. malaccensis* are unable to replace opportunistic pioneers due to a lack of larger broodstock. If left unchecked, the stand structure will transition to the dominance of secondary species with lesser ecological and economic value, accelerating forest degradation. Thus, statistical analysis not only supports the diameter structure pattern, but also stresses the need of retaining large-diameter trees to enable a continuous regeneration cycle (Sakai et al. 2022). To ensure the viability of the BTPF stand structure, management techniques should focus on safeguarding parent trees, decreasing anthropogenic pressure, and undertaking assisted natural regeneration and enrichment planting of essential species.

The vertical structure of a forest refers to the arrangement of plant layers from the forest floor to the upper canopy, formed by trees, shrubs, and undergrowth of varying heights. This structure reflects the stratification of the vegetation canopy, typically divided into several main layers: seedlings (0-1 m), saplings (1-4 m), poles (5-10 m), mid-trees (10-30 m), and canopy/emergent trees (>30 m). The presence of these various strata is an important indicator in assessing ecosystem complexity, the sustainability of natural succession, and the forest's ability to support biodiversity and other ecosystem functions.

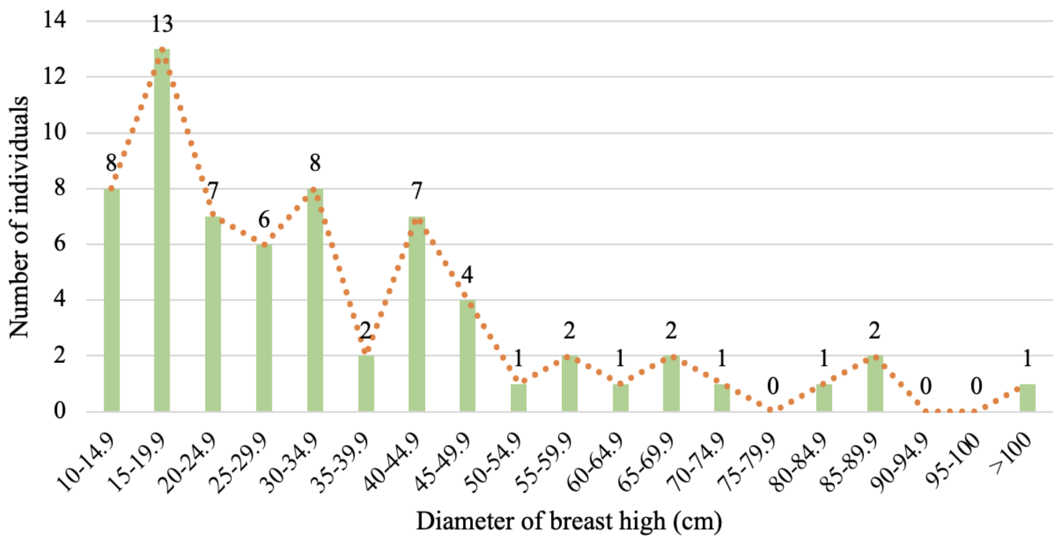

Figure 3. Horizontal forest structure in research area

Figure 4 shows a predominance of individuals in the lower strata, particularly in strata E (0-1 m) with 92 individuals and strata D (1-4 m) with 55 individuals. Meanwhile, the number of individuals in higher strata decreased drastically, with only 27 individuals in strata C (4-10 m), 14 individuals in strata B (10-30 m), and only 5 individuals in strata A (>30 m). This pattern shows active natural regeneration, as there are numerous juvenile individuals capable of replacing mature trees. However, the low quantity of large-diameter trees suggests that the primary canopy stock is rather limited, implying that the forest's continuing structural function is dependent on the successful transition of small individuals to higher strata.

The Kruskal-Wallis test indicated a significant difference in strata values (χ^2: 15.72, df: 3, p: 0.0012). Dunn's post-hoc analysis with Holm's correction revealed significant differences in dominance between tree and seedling strata (p: 0.002) and between trees and saplings (p: 0.015), however the tree-sapling stratum did not exhibit significant differences (p: 0.118). This pattern aligns with the vertical structure histogram, indicating a greater number of individuals in the lower strata, whereas ecological dominance is predominantly situated in the tree strata. Ecologically, a complete vertical structure from bottom to top represents the complexity of the environment and promotes high biodiversity. The lower strata (seedlings and saplings) provide habitat for understory species, whilst the top strata (canopy trees) control microclimate conditions, light availability, and nutrient cycling. If any of the strata, particularly the canopy strata, are severely decreased, ecological stability may be threatened. The viability of the canopy community is thus not reflected in the mix of dominating species in the lower strata, even in the face of regeneration. The role of dominant species in sustaining ecosystem structure, including the canopy's function as habitat providers, microclimate regulators, and water cycle controllers, is indicated by high IVI values in the tree strata from an ecological perspective. Nevertheless, the potential to establish a "regeneration bottleneck" in the seedling and sapling strata due to the restricted regeneration representation of these dominant species has the potential to diminish structural diversity in the long term. This finding is consistent with research conducted in dipterocarp forests in Malaysia (Sawada et al. 2021), which demonstrated a discrepancy between the dominance of tree strata and the regeneration of the understory. Studies in Southeast Asian dipterocarp forests show similar patterns. Okuda et al. (2019) found a reverse-J distribution in post-logging forests in Vietnam, indicating active regeneration but a significant reduction in the number of large canopy trees. Ng et al. (2022) emphasized that imbalances between strata are indicators of ecological stress, where the abundance of seedlings does not necessarily guarantee continued regeneration due to light limitations and competition. Ganivet et al. (2022) showed that dipterocarp seedlings vary in shade and drought tolerance, so high seedling numbers are only meaningful if the species can survive through the tree strata. Meanwhile, Prohaska et al. (2023) noted that low numbers of large trees in dipterocarp forests indicate long-term structural degradation, which could shift community dominance in the absence of management interventions. Thus, while the vertical structure found in this study implies high quantitative regeneration, it still creates problems with ensuring the sustainability of canopy species populations.

Sawada et al. (2021) found that changes in stand structure in forests dominated by *Shorea* species occurred as a result of forest degradation, with the number of seedlings falling dramatically as the canopy changed. Canopy trees, particularly those from the Dipterocarpaceae family, have a high IVI in the mature tree strata. The loss of large individuals leads to a decline in the IVI in the upper strata, thus diminishing their ecological role in regulating the microclimate, nutrient cycling, and providing habitat for canopy fauna. High regeneration in the lower strata (seedlings-saplings) may reflect recovery potential, but other studies in dipterocarp forests have shown that seedling dominance does not necessarily guarantee long-term regeneration success (Okuda et al. 2019; Ediriweera et al. 2020). This is especially true in degraded habitats, where many seedlings are dominated by pioneer species and are less able to replace the ecological role of canopy dipterocarps. Ng et al. (2022) even emphasize that imbalances between strata are often an indicator of ecological stress due to habitat degradation. If habitat destruction continues, this regeneration may not produce an intact dipterocarp forest, but rather become dominated by small-diameter pioneer species. As a result, management measures are required, such as safeguarding remaining large trees, controlling disturbance, and restoring habitat so that regeneration can lead to the re-establishment of a stable vertical structure.

Ecological index in Bukit Tiban Protected Forest

Species diversity is controlled by various factors, including the quality of resources consisting of habitat and the number of niches controlled by resources (Ruziman et al. 2022). Among the many ways that can be done to evaluate species diversity in an ecosystem, the diversity index, species evenness index, and species richness index are the most commonly used. Based on the results of the study, it is known that the diversity of tree species in the BTPF varies at all stages of growth (Figure 5).

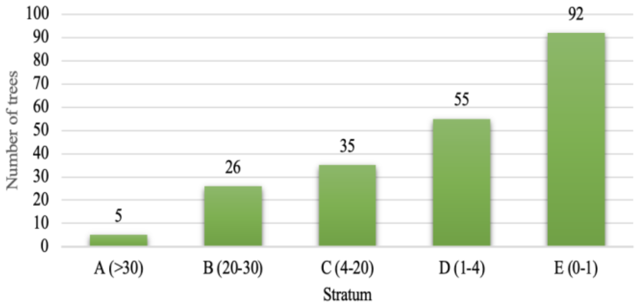

Figure 4. Vertical forest structure in research area

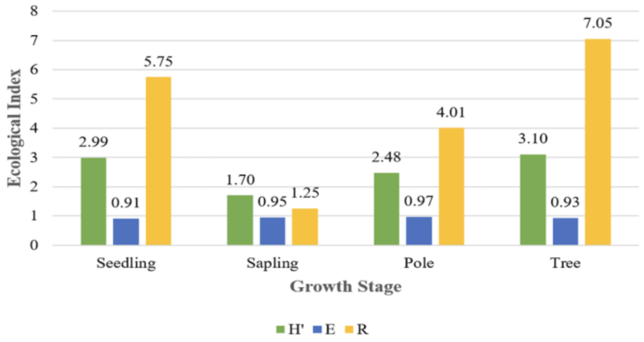

Figure 5. Ecological index of tree species in research location

Research reveals distinct variations in ecological index values across different growth strata. The tree strata exhibited the highest species diversity (H': 3.10), suggesting a complex and stable community structure within the canopy layer. In contrast, the sapling strata recorded the lowest diversity (H': 1.70), reflecting limited natural regeneration during the middle growth phase. The evenness index (E) ranged from 0.91 to 0.97 across all strata, signifying a relatively uniform distribution of individuals among species, with no excessive dominance by any single species. The species richness index (R) exhibited variation, with the maximum value recorded in the tree strata (R: 7.05) and the minimum in the sapling strata (R: 1.25), thereby reinforcing the presence of a regeneration gap during the middle growth stage. When compared to other research conducted in Asian tropical forest ecosystems, the H' value in the tree strata at Tiban (3.10) exceeds the value range documented by Visitthisath et al. (2021) in tropical forests of Thailand (3.28) and Thanh et al. (2021) in dry forests of Vietnam. The higher H' value suggests that, despite anthropogenic pressures, the mature tree community at Bukit Tiban maintains considerable species diversity. The evenness value (E: 0.91-0.97) exceeds the range reported by Muluneh et al. (2021) for dipterocarp regeneration (0.80-0.92), suggesting a more balanced species distribution at the study site. The species richness in the tree strata (R: 7.05) surpasses the general range documented in comparable studies (2-6), thereby emphasizing the significance of this ecosystem as a crucial habitat for diverse forest species.

The ecological interpretation of these findings indicates that dominant species, especially *R. curtisii*, significantly influence community structure, as evidenced by elevated IVI values across nearly all strata. The observed discrepancy between high diversity in the tree strata and low diversity in the regenerative strata indicates a potential barrier to regeneration. This aligns with findings from a study conducted in a Vietnamese dipterocarp forest (Thanh et al. (2021), indicating that anthropogenic disturbance may diminish regenerative diversity, despite the mature community maintaining its diversity. Milodowski et al. (2021) found that a balanced community structure in the upper strata does not ensure ongoing regeneration, particularly when light competition restricts the growth of canopy species The importance of keystone species with high IVI in preserving the stability of tropical ecosystems is highlighted by several recent research (Prohaska et al. 2023). Failure of dominant species to regenerate will result in shifts in community structure and a decline in long-term ecological function. These results highlight the necessity of enrichment planting strategies within conservation efforts to enhance regenerative strata, especially involving ecologically significant Dipterocarp species like *R. curtisii.* Protecting natural habitats, restoring canopy strata, and involving the community in management are essential measures for ensuring sustainable ecosystem function in Bukit Tiban.

This condition demonstrates the role of *R. curtisii* as a dominant species with a high IVI that sustains community structure. The limited diversity and richness in the regenerative strata indicate a hindrance to regeneration, which, if unaddressed, may diminish the ecological function of *R. curtisii* over time. Consequently, pertinent conservation strategies involve safeguarding existing mature trees, rehabilitating the canopy layers, and implementing enrichment planting with ecologically significant Dipterocarp species to enhance the regenerative layers. Involving local communities in habitat monitoring and management is essential for mitigating anthropogenic pressures, including fragmentation (Prohaska et al. 2023). Ecological index values reflect the structural condition of the community and offer straightforward recommendations for sustainable conservation strategies.

Threat in Bukit Tiban Protected Forest

Based on interview with stakeholder *R. curtisii* in BTPF faces significant anthropogenic threats, primarily habitat fragmentation, increased disease susceptibility, altered forest structure and composition, and low natural regeneration rates. This finding also supported by previous research conducted by Firdamayanti et al. 2024, which found anthropogenic activities such as conversion for settlements, agriculture and industry. Furthermore, Yuliasrin (2019) also found another activity such as land clearing on steep slopes has created critical land, increasing the risk of landslides in the area also has caused significant forest degradation (Fambayun and Kalima 2020). The BTPF also close to human settlement, this condition has led to agricultural land conversion, reducing forest cover and depleting tree structure, particularly at the sapling and pole stages, due to harvesting for construction materials. Some part of research location also ecotourism area. Recreational activities such as hiking and camping, managed by local communities, contribute to ecosystem degradation. Visitors often unintentionally damage trees through machete cuts and trail clearing, leading to seedling loss along hiking paths. The lack of clear regulations governing these activities further exacerbates forest resource exploitation, increasing physical damage to trees and heightening disease vulnerability. Another major concern is the absence of mass flowering events in *R. curtisii* since 2015. In Dipterocarpaceae, mass flowering is typically triggered by low temperatures and extreme drought linked to El Niño cycles (Yasuda et al. 1999; Numata et al. 2003). However, small islands experience frequent climate anomalies, disrupting flowering patterns and limiting reproductive opportunities. Without flowering events, seed production and dispersal are severely constrained, impeding natural regeneration. Furthermore, even when fruiting occurs, regeneration remains challenging due to high seed predation by wildlife, further reducing germination success. Agricultural expansion has also significantly impacted pollinator populations, particularly through pesticide use, which may disrupt natural pollination processes. Further research is needed to assess these ecological disruptions and develop conservation strategies to sustain *R. curtisii* regeneration amid dynamic environmental changes.

Sustainable management strategies must be implemented for managing the species. Strengthening legal protections against illegal logging and land conversion, along with designating core conservation zones, is critical. Reforestation and assisted regeneration efforts should focus on replanting dominant and threatened species, supported by community-based nurseries. Fire management measures, including firebreaks and controlled burns, can mitigate wildfire risks, while community engagement through education and eco-tourism initiatives can reduce dependency on forest resources. Climate adaptation strategies, such as monitoring flowering patterns and restoring pollinator habitats, are essential to enhance resilience. Finally, long-term research and monitoring of soil recovery, species interactions, and ecological indices will help track restoration progress. By integrating these approaches, the BTPF can recover its biodiversity, ensuring the survival of keystone species and the ecological services they provide for future generations.

ACKNOWLEDGEMENTS

The author is grateful to thank the Universitas of Sumatera Utara, Indonesia, for supporting this research under the Talenta Research Grant through International Collaborative Research Scheme Number: 18589/UN5.1.R/PPM/2024, dated May 30, 2024. The author would also like to thank the Bukit Tiban Protected Forest Management, Riau Islands, Indonesia, for providing permission to conduct the research.

REFERENCES

Anjos LJS, Pereira NGP, Alves STM, Jose RDSA, de Souza LA. 2021. Forest remnants in private lands are critical to the persistence of endangered birds in an Amazonian hotspot. J Nat Conserv 61: 1617-1381. DOI: 10.1016/j.jnc.2021.125984.

Appanah S, Turnbull JM. 1998. A Review of Dipterocarps: Taxonomy, Ecology and Silviculture. Center for International Forestry Research (CIFOR) Indonesia, Bogor.

Ashton PS, Heckenhauer J. 2022. Tribe *Shoreae* (Dipterocarpaceae subfamily Dipterocarpoideae) finally dissected. Kew Bull 77: 885-903. DOI: 10.1007/s12225-022-10057-w.

Ashton PS. 1982. Dipterocarpaceae. In: Steenis (eds). Flora Malesiana Series I. National Herbarium of the Netherlands, Netherlands.

Awoke N, Nemomissa S, Bekele T. 2022. Floristic composition and diversity of vascular plants in remnant patches of arsi-bale massif of Oromia Regional State, Ethiopia. Intl J Ecol 2022 (2): 1-11. DOI: 10.1155/2022/6693679.

Barstow M. 2018. *Shorea curtisii*. The IUCN Red List of Threatened Species 2018: e.T33463A68073253. DOI: 10.2305/IUCN.UK.2018-1.RLTS.T33463A68073253.en.

Bodo T, Gimah BG, Seomoni KJ. 2021. Deforestation and habitat loss: human causes, consequences and possible solutions. J Geogr Res 4: 22-30. DOI: 10.30564/jgr.v4i2.3059.

Both S, Riutta T, Paine CET, Elias DMO, Cruz RS, Jain A, Johnson D, Kritzler UH, Kuntz M, Majalap N, Mielke N, Montoya-Pfeiffer P, Ostle NJ, Arn Teh YH, Arnold A, Burslem DFRP, Majalap-Lee N, McGuire KL, Turner BL. 2019. Logging and soil nutrients independently explain plant trait expression in tropical forests. New Phytol 226 (3): 571-582. DOI: 10.1111/nph.16392.

Cadavid-Florez L, Laborde J, Mclean DJ. 2020. Isolated trees and small woody patches greatly contribute to connectivity in highly fragmented tropical landscapes. Landsc Urban Plan 196: 103745. DOI: 10.1016/j.landurbplan.2020.103745.

Conover WJ. 1999. Practical Nonparametric Statistics (3rd ed.). John Wiley & Sons. New York.

Curtis JT, McIntosh RP. 1950. The interrelations of certain analytical and synthetic phytosociological characters. Ecology 31 (3): 434-455. DOI: 10.2307/1931497.

De Frenne P, Lenoir JRMH, Luoto M, Scheffers B, Zellweger F. 2021. Forest microclimates and climate change: Importance, drivers and future research agenda. Glob Change Biol 27 (11): 2279-2297. DOI: 10.1111/gcb.15569.

Di Biase L, Pace L, Mantoni C, Fattorini S. 2021. Variations in plant richness, biogeographical composition, and life forms along an elevational gradient in a Mediterranean Mountain. Plants 10: 2090. DOI: 10.3390/plants10102090.

Do HTT, Grant JC, Zimmer HC, Trinh BN, Nichols JD. 2020. Site conditions for regeneration of climax species, the key for restoring moist deciduous tropical forest in Southern Vietnam. Plos One 15 (5): e0233524. DOI: 10.1371/journal.pone.0233524.

Ediriweera S, Bandara C, Woodbury DJ, Gunatilleke XMIAUN, Gunatilleke CVS, Ashton MS. 2020. Changes in tree structure, composition, and diversity of a mixed-dipterocarp rainforest over a 40-year period. For Ecol Manag 458: 117764. DOI: 10.1016/j.foreco.2019.117764.

Fambayun RA, Kalima T. 2020. Population of potential rattan in Bukit Tiban protection forest, Batam, Indonesia. Jurnal Penelitian Kehutanan Wallacea 9 (2): 93-109. DOI: 10.18330/jwallacea.2020.vol9iss2pp93-109.

Faria D, Delabie JHC, Dias MH. 2021. The Hileia Baiana: An assessment of natural and historical aspects of the land use and degradation of the Central Corridor of the Brazilian Atlantic Forest. In: Marques MCM, Grelle CEV. The Atlantic Forest (eds). Springer International Publishing, Cham.

Firdamayanti A, Alawy F, Saifimar C, Galingging A, Wijaya K, Rachmat HH. 2024. Fragmented forests in highly industrialized Batam Island, Indonesia: A vital pocket for biodiversity conservation of *Rubroshorea curtisii* (Dipterocarpaceae). IOP Conf Ser Earth Environ Sci 1433: 012009. DOI: 10.1088/1755-1315/1433/1/012009.

Ganivet E, Unggang J, Bodos V, Demies M, Ling CY, Sang J, Bloomberg M. 2020. Assessing tree species diversity and structure of mixed dipterocarp forest remnants in a fragmented landscape of north-western Borneo, Sarawak, Malaysia. Ecol Indic 112: 106117. DOI: 10.1016/j.ecolind.2020.106117.

Hammer Ø, Harper DAT, Ryan PD. 2001. PAST: Paleontological statistics software package for education and data analysis. Palaeontol Electron 4 (1): 1-9.

Hammond ME, Pokorný R, Okae-Anti D, Gyedu A, Obeng IO. 2021. The composition and diversity of natural regeneration of tree species in gaps under different intensities of forest disturbance. J For Res 32: 1843-1853. DOI: 10.1007/s11676-020-01269-6.

Hayward RM, Banin LF, Burslem DFRP, Chapman DS, Philipson CD, Cutler MEJ, Reynolds G, Nilus R, Dent DH. 2021. Three decades of post-logging tree community recovery in naturally regenerating and actively restored dipterocarp forest in Borneo. For Ecol Manag 488: 119036. DOI: 10.1016/j.foreco.2021.119036.

Kowarik I, von der Lippe M. 2017. Plant population success across urban ecosystems: A framework to inform biodiversity conservation in cities. J Appl Ecol 55: 2354-2361. DOI: 10.1111/1365-2664.13144.

Lepun P, Heng RKJ. 2020. Floristic and forest structure of hill mixed dipterocarp forest at Bukit Kana National Park, Sarawak, Malaysia. Malays For 83 (2): 259-280.

Magurran AE. 2004. Measuring Biological Diversity. Blackwell Publishing, Oxford.

Malik S. 2025. Comparative analyses and phylogenetic dependence in traits and trends of Dipterocarpaceae. Ecol Evol 15 (3): e70784. DOI: 10.1002/ece3.70784.

Malkinson D, Kopel D, Wittenberg L. 2018. From rural-urban gradients to patch-matrix frameworks: plant diversity patterns in urban landscapes. Landsc Urban Plan 169: 260-268. DOI: 10.1016/j.landurbplan.2017.09.021.

Matsuo T, Martínez-Ramos M, Bongers F, van der Sande MT, Poorter L. 2021. Forest structure drives changes in light heterogeneity during tropical secondary forest succession. J Ecol 109: 2871-2884. DOI: 10.1111/1365-2745.13680.

Milodowski DT, Coomes DA, Swinfield T, Disney MI, Muir J, Burt A, Wilkes P, Nilus R, Bin Jami J, Cutler MEJ. 2021. The impact of logging on vertical canopy structure across a gradient of tropical forest degradation intensity in Borneo. J Appl Ecol 58 (8): 1764-1775. DOI: 10.1111/1365-2664.13895.

Muluneh MG, Feyissa MT, Wolde TM. 2021. Effect of forest fragmentation and disturbance on diversity and structure of woody species in dry Afromontane forests of northern Ethiopia. Biodivers Conserv 30: 1753-1779. DOI: 10.1007/s10531-021-02167-x.

Nath B, Wang Z, Ge Y, Islam K, Singh RP, Niu Z. 2020. Land use and land cover change modeling and future potential landscape risk assessment using Markov-CA model and analytical hierarchy process. ISPRS Intl J Geo-Inform 9 (2): 134. DOI: 10.3390/ijgi9020134.

Newman MF, Burgess PF, Whitmore TC. 1999. Pedoman Identifikasi Pohon-Pohon Dipterocarpaceae Sumatera. Prosea Indonesia, Bogor. [Indonesian]

Ng CH, Ng KKS, Lee SL, Alias NN. 2024. Genetic diversity and population structure of *Rubroshorea curtisii*, a hill dipterocarp species: implications for conservation. Tree Genet Genom 20: 16. DOI: 10.1007/s11295-024-01649-7.

Ng F, Zubir I, Zurfatiha Z, Sharma D, Latiff A, Hazrina Z, Shahira A, Shah-Redza H. 2022. Speciation in the Malay Peninsula in relation to dipterocarp forest height, structure, turnover, and reproductive biology. J Trop For Sci 34 (1): 103-113.

Numata S, Kachi N, Okuda T, Manokaran N, Ashton PS. 2003. Temporal and spatial patterns of mass flowerings on the Malay Peninsula. Am J Bot 90 (7): 1025-1031. DOI: 10.3732/ajb.90.7.1025.

O'Connor MI, Gonzalez A, Byrnes JEK, Cardinale BJ, Duffy JE, Gamfeldt L, Griffin JN, Hooper D, Hungate BA, Panquette A, Thompson PL, Dee LE, Dolan KL. 2017. A general biodiversity function relationship is mediated by trophic level. Oikos 126 (1): 18-31. DOI: 10.1111/oik.03652.

Okuda T, Yamada T, Hosaka T, Miyasaku N, Hashim M, Lau AMS, Saw LG. 2019. Canopy height recovery after selective logging in a lowland tropical rain forest. For Ecol Manag 442: 117-123. DOI: 10.1016/j.foreco.2019.03.045.

Paul A, Khan MF, Das AK. 2019. Population structure and regeneration status of rhododendrons in temperate mixed broad-leaved forests of western Arunachal Pradesh, India. Geol Ecol Landsc 3: 168-186. DOI: 10.1080/24749508.2018.1525671.

Prohaska A, Seddon AWR, Rach O, Smith A, Sachse D, Willis KJ. 2023. Long-term ecological responses of a lowland dipterocarp forest to climate changes and nutrient availability. New Phytol 240: 2513-2529. DOI: 10.1111/nph.19169.

R Core Team. 2023. R: A Language and Environment for Statistical Computing. https://cran.r-project.org/doc/manuals/r-release/fullrefman.pdf.

Romadini NP, Indrioko S, Widiyatno, Faridah E. 2022. Regeneration failure and seedling growth of Dipterocarpus gracilis, a vulnerable dipterocarp in a tropical monsoon forest in Central Java, Indonesia. J Trop For Sci 32 (1): 45-58. DOI: 10.13057/biodiv/d230963.

Rüger N, Schorn ME, Kambach S, Chazdon RL, Farrior CE, Meave JA, Muñoz R, vanBreugel M, Amissah L, Bongers F, Craven D, Hérault B, Jakovac CC, Norden N, Poorter L, van der Sande MT, Wirth C, Delgado D, Dent DH, Lopez OR. 2023. Successional shifts in tree demographic strategies in wet and dry Neotropical forests. Glob Ecol Biogeogr 32: 1002-1014. DOI: 10.1111/geb.13669.

Ruziman HH, Ismail A, Pardi F, Mohd-Hatta SK. 2022. Edaphic influences on tree species composition and community structure in a secondary-lowland Dipterocarp forest of Kota Damansara Forest Reserve, Selangor. Malays Appl Biol 51 (4): 11-21. DOI: 10.55230/mabjournal.v51i4.06.

Sadeghi SM, Hanum IF, Razali WMW, Kudus KA, Hakeem KR. 2014. Tree composition and diversity of a hill dipterocarp forest after logging. Malay Nat J 66 (4): 1-5.

Sakai A, Arbain A, Sugiarto S, Rahmawati K, Mirmanto E, Takahashi M, Ueda A. 2022. Composition and diversity of tree species after fire disturbance in a lowland tropical forest in East Kalimantan, Indonesia. Biodiversitas 23 (3): 1576-1587. DOI: 10.13057/biodiv/d230348.

Sawada Y, Imai N, Takeshige R, Kitayama K. 2021. Relationships between tree-community composition and regeneration potential of Shorea trees in logged-over tropical rain forests. J For Res 27 (3): 222-229. DOI: 10.1080/13416979.2021.2015830.

Shu H, Xiao C, Ma T, Sang W. 2021. Ecological health assessment of Chinese National Parks based on landscape pattern: A case study in Shennongjia National Park. Intl J Environ Res Public Health 18: 11487. DOI: 10.3390/ijerph182111487.

Soepadmo E, Wong KM. 2004. Tree Flora of Sabah and Sarawak (Vol. 5: Dipterocarpaceae). Forest Research Institute Malaysia, Kuala Lumpur.

Sosef MSM, Dauby G, Blach-Overgaard A et al. 2017. Exploring the floristic diversity of tropical Africa. BMC Biol 15 (1): 15. DOI: 10.1186/s12915-017-0356-8.

Suhaimi AH, Kobayashi MJ, Satake A, Ng CC, Lee SL, Muhammad N, Numata S, Otani T, Kondo T, Tani N, Yeoh SH. 2023. An ecological transcriptome approach to capture the molecular and physiological mechanisms of mass flowering in Shorea curtisii. PeerJ 11: e16368. DOI: 10.7717/peerj.16368.

Susilowati A, Rachmat HH, Yulita KS, Wijaya K. 2023. Floristic composition and structure of *Eurycoma longifolia* habitat in Muka Kuning Nature Tourism Park, Riau Islands, Indonesia. Biodiversitas 24 (5): 2836-2842. DOI: 10.13057/biodiv/d240537.

Susilowati A, Wijaya K, Mawazin, Rachmat HH, Lismayati Y, Kurniawan H, Ginting IM. 2024. Conserving dipterocarps biodiversity in remnant forests of small islands in Batam Island, Indonesia. Biodiversitas 25 (2): 726-733. DOI: 10.13057/biodiv/d250232.

Symington CF. 1943. Foresters' Manual of Dipterocarps. Malayan Forest Records No. 16. Forest Department, Federated Malay States, Kuala Lumpur.

Tani N, Ng CH, Lee SL, Lee CT, Muhammad N, Kondo T, Tsumura Y, Sugiyama S, Niiyama K, Ripin A, Kassim AR, Musa S. 2025. Negative frequency-dependent selection through variations in seedling fitness due to genetic differentiation of parents' pair in a tropical rainforest tree, *Rubroshorea curtisii* (Dipterocarpaceae). Front Genet 16: 1552024. DOI: 10.3389/fgene.2025.1552024.

Thanh-Tuan N, Quang Bao T, Rodríguez-Hernández DI, Gliottone I. 2021. Tree diversity and species composition of tropical dry forests in Vietnam's Central Highlands Region. For Stud 75 (1): 80-103. DOI: 10.2478/fsmu-2021-0013.

Tilahun D, Gashu K, Shiferaw GT. 2022. Effects of agricultural land and urban expansion on peri-urban forest degradation and implications on sustainable environmental management in Southern Ethiopia. Sustainability 14 (24): 16527. DOI: 10.3390/su142416527.

Trozzo K, Munsell J, Niewolny K, Chamberlain JL. 2019. Forest food and medicine in contemporary Appalachia. Southeast Geogr 59: 52-76. DOI: 10.1353/sgo.2019.0005.

Turkis S, Elmas E. 2018. Tree species diversity and importance value of different forest communities in Yenice Forests. Fresenius Environ Bul 27: 4440-4447.

Van Steenis CGGJ. 1983. Flora Malesiana. Noordhoff-Kolff, Jakarta.

Visitthisath K, Setthapun W, Jansri SN. 2021. Vegetation structure diversity and value-added guideline corresponding to biodiversity for Phu Hung community forest, Thum Tong Subdistrict, Mueang Nan District, Nan Province, Thailand. Turk J Comput Math Educ 12 (8): 2739-2746.

Wardhana HD, Mutaqqin T, Aryanti NA, Kurniawan I. 2022. Potential feeding plants of Javan Langur (*Trachypithecus auratus*) in the eastern slope of Biru Mountain, Batu City, East Java, Indonesia. Biodiversitas 23 (8): 4216-4222. DOI: 10.13057/biodiv/d230845.

Wintle BA, Kujala H, Whitehead A, Cameron A, Veloz S, Kukkala A, Moilanen A, Gordon A, Lentini PE, Cadenhead NCR, Bekessy SA. 2019. Global synthesis of conservation studies reveals the importance of small habitat patches for biodiversity. Proc Natl Acad Sci 116: 909-914. DOI: 10.1073/pnas.1813051115.

Wu MM, Liang Y, He HS, Liu B, Ma T. 2022. Remnant trees location and abundance play different roles in forest landscape recovery. For Ecol Manag 511: 120154. DOI: 10.1016/j.foreco.2022.120154.
Yang J, Yang J, Xing D, Luo X, Lu S, Huang C, Hahs AK. 2021. Impacts of the remnant sizes, forest types, and landscape patterns of surrounding areas on woody plant diversity of urban remnant forest patches. Urban Ecosyst 24: 345-354. DOI: 10.1007/s11252-020-01040-z.
Yasuda M, Matsumoto J, Osada N, Ichikawa S, Kachi N, Tani M, Okuda T, Furukawa A, Nik AR, Manokaran N. 1999. The mechanism of general flowering in Dipterocarpaceae in the Malay Peninsula. J Trop Ecol 15 (4): 437-449. DOI: 10.1017/S0266467499000930.
Yatar C, Thinka S, Sungkaew S, Wachrinrat C, Asanok L, Kamyo T, Hermhuk S, Kachina P, Thongsawi J, Phumphuang W, Yarnvudhi A, Waengsothorn S, Cheysawat S, Marod D. 2024. The dynamics of deciduous dipterocarp forest in relation to climate variability in the Sakaerat Biosphere Reserve, Northeastern Thailand. Biodiversitas 25 (9): 3088-3098. DOI: 10.13057/biodiv/d250914.
Yuliastrin A. 2020. Ancaman Bencana Longsor di Hutan Lindung Bukit Tiban, Batam. Universitas Terbuka UPBJJ-UT, Batam.
Zhou C, Ding Y, Zang R. 2023. Assessing the recovery in species, size and location diversities of a lowland tropical rainforest after shifting cultivation by multiple indices at stand and neighborhood scales. J Environ Manag 341: 118089. DOI: 10.1016/j.jenvman.2023.118089.