

UNIVERSITI PUTRA MALAYSIA

FLORISTIC COMPOSITION, TREE GROWTH AND ABOVEGROUND BIOMASS ALONG AN ALTITUDINAL GRADIENT THIRTY YEARS AFTER SELECTIVE LOGGING AT PAYEH MAGA HIGHLAND, SARAWAK, MALAYSIA

RENEE SHERNA ANAK LAING

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By

RENEE SHERNA ANAK LAING

Thesis Submitted to the School of Graduate Studies, Universiti Putra Malaysia, in Fulfilment of the Requirements for the Degree of Master of Science

November 2020

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Abstract of thesis presented to the Senate of Universiti Putra Malaysia in fulfilment of the requirement for the degree of Master of Science

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November 2020

Chair : Associate Professor Ong Kian Huat, PhD Faculty : Agricultual Science and Forestry (Bintulu Campus)

The tropical mountain forests ecosystem is a hotspot of biodiversity and provides a significant contribution to the humans' livelihoods and is vital for global carbon cycle. However, there is still a lack of information regarding floral diversity and their altered biomass impact on how it has been affected due to a logging operation in Sarawak. A study was conducted to determine vegetation recovery, tree growth rate, above ground biomass (AGB) on the Payeh Maga Highland, Sarawak, Malaysia of logged and unlogged forests at different elevations: 750-1200 m (S1), 1200–1500 m (S2) and >1500 m (S3) in 30 plots (20 x 20 m). Tree height and *dbh* of tree with *dbh* of 10 cm and above were tagged with numbered metal plates and measured. The stand and growth dynamics were assessed for a 12 month duration. A total of 891 trees were enumerated. The trees comprised 55 families, 98 genera and 203 species. The lowest elevation site, S1, contained the highest number of species (117) and families (42), while the highest elevation site, S3, contained the lowest diversity (25 families and 59 species). The most speciose families in S1 were Dipterocarpaceae (11 species) and Lauraceae (10species), while Lauraceae and Fagaceae were the dominant tree families at S2 and S3. Secondary tree species are still abundantly found in the logged plots, especially at S2 (45.5%), indicating that after 30 years, these stands are still in the recovery stages. Significant increments in diameter at breast height (*dbh*), height, basal area and AGB were observed in the S3 between logged-over and primary plots. As elevation increases, diameter at breast height (dbh), basal area, height and stand AGB are decreasing. Lower concentrations of soil nutrients, especially N, may be the main reason for the slow recovery. Basal area and AGB exhibited negative relationships with elevation and rainfall but positively correlated with temperature, soil P and K. In conclusion, the results suggested that the selectively logged forest at the Payeh Maga Highland required more than three decades to recover. Meanwhile stand dynamic variations favored recruitment and increases in density and basal area as well as AGB indicated that the Payeh Maga Highland community is experiencing the construction phase.



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Abstrak tesis yang dikemukakan kepada Senat Universiti Putra Malaysia sebagai memenuhi keperluan untuk Ijazah Master Sains

KOMPOSISI FLORISTIK, BIOJISIM TUMBUHAN DAN PRODUKTIVITI DIRIAN SEPANJANG SATU GRADIEN KETINGGIAN DI TANAH TINGGI PAYEH MAGA, SARAWAK, MALAYSIA

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Ekosistem hutan gunung tropika merupakan kawasan yang mempunyai kepelbagaian biodiversiti dan memberikan sumbangan penting kepada kehidupan manusia dan kitaran karbon dunia. Walau bagaimanapun, masih terdapat kekurangan maklumat mengenai kepelbagaian flora dan mengenai perubahan biomas akibat daripada operasi pembalakan di Sarawak. Satu kajian telah dijalankan untuk menentukan pemulihan tumbuh-tumbuhan, kadar pertumbuhan pokok dan biomas di atas tanah (AGB) di Payeh Maga Highland, Sarawak, Malaysia di hutan yang belum pernah teroka dan hutan yang telah dibalak di ketinggian yang berbeza: 750-1200 m (S1), 1200-1500 m S2) dan > 1500 m (S3) di 30 plot (20 x 20 m). Ketinggian dan diameter pokok yang berukuran 10 cm ke atas telah ditag dengan plat besi yang bernombor dan diukur. Dirian dan pertumbuhan dinamik pokok telah dinilai untuk tempoh selama 12 bulan. Sebanyak 891 pokok telah direkodkan. Pokok-pokok tersebut terdiri daripada 55 famili, 98 genera dan 203 spesies. S1 yang terletak di altitud paling rendah mencatatkan bilangan spesies pokok (117) dan family pokok (42) yang tertinggi, manakala S3 yang terletak di altitud paling tinggi mencatatkan taburan pokok terendah (25 famili dan 59 spesies). Famili yang paling banyak dijumpai di S1 terdiri daripada Dipterocarpaceae (11 spesies) dan Lauraceae (10 spesies). Sementara itu, Lauraceae dan Fagaceae adalah famili pokok dominan di S2 dan S3. Spesies pokok sekunder masih banyak dijumpai di plot yang telah dibalak, terutamanya di S2 (45.5%), menunjukkan selepas 30 tahun, pokokpokok tersebut masih dalam proses pemulihan. S3 menunjukkan, peningkatan yang ketara bagi diameter pada ketinggian dada (dbh), ketinggian pokok, keluasan basal pokok dan AGB di antara plot yang telah dibalak dan plot hutan yang belum pernah diteroka. Peningkatan altitud menyebabkan diameter pada ketinggian dada (dbh), keluasan basal pokok, ketinggian pokok dan AGB pokok semakin berkurangan. Kandungan nutrien tanah yang rendah terutamanya N,

mungkin menyebabkan pemulihan hutan menjadi perlahan. Keluasan basal pokok dan AGB menujukkan hubungan negatif dengan altitud dan hujan tetapi berkorelasi positif dengan suhu, tanah P dan K. Kesimpulannya, keputusan yang diperolehi menunjukkan kawasan hutan yang dibalak di Payeh Maga Highland memerlukan tempoh masa lebih tiga dekad untuk pulih. Sementara itu, variasi pertumbuhan yang dinamik menggalakkan peningkatan dan penambahan pokok dan keluasan basal pokok serta AGB menunjukkan bahawa komuniti Payeh Maga Highland sedang mengalami fasa pertumbuhan.



First and foremost, praise to God for His blessing that I was able to finish my research work with great success. Next, I would like to express my special thanks of gratitude to my supervisor and co-supervisor, Associate Professor Dr. Ong Kian Huat and Associate Professor Dr. King Jie Hung who gave me the opportunity to do my research work under their guidance and support and most of all patience throughout the entire research.

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I certify that a Thesis Examination Committee has met on 24th November 2020 to conduct the final Examination of Renee Sherna Anak Laing on her thesis entitled "Floristic composition, tree growth and aboveground biomass along an altitudinal gradient thirty years after selective logging at Payeh Maga Highland, Sarawak, Malaysia" in accordance with the Universities and University College Act 1971 and the Constitution of the Universiti Putra Malaysia [P.U. (A) 106] 15 March 1998. The committee recommends that the student be awarded the Master of Science.

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LIST OF ABBREVIATIONS

AGB	Aboveground Biomass
Са	Calcium
dbh	Diameter at breast height
E	Pielou's index
FAO	The Food and Agriculture Organization
H'	Shannon-Weiner Index
НоВ	Heart of Borneo
IPCC	Intergovernmental Panel on Climate Change
ΙΤΤΟ	International Tropical Timber Organization
IUCN	International Union for Conservation of Nature
IVI	Importance Value Index
к	Potassium
LIP	Llyod Index of patchiness
meq	One-thousandth of an equivalent of a chemical
Mg	Magnesium
mg	Milligrams
MUS	Malayan Uniform System
Ν	Nitrogen
Р	Phosphorus
REDD	Reducing Emissions from Deforestation and Degradation in Developing Countries
RIL	Reduced Impact Logging
RIL-C	Reduced Impact Logging for Climate Change Mitigation
S1	Site 1
S2	Site 2

S3 Site 3

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- SMS Selective Management System
- TmFO Tropical managed Forests Observatory
- TMFs Tropical Montane Rainforest
- VMR Variance to mean ratio
- WWF World Wide Fund for Nature



CHAPTER 1

GENERAL INTRODUCTION

Tropical forests cover more than 50% of total forests present on the Earth's surface and they contribute ecological, social and livelihood support to the indigenous peoples (Romijn *et al.*, 2015). Over 20% of all tropical forest areas are in Southeast Asian countries namely, Indonesia, Malaysia, Myanmar, Laos, Papua New Guinea and Thailand and Oceanic regions (Butler, 2015). However, tropical forests in Southeast Asia are on the brink of depletion due to logging activities, development of new agricultural fields, development of roads and trails, agroforestry practices and forest fires (Laurance, 2007).

In Malaysia, there are seven major forest types which among it consist of upper dipterocarp forest, montane oak-laurel forest (lower montane) and montane ericaceous forest (upper montane). The montane and sub montane of Malaysia differ according to elevation in their appearance, structure and floral and faunal composition. The upper dipterocarp forest is located at 750 – 1,200 m while the lower montane forest regularly begins at elevation of 1,200 – 1,500 m whereas the upper montane usually starts to take place at elevation > 1,500 m (5th National Report to Convention on Biological Diversity, 2015).

In last few decades, tropical rainforest of Borneo is undergoing speedy depletion including loss of wildlife and biodiversity due to intense logging for timber production and oil palm cultivation (Hansen *et al.*, 2008), which Sarawak and Sabah currently possess only 3% and 8% of pristine forest today respectively (Bryan *et al.*, 2013).

Montane vegetation of Borneo is one of the most diverse montane habitats on Earth as the flora is derived from both Asian and Australian families. Araucariaceae, Clethraceae, Podocarpaceae, Symplocaceae and Theaceae are all families generally found in montane forests. Lower montane elevations start at 1,200 m where at this level the dominance of Fagaceae, Myrtaceae and Lauraceae family take place. Oaks (*Quercus* and *Lithocarpus* spp.) and chestnut (*Castanopsis* spp.), laurels, *Podocarpus, Syzygium* and *Calophyllum* species are common (WWF, 2014). Rhododendrons belongings to Ericaceae family are found on acidic and peat soils upper montane area where it's adapted to harsh upper montane environments (MacKinnon, 1997). According to Richards (1936), upper montane forests share many common species and features of structure and appearance with heath forests.

Nowadays few studies have shown that selectively logging forest can retain it similar biodiversity and ecosystem as primary forest. The processes of tropical forest recovery have been studied for decades, with many authors making the

major conclusion that recovery patterns are wide-ranging, and depend on the initial environmental factors and disturbances (Walker *et al.*, 2010; Walker & del Moral, 2011; Chazdon, 2014). Most studies on tropical forests cover relatively short periods of time after logging (Cannon *et al.*, 1998; Edwards *et al.*, 2012) and the recovery of logged forests usually in need of several decades (Chazdon, 2014). Yet, Zang *et al.* (2010) mentioned that the forest recovery rate of community structure and species composition after selective logging is relatively fast because, to some extent, the forest structure is sustained. However, with intensive logging or repeated logging, the forest recovery process can be deteriorated (Putz *et al.*, 2012; Martin *et al.*, 2015).

Realizing that Bornean highlands are facing multiple threats and lack of information regarding recovery status of highlands forest after logging, WWF initiated a Heart of Borneo (HoB) initiative bring together three countries (Brunei Darussalam, Malaysia (Sabah and Sarawak) and Indonesia) to safeguard and manage forested areas in Borneo highlands located at the centre of Borneo Island covering an area up to 240,000 km². In HoB forest cover area, Sarawak makes up 1.83 million ha of it. The upland rainforest is the largest ecosystem of the HoB by covering an area up to 7.78 M ha. Meanwhile the montane forest in HoB which has been sub-divided into lower montane, upper montane and sub-alpine zones is covering up to 5.30 M ha of area. These forest ecosystems have experienced cover loss around 900,000 ha between the years of 2007 – 2012 (WWF, 2014).

It is acknowledged that adequate information of plant diversity at lowland forest has been widely documented but not at mountain forest, therefore a detailed list of plant diversity at different elevation at highland forest after logging are required to be used as guidance for rehabilitation purpose in addition to ensure the sustainability of highland forests biodiversity. Hence, through this research it is expected to provide better and useful information regarding floristics changes along with the altitudinal after logging activities as only few studies have been done previously at mountain tropical rainforest in Borneo. This study is trying to answer following questions: (1) is there any variation in plant species diversity and stand productivity 30 years after selective logging; and (2) what is AGB, growth and regeneration status at Payeh Maga Highland in both primary and logged-over forest conditions?

Objectives

The general objective of this study is to investigate the plant species diversity at two different forest conditions (logged and unlogged) and tree growth along an altitudinal variation. Thus, the specific aims of this study are:

i. To investigate the altitudinal variation in plant species diversity and stand productivity due to selective logging after 30 years and;

ii. To provide an estimation of AGB, growth and regeneration status for the Payeh Maga Highland at different elevation in primary and logged-over forest conditions.

Hypothesis

It is predicted that there are differences in the presence of tree species between both logged-over and primary forest conditions taking into account the altitude factor. This study also hypothesized that there are changes in plant diversity along an increasing altitude same as mention by Whitmore (1998). These changes in vegetation are related to the altitudinal gradient of climatic factor along the mountain slope. Aside that, it been stated by previous studies that an area that subjected to logging still retained the secondary forest structural appearances even long after logging activities are done. Furthermore, there are studies that mentioned aboveground biomass is decreasing as elevation increasing and this pattern is linked with the tree height-diameter allometry which is associated with trees species, precipitation and temperature (Feldpausch *et al.*, 2011, Banin *et al.*, 2012, Leuschner *et al.*, 2013, Girardin *et al.*, 2014, Fayolle *et al.*, 2016).

CHAPTER 2

LITERATURE REVIEW

2.1 Tropical Highlands: Distributions, Importance and Threats

Tropical rainforest can be found in America, African and Indo-Malesian which represents India, Thailand, Malaysia, Indonesia, Philippines and Papua New Guinea (Flenley, 1981; Rietbergen, 1993). The tropical rainforest can be recognized by their notable features of evergreen canopy dominated by woody trees, shrubs, herbaceous and epiphytes. Tropical montane rainforest (TMFs) stated as forests between 23.5°N and 23.5°S above 1000 m.a.s.l., which make up 8% of the world's tropical forests (Spracklen & Righelato, 2014). This type of forest embraces the most complex biodiversity ecosystems and has a high level of endemism of all (Mittermeier *et al.*, 2004, Bruijnzeel *et al.*, 2011).

The Malaysian states of Sabah and Sarawak are in the heart of the Sundaland biodiversity hotspot, in the island of Borneo. The island is known as the center of diversity for the family Dipterocarpaceae. Of the 390 species in Southeast Asia, 265 are found in the island, and 155 are endemic (Ashton, 1982). According to Appanah & Ratman (1992), the diversity of animals is high too, and so the endemism. Of the 196 mammal species, 40 are endemic. While in 2005, WWF found that Borneo has a number of 44 endemic mammals.

Originally, Ohsawa (1995) proposed a mountain-vegetation zonation for Southeast Asian forests based on change of temperature with altitude with lowland forest distributes under 1,000 m elevation, lower montane forest occurs between 1,000 - 2,500 m elevation, and upper montane forest is distributing on 2,500 - 3,800 m elevation (Figure 1).

Meanwhile, Culmsee (2010) suggested that at the Central Sulawesi, Indonesia there is different vegetation zonation; submontane forest (1,050 m), lower montane forest (1,400 m), mid-montane forest (1,800 m) and upper montane forest (2,400 m). Nakashizuka *et al.* (1992) proposed rainforest classification in the Malay Peninsula based on altitudinal i.e., the lowlands are below 700 m above sea level, the transition 700 - 1,100 m, the lower montane 1,100 - 1,500 m and the upper montane 1,500 - 1,700 m. However, in Costa Rica, Holdridge life zones was used to classify their forest where tropical wet forest can be found at 35 - 600 m, pre-montane rainforest at 600 - 1,450 m, lower montane rainforest at 1,450 - 2,500 m, and montane rainforest at 2,500 - 2,900 m (Kluge & Kessler, 2006). Zhu *et al.* (2015) categorized the Southern Yunnan forest in China as lowland forest (< 1,100 m); lower montane forest (1,100 - 1,600 m) and above 1,600 m is middle montane forest.



Figure 1 : Altitudinal formation zones in typical high areas of south/southeast/east Asian mountains overlaid on a mountain vegetation template for humid Asia (Ohsawa, 1995)

Tropical forests are known for their global mega biodiversity (Baraloto *et al.*, 2013) and provide environmental and economic services. Despite their importance, the rapid disappearing of these forests resulted from exploitation of its resources is creating major environmental and economic problems (Mani & Parthasarathy, 2006; Michalski & Peres, 2013). The rate of tropical forest disappearing was the highest among different forest types, at a rate of 6.38 M ha yr⁻¹ for 2010–2015 (Keenan *et al.*, 2015). Recently Southeast Asia has lost 293,000 km² of forest between 2000 and 2014 more than 11% of the total forest covers in 1999 (Zeng, 2018). Approximately 82,000 km² or 28% of area cleared for farming was found in highland and this clearing mainly involved the mature forests. Forests cleared for growing crops occurred mainly in the higher elevations of mainland Southeast Asia and primarily in the lowlands in maritime areas, mainly occurred in Malaysia and Indonesia (Zeng, 2018).

Today most of the remaining forests in Borneo are classified as production forests (Gaveau *et al.*, 2014) and are thus open for logging. On the island of Borneo, almost 50% (approx. 26.3 M ha) of the original forest cover has been lost and another 25% (approx. 13.6 M ha) has been selectively logged, while an additional 42% (approx. 6.6 M ha) of the remaining primary forest cover is expected to be logged in the near future (Gaveau *et al.*, 2014). In 1973, the highlands forest area of Borneo covered up to 29,456 km² and in year of 2010, 18,639 km² has been logged. By year of 2010, the highland forest area has loss 1,104 km² which is 3.75% of the forest area from in the year of 1973 (Gaveau *et al.*, 2014).

As the biodiversity conservation area is becoming increasingly limited (Rodrigues *et al.*, 2004), more attention and interest is focused on in the conservation value of degraded tropical forests (Berry *et al.*, 2010; Edwards *et al.*, 2010). This is because according to Lawton *et al.* (1998) and Gibson *et al.* (2011), degraded tropical forests usually have higher species richness than monoculture plantations and agricultural lands. Besides that, understanding the structure, diversity and ecosystem processes of disturbed forests is gradually more important as they play a key role in the maintenance of carbon stocks, biodiversity, and other services (Chazdon *et al.*, 2009; Gardner *et al.*, 2009; Kauffman *et al.*, 2009; Sodhi *et al.*, 2009; Berry *et al.*, 2010; Edwards *et al.*, 2011; Yosi *et al.*, 2011; Clark & Covey, 2012; Picard *et al.*, 2012; Putz *et al.*, 2012; Yamada *et al.*, 2013; Sist *et al.*, 2015).

The mountain has many important roles as most of the livings are depending on the mountain ecosystem especially it played as the major role in providing clean water sources for human population as the mountain range also is part of important roles in water cycle. Mountain forest in Borneo plays an important role as water catchment area as most of the major rivers begin their journey from the HoB. Besides being the significant role for water catchment area, mountain areas also home for estimation of almost one billion of peoples in the world that live in mountain areas. Half of the human populations in the world are depending on the mountain for water and food. Furthermore, due to the mountain ranges are naturally beautiful scenery and have variety of landscapes indirectly it has become the tourism areas which act as the source of income for the country and people that are living on it. Yet, the mountains are facing threat from various factors such as climate change, logging, natural disasters and land clearing for cropland and pasture. As a result of these changes, disasters such as soil erosion, flood and landslides are happening since the mountain ranges also act as natural barrier from preventing these happen by reducing speed of the wind and soils runoff (FAO, 2018). Due to its importance, clearing highland areas will have extensive impact on the existing ecosystem, soils, water quality, and flora and fauna composition (Zeng, 2018).

Thus, the mountain ranges have very important roles at varies scales towards human and the environment. Therefore, the preservation of mountain areas is important to ensure that the Earth has prolonged supply of clean freshwater and food while also important in sustaining the mountain biodiversity. By analyzing available quantitative censuses, ecological important tree species as well as special concern species can be identified and assist conservation efforts in order to manage and maintain forest biodiversity sustainability. Eventually, this will enhance the understanding of the ecological impacts of logging. Thus, the Tropical managed Forests Observatory (TmFO), a network in understanding the long-term effects of logging on tropical rainforest ecosystems was initiated with the objective of gathering long-term data on the resilience of logged tropical forests at regional and global scales (Sist *et al.*, 2015).

2.2 Tree Diversity and Floristic Composition in Highland Forest

Ashton (1995) provided a detailed description of the altitudinal forest profile in Sabah and Sarawak, suggesting that there is no clearly distinguishable altitudinal zonation, as the changes in floristic composition of the forests with altitude was gradual and continuous while other researchers identified there are distinguishable types of montane forests on Mount Mulu (2,371 m, Sarawak), Gunung Silam (890 m, Sabah) and Mount Kinabalu (4,095 m, Sabah), respectively based on forest structure and floristic composition. (Proctor *et al.*, 1988; Kitayama, 1992; 1995; Martin, 1997). Besides being hotspots of diversity, tropical mountain forest (TMF) is also hotspots of botanical knowledge gaps. Therefore, biodiversity field studies in underexplored TMF are needed to fill essential knowledge gaps about plant diversity (Wilson, 2017).

Based on the Slik *et al.* (2003) study, they found that there is variation between lowland and highland tropical forests in term of species composition. It has been supported by several studies in Borneo that the tree flora does in fact gradually change along altitudinal gradients, and most characteristic lowland tree species become uncommon above 800 m altitude (Kitayama, 1992; Adam, 2001).

In lower mountain forest of Mount Kinabalu (1,700 m), top three dominant species are *Tristaniopsis clementi*, *Payena microphylla* and *Dacrydium pectinatum* meanwhile on sedimentary substrates the dominant tree are *Lithocarpus confertus*, *Ternstroemia coriacea*, *Syzygium castaneum*, *Tristaniopsis elliptica*, *Agathis kinabaluensis*, *Xanthophyllum tenue* and *Podocarpus gibbsii*. The upper montane (2,700 m), is dominated by *Syzygium punctilimbum*, *Magnolia carsonii*, *Olea decussate*, *Lithocarpus havilandii* and *Ilex zygophylla* (Aiba *et al.*, 2002). Aiba *et al.* (2002) stated that overall, the higher elevation forest on Mount Kinabalu is characterized by the dominanace of Myrtaceae and Coniferales.

Castanopsis endertii, Castanopsis hypophoenicea and Castanopsis oviformis are endemic to Borneo. This genus is distributed from sea level up to 2,500 m while *Lithocarpus* is distributed from sea level up to 3,000 m. *Lithocarpus bullantus* and *Lithocarpus echinifer* were reported to be endemic in Borneo (Soepadmo et al., 1995). *Syzygium multibracteolatum* is endemic to Borneo and are dominance on montane forest (Ashton, 2011). *Tristaniopsis microcarpa* and *Tristaniopsis beccarii* are endemic to Borneo and *T. microcarpa* is commonly be found in mixed dipterocarp forests up to an altitude of 1,000 m (Ashton, 2011). *Gymnostoma sumatranum* is confined to hill, ridge and lower montane forests at altitude 600 – 1,800 m (Pungga, 1995). In the lower montane forest, oak (*Quercus* spp. and *Lithocarpus* spp.) and chestnut (*Castanopsis* spp.) are dominant. *Tristaniopsis* and *Rhodamnia* are common in the upper montane forest (WWF, 2014). Based on study by Ipor et al. (2002) on floristic composition of forest formation at Mahua, Crocker Range National Park (>1,500 m) at Sabah, it was found that the dominance species in term of above ground biomass and important value was Duabanga molucanna from family Lythraceae. This followed by Knema ashtonii, Agathis lanceolata, Lithocarpus cantleyanus and Litsea ochracea from family Myristicaceae, Araucariaceae, Fagaceae and Lauraceae, respectively which is in sync with WWF (2014) findings where those family are representing montane forest vegetation in Borneo. At tropical mountain Southern Yunnan, China, the lowland forest (<1,100 m) is dominated by Euphorbiaceae, Sapotaceae, Lauraceae, Meliaceae, Rubiaceae and Sapindaceae; the lower montane (1,100 - 1,600 m) is dominated by Fagaceae. Theaceae. Euphorbiaceae and Lauraceae; middle montane (>1,600 m) is dominated by Lauraceae, Magnoliaceae, Cornaceae, Euphorbiaceae and Fagaceae (Zhu et al., 2015). The findings by Zhu et al. (2015) at the lower montane forest in Southern Yunnan, China show a slight difference in the family dominating the area as the important families that dominated the lower montane forest of Borneo are Fagaceae, Myrtaceae, Clusiaceae and Euphorbiaceae (Soepadmo, 1995). Table 1 provided a summary on floristic composition data from several researches that has been conducted in the tropical mountain rainforest.

2.3 Tree Growth and Its Dynamic in Highland Forests

It has been reported by several studies that the patterns of increase of stem density, diameter at breast height (*dbh*, 1.3 m above the ground on the uphill side of stem), and basal area with increasing altitude in Hawaii (United States), Mount Kinabalu (Malaysia), Udzungwa Mountains (Tanzania) and the Andes (Herbert & Fownes, 1999; Takyu *et al.*, 2002; Lovett *et al.*, 2006; Girardin *et al.*, 2014). The decrease pattern of basal area with increasing altitude was widely found in tropics for example Mount Elgon at Kenya-Uganda (Hamilton & Perrott, 1981) or no trend between stand basal area and altitude (Girardin *et al.*, 2014) as limitation of soil nutrient supply at higher and cooler sites (Ohsawa, 1995; Aiba & Kitayama, 1999; Kitayama & Aiba, 2002; Moser *et al.*, 2007; 2011).

This is due to elevational zones as it becomes cooler and temperature drops faster at elevations higher than 2,000 m in the tropics than below 2,000 m elevations (Ohsawa, 1995; Kitayama & Aiba, 2002). Eventually it leads to slower litter decomposition and thus leading to limitation of soil nutrient supply at higher altitudes (Kitayama & Aiba, 2002; Takyu *et al.*, 2003). Aside from basal area, canopy height also is decrease with increasing elevation which has been reported in other studies at tropical evergreen broadleaf forests in Southeast Asia and in Ecuadoran tropical forests (Kitayama & Aiba, 2002; Takyu *et al.*, 2003; Moser *et al.*, 2007). However, Do *et al.* (2015) stated that there is increasing of canopy height with increasing altitude. Same goes to leaf area index (LAI) as it tends to be declining as the elevation increasing (Leuschner *et al.*, 2007; 2013; Moser *et al.*, 2007; Unger *et al.*, 2012; 2013) although Fischer *et al.* (2013) found little or no obvious trend. Some studies along elevational transects also reported increasing stem density with elevation (Lieberman *et al.*, 1996; Takyu *et al.*, 2003), but no observable trends in basal area was recorded

(Girardin *et al.*, 2014a). In contrast, decreasing stem density and increasing basal area with increasing elevation have been found in the Andes in Ecuador (Unger *et al.*, 2012). Biomass increases exponentially with tree diameter, average tree diameter, large tree density and stand basal area tend to be better predictors of above-ground biomass (AGB) than overall tree density (Slik *et al.*, 2010; Lewis *et al.*, 2013; Poorter *et al.*, 2015).

Altitude (m)	Number of stems (ha)	Species	Plots size (ha)	References
901 – 1,000	366	93	0.5	Torres et al. (2019)
500 - 1,000	520	158	3.5	
1,000 – 1,500	552	<mark>1</mark> 08	3.5	Do et al. (2017)
1,500 – 1,800	740	70	3.5	
1,600	n.a.	<mark>1</mark> 51	1.0	Heng et al. (2016)
500 – 1,000	n.a.	154	2.0	Van Do et al. (2015)
>1,000	n.a.	98	0.8	Van Do <i>et al.</i> (2015)
1,650 – 1,780		62 – 70	0.25	Zhu et al. (2015)
1,050	520	104	1.44	Culmsee & Ripotang (2009)
1,400	592	60	0.72	Culmsee & Ripotang (2009)
1,100 – 1,200	n.a.	150	1.0	Kessler et al. (2005)
1,700 (1,560)	778	84	0.5	Aiba & Kitayama (1999)
1,700 (1,860)	1420	32	0.2	Aiba & Kitayama (1999)
900	693	228	1.0	Wright <i>et al</i> . (1997)

Table 1 : Data from several studies on mo	ountain range
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Quite a large amount of data regarding AGB storage with factors affecting it in lowland tropical forests have been accessible for these past few years (Malhi et al., 2006; Slik et al., 2010; Quesada et al., 2012; Lewis et al., 2013; Poorter et al., 2015; Fayolle et al., 2016). However less information on aboveground biomass in tropical montane forests is obtainable, even though it has been highlighted in many studies on their potential to accumulate and sequester substantial amounts of carbon (Spracklen & Righelato, 2014). According to the Forest Resources Assessment 2005 report (FAO, 2006), forest biomass is categorized as organic material both above and below ground and both living and dead. This forest biomass acts as a carbon pool and according to the Intergovernmental Panel on Climate Change (IPCC), this pool in the terrestrial ecosystem can be divided into AGB, below-ground biomass, the death mass of litter, woody debris, and soil organic matters (Vashum & Jayakumar, 2012). The tropical forest ecosystem stored approximately 50% of carbon in biomass while another 50% can be found in the soil (IPCC, 2006). IPCC described AGB as all living biomass above the soil including stem, stump, branch, bark, seeds, and foliage (IPCC, 2006). Among the above-mentioned carbon pools, AGB of the tree are mainly the largest carbon pool in tropical forests (Vashum & Jayakumar, 2012). In tropical forests, the AGB are varies widely due to regional differences in climate, species diversity, stem density, stem size distribution, edaphic conditions, topography, and disturbance history (Urguiza-Haas et al., 2007; Do et al., 2010; Con et al., 2013; Ngo et al., 2013; Mohandass et al., 2016).

In a reported document of Global Forest Resources Assessment 2010, FAO (2010) found that the South and Southeast Asia forests contained 51.933 Pg of biomass with approximately 176.4 Mg ha⁻¹. In the same assessment found that there are reductions in biomass stock per hectare in the South and Southeast Asia from the period of 1990 to year of 2010 (FAO, 2010). The tropical forests in Borneo, on average have an AGB about 60% higher compared to similar ecosystems in other regions (Slik et al., 2010). Shanmughavel (2001) mentioned that total AGB in mixed dipterocarp – dense stocking mountainous of Sarawak accumulated around 330 - 405 Mg ha⁻¹ of biomass whereas at upper hill of Peninsular Malaysia, 275 Mg ha⁻¹ of biomass is recorded. Poorter et al., (2015) mentioned that changes of AGB with increasing elevation has been linked to changes in tree species richness. As higher species richness can increase the alteration in species traits in the community and lead to niche complementarity which result in higher resource capture that leads to more efficient resource use and higher productivity. In addition, tree diameter, stand basal area and large tree density are better estimators for increment in biomass compared to overall tree density (Slik et al., 2010; Lewis et al., 2013; Poorter et al., 2015).

The accurate measurement and mapping of biomass is necessary as the carbon sequestration and accumulation in the forests plays important role in studying carbon stock quantification, mitigating climate change, forest fuel accumulation in terrestrial ecosystem (Ryu *et al.*, 2004; Kumar & Mutanga, 2017). Besides that, accurate estimation of AGB for development of carbon storage baseline is also in line within the framework of the "Reducing Emissions from Deforestation and Degradation in Developing Countries" scheme (REDD) since deforestation is one of the main contributors in greenhouse gas emissions (Gibbs *et al.*, 2007; Angelsen, 2009). Table 2 provided with some AGB data from several studies in Indonesia and Sabah, Malaysia.

2.4 Selective Logging, Impact and Recovery on Forest Stands

Selective logging can be defined as 'the removal of mature, over-mature and defective trees in such a way as to leave an adequate number and volume of uninjured and healthy residuals of commercial species and other tree species, essential to assure a future crop of timber and forest cover for the protection of soil and water (Rapera, 1977). In simplest context, selective logging was designed to mimic gap produces due to natural tree dead (Webb, 1997). Thus, under supervised condition, no significant changes in forest structure will be expected (Matthews, 1989), at the same time due to gap formation, natural regeneration and residual tree growth will be enhanced (Hartshorn, 1989). This has been a familiar logging method used in tropical rainforests, especially in Southeast Asia (Bischoff *et al.*, 2005, Okuda *et al.*, 2003).

Table 2 : The collection of data for aboveground biomass with elevation	IS
at different locations in Indonesia and Sabah, Malaysia	

Location	Elevation (m)	AGB (t ha ⁻¹)	Plot size (ha)	Reference
Mount Kinabalu, Sabah, Malaysia	1,560	294	0.5	Kitayama & Aiba (2002)
Lore Lindu National Park, Sulawesi, Indonesia	1,050	286	0.96	Hertel et al. (2009)
Montane tropical forest, Lore Lindu	1,050	308.7	1.44	Culmana at $al (2010)$
National Park, Sulawesi, Indonesia	1,400	304.6	0.72	Cullisee et al. (2010)
Long Barang, East Kalimantan	1,026	359.1	1.26	Slik et al. (2010)
Puak Highlands, East Kalimantan	1,349	765.5	4.5	Slik <i>et al.</i> (2010)
	1,000	110	0.1	
	1,000	160	0.1	
	1,000	230	0.1	
	1,200	210	0.1	
	1,200	240	0.1	
Mt Binioni Lombok	1,200	270	0.1	Decae at $al (2012)$
Wit. Kinjani, Lombok	1,400	205	0.1	Dossa et al. (2013)
	1,400	245	0.1	
	1,400	255	0.1	
	1,600	215	0.1	
	1,600	140	0.1	
	1,600	110	0.1	

Selective logging was first promoted in the forests of Malaysia, India, Burma, and the West Africa during the period of the British and French colonial systems (Puettmann et al., 2015). In Peninsular Malaysia, a logging regime named the "Malayan Uniform System" (MUS) has been practiced since 1948 to manage lowland dipterocarp forest. It was commonly used until the 1970s. A part of the Pasoh Forest Reserve was originally logged under this system from 1954 to 1959 (Manokaran, 1996; Latiff, 2011). Due to ineffectiveness of MUS in managing hill dipterocarp forest (Shaharuddin, 1998), therefore, in 1978 the selective management system (SMS) logging regime was being introduced and has been started in hill dipterocarp forest in Peninsular Malaysia (Latiff, 2011). In Borneo, the logging industry began with the arrival of European colonizers in the middle of the 19th century. The widening use of machines such as crawler tractors and chainsaw start occupying the Borneo's forest because of increasing global demand for the wood products (Dawkins & Philip, 1998). Nowadays, mostly all the forests in Malaysia are managed by SMS, in which selected tree species above 50 cm dbh had been felled to reduce the impact of logging and the rest are left to form the next cut in 30 years (Thang, 1987; Sheil & van Heist, 2000; Jennings et al., 2001).

SMS is least detrimental to wildlife species in tropical forests (Grove, 2002; Meijaard & Sheil, 2008) and is considered as more flexible management system to maintain highly variable forest in the hilly terrain (Appanah, 1998), to ensure the forest renewal and obtained yield on sustainable basis. In addition, few mother trees such as advanced relicts have been left to provide a sufficient seed and to enhance the seedling regeneration (Appanah & Weinland, 1990). However, this system has been questioned for its effectiveness in managing Malayan hill dipterocarp forest (Wyatt-Smith 1988, Appanah & Weinland, 1990) because according to some studies, there is no evidence of long-term success

of this management system (Whitmore, 1984; 1990) and there is evidence of biodiversity decline after second and following cuts (IUCN, 1992) since the system does not only focuses on the removal of trees, but there is also the construction of logging roads that spread like a spider's web around a forest. This type of logging contributes significantly to forest degradation (Shearman *et al.*, 2012; Brandt *et al.*, 2016; Griscom *et al.*, 2017) which it conflicts with the requirements to accommodate biodiversity conservations (Ashton, 2008). In tropical forests, approximately 12-25% of the forest area is covered by logging roads, skid trails, and log yards (Uhl & Vieira, 1989, Pinard *et al.*, 2000, Jackson *et al.*, 2002).

The effects of logging operations on forest structure and dynamics vary according to logging intensity and applied forest management regime (Blanc *et al.*, 2009; Burivalova *et al.*, 2014; West *et al.*, 2014). Sometimes, the actual extent and degree of disturbance on forest structure, composition, and diversity and regeneration dynamics can be noticeably huge (Okuda *et al.*, 2003; Rutishauser *et al.*, 2016). The direct impacts of harvested trees include damaging and killing remaining adjacent residual stems and on direct failing path (Ho *et al.*, 2004) where this directly or indirectly reduces the habitat suitability of forest fauna (Potts, 2011) such as butterflies, fish, and mammals (Brooke *et al.*, 2003), birds (Sehgal, 2010) and monkeys (Collins, 2008).

Site productivity will be reduced due to soil disturbance because of felling trees and logging roads and skid trails construction (Pinard et al., 2000) which indirectly it also reduces the above ground biomass, which affects the soil environment and the forest ecosystem (Gullison & Hardner, 1992; Sidle et al., 2004). This eventually can alter the forest regeneration progress as seeds in logged forests may be exposed to increased light penetration and hotter, drier air, and soil conditions due to the relatively open canopy (Hardwick et al., 2015). Seedling recruitment of canopy trees is vital for forest regeneration and recovery (Chazdon, 2003; Bagchi et al., 2011). The removal of the largest and most reproductively active trees (Fisher et al., 2011) may directly reduce potential seed sources supply and superior genetic pools for regeneration for total seed production at the scale of the landscape (Ho et al., 2004; Bagchi et al., 2011). Curran et al. (1999) and Tuomela et al. (1996) reported reduced regeneration potential in logged forests in Kalimantan because most of the seed-bearing trees (>50 cm dbh, Nguyen-The & Sist, 1998) of commercial timber species had been harvested.

The remaining trees may also become spatially isolated from reproductive conspecifics, which may reduce crosspollination and, indirectly decrease seed crop size at the scale of individual trees (Murawski *et al.*, 1994; Ghazoul *et al.*, 1998). Cannon *et al.* (1994) found a reduced density of small dipterocarp trees due to logging induced mortality, suggesting that regeneration in their logged sites in West Kalimantan was concentrated in a few generally non dipterocarp species, whilst other studies (Silva *et al.*, 1995; Pelissier *et al.*, 1998; Carvalho *et al.*, 2004) found reductions in shade tolerant species and stimulation of light-demanding timber species. Many of the commercially important trees, notably

dipterocarps, require shade to regenerate successfully (Whitmore, 1978) and will not regenerate until a closed canopy is recreated.

Undeniably selective logging has been suggested as a major driver of tropical forest degradation (Asner *et al.*, 2009; Gatti *et al.*, 2015; Zimmerman & Kormos, 2012), yet the logged forest still can retain similar biodiversity and ecosystem processes as primary forest since it less environmentally severe (Berry *et al.*, 2010; Edwards *et al.*, 2012a; 2012b; Ewers *et al.*, 2015; Mazzei *et al.*, 2010; Yamada *et al.*, 2014) depending on the logging intensity and absence of further logged-forest disturbances (Burivalova *et al.*, 2014; Edwards *et al.*, 2014c; Lee-Cruz *et al.*, 2013; Luke *et al.*, 2014). Thus, the ecological value of logged forests should not be overlooked since functioning forest stands are important in maintaining species and habitat diversity, and trees are the main component of a forest stand. Therefore, constant monitoring and management are required to assess their succession processes (Attua & Pabi, 2013; Ewers *et al.*, 2015).

Forest recovery is a process involves regrowth of remnant plants, recolonization by species previously present, and colonisation by new species (Gleen-Lewin & van der Maarel, 1992; Chazdon, 2014). Variations in the habit quality of different sites might influence forest recovery. One of the factors that impacts tropical forest restoration is the distance between disturbed lands and remnant intact of old growth forests (Chazdon, 2014) as species richness and seed dispersal can impact on the migration capacity of species which play as main roles on recovery of disturbed forest (Clark & Clark, 2000; Boukili & Chazdon, 2017). Usually, old growth forests at the ridges are undisturbed and have many parent seed trees. These seeds of old-growth forest species will become the source to spread seeds directly into clear-cut areas. Occasionally, seed trees are intentionally left after clear-cutting to provided seeds and escalate the natural recovery process (Zahawi et al., 2013). Therefore, it is advisable to logging company when conducting the logging practices, some patches of old growth forests were left unlogged for various reasons, such as a rugged topography, conditions of log transportation, or for animal use (Ding et al., 2016).

Studies in Indonesia shows that after only 15 years of recovery a selective logging forest in Indonesia that lost most mature trees still can retained the tree abundance, number of species, and diversity indices are same level as an old growth forest (Slik *et al.*, 2002) whereas Cannon *et al.* (1998) reported that post commercial selective logging needed 8 years of recovery. Then, study in a Brazilian tropical forest also shows the similar result which required 5 years of recovery after intensive logging of trees with \geq 30 cm *dbh* (Villela *et al.*, 2006).

Results from other studies also supported earlier findings that Asian tropical forest is slower in recovery as compared to South American tropical forest due to climate (Brearley *et al.*, 2004; Lewis *et al.*, 2009; Cole *et al.*, 2014). The evergreen forest recovery process could be slower and favour deciduous species because of the extreme climate events and changes in the climate regime as nowadays the wet seasons have become shorter and wetter; the dry

seasons have become hotter, longer, and drier (Dai *et al.*, 2004; Koh & Gardner, 2010; Wang *et al.*, 2015; Rodrigues *et al.*, 2015; Schmidt-Thome *et al.*, 2015; Andresen *et al.*, 2016; Bose *et al.*, 2017). Incorporating floristic composition, stand structure and soil quality of logged-over forest after logging are pivotal elements in considering forest recovery procedures and planning of forest management policies in various disturbance regimes (Gutierrrz & Huth 2012; Kalaba *et al.*, 2013). Moreover, forest regeneration is depending on forest types as different harvesting methods such as clear cutting and selective logging have different effects (Chazdon, 2014; Martin *et al.*, 2015). Furthermore, the alteration in remainder species and community composition after logging activities also expected to be a reason for the recovery process of ecological systems (Faria *et al.*, 2009; Kusumoto *et al.*, 2015; Xu *et al.*, 2015). In addition, quantification of biomass and carbon storage could also give an indication in the improvement of the forest ecosystem services after recovery (Baker *et al.*, 2010).

2.5 Reduced Impact Logging

Reduced-impact logging (RIL) is a set of enhanced timber harvesting guidelines for natural forests that are selectively logged. The term "reduced-impact logging" and its acronym (RIL) were seemingly first used in 1993 and first applied to an improved forest management project in Malaysia (Putz & Pinard, 1993). Detailed RIL guidelines were developed in Australia (Queensland Department of Forests, 1988) served as templates for developing more locally tailored sets in several regions in the tropics (Dykstra & Heinrich, 1996). RIL codes-of-practice were subsequently developed in Asia and the Pacific (FAO, 1999), Brazil (Sabogal et al., 2000), Malaysia (Forest Department Peninsular Malaysia, 2003), and West and Central Africa (FAO, 2003). Most RIL Guidelines are also components of most forest management plans and start with recommendations related to designation of forest management units. Central to all RIL guidelines is the use of felling techniques that increase worker safety, reduce wood waste, and direct the fall of trees (Putz et al., 2008). RIL can be described by trained workers as intensively designed and carefully managed timber harvesting in ways that reduce the deleterious effects of logging (FAO, 2004).

Studies on tropical forestry over the past decades reported that RIL in experimental cutting blocks reduced emissions from logging by 30–50% (Johns *et al.*, 1996; Pinard & Putz, 1996; Mazzei *et al.*, 2010; Medjibe *et al.*, 2013; Vidal *et al.*, 2016). There is evidence that emissions benefits of some attempts at RIL disappear when logging intensities are considered (Sist *et al.*, 2003; Martin *et al.*, 2015). It is also not clear whether the potential climate mitigation benefits of RIL are associated with forest certification (Miller *et al.*, 2011; Griscom *et al.*, 2014; Martin *et al.*, 2015), which may suggest that certification systems, such as the Forest Stewardship Council (FSC), even though requiring many RIL practices to be employed but were not designed to achieve emissions reductions (Romero & Putz, 2018).

Uncertainty about the scalable and verifiable climate benefits of RIL motivated development of RIL-C, a set of improved selective logging practices focused on verifiable emission reductions (Griscom et al., 2014). The Reduced Impact Logging for Climate (RIL-C) Protocol tested in Indonesia focuses on reducing carbon emissions (Griscom et al., 2014; Ellis et al., 2016) to represent increased climate change issues and increased opportunities to benefit financially from these reductions. RIL-C practices are defined by their capacities to deliver measurable climate change mitigation outcomes without reductions in timber yields (Ellis et al., 2019). Examples of RIL-C practices include improved felling and bucking for greater wood utilization (thus reducing waste), directional felling to avoid collateral damage, skid trail planning, long-line winching, and narrower haul road construction. In 2016, The Nature Conservancy and partners published a methodology for measuring and validating RIL-C outcomes (The Nature Conservancy and TerraCarbon LLC, 2016); in 2018, this methodology was recommended for making carbon claims as part of the Forest Stewardship Council's Ecosystem Services Procedure (FSC, 2018).

CHAPTER 3

STAND STRUCTURE, FLORISTIC COMPOSITION AND SPECIES DIVERSITY ALONG ALTITUDINAL GRADIENTS OF A BORNEAN MOUNTAIN RANGE 30 YEARS AFTER SELECTIVE LOGGING

3.1 Introduction

Logging activities and their spin-off effects are of high concern. It has been reported that a total of 165 M ha out of 403 M ha or 41% of the tropical production of permanent forest estates were earmarked for logging in the International Tropical Timber Organization (ITTO) producer member countries in 2010 (Blaser *et al.*, 2011). Approximately 85% of tropical forests were found in the 33 ITTO member countries. Southeast Asia is commonly viewed as one of the most threatened hotspots (Sodhi *et al.*, 2004). This is because the deforestation rate and timber harvesting were the highest among other tropical regions (Sodhi *et al.*, 2004; Hansen *et al.*, 2008). Cleary *et al.* (2007) found that tropical timber exports from Borneo Island alone exceeded those from Africa and Latin America combined. Asner *et al.* (2009) estimated that the rate of selective logging in tropical forests was 20-fold higher than the deforestation rate.

In Malaysia, a selective management system (SMS) was introduced in 1978 with polycyclic rotations of 25–30 years (Thang, 1987). SMSs and other selective logging practices are considered a sustainable approach to managing forests due to their dual potential for the continued production of timber and the protection of ecosystem integrity (Edwards *et al.*, 2014b). Many authors have reported that tropical forests still retain diverse species and considerable functional diversity (Berry *et al.*, 2010; Edwards *et al.*, 2014b) when selectively logged and more so when using reduced-impact techniques (Bicknell *et al.*, 2014) at low intensity (Burivalova *et al.*, 2014) or under a land sparing approach (Edwards *et al.*, 2014a). Thus, the value of logged tropical forests for conservation is increased (Edwards *et al.*, 2014b). The selective management system is arguably the best silvicultural intervention for tropical forests where only a selected number of trees will be harvested, usually in a range of 8–15 trees ha⁻¹ (Sist *et al.*, 1998; Pinard *et al.*, 2000).

Okuda *et al.* (2003) highlighted the importance of knowledge of the effects of logging on stand structure, floristic composition, and species diversity to sustainably manage a forest silviculturally and ecologically. In the case of Malaysia, selective logging may result in damage to 8–47% of trees (Borhan *et al.*, 1987; Saiful & Latiff 2014), while destroying up to 55% of the stand basal area (Ho *et al.*, 2004). A mortality rate of 35–40% was recorded for trees initially suffering from minor injury after a harvesting operation (Saiful & Latiff, 2014). This observation was in line with Wyatt-Smith's (1954) report that many trees die a few years after logging. Logging damage was observed across diameter size

classes, especially those in the largest and smallest ranges (Ho *et al.*, 2004). It is expected that conditions for regenerated logged forests to grow in similarity to primary forests remain unsuitable for at least 15 years (Pinard *et al.*, 1996), 40 years (Shima *et al.*, 2018), 250 – 500 years (Kartawinata, 1994) and even centuries (Whitmore, 1991).

The diversity of tree species is an important characteristic of tropical forest biodiversity (Evariste *et al.*, 2010). Climate, topographic and land aspects, soil and other geomorphological characteristics, species composition and stand structure are the main factors influencing tree species diversity in a forest. Studying forest stand structure will assist in understanding the dynamics of forest ecosystems and the characteristics of stand biodiversity (Ozcelik, 2009). In addition, it provides a good indicator for envisaging the ecologically soundness and sustainability of a logging operation (Clark *et al.*, 1996). To improve the biodiversity conservation value of degraded and deforested areas, information on diversity from tree species census is vital (Baraloto *et al.*, 2013).

The highland forest provides an essential water supply to almost one-tenth of the human population in the world. In addition, downstream industries, settlements, and agriculture are also dependent on it. However, the highest rate of deforestation in any biome occurs in tropical upland forests at 1.1% annually (WWF, 2014). This is due to sophisticated logging machinery, which makes it easier to access highland areas, so that logging activities can be carried out in the area. Borneo Island, the main area with vast diversity of flora and fauna, can easily be found in the central part of Borneo. Therefore, a trans-boundary project, Heart of Borneo (HoB), was launched to safeguard and manage forested areas in Borneo highlands located at the centre of Borneo Island. HoB is an initiative with the collaboration of three countries: Brunei Darussalam, Malaysia, and Indonesia (WWF, 2014).

The Payeh Maga Highland, Long Tuyo, is part of the Murud complexes and is located on the northern east of Lawas, Sarawak, Malaysia on Borneo Island. This area is one of the areas dedicated under HoB initiatives. It consists of four major vegetation types: hill dipterocarp forest, sub-montane forest, heath forest and mossy forest. Payeh Maga refers to swampy highlands in the Lun Bawang language. This land is classified as state land, which is currently under a logging concession. Despite this, this area has been utilized by the local Lun Bawang community as a source of their livelihoods for generations. The Payeh Maga Highland also acts as an important water catchment area because there are two major rivers drained from Gunung Tuyo and Gunung Matallan (Ampeng *et al.*, 2013).

The lack of information on floral diversity in the highlands of Sarawak, as well as the post-logging effect, has resulted in a vegetation survey within the HoB area. Information on floral and faunal diversity is important as baseline information for managing forests sustainably according to forest types. Previous studies of tropical montane vegetation have been conducted in tropical regions (Aiba &

Kitayama, 1999; Culmsee & Pitopang, 2009; Clark *et al.*, 2015). In these studies, tree composition and diversity decreased with elevation. This trend is probably related to differences in climate and nutrient availability. However, general patterns still need to be further studied (Spracklen & Righelato, 2013). Accurate estimates of selective logging effects on tree flora are not readily available for many tropical habitats, thus making it difficult to predict how ecosystems respond to human disturbance. As part of the Payeh Maga area has been logged at the lowland area in the early 1980s, the upper forest remains reasonably intact (Ampeng *et al.*, 2013). Therefore, these two types of forest conditions at different altitudes can act as a reference to differentiate tree species composition and structure under both stand conditions. Thus, the main objectives of this research were to investigate the altitudinal variation in plant species diversity and stand recovery due to selective logging after 30 years.

3.2 Materials and Methods

3.2.1 Study Area

The study site is located at the Payeh Maga Highland, Lawas, Sarawak, Malaysia. The Payeh Maga Highland is located approximately 70 km from Lawas town, which lies within 4°51' 40.66" N and 115°24'24.33" E (Figure 2). The topography of Payeh Maga Highland is hilly with river valleys and hills until reaching the upper montane habitats. The data were collected from three different sites, namely, Site 1 (S1), Site 2 (S2), and Site 3 (S3). The soil series at these sites is Merit, a group of red yellow podzolic soils. Nutrient availability varies between sites (Table 3) in logged-over only N and Ca vary significantly whilst in unlogged forest, only P and K vary significantly. The average temperature and annual rainfall for S1 were 24.6°C and 2,869 mm, for S2 were 20.6°C and 3,737 mm and for S3 were 19.3°C and 3,911 mm, respectively. The areas were logged 30 years ago using the selected logging method where trees above 50 cm in diameter were harvested.

3.2.2 Plot Establishment

Plots have been established in two stand conditions: unlogged and logged forests at three different altitudes: 750 - 1,200 m, 1,200 - 1,500 m and >1,500 m (Figure 2). The logged areas were identified by the presence of tree stumps. At each elevation range (e.g., 750 - 1,200 m), 5 plots were established in the logged area, and another 5 plots were established in the unlogged forest. To minimize the impacts of pre-logging differences in stand structure, floristic composition, and species diversity between the two plots, these plots were established within 250 m of each other. Each paired plot (unlogged and logged) was set on similar elevation, topographic and edaphic conditions to minimize the induced effect of these factors on stand structure, floristic composition, and species diversity plot sizes was 20×20 m.


Figure 2 : Location of study area with three different studies sites (S1, S2 and S3) in Sarawak, Malaysia on Borneo Island

Table 3 :	Nutrient	concentration	at three	different	sites	of the	Payeh	Maga
Highland							-	-

Nutrient	Sito	Fores	st condition
Numerit	Olle	Logged	Unlogged
	S1	0.083 ^a	0.141ª
N (%)	S2	0.079 ^b	0.156ª
	S3	0.275 ^a	0.177ª
	S1	747.28 ^a	574.05ª
P (mg/kg)	S2	374.88 ^a	282.32ª
	S3	240.91ª	84.85 ^b
	S1	2.49 ^a	3.42ª
K (meq/100g)	S2	2.53ª	2.27ª
	S3	1.54 ^a	1.23 ^b
	S1	0.40 ^a	0.15ª
Ca (meq/100g)	S2	1.46ª	1.40ª
	S3	0.65 ^b	1.58ª
	S1	0.38ª	0.47 ^a
Mg (meq/100g)	S2	0.38ª	0.45 ^a
	S3	0.41ª	0.42ª

The main logging activities at the Payeh Maga Highland covered an area up to 1,000 m in elevation. Logging above 1,000 m was difficult due to bad road conditions, steep slopes, and a low number of large trees. Only small portions and patches of the area were logged, and the area was later abandoned. Hence, only a small size plot of 20×20 m was possible in the logged stands at elevations over 1,200 m. In addition, the hilly topographic condition of this area eventually made it harder to establish a larger sized plot in this locality.

All trees 10 cm and above in *dbh* were tagged and measured using diameter tape. Tree total height was determined using a clinometer and measuring tape. Tree species identification was performed at the time of data collection. For unidentified tree species, leaf specimens were collected, and identification was made through voucher specimens that can be found in the Herbarium of the Forest Department, Sarawak, Malaysia.

3.2.3 Data Analysis

To understand the differences between study sites, the overall tree data were quantitatively analysed for the total number of stems and basal areas categorized into family, genus and species levels according to different elevations and stand conditions. The importance value index (IVI) of tree species was determined as the sum of the relative density, relative frequency, and relative dominance (Curtis & McIntosh, 1951). The following five indices of species diversity were used to determine the tree species diversity and distribution: (1) Shannon-Wiener index (Pielou, 1969); (2) Pielou's index (Pielou, 1969); (3) Fisher's α index (Whittaker, 1960); (4) Variance to mean ratio (Greig-Smith, 1983); and (5) Lloyd index of patchiness (Lloyd, 1967). Sørensen's similarity index was used to compare each of the forest types and altitudes (Sørensen, 1948). The beta diversity was determined by regressed Sørensen's similarity index values with the natural logarithm of the distances (Condit et al., 2002). The presence of endemic species in the study sites was also determined and cross-checked with the International Union for Conservation of Nature Red List, Tree Flora of Sabah and Sarawak and Wild Life Protection Ordinance 1998.

3.2.4 Statistical Analysis

Differences between mean values of average diameter and height, basal area and indices of species diversity were tested for their significance using Student's t-test. These analyses were carried out using SAS 9.4 (SAS Institute Inc., NC, USA).

3.3 Results

3.3.1 Tree Diversity and Distribution

A total of 203 species were enumerated from three different elevation study sites. Of these, 157 were identified at the species level and 46 were identified at the genus level. S1 (940 – 980 m) recorded the highest family, genera and species richness compared to higher altitude sites. A total of 117 tree species were found in S1, while only 82 and 59 species were recorded in S2 (1,300 – 1,420 m) and S3 (1,630 – 1,660 m), respectively. Species diversity analysis using Fisher's α index showed significantly higher values for S1 compared to S2 and S3 (Table 4), indicating that S1 is more diverse than higher altitude sites. The rate of beta diversity was enhanced by distance (Table 5), where the Sørensen's similarity index values decreased significantly with distance between study plots.

3.3.2 Stand Floristic Composition

A total of 55 families were recorded from all three sites of the Payeh Maga Highland. Only a few tree families were speciese among the sites, with 18% represented by four or more species. Dipterocarpaceae and Lauraceae were the most speciose families in S1. In both S2 and S3, Lauraceae and Fagaceae were the dominant tree families based on the number of species per family.

Myrtaceae, Fagaceae, Dipterocarpaceae and Lauraceae were the four dominant tree families at S1 (Table 6). Although Myrtaceae and Pentaphylacaceae were the most common tree families at S2 (Table 6), the most speciose tree families, nevertheless, were the Lauraceae (9 species) and Fagaceae (7 species). Meanwhile, at S3, Fagaceae and Myrtaceae were the most dominant tree families, while the most diverse families were the Fagaceae and Lauraceae (Table 6).

Species composition and number of individuals deviated among the three different sites (Table 6). Based on the number of stems in each plot, *Syzygium* sp. was the most abundant species across the sites (Table 6). The tree compositions of S2 and S3 were the most similar when compared between different elevations sites with a Sørensen similarity index of 35.2%. *Lithocarpus conocarpus* and *Calophyllum* sp. were another two dominant species found at S1 (Table 6). *Lithocarpus* species were also found to be abundant at S2 and S3 (Table 6).

Variable	S	51		S2		S3
Vallable	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged
Elevation (m)	995 - 1069	991 - 1073	1370 -	1374 -	1637 -	1633 -
No. of families	36	32	21	29	19	22
No. of genera	56	51	21	41	29	27
No. of species	77	70	33	59	39	37
Average diameter (cm)	19.96 (10.1 - 66.3)ª	22.67 (10.1 - 65.3)ª	18.57 (10.0 - 50.8)ª	21.89 (10.0 - 83) ^b	16.28 (10.0 - 35.5)ª	17.36 (10.0 - 46.0)ª
Average height (m)	18.22 (8.45 - 8.38) ^a	19.82 (9.49 - 35.49)ª	14.78 (5.21 - 9.80) ^a	17.28 (3.92- 35.88) ^b	12.99 (4.90- 31.27) ^a	13.17 (5.41 - 27.98) ^a
Stem density (0.2 ha)	169	142	130	156	140	154
Basal area (m²/0.2 ha)	33.45ª	35.94ª	19.47ª	39.82 ^b	16.67ª	21.84ª
Shannon-Weiner Index (H')	2.85ª	2.84ª	1.92ª	2.60 ^b	2. <mark>3</mark> 7ª	2.26ª
Fisher's α Index Pielou's Index (E)	30.75ª 0.94ª	35.21ª 0.95ª	8.24 ^a 0.85 ^a	16.17ª 0.90ª	13.66ª 0.91ª	10.34ª 0.85ª
Llyod Index of patchiness (LIP)	3.67ª	4.37ª	2.34ª	3.70 ^b	1.9 ^{9ª}	2.82ª
Variance to mean ratio (VMR)	0.8 <mark>2ª</mark>	0.69ª	2.75ª	1.94ª	0.65ª	2.28 ^b

Table 4 : Details of floristic composition and diversity indices between logged and unlogged condition at S1, S2, and S3 at the Payeh Maga Highland, Sarawak

Different letters indicate significant difference at p < 0.05 between between primary and logged forests (T-test)

Table 5 : Relationship between Sørensen's similarity index and log distance (km) in Payeh Maga Highland

Site	Sørensen's similarity index	Log distance (km)
S1 vs S3	0.181	0.48
S1 vs S2	0.249	0.20
S2 vs S3	0.352	0.15

A total of 19 endemic species of Borneo were found in this study area (Table 7), with 11 species recorded at S1, seven species at S2 and another four species at S3. *Lithocarpus andersonii* and *Alseodaphne oblanceolata* were recorded at both S1 and S2. *Tristaniopsis beccarii* was recorded at S2 and S3. There were four endemic species of Dipterocarpaceae (*Shorea crassa, Shorea isoptera, Shorea parvistipulata* and *Vatica micrantha*) recorded in this study area (Table 7).

		24		20	63	
		51	5	52	5	3
	Logged	Unlogged	Logged	Unlogged	Logged	Unlogged
	Family	Family	Family	Family	Family	Family
	Fagaceae (22)	Myrtaceae (25)	Pentaphylacaceae (37)	Myrtaceae (41)	Fagaceae (38)	Fagaceae (38)
	Myrtaceae (15)	Dipterocarpaceae (12)	Fagaceae (22)	Fagaceae (16)	Myrtaceae (24)	Myrtaceae (28)
	Dipterocarpaceae (11)	Lauracea (11)	Euphorbiaceae (16)	Lauraceae (14)	Lauraceae (13)	Lauraceae (19)
a)	Lauracea (10)	Fagaceae (10)	Asteraceae (8)	Ebenaceae (12)	Casuarinaceae (12)	Calophyllaceae (9)
Ë	Polygalaceae (9)	Calophyllaceae (9)	Lythraceae (8)	Elaeocarpaceae (9)	Elaeocarpaceae (10)	Podocarpaceae (8)
0.2	Species	Species	Species	Species	Species	Species
/se		. (22)			Lithocarpus bennettii	
tree	Syzygium sp. (15)	Syzygium sp. (22)	Adinandra excels (26)	Syzygium sp. (35)	(14)	Lithocarpus sp. (26)
~	Lithocarpus conocarpus	Colonbullum on (7)	Macaranga pachyphylla		S_{1}	Alseodaphne sp.
sit	(8)	Calophyllum sp. (7)	(12)	Diospyros evena (10)	Syzygium sp. (14)	(14)
Der	Saurauia sp. (7)	Memecylon sp. (6)	Lithocarpus conocarpus	Garcinia sp. (6)	Lithocarpus	Tristaniopsis
	Gauladia Sp. (1)	Memocylon sp. (6)	(11)	Garcinia sp. (6)	conocarpus (10)	microcarpa (14)
	Lithocarpus gracilis (7)	Lithocarpus conocarpus	Adinandra dumosa (11)	Lithocarpus andersonii	Gymnostoma sp. (11)	Svzvaium sp. (11)
		(4)		(6)	T ((()	c)_jg cp. ()
	Schoutenia accrescens	Shorea platyclados (4)	Vernonia arborea (8)	Tristaniopsis beccarii (5)	Tristaniopsis	Calophyllum sp. (9)
	(7)				merguensis (8)	
	Family	Family	Family	Family	Family	Family
-	Fagaceae (1.39)	Fagaceae (1.11)	Fagaceae (0.74)	Myrtaceae (2.08)	Myrtaceae (0.81)	Fagaceae (0.83)
ha)	Dipterocarpaceae (0.56)	Myrtaceae (0.86)	Pentaphylacaceae (0.68)	Fagaceae (1.11)	Fagaceae (0.71)	Myrtaceae (0.79)
2	Myrtaceae (0.54)	Dipterocarpaceae (0.67)	Asteraceae (0.51)	Dipterocarpaceae (0.71)	Casuarinaceae (0.27)	Lauraceae (0.49)
2/0	Calophyllaceae (0.47)	Calophyllaceae (0.55)	Euphorbiaceae (0.46)	Elaeocarpaceae (0.68)	Lauraceae (0.27)	Podocarpaceae
<u> </u>					Dentenhylagooog	(0.47) Calaphyllagogo
ea	Myristicaceae (0.39)	Lauraceae (0.51)	Lythraceae (0.29)	Lauraceae (0.57)		
Ar	Species	Species	Species	Species	(0.23) Species	(0.41) Species
sal	Species	Species	Species	Species	Species	
ga	Syzygium sp. (0.54)	Syzygium sp. (0.60)	Adinandra dumasa (0.53)	Syzygium sp. (1.73)	<i>Syzygium</i> sp. (0.38)	Liinocarpus sp.
_	Lithocarous gracilis				Lithocarnus hennettii	(0.03) Tristanionsis
	(0 41)	Calophyllum sp. (0.46)	Vernonia arborea (0.51)	Shorea crassa (0.54)	(0.27)	microcarna (0.52)
	(0.71)				(0.21)	(0.0Z)

Table 6 : Top five families and species in logged and unlogged condition at S1, S2, and S3 for density, basal area and importance value index (IVI)

C

Calophyllum sp. (0.40)	Gluta aptera (0.33)	Lithocarpus conocarpus (0.43)	Lithocarpus sundaicus (0.41)	<i>Gymnostoma</i> sp. (0.25)	Dacrydium elatum (0.47)
Lithocarpus cantleyanus	Castanopsis oviformis	Duabanga moluccana	Lithocarpus lucidus	Tristaniopsis merguensis (0.25)	Calophyllum sp. (0.41)
Horsfieldia grandis (0.35)	Shorea parvistipulata (0.27)	Macaranga pachyphylla (0.27)	Tristaniopsis beccarii (0.35)	Calophyllum sp. (0.23)	Alseodaphne sp. (0.38)
Species	Species	Species	Species	Species	Species
<i>Syzygium</i> sp. (19.73)	<i>Syzygium</i> sp. (28.85)	Adinandra excels (44.52)	Syzygium sp. (49.60)	<i>Syzygium</i> sp. (26.11)	<i>Lithocarpus</i> sp. (37.25)
Lithocarpus conocarpus (11.47)	Calophyllum sp. (15.30)	Macaranga pachyphylla (22.18)	Diospyros evena (14.42)	<i>Lithocarpus bennetti</i> (24.21)	Tristaniopsis microcarpa (26.94)
Calophyllum sp. (10.98)	<i>Memecylon</i> sp. (10.88)	Vernonia arborea (22.08)	Lithocarpus andersonii (12.25)	<i>Gymnostoma</i> sp. (18.59)	Alseodaphne sp. (23.69)
<i>Saurauia</i> sp. (10.92)	Shorea parvistipulata (7.89)	Lithocarpus conocarpus (21.55)	Garcinia sp. (11.16)	Lithocarpus conocarpus (16.54)	Calophyllum sp. (21.30)
Lithocarpus gracilis (10.56)	Madhuca sericea (7.36)	Syzygium sp. (21.06)	Lithocarpus lucidus (10.18)	Tristaniopsis merguensis (16.18)	Dacrydium elatum (20.34)

(%) I/I

No.	Family	Endemic Species	Location
1	Anacardiaceae	Melanochyla castaneifolia	S1 Unlogged
2	Convolvulaceae	Erycibe borneensis	S1 Logged &
3	Dipterocarpaceae	Shorea crassa	S2 Unlogged
4	Dipterocarpaceae	Shorea isoptera	S1 Logged
5	Dipterocarpaceae	Shorea parvistipulata	S1 Unlogged
6	Dipterocarpaceae	Vatica micrantha	S1 Unlogged
7	Fagaceae	Castanopsis oviformis	S1 Unlogged
			S1 Logged &
Q	Fagacoao	Lithocarpus andorsonii	Unlogged
0	l'agaceae	Ennocarpus andersonni	S2 Logged &
			Unlogged
9	Fagaceae	Quercus pseudoverticillata	S2 Unlogged
10	Lauraceae	Alseodaphne borneensis	S2 Unlogged
11	Lauraceae	Alseodaphne oblanceolata	S1 Logged
	Ladiaoodo	neoodaphino osianooolata	S2 Logged
12	Lauraceae	Endiandra ochracea	S3 Logged
13	Myristicaceae	Gymnacranthera contracta	S1 Logged &
			Unlogged
14	Myristicaceae	Myristica borneensis	S2 Unlogged
15	Myrtaceae	Syzygium multibracteolatum	S3 Unlogged
16	Mvrtaceae	Tristaniopsis beccarii	S2 Unlogged
47			S3 Logged
1/	Nyrraceae	I ristaniopsis microcarpa	S3 Unlogged
18	Pentaphylacaceae	Adinandra exceisa	STLogged
19	Rubiaceae	Timonius lasianthoides	S1 Logged

Table 7 : Number of endemic species in Borneo that can be found in the study area

3.3.3 Diameter, Height and Basal Area Distribution

A total of 891 individuals with $dbh \ge 10$ cm were recorded from all three different elevation sites. S1 had the highest number of individuals, followed by S3 and S2. There was no apparent difference in tree density between the three sites (Table 4). All sites showed a similar inverted J-curve trend, with the highest number of individuals having smaller tree dbh (Figure 3). The highest dbh was recorded at S1, while the lowest was 16.9 m, from S3 (Table 4), which was 19.1% smaller than that at S1.

As the elevation increased from 940 to 1,670 m, the height of the tree decreased (Figure 4). The average tree height was 16.1 m. S1 had three strata with an average height of 18.9 m (Table 4), while S2 had two strata with a height of 16.1 m (Table 4) and only one tree stratum was observed at S3 with an average height of 13.2 m (Table 4). Payeh Maga Highland had a total basal area of 83.6 m² ha⁻¹, with S1 having the highest value of 34.7 m² ha⁻¹ and S3 having the smallest basal area (Table 6).



Figure 3 : Tree distributions among different diameter size class of three different sites [(a) S1, (b) S2 and (c) S3] at the Payeh Maga Highland





3.3.4 Effects of Logging and Vegetation Recovery

It is interesting to note that after 30 years of logging, the number of genera and species was found to be higher in the logged areas than in the unlogged stands for S1 and S3 (Table 4). However, the opposite trend was observed at S2; the unlogged area had a significantly higher number of tree species than the logged stands (Table 4).

The unlogged forest recorded higher average stand dbh and height compared to the logged areas (Table 4). However, only S2 showed significant differences between the two different stand conditions (Table 4). Similar to other physical parameters, only the stand basal area of S2 showed a significant difference between stand conditions. The logged forest was 51.1% smaller than the unlogged areas (Table 4). The logged forest of S1 recorded a 19% higher stand density compared to the unlogged forest (Table 4). Meanwhile, S2 and S3 showed opposite trends (Table 4), where unlogged stands had higher stand densities.

The Shannon–Wiener index and Fisher's α index of logged S1 and S3 plots were similar to that of the unlogged stands (Table 4); however, their compositional similarity, as indicated by the Sørensen index between the two different forest types, was only 35.4% and 44.7%, respectively. Meanwhile, at S2, higher diversity indices were recorded in the unlogged plots compared to the logged stands (Table 4). However, only the value of the Shannon–Wiener index was significantly different.

At S1, trees in both forest types were evenly (Pielou's index (E) of 0.94–0.95), under-dispersed (variance to mean ratio (VMR) was less than 1) and clumped or aggregated (Lloyd Index of patchiness (LIP) was more than 1) in distribution (Table 4). The distribution of individual trees in logged plots and unlogged stands in S3 showed different patterns (Table 4), where the logged stands were evenly distributed (E more than 0.85), aggregated (LIP more than 1) and under-dispersed (VMR less than 1). Meanwhile, the unlogged stands were evenly distributed and aggregated but over-dispersed (VMR more than 1). Individual trees at S2 in both forest conditions were evenly distributed, aggregated, and over-dispersed (Table 4).

At S1, *Syzygium* sp. was the most common tree species under both forest conditions (IVI of 19.73 vs 28.85 in logged and unlogged stands, respectively, Table 6). At S2, logged plots were dominated by pioneer species of *A. dumasa, Macaranga pachyphylla* and *V. arborea* with IVI values of 44.52, 22.18 and 22.08, respectively (Table 6). Meanwhile, *Syzygium* sp. (IVI of 49.6) was the most abundant species in the unlogged stands (Table 6). *Lithocarpus* sp., *Tristaniopsis microcarpa, Alseodaphne* sp., *Calophyllum* sp. and *Dacrydium elatum,* were the five species showing higher ecological weights with IVI values of more than 20 in the unlogged stands of S3 (Table 6). Despite this, only two

species, i.e., *Syzygium* sp. and *Lithocarpus bennettii* had IVI values higher than 20 within the logged plots (Table 6).

3.4 Discussion

The elevation of the research area ranged from 900 - 1670 m. Three sites were selected for plot establishment to assess the recovery of tree diversity after 30 years of logging activities. Based on the general classification of forest types by Symington (1943) and Whitmore (1984) for forests in Peninsular Malaysia, these three sites (S1, S2 and S3) represent upper dipterocarp forest (750 - 1,200 m), montane oak-laurel forest (lower montane: 1,200 - 1,500 m) and montane ericaceous forest (upper montane: >1,500 m), respectively.

3.4.1 Stand Structure, Floristic Composition, and Tree Diversity

In this study, there was a linear decrease in the number of species per stand with increasing altitude. This is similar to the observation made by Kitayama (1992) and Aiba & Kitayama (1999) at Mount Kinabalu, Sabah, Malaysia. Comparatively, Payeh Maga Highland had higher species richness than Mount Kinabalu. At similar elevations, only 93, 70 and 58 tree species were enumerated at Mount Kinabalu, respectively (Kitayama, 1992). Clark et al. (2015) also made similar observations in Costa Rica and found that tree species decrease as altitude increases. At Mt. Pangrongo, West Java, a total of 57 tree species were reported at elevations greater than 1,400 m in a one ha plot (Yamada, 1975). Meanwhile, in Central Sulawesi, Culmsee & Pitopang (2009) found 104 and 60 tree species at elevations of 1,050 and 1,400 m, respectively. In a more recent survey covering an area of 1.5 ha on Mount Kinabalu, Aiba et al. (2006) recorded 122 tree species at elevations of 1,550 - 1,625 m. Similarly, Kueh et al. (2017) recently recorded 151 species in an unlogged area totalling one ha at the Payeh Maga Highland with elevations ranging from 1,550 - 1,650 m. In the current study, only a 0.4 ha area was inventoried and only approximately one third of the number of species reported by Kueh et al. (2017) was recorded. This is not a surprise because tropical forests are heterogeneous in nature (Condit et al., 2002), where species continue to be found as the area covered increases. This is due to unique species that are usually represented by a single individual. Kueh et al. (2017) found that 44% of the species were rare in their study. In general, decreasing of tree species richness with increasing altitude, is due to a greater role of environmental filtering at higher elevations for species that are able to withstand the cooler temperatures, fog, reduced light incidence and higher relative humidity (Dossa et al., 2013; Sassen & Sheil, 2013; Girardin et al., 2014).

Family richness in tropical mountain areas is predictable (Pipoly & Madulid, 1998; Aiba *et al.*, 2006; Saw, 2010). At S1, Myrtaceae and Fagaceae were the most dominant tree families, and Dipterocarpaceae and Lauraceae only ranked third and fourth, respectively (Table 5). However, Dipterocarpaceae was consistently the most speciose tree family, as indicated for this type of forest

(Saw, 2010). The current results indicated that Fagaceae and Lauraceae were more abundant in the tropical montane zone, similar to other findings in South East Asia (Ohsawa, 1991; Tagawa, 1995; Sri-Ngernyuang *et al.*, 2003). This type of vegetation is also known as oak-laurel forest, the term used by Kochummem (1989) and Tagawa (1995).

Species composition and number of individuals diverged between sites (Table 6). A Sørensen similarity index between sites revealed that the compositions of S2 and S3 were the most similar, with a value of 35.2%. *Syzygium* sp. was a generalist on Payeh Maga Highland, having broad altitudinal ranges extending from 900 to 1,700 m (Table 6). Similar observations have also been reported by Kitayama (1992) and Aiba *et al.* (2006) at Mount Kinabalu at similar elevations. *Lithocarpus conocarpus* and *Calophyllum* sp. were two other dominant species at S1 (Table 6). Saw (2000) also reported that *Calophyllum* sp. was easily found in the upper dipterocarp forests of Peninsular Malaysia. *Lithocarpus* species (*L. conocarpus, L. andersonii, L. bennettii* and *Lithocarpus* sp.) were also found abundantly at S2 and S3 (Table 6), consistent with the dominance of its family in these areas.

In this study, a total of 19 endemic species of Borneo were recorded (Table 7). The highest number of endemic species was found in S1 followed by S2 and S3. Four endemic Dipterocarpaceae (*Shorea crassa* at S2; *Shorea isopteran, Shorea parvistipulata* and *Vatica micrantha* at S1) were found in this study area (Table 7). Three endemic species each from Fagaceae, Lauraceae and Myrtaceae were also documented during this study (Table 7). At a similar elevation range to S3, Kueh *et al.* (2017) recorded 12 endemic species in their study.

This enumeration successfully recorded 891 individuals with $dbh \ge 10$ cm from all three different elevation sites. However, there was no significant difference in tree density between the three elevation sites (Table 4). This result was similar to Kitayama's (1992) report on Mount Kinabalu, where no differences in stand density were observed at similar altitudes to those of the current study. However, the current study recorded a much higher number of stems compared to those from Mount Kinabalu (1,430 trees ha⁻¹ compared to 759 tree ha⁻¹ at S2 or at 1,400 m).

The inverted J-curve trend was observed across all sites, with the lowest number of individuals having larger tree dbh (Figure 3). This is a common pattern in tropical forests, with only a few individuals in higher diameter classes and numerous individuals in smaller diameter classes. Sahu *et al.* (2012) suggested that this trend is an indicator of the healthy forest recruitment process. A larger number of smaller stems will ensure the replacement of trees with larger diameters when the latter perish. S3 trees exhibited the lowest average *dbh* (16.9 cm) among the three sites in the Payeh Maga Highland (Table 4). The diameter distribution of trees has often been used to represent the population structure of forests (Rao *et al.*, 1990).

The height of the tree decreases at Payeh Maga Highland with increases in elevation (Figure 4). Three strata were observed in S1, while S2 had two strata and only one tree stratum was observed in S3. These three stands can be considered lowland dipterocarp forest, lower montane forest, and upper montane forest, respectively (van Steenis, 1984; Whitmore, 1984). According to their classification, lowland forest has three strata with a height of 25 - 45 m; lower montane forest generally has two strata with a height of 15 - 33 m; and upper montane forest has one stratum with a height of 1.5 - 18 m.

The total stand basal area at Payeh Maga Highland was $83.6 \text{ m}^2 \text{ ha}^{-1}$, with S1 having the highest value of $34.7 \text{ m}^2 \text{ ha}^{-1}$ (Table 6). Several authors (Kitayama, 1992; Aiba & Kitayama, 1999; Culmsee & Pitopang, 2009; Clark *et al.*, 2015) did not find any correlation between basal area and elevation, especially at altitudes of 900 - 1,700 m. In the current study, the basal area of the three different sites decreased as elevation increased (with S3 having only 19.3 m² ha⁻¹).

Based on the analyses, it can be suggested that increasing altitude results in declining tree species diversity, average dbh, height and basal area under both conditions across the three sites (Table 5). This similar trend is in accordance with Yamada (1977), who suggested that the abovementioned forest structure also decreases with increasing altitude in tropical mountains. Even though elevation was the mainly prominent environmental related for the diversity and composition of tree communities in elevational transect study, a noteworthy relationship of the variation in tree diversity on Borneo was explained by soil properties (Slik *et al.*, 2009). It shows that significant contrasts in edaphic conditions, such as the difference between soils on silicate, calcareous, or ultramafic rocks, may in particular cases lead to larger differences in tree diversity and community structure than the elevation gradient itself (Aiba *et al.*, 2015).

3.4.2 Effects of Logging and Vegetation Recovery

Currently, large highland forests are disturbed due to forestry operations and agricultural activities. Hence, understanding the structure, diversity, and ecosystem processes of this type of forest is becoming increasingly important. This is because these forests play a vital role in the trade-off between their inherent functions, such as maintenance of biodiversity and carbon stocks, provision of goods, and other services (Picard *et al.*, 2012; Yamada *et al.*, 2013; Sist *et al.*, 2015). Knowledge regarding the impacts of logging and the rate of recovery of forests is still insufficient, particularly in the mountainous region of Borneo, thus causing substantial uncertainty about those values (Yosi *et al.*, 2011).

Recovering species richness due to logging is complex and depends on the prior forest status (Tilman, 1999) and on the extension and distribution of disturbed areas across a landscape (Sheil & Burslam, 2003; Cadotte, 2007; Bongers *et al.*, 2009). Reports have indicated that species richness may increase (Plumptre, 1996; Cannon *et al.*, 1998; Berry *et al.*, 2010), decrease (Makana & Thomas, 2006; Gutiérrez-Granados *et al.*, 2011; Clark & Covey, 2012) or remain unchanged (Verburg & van Eijk-Bos, 2003; Bischoff *et al.*, 2005; Medjibe *et al.*, 2011) in response to selective logging in tropical forests. In an area where recurrent disturbances that are able to prevent competitive elimination of species occur frequently but not so frequently as to exclude most species, higher diversity can be expected (Connell, 1978; Hubbell *et al.*, 1999; Whittaker *et al.*, 2001).

Although commercial logging in Sarawak has been operating since the 1970s, the effects on tree diversity recovery remain largely unknown due to a lack of field information. The effects on tree species diversity are complex. Analyses from this study suggested that after 30 years of logging, the number of species was found to be higher in the logged areas than in the unlogged stands at S1 and S3 (Table 4). Despite this, the unlogged area of S2 has a significantly higher number of tree species than the logged stands (Table 4).

The presence of secondary species within the logged stands can be an indicator of the degree of degradation and its recovery rate for a particular site. After more than 30 years, five secondary tree species are still recorded in logged plots at S1 (*Adinandra dumosa, Adinandra* sp., *Euodia* sp., *Glochidion* sp. and *Vernonia arborea*) and S3 (*A. dumosa, Adinandra* sp., *Cratoxylum* sp., *Saurauia* sp. and *Syzygium* sp.). In S2, 15 out of 33 tree species found are secondary species in the logged stands, indicating that the forest area is still in the early recovery stages. Generally, the topography of S2 is hilly with multiple slopes of 30 - 60% and a rocky surface with a thin soil layer (most trees grow on the rock boulders). Once this area is subjected to logging, site degradation is high, and residual stand damage can be substantial. This resulted in a slower recovery site and eventually delayed the process of plant succession. In addition, lower N content recorded in the logged areas will continue to affect the growth of the residual stand and newly established seedlings (Table 3). This adds to the delay of overall forest recovery.

Unique species with fewer individuals were significantly higher in the unlogged plots of S2, which resulted in significantly higher Shannon index values when compared to the logged stands (Table 4). Moreover, the logged stands of S2 displayed a species oligarchy (80% of the respective plot density was shared by just 10 secondary species in logged plots). There was not much difference in species richness when considering trees of 20 cm *dbh* and above at smaller plot sizes of 400 m² between logged and unlogged stands. As indicated earlier, the existence of differences in species richness at larger scales does exit (for example, at S3, Kueh *et al.*, 2017), similar to other reported results (Berry *et al.*, 2008; Imai *et al.*, 2012; Jayakumar *et al.*, 2017).

These findings suggest that the response of species richness varies depending on sample size. At the larger scale, the difference is high due to the inclusion of varieties of stand patches (higher species composition due to different microclimate conditions, which encourage the growth of different species, and more heterogeneity in vegetation structure) that are affected by different intensities of disturbance (volume of timber extraction, severity of damage on soil and residual standing trees, intensity of roads and trails, etc.).

It is difficult to generalize the effects of logging on stand density and basal area. The stand recovery at S1 indicated that the logged area had 19% more stems than the unlogged stands. Despite this, data analyses suggested that the basal area was lower compared to unlogged plots (Table 4). This is because logged stands had trees with smaller diameters at S1 (Figure 3). Meanwhile, logged stands at S2 recorded 17% less stems than unlogged plots (Table 4). The basal area of the logged area was only half of that of the unlogged stands (Table 4). A similar trend was observed for S3, where a lower number of stems and basal areas (ca. 9.1 and 23.7%, respectively) were recorded for logged plots (Table 4). These results indicated that the effects of selective logging on stem density and stand basal area are still prevailing even after 30 years, especially at S2 and S3. Jayakumar *et al.* (2017) also reported similar observations in a 27-year-old forest after a selective logging operation in Western Ghats, India.

The low tree density in logged plots at S2 in the current study (Table 4) was due to the low regeneration of primary species. The composition of primary species in the logged plots was poor, with only 24% of tree species recorded. Okuda *et al.* (2003), Villela *et al.* (2006) and Jayakumar *et al.* (2017) reported that the number of light-demanding secondary species increased while shade-tolerant species were reduced as a result of selective logging. Results of the current study, especially observations from S2, confirm with their results. The majority of trees in the logged stands of S2 have heights of 10.0 - 19.9 m (Figure 4), while 40.8% have canopies with diameters of 5 - 10 m (Figure 5). These created stands occupied by intermediate strata species rather than canopy strata species as observed in the unlogged plots. Approximately 45.5% of the recorded species were secondary species with low density wood. Saldarriaga (1987) and Jayakumar *et al.* (2017) reported similar results in which logging promoted lower strata species with light wood densities.

Succession at S1 and S3 was more rapid than at S2, where there were accumulations of 93.1% and 76.3%, respectively, of the basal area of the unlogged stands only after three decades (Table 4). The rapid accumulation might be due to more fertile soils at S1 and S3. When assessing the succession rate of secondary forests in Neotropical areas, Guariguata & Ostertag (2001) found that rapid accumulation of basal area was obtained from areas with high soil fertility. The remnant of large trees in logged plots also contributes to the higher basal area values obtained in the analysis.

Logging activities did not contribute to the similarity of tree communities among the study plots. The Sørensen similarity index indicated that S3 had the highest species similarity (44.7%) between logged and unlogged stands, followed by S1 (35.4%) and S2 (21.5%). The direction and stage of forest recovery could only be studied after 30 years because the pre-logging state of the selectively logged plots was unknown.

The current study sites retained many of the structural appearances typical of secondary forests, such as lower stem height, smaller canopy, and basal area (Brown & Lugo, 1990), compared to unlogged or primary forests. Forest size class structures among the three different sites were similar. In general, many structural characteristics of these three forests in logged plots were approximately 75% of the unlogged plot values (Figure 3, 4 and 5). Only S2 showed significant differences in dbh and height between unlogged and logged plots (Table 4). The lack of statistically significant differences in S1 and S3 might be due to the limited number of plots (five) in each forest type.

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At the landscape level, there is an advantage of rapid recovery, particularly for species floristic composition, if disturbed forest areas are in juxtaposition to unlogged areas. However, the current study revealed that the residual impacts on stand structure, composition, density, and basal area are still noticeable three decades after logging, especially at S2 (Table 5), even though seed sources from unlogged stands were in the close vicinity. The recovery of important species in the logged plots indicates that those species were in various degrees of recovery at different elevations and sites with higher recovery at S3 and S1 and poorer recovery at S2 (Table 5). These results suggested that the selectively logged forests would require a longer period of recovery time than the predicted forests, which is in agreement with other recent studies (Chua *et al.*, 2013; Yamada *et al.*, 2013; Jayakumar *et al.*, 2017).





Figure 5 : Tree distributions among tree canopy diameter class of different sites [(a) S1, (b) S2 and (c) S3] at the Payeh Maga Highland

The species richness (as gauged by the number of species, Shannon–Wiener index and Fisher's α index) of the logged S1 and S3 plots was similar to that of the unlogged stands; however, their compositional similarity (Sørensen index) between the two different forest types was only 35.4% and 44.7%, respectively. Ferreira & Prance (1999) and Brearley *et al.* (2004) reported similar findings where more than 30 years after logging, the forest still exhibited major differences in terms of floristic composition, although difference in basal area was insignificant.

However, at S2, the logged areas recorded significantly lower H' values compared to unlogged plots (Table 4). The difference was due to the huge gap in the number of species found in both areas. The majority of species recorded in the logged plots were categorized as pioneer species, and a slow regeneration process was clearly observed in those plots. When diversity index values were compared to those reported for the lowland tropical forests (Kueh *et al.*, 2017), the current study recorded lower values of 1.92–2.60 (Table 4). At similar elevations, the H' values recorded in this study were in the lower range of the values reported by Aiba & Kitayama (1999), Do *et al.* (2015), and Zhu *et al.* (2015). The H' values at S3 (Table 5) were within the ranges reported by Kueh *et al.* (2017).

The recovery of logged stands at the structural level depends on the proximity of the seed sources from unlogged areas as well as the remnant trees within the stand that serve as mother trees. The severity of logging damage to the existing stand and site were also important to determine the rate of stand recovery. The current study on floristic composition indicated the importance of logged forests as basins of biodiversity in tropical mountain landscapes. Continuous monitoring of the established permanent plots will provide vital data on forest recovery and help to determine factors affecting those processes.

3.5 Conclusions

Tree diversity, floristic composition and stand structure at different altitudes of the Payeh Maga Highland were examined. A total of 891 tree stems were enumerated and 203 tree species were recorded at three different elevations. A linear relationship between elevation and tree diversity was observed. Higher species diversity was found at the lower altitude and decreased significantly with increasing elevation which shows the relationship between elevation and tree diversity. There were 55 tree families recorded with Dipterocarpaceae and Lauraceae dominating lower elevations, while Lauraceae and Fagaceae dominated higher elevations. *Syzygium* sp. was the most abundant species across three sites. In addition, 19 Bornean endemic tree species were found during this study, and the lowest elevation site had the highest number of species. However, no differences were observed in stand density between different study sites. An inverted J-curve trend was observed at all three sites, with only a few individuals at higher diameter classes. Increasing altitude resulted in a reduction of average stand dbh, height and total basal area. Stands that were subjected to selective logging still retained secondary forest structural appearances after 30 years of succession. Generalization of stand diversity recovery at this study is difficult. Stands at different altitudes respond differently when subjected to logging, which might be related to logging intensity, logging damage, site topographic conditions, soil properties and nutritional status, proximity of seed sources and the remnant trees within the stand that serve as mother trees. The presence of secondary light-demanding species in logged stands can be an indicator of the recovery rate of a particular site.

This study provides information that can be used as a benchmark for future research in the highland area of Borneo. The current study on floristic composition indicated the importance of logged forests as banks of biodiversity in tropical mountain landscapes. Continuous monitoring of the established permanent plots will provide vital data on forest recovery and help to determine factors affecting these processes.

CHAPTER 4

TREE GROWTH AND ABOVEGROUND BIOMASS IN A TROPICAL MOUNTAIN FOREST THIRTY YEARS AFTER SELECTIVE LOGGING IN SARAWAK, BORNEO

4.1 Introduction

Tropical rainforests play an important role in the global carbon cycle, and the destruction of tropical rainforests has been reported to be a major source of greenhouse gases, such as carbon dioxide, methane and nitrous oxide (Kanninen et al., 2007; Nabuurs et al., 2007). Lowland tropical rainforests are rapidly being explored and logged, and these activities have reached tropical upland forests. Zeng et al. (2018) noted that the conversion of forested mountainous regions into cropland can have major environmental impacts, ranging from soil retention to water quality issues. Greater reductions in AGB in logged forest indicate a greater degree of forest degradation, which is detrimental to the ecosystem services of mountain forests, such as water and soil conservation. Borneo is covered with a large mountainous interior; however, this area is becoming increasingly affected by anthropogenic activities, especially by logging and agricultural expansion. Indeed, the WWF (2016) reported that upland (300 - 800 m) and montane forests (800 - 4,000 m) covering 800,000 ha and 200,000 ha, respectively, were lost between 2005 and 2015.

In Malaysia, selective logging, wherein trees above certain cutting limits (typically 45 cm in diameter) are logged, has been practised since 1978 (Thang, 1987). Selective logging is considered a sustainable approach for managing forests and thus is an essential part of forest certification. Therefore, all forest management units in Sarawak, Malaysian Borneo, use this management practice. Generally, any logging activities can damage residual stands; in Malaysia, 8-47% of standing trees have been reported to sustain damage (Saiful & Latiff, 2014), and an estimated 55% of the basal area has been reported to be destroved by logging activities (Ho et al., 2004). Saiful & Latiff (2014) found that the extraction of 27 trees per hectare reduced the number of individuals >1.5 m height by 47.7% six months to one year after logging. The recovery of the logged-over forest to climax conditions might take decades or even centuries, as suggested by Whitmore (1991), Kartawitana (1994) and Shima et al. (2018). Forest recovery in terms of basal area and AGB is seldom reported for mountainous regions, including Borneo. Changes in basal area, AGB, and carbon balance within an ecosystem can be estimated through knowledge of the dynamic processes of forest (Lewis et al., 2009). Information on tropical mountain rainforest dynamics is limited and is urgently needed to understand changes in tree populations due to natural and man-made disturbances as well as climate change.

Given the lack of studies on forest biomass, growth and regeneration in the highlands of Sarawak, the objective of this research was to estimate the AGB and growth status of the Payeh Maga Highland at different elevations in primary and logged-over forests. This study provides some baseline information on the growth rate (*dbh*, basal area and AGB), the status of forest recovery 30 years after selective logging and the factors influencing forest recovery. Estimates of AGB can provide indicators of forest degradation, which is detrimental to ecosystem services, such as water and soil conservation.

4.2 Materials and Methods

4.2.1 Site Description

This research was carried out in a tropical mountain range in Sarawak, northern Borneo, Malaysia. The Payeh Maga Highland is located near the town of Lawas (4°50'0"N, 115°24'0"E) in the Limbang Division of northern Sarawak, Malaysia. The highland spans an elevation range from 500 and 1,800 m on the southern side of the Meligan Range and is located approximately 70 km from Lawas. This area is a part of the Heart of Borneo initiative, a transboundary project of Brunei Darussalam, Malaysia and Indonesia that maintains and manages forested areas in the Borneo highlands (WWF, 2014).

The forest is a part of a larger forest management unit (FMU) that is managed primarily for timber production under a sustainable forest management scheme. The site consists of regenerated logged-over and intact primary forests. The highland covers an area of approximately 10,000 ha (within 4°11'30"–4°33'30"N and 115°29'30"–115°39'30"E). The annual temperature decreases from 24.6°C at 960 m to 20.6°C at 1,350 m and 19.3°C at 1,650 m, and the annual mean rainfall increases with elevation from 2,869 mm yr⁻¹ at 960 m to 3,737 mm yr⁻¹ at 1,350 m and 3,911 mm yr⁻¹ at 1,650 m. Climatic information (temperature using a temperature/RH smart sensor and rainfall using a 0.01" rain gauge smart sensor) was collected hourly using HOBO RX3000 weather station kits (Onset Computer Corporation, Bourne, MA) that were installed in each forest type throughout the study for 12 months. At each study location, the weather station was installed in an open area (no canopy cover within at least a 15-m radius), and sensors (temperature/RH and rain gauge sensors) were mounted to different poles at a height 1.5 m above the ground surface.

The Payeh Maga Highland has a complex topography of multiple slopes ranging from gently rolling to hilly (class IV terrain >35°) along its altitudinal gradient. The geological substrate of the area consists of shallow marine formations. This area was formed during the Late Oligocene to Early Miocene of the Meligan Formation, which is predominantly composed of sand (James, 1984). Red-yellow podzolic soil of the Merit soil series is the main soil type in the area.

4.2.2 Plot Establishment

Stands and growth dynamics were assessed for 12 months. The forests were classified into logged-over and primary forest stands. The forest areas were selectively logged 30 years ago, where trees larger than 50 cm in *dbh* were harvested and then left to regenerate naturally. Generally, 8 - 15 trees ha⁻¹ are harvested during logging operations based on available forest resource assessment information (Pinard *et al.*, 2000). This unsupervised conventional logging activity includes the use of chainsaws for felling and crawler bulldozers for skidding. The land-use history was provided by the timber company and confirmed by the local community.

Study plots were established at three different altitudes: 940 - 980 m (upper dipterocarp forest, S1), 1,300 - 1,420 m (lower montane oak-laurel forest, S2), and 1,630 - 1,660 m (upper montane forest, S3). At each elevation, logged-over and primary forest areas were determined (Figure 6). The presence of tree stumps was used as an indicator of past logging operations. Five 20×20 m plots were established in the logged-over forest area with one tree stump in the center of the plot at each elevation. A 20×20 m plot size was used because of the difficulty of locating large logged areas, especially in S2 and S3. Above 1,000 m, logging is difficult because of the challenging terrain, the high maintenance costs for logging roads caused by unstable soil and slope conditions and the smaller number of large trees. Thus, logged areas were often small, and patches were logged before they were abandoned. The hilly and steep topographic conditions of S2 made establishing larger plots in this area difficult. S2 plots were located on a steeper slope (up to 30°) with a high incidence of surface boulders. As a result, this area has thin-layer soil and forest litter cover.

At each elevation, another five plots were also established in the primary forest area. These plots had similar elevation, topographic and edaphic conditions. Plots within primary and logged-over forests were established at least 40 m apart from each other, as this distance is thought to be sufficient for offsetting any effect of logging disturbance, as the harvested logs rarely exceeded 40 m in length, and a low harvesting density was employed during the selective logging operation. All plots were established far away from any known logging roads or trails. A global positioning system navigation device was used to identify each plot location. The slope condition and elevation of each plot were also recorded.

4.2.3 Stand Measurement

Tree *dbh* (1.3 m above the ground on the uphill side of the trunk) was measured for trees of 10 cm *dbh* and greater. All measured trees were tagged with a numbered metal plate. The tree was tagged to ensure accurate reidentification. A band was painted at breast height, and diameter measurements were made on this painted band. During the second census, *dbh* was determined by

measuring the painted band. Diameter tape was used to measure *dbh*. Identification of tree species was performed on-site. For unidentified tree species, leaf specimens were gathered, and identification was performed with voucher specimens in the Herbarium of the Forest Department Sarawak, Malaysia. Initial measurements were taken in April–May 2016 and final measurements were collected in April–May 2017. Newly recruited trees (i.e., those with *dbh* of at least 10 cm) were painted, tagged, identified, measured and recorded at the second census. Trees were confirmed as dead only when the trunk lacked sprouts and rot had begun above breast height. Detailed information regarding stand structure, floristic composition and species diversity of the current research sites is provided in Laing *et al.* (2019). The average *dbh* for logged-over stands of S1, S2 and S3 was 19.96, 18.57, and 16.28 cm, respectively. The average *dbh* values for primary stands were 22.67, 21.89, and 17.36 cm for S1, S2 and S3, respectively.



Figure 6 : Location of the study area with three different study sites (S1, S2, and S3) in Sarawak, Malaysia, on Borneo

4.2.4 Biomass Estimation

A nondestructive method was used to determine tree biomass. As there are no specific allometric equations available for montane forests in Sarawak, a generic allometric equation was used in this study. AGB was estimated using the existing empirical allometric equation based on Brown *et al.* (1989) for tropical forests in moist life zones because only *dbh* was measured in this study.

AGB (kg) =
$$38.4908 - 11.7883 \ dbh + 1.1926 \ dbh^2$$
 (1)

4.2.5 Data Analysis

Differences in *dbh* growth and changes in basal area and AGB between primary and logged-over forests at each site, and total basal area and total AGB (second census) were tested using Student's t-tests (*P*<0.05). One-way ANOVA was used to determine whether there were significant differences for between forest types (S1, S2 and S3), and Tukey's Studentized Range Test was used for the mean separation test. These analyses were carried out using the SAS 9.4 package (SAS Institute Inc., NC, USA).

4.3 Results

4.3.1 Stand Density, Total Basal Area and Aboveground Biomass

In an area of 2000 m², a total of 160, 130, and 140 trunks were recorded in the logged-over stands of S1, S2 and S3, respectively, and 142, 159, and 154 trunks were recorded in the primary stands, respectively (Table 8).

The mean total basal area was not significantly different between primary and logged forests in S1 and S3 (Table 8). A significant difference between primary and logged-over forests was only observed for S2 (Table 8). Selective logging led to a 51.0% reduction in the mean total basal area in the S2 logged-over plots. All of the logged-over plots had lower AGB values than the primary plots in the different forest types; however, the difference was only significant for S2 (Table 8). A decline of 61.3% in AGB was recorded in the S2 logged-over plots compared with S2 primary plots.

Table 8 : Total basal area and aboveground biomass at the Payeh Maga Highland

Site	Forest type	Stand density (0.20 ha)	Basal area (m ² ha ⁻¹)	AGB (Mg ha-1)
C1	Logged	160	33.4 (±13.0) ^{a1}	340.9 (±135.3) ^{a1}
31	Primary	142	35.9 (±2.5) ^{a1}	387.7 (±44.2) ^{a1}
60	Logged	130	19.5 (±2.9) ^{b12}	181.5 (±33.6) ^{b2}
32	Primary	159	39.8 (±9.3) ^{a1}	432.4 (±102.6) ^{a1}
62	Logged	140	16.7 (±8.3) ^{a2}	144.5 (±53.9) ^{a2}
33	Primary	154	21.8 (+7.5) ^{a2}	201.6 (+101.1) ^{a2}

Means (n = 5) with standard deviation represented by different superscript lowercase letters indicate significant differences between primary and logged forests by Student's t-test, and different superscript numbers indicate significant differences between forest types by Tukey's Studentized Range Test (P<0.05).

4.3.2 Growth Response after Selective Logging

An obvious unimodal pattern in AGB growth (the greatest value was recorded at S2) was observed in the primary plots along the altitudinal gradient of the Payeh Maga Highland (Table 9). A similar pattern was also observed in the basal area; however, the changes were subtle (Table 9). The increases in tree *dbh*, basal area and AGB did not differ significantly between the logged-over plots and the primary stand plots in either S1 or S2 (Table 9). However, there were significant differences in *dbh*, basal area and AGB increment in S3 between logged-over and primary plots (Table 9). Higher *dbh*, basal area and AGB growth were recorded in the S3 primary plots than in the logged-over stands.

A unimodal outline in *dbh* growth rates was observed with middle size trees (*dbh* 30-49.9 cm) recorded the highest rate of growth except for the S3 logged-over plots which recorded no increment (Figure 7). However small trees (*dbh* 10.0 - 29.9 cm) contributed substantially to AGB increases than middle size trees in most types of forests and conditions except for the S1 primary plots (Figure 8) due to the high number of trunks found in this *dbh* class.

Five tree species had *dbh* growth rates of 0.5 cm yr⁻¹ and above in the S1 (Table 10). A similar growth rate was observed for six species in both S2 and S3 (Table 10). In all three forest sites in this study, *Lithocarpus* species, especially *L. conocarpus*, were among the best performers in terms of *dbh* growth (Table 10). However, *Suarauia* sp. had the highest *dbh* growth rates (0.8 cm yr⁻¹) in S1, *Nauclea* sp. (1.3 cm yr⁻¹) in S2 and *Elaeocarpus* sp. (0.7 cm yr⁻¹) in S3 (Table 10). Fagaceae was one of the most rapidly growing tree families in the Payeh Maga Highland. This tree family consistently had a *dbh* growth rate of 0.5 cm yr⁻¹ and above at all three different elevations (Table 10). However, Actinidiaceae and Fagaceae had the highest *dbh* growth rate in S1 (0.7 cm yr⁻¹), whereas in S2 and S3, Rubiaceae (1.2 cm yr⁻¹) and Podocarpaceae (0.7 cm yr⁻¹) had the highest *dbh* growth rates, respectively. In S1, Dipterocarpaceae (23 individuals) had a *dbh* growth rate of 0.4 cm yr⁻¹.

Assuming during the next cutting cycle, ten trees with *dbh* 50 cm and above (irrespective of tree species) will be harvested in one hectare, a projection was conducted to estimate the duration it will take for this number of harvestable trees to become in logged plots. According to the simulation, another 5 years is needed for S2 logged-over plots. Meanwhile S3 logged-over needs for more than 200 years.

Table 9 : Mean values of mean *dbh* growth rates and increments of stand-level basal area and aboveground biomass

Site	Forest type	dbh (cm yr ⁻¹)	Basal area (m² ha⁻¹ yr⁻¹)	AGB (Mg ha ⁻¹ yr ⁻¹)
C 1	Logged	0.37 (±0.10) ^{a2}	0.90 (±0.34) ^{a1}	10.5 (±4.0) ^{a1}
31	Primary	0.27 (±0.06) ^{a1}	0.75 (±0.05) ^{a1}	9.2 (±0.5) ^{a1}
60	Logged	0.59 (±0.11) ^{a1}	1.15 (±0.27) ^{a1}	13.1 (±2.9) ^{a1}
32	Primary	0.42 (±0.29) ^{a1}	1.03 (±0.41) ^{a1}	12.3 (±5.0) ^{a1}
62	Logged	0.18 (±0.09) ^{b3}	0.27 (±0.22) ^{b2}	2.6 (±2.1) ^{b2}
33	Primary	0.47 (±0.24) ^{a1}	0.96 (±0.25) ^{a1}	10.7 (±3.2) ^{a1}

Means (n = 5) with standard deviation represented by different superscript lowercase letters indicate significant differences between forest conditions by Student's t-test, and different superscript numbers indicate significant differences between forest types by Tukey's Studentized Range Test (P<0.05).

4.4 Discussion

4.4.1 Basal Area and Aboveground Biomass in Primary Forests

In the primary plots, a unimodal pattern for both basal area and AGB was observed, and the highest values were recorded at S2 (Table 8). A similar unimodal pattern was reported by Kitayama (1992) and Aiba & Kitayama (1999) on Mount Kinabalu. The basal area and AGB in the primary plots at the highest elevation (S3) were significantly lower than those in the S1 and S2 plots (Table 8). Similar observations were reported by Kitayama (1992), Aiba & Kitayama (1999), Asner et al. (2012; 2014) and Spracklen & Righeloto (2014), wherein basal area and AGB at higher altitudes were lower than those at lower elevations. The significantly lower basal area and AGB in this study might be related to available soil nutrient concentrations and physiological stress stemming from frequent cloud cover. S3 plots had significantly lower soil available P and exchangeable K concentrations compared with S1 and S2 plots (cf. Laing et al., 2019), which might be the main reason for the striking differences in basal area and AGB between forest types. During our study, S3 was frequently under cloud cover compared with S1 and S2. Such conditions could lead to reduced transpiration and photosynthetically active radiation, thus leading to physiological stress and reduced tree growth (Kitayama, 1992). The reduced transpiration rate might decrease nutrient uptake (which is already lower in S3 than in other forest types) by the plants, and the lower availability of photosynthetically active radiation reduces the rate of photosynthesis (Kitayama, 1992). This might be another reason why the basal area and AGB of S3 stands in the current study were significantly lower than the basal area and AGB of S1 and S2.

4.4.2 Basal Area and Aboveground Biomass in Logged-Over Forests

Generally, selective logging practices reduce forest stand basal area and AGB. Under these practices, trees larger than 50 cm in *dbh* were extracted. These trees are the largest contributors to basal area and AGB in primary forests (Berenguer *et al.*, 2014). The lower number of trunks (*dbh* >50 cm) in the loggedover plots in the current study (cf. Laing *et al.*, 2019) are indicators of past selective logging activities.

The basal area and AGB recovery after 30 years were different among the three different altitudinal forest sites. Even after 30 years, the effects of selective logging on basal area and AGB were still obvious in S2. Basal area and AGB were significantly lower in the S2 primary stands by 51.0% and 58.0%, respectively, relative to the basal area and AGB of the logged-over plots of S2 (Table 8). S2 logged-over stands experienced the greatest disturbance intensity compared with the two other logged forests, as indicated by the presence of a high number of light-demanding pioneer species within those plots (cf. Laing et al., 2019). This disturbance probably resulted from unsupervised logging operations, such as using untrained chainsaw operators in the felling procedure and the lack of pre-harvest climber cutting. Topographic conditions also increase the impact on residual forest stands. Matangaran et al. (2019) observed higher damage in residual stands at steeper slope plots that were subjected to selective logging operations in Indonesia. The current S2 plots were situated on steeper slopes with a higher number of surface boulders; thus, a higher logging impact is expected in this forest area compared with S1 and S3. The significant differences in basal area and AGB between forest conditions (logged-over and primary) were also related to the availability of large trees. Only two trees larger than 40 cm (Adinandra sp. and Vernonia arborea) were recorded in S2 loggedover plots, whereas the primary plots of S2 had 14 trees in the same diameter category.

A similar observation was reported by Saiful & Latiff (2014) in a hill dipterocarp forest in Kedah, Malaysia, where a greater number of pioneer species was found in areas experiencing higher damage and mortality rates after logging. This result indicated that a longer period is required for basal area and AGB recovery at S2. This finding is consistent with the results of other studies in logged-over forests of West Kalimantan (Cannon *et al.*, 1998), Central African Republic (Hall *et al.*, 2003; Gourlet-Fleury *et al.*, 2013), Peninsular Malaysia (Okuda *et al.*, 2003) and the Eastern Amazon (Sist *et al.*, 2014). Hence, artificial planting using locally available indigenous species should be implemented in S2 logged-over areas to increase their basal area and AGB.

There are no differences between the logged-over and primary plots for both S1 and S3 (Table 8) because a greater number of large relict trees can still be found in the logged-over S1 and S3, respectively. Thus, both forest areas experienced a smaller impact of logging than S2.

	S1		S2		S3	
No.	Species*	<i>dbh</i> (cm yr⁻¹)	Species	<i>dbh</i> (cm yr⁻¹)	Species	<i>dbh</i> (cm yr⁻¹)
1					Elaeocarpus sp.	
2	Saurauia sp. (7) Lithocarpus	0.8	Nauclea sp. (8) Vernonia arborea	1.3	(13) Dacrydium elatum	0.7
3	conocarpus (11) Lithocarpus ewyckii	0.7	(9) Lithocarpus	1.1	(9) Lithocarpus	0.6
4	(5) Lithocarpus gracilis	0.6	conocarpus (12) Lithocarpus	0.9	blumeanus (7) Lithocarpus	0.6
5	(7) Shorea platvclados	0.5	sundaicus (8) Lithocarpus lucidus	0.8	conocarpus (12)	0.6
6	(5)	0.5	(5)	0.6	(27)	0.5
7	accrescens (9)	0.4	(11)	0.5	Macaranga sp. (5)	0.5
'	(6)	0.4	Adinandra sp. (26)	0.4	sumatranum (6)	0.4
8	(12)	0.3	pachyphylla (12)	0.4	Adinandr <mark>a</mark> sp. (6)	0.3
9	Macaranga hypoleuca (7)	0.3	Duabanga moluccana (8)	0.3	Garcinia sp. (12)	0.3
10	Madhuca sericea (5)	0.3	Lithocarpus bennettii (2)	0.3	Lithocarpus bennetti (14)	0.3
1	Family*		Family*		Family* Podocarpaceae	
2	Actinidiaceae (8)	0.7	Rubiaceae (9)	1.2	(10) Elaeocarpaceae	0.7
-	Fagaceae (30)	0.7	Asteraceae (8)	1.1	(16)	0.6
3	Anacardiaceae (6)	0.6	Actinidiaceae (5)	0.9	Fagaceae (77)	0.5
4	Elaeocarpaceae (8)	0.6	Fagaceae (38)	0.7	Sapotaceae (7)	0.4
5 6	Asteraceae (6) Dipterocarpaceae	0.4	Anacardiaceae (5) Elaeocarpaceae	0.6	Clusiaceae (14) Thymelaeaceae	0.3
7	(23) Calophyllaceae	0.4	(11) Melastomataceae	0.5	(5) Calophyllaceae	0.3
8	(16)	0.3	(7)	0.5	(12) Casuaripaceae	0.2
a	Lauraceae (21)	0.3	Phyllanthaceae (5)	0.5	(17)	0.2
9 10	Malvaceae (11)	0.3	(39)	0.4	(7)	0.2
10	(14)	0.3	Lythraceae (8)	0.3	Myrtaceae (52)	0.2

Table 10 : Rate of change in *dbh* of the top 10 tree species and families inPayeh Maga Highland

* Only species with \geq 5 individuals are shown. Numbers in brackets indicate the number of individuals detected

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Figure 7 : Mean values of mean *dbh* growth rates for *dbh* classes 10.0–29.9, 30.0-49.9 and ≥ 50.0 cm



Figure 8 : Mean values of increments of above ground biomass for *dbh* classes 10.0–29.9, 30.0–49.9 and \geq 50.0 cm

4.4.3 Mean Stand Growth

During secondary succession, approximately 5–10 Mg ha⁻¹ yr⁻¹ of biomass can accumulate in tropical forests (Brown & Lugo, 1990; Alves *et al.*, 1997; Steininger, 2000). Brown & Lugo (1990) noted that the rate of increase is faster during the early stage of succession (up to 15 years). Mazzei *et al.* (2010) reported that the average growth rate of logged stands was double that of primary stands four years after logging.

After 30 years of recovery, the growth rate of logged-over stands was similar to that of primary stands except for S3, where primary plots recorded significantly higher *dbh*, basal area and biomass rate of growth than did logged-over forests (Table 9). A total of 23 trunks in S3 primary plots had *dbh* growth rates of 1.0 cm yr^{-1} and above compared with only 5 trunks in the logged-over stands of S3. Thus, there were clear differences between these two forest conditions.

The growth rate of trees in S3 was not measured immediately after logging and is likely consistent with the pattern described by Brown & Lugo (1990), wherein growth rates were accelerated during the early succession period, especially the first 15 years. During these successional stages, gaps created by logging activities allowed once shaded and understory trees to increase their growth rates and created intense competition to colonize and dominate the canopy layer. Coomes & Allen (2007) suggested that plants associated with these stressed sites have traits that allow them to survive harsh conditions but also result in a slow growth rate. This could explain why the growth rates of trees in the S3 logged-over stand were lower than in the primary forest. This pattern is similar to that documented by Coomes & Allen (2007), who suggested that competition for both light and nutrients suppresses the growth of trees in mountain beech forests. Meanwhile, the supply of resources to trees is spatially heterogeneous because gaps created by the death of large trees allow light to penetrate the forest floor and sometimes result in the release of nutrients (Coomes & Grubb, 2000); thus, tree growth in natural forests, such as S3 primary forests, is immensely variable (Canham et al., 2004).

Plants are generally thought to grow faster in warmer climates; consistent with this expectation, S1 logged-over plots recorded significantly higher *dbh* growth rates than S3 plots (Table 9). However, logged-over S2 had a significantly higher *dbh* growth rate than both S1 and S3 (Table 9), as a total of 77 trunks (or 59.2% of total trunks) in the S2 logged-over stand were pioneer species, and 36 trunks had a *dbh* growth rate of 1.0 cm yr⁻¹ and above. Meanwhile, only 13 and 5 trunks recorded similar rates of growth in S1s and S3s, respectively. The high number of trunks with high *dbh* growth rates in S2 compared with S1 and S3 contributed to the substantial difference observed in the rates of growth.

There was no difference in basal area or AGB between logged-over S1 and S2 (Table 9) because of the presence of a higher number of large relief trees (*dbh*

more than 30 cm) in S1 than in S2. The growth increment in larger trunks contributes more in terms of basal area growth than smaller trees. Since AGB was calculated based on *dbh*, a similar result is expected.

The mean *dbh* growth rates and increments of stand-level basal area and AGB of primary forests did not significantly vary among sites (Table 9). Interestingly, forest stands experiencing warmer conditions had the lowest growth increments, which is related to the higher number of pioneer species with higher growth increments in the primary plots of S2 and S3. A total of 3, 10 and 16 trunks had *dbh* growth rates of 1.0 cm yr⁻¹ and above for the primary plots of S1, S2 and S3, respectively.

Supposing that the current recorded growth rates observed is maintained in the logged-over forests, the projection suggested that in order to be able to harvest ten trees with *dbh* of 50 cm and above in one hectare, require at least 30 years for S1, 35 years for S2 and centuries of regrowth for S3. These simulated values are much longer than the 25–30 years cutting cycle stated in the current selective logging practices for S2 and S3. Given that the observation duration in the current study is only 12 months, these information obtained may not accurately reflect long term recovery of such forests and need to be treated with caution as each year provides different growing conditions and different tree species will respond differently to changing weather patterns.

4.4.4 Management Implications

The results of this study may be useful for improving existing harvesting guidelines implemented in the highlands of Borneo to maintain the sustainability of the timber production of these highland forests. Previously used selective management practices could be improved in several ways. First, they could be improved by incorporating reduced-impact logging techniques. This method has gained popularity since the 1990s because it managed to reduce up to 50% of damage to residual stands compared with conventional logging methods (Putz et al., 2008). The fundamental technical elements in this technique include proper inventory and tree location mapping, planning of roads and skid trails, vine cutting prior to harvesting, directional felling with low stump cutting, efficient utilization of felled logs, optimal preparation of roads and skid trails so winched logs minimize forest floor disturbance, and utilization of slash management (e.g., Pinard et al., 1995; ITTO, 1996). Second, silvicultural treatments, such as enrichment planting using indigenous tree species (such as Lithocarpus species), can be applied, and the number of trees over 50 cm dbh to be retained could be increased to support the reproduction of timber species after logging. Third, in steep topographic conditions with a high incidence of surface boulders, such as S2 in the current study, logging operations should be avoided.

4.5 Conclusion

A unimodal distribution was observed for stand basal area and AGB in the primary plots of this study, with the highest values observed in S2. However, the growth rates among the primary forest types in this study were similar. After 30 years of selective logging, the stand basal area and AGB of the S2 logged-over plots were significantly lower than the stand basal area and AGB of the primary stands, indicating that a longer recovery period is required. However, significant differences in *dbh*, height and basal area increases were only observed in S3 between primary and logged-over forests. These uncertainties could stem from the short period of the study and the number of plots; thus, prolonged monitoring of the sites is needed, in addition to measurement of more parameters, especially competition with neighboring trees and canopy openness.

CHAPTER 5

CONCLUSION, STUDY LIMITATIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

5.1 Conclusion

The current study describes about the tropical mountain forests ecosystem regarding its flora diversity and stand dynamics. The tropical mountain forests ecosystem is a hotspot of biodiversity and provides a significant contribution to the humans' livelihoods and is vital for global carbon cycle. Regardless of its importance there is still a lack of information regarding floral diversity and their biomass as well as impact of logging operations on both tree diversity and biomass especially in Borneo Island of Sarawak.

The study was conducted at the Payeh Maga Highland, Sarawak, Malaysia. The plots (20×20 m each) were established both in the logged-over and unlogged forests at three different elevations: 750–1,200 m (S1), 1,200–1,500 m (S2) and >1,500 m (S3) to gather all the necessary information (tree height and *dbh*) to determine the flora composition, plant biomass and stand productivity in response to logging operation 30 years ago. The stand growth dynamics were also assessed at 12 months interval. Identification of tree species was performed on site and for any unidentified tree species, leaf specimens were gathered, and identification was made with voucher specimens in the Herbarium of the Forest Department Sarawak, Malaysia.

As a conclusion, a total of 891 individuals with *dbh* ≥10 cm were recorded from all three different elevation sites. The trees comprised 55 families, 98 genera and 203 species. The lowest elevation site, S1, contained the highest number of species (117) and families (42), while the highest elevation site, S3, contained the lowest diversity (25 families and 59 species). The most speciose families in S1 were Dipterocarpaceae (11 species) and Lauraceae (10 species), while Lauraceae and Fagaceae were the dominant tree families at S2 and S3. Syzygium sp. was the most abundant species across the sites. A total of 19 endemic species to Borneo were found in this study area, with 11 species recorded at S1, seven species at S2 and another four species at S3. Secondary tree species are still abundantly found in the logged plots, especially at S2 (45.5%), indicating that after 30 years, these stands are still in the recovery stages. The logged plots retained typical secondary forest appearances with lower canopy diameter and height as well as *dbh* than the unlogged forest. Although the species richness approached that of the unlogged plots, especially in S1 (90.9%) and S3 (94.9%), there were still major differences in the floristic compositions between logged and unlogged plots indicated by the Sørensen index (35.4%, 21.5% and 44.7% for S1, S2 and S3, respectively).

In the primary plots, a unimodal pattern for both basal area and AGB was observed, and the highest values were recorded at S2. The basal area and AGB in the primary plots at the highest elevation (S3) were significantly lower than those in the S1 and S2 plots. The mean total basal area was not significantly different between primary and logged forests in S1 and S3. A significant difference between primary and logged-over forests was only observed for S2. Selective logging led to a 51.0% reduction in the mean total basal area in the S2 logged-over plots. All of the logged-over plots had lower AGB values than the primary plots in the different forest types; however, the difference was only significant for S2. A decline of 61.3% in AGB was recorded in the S2 logged-over plots compared with S2 primary plots.

In all three forest sites in this study, *Lithocarpus* species, especially *L. conocarpus*, were among the best performers in terms of *dbh* growth. However, *Suarauia* sp. had the highest *dbh* growth rates (0.8 cm yr⁻¹) in S1, *Nauclea* sp. (1.3 cm yr⁻¹) in S2 and *Elaeocarpus* sp. (0.7 cm yr⁻¹) in S3. Fagaceae was one of the most rapidly growing tree families in the Payeh Maga Highland. However, Actinidiaceae and Fagaceae had the highest *dbh* growth rate in S1 (0.7 cm yr⁻¹), whereas in S2 and S3, Rubiaceae (1.2 cm yr⁻¹) and Podocarpaceae (0.7 cm yr⁻¹) had the highest *dbh* growth rates, respectively.

In this study, stands that were subjected to selective logging still retained secondary forest structural appearances after 30 years of succession. Generalization of stand diversity recovery at this study is difficult. Stands at different altitudes respond differently when subjected to logging, which might be related to logging intensity, logging damage, site topographic conditions, soil properties and nutritional status, proximity of seed sources and the remnant trees within the stand that serve as mother trees. The presence of secondary light-demanding species in logged stands can be an indicator of the recovery rate of a particular site. Therefore, awareness upon the ecological processes in tropical rainforests following selective logging is extremely important for improving predictions of forest development and guidelines for sustainable forest management.

Furthermore, understanding of tropical rainforest's responses to different levels of anthropogenic and natural disturbance and their potential consequences for ecosystem functions is crucial for an insight of changes in ecosystem properties and stability. Interactions between forest landscape and parameters of forest dynamics such as rates of tree growth, mortality and recruitments rates must be clarified to improve knowledge of ecosystem controls.

This study provides information that can be used as a benchmark for future research in the highland area of Borneo. The current study on floristic composition indicated the importance of logged forests as banks of biodiversity in tropical mountain landscapes. Continuous monitoring of the established

permanent plots will provide vital data on forest recovery and help to determine factors affecting these processes as assessment of forest recovery after selective logging require longer study duration since the stand dynamics in the studied stands have reached a stable growing condition.

5.2 Study Limitations

Although this study has found a lot of useful information regarding tree flora diversity and stand dynamics 30 years after logging, it still subjected to several limitations. First is the low number of designated plots and small plot size. Due to smaller and low numbers of plots, the result from the data seems biased and did not show much of significant differences between two different forest conditions.

There is only one yearly observation (12 months interval) was conducted and probably too short to gather information related to tree growth response due to weather changes. This limitation was due to financial, labor and logistics constraints. A longer study duration perhaps can give a better result in examine the growth variation due to climatic changes (as moisture, light and temperature have their roles in determining tree growth).

5.3 Recommendations for Future Research

Therefore, to overcome these limitations, a few recommendations are suggested for future research at this study sites:

- (i) Establishing larger plot of 50×50 m and maintain at least 5 of those plots in each study site covering an area of 1.25 ha. This is because larger number of plots can generate more information in quantitative study, higher number of tree species and families will be enumerated (Kueh *et al.*, 2017).
- (ii) A longer study duration (yearly observation) of up to three years to catch any variation in weather and observe growth responses due to those environmental changes.
- (iii) More data loggers should be installed within each forest conditions and elevations. Two to five units should be included in each forest type at each elevation to monitor the environmental changes within the plots. By doing so, it can yield more data on environmental factors thus able to determine whether these parameters are confounding or otherwise. At the same time microclimate information will be make available as well, and
- (iv) Nutrient cycling study should be carried out such as collecting litter fall, determining forest floor decomposition and litter accumulation as these information play a vital part in determining the nutrient availability and understanding its influences on stand dynamics.

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APPENDICES



Figure 9 : Trees were tagged with numbered plate and measured



Figure 10 : Location of data logger at Site 2 (S2)



Figure 11 : Presence of tree stump as indicator for logged plots



Figure 12 : Conditions in unlogged plots

6



Figure 13 : Map of location of Lawas town and the Payeh Maga Highland

BIODATA OF STUDENT

Renee Sherna Anak Laing was born on 1st June 1991 at Miri, Sarawak. She received her primary education at Sekolah Jenis Kebangsaan (C) Chung San, Miri. Then, she attended her secondary education at Sekolah Menengah Kebangsaan Riam, Miri. She then continued her diploma level at Universiti Putra Malaysia Bintulu Campus and was awarded Diploma in Forestry in 2012. In 2016, she was awarded of Bachelor of Forestry Science with First Class Honours from Universiti Putra Malaysia. Afterwards in March 2016, she was enrolled as a full time Master of Science student at Universiti Putra Malaysia Bintulu Campus.



LIST OF PUBLICATION

- Laing RS, Ong KH, Kueh RJH, Mang NG, King PJH, Sait M. 2019. Stand structure, floristic composition and species diversity along altitudinal gradients of a Bornean mountain range 30 years after selective logging. Journal Mountain Sci. 16: 1419–1434.
- Laing RS, Ong KH, Kueh RJH, Mang NG, King PJH. 2021. Tree growth and aboveground biomass in a tropical mountain forest thirty years after selective logging in Sarawak, Borneo. Glob Ecol Conserv. 26: e01461.





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