# DIVERSITY AND COMPOSITION OF FERNS (MONILOPHYTES) OF PRIMARY AND LOGGED-OVER FOREST IN DANUM VALLEY AND ULU SEGAMA FOREST COMPLEX

**FLORINA ANTHONY**

**INSTITUTE FOR TROPICAL BIOLOGY AND CONSERVATION**
**UNIVERSITI MALAYSIA SABAH**
**2017**

# DIVERSITY AND COMPOSITION OF FERNS (MONILOPHYTES) OF PRIMARY AND LOGGED-OVER FOREST IN DANUM VALLEY AND ULU SEGAMA FOREST COMPLEX

FLORINA ANTHONY

THESIS SUBMITTED IN FULFILMENT FOR
THE DEGREE OF MASTER OF SCIENCE

INSTITUTE FOR TROPICAL BIOLOGY AND
CONSERVATION
UNIVERSITI MALAYSIA SABAH
2017

# UNIVERSITI MALAYSIA SABAH

BORANG PENGESAHAN STATUS TESIS

JUDUL : **DIVERSITY AND COMPOSITION OF FERNS (MONILOPHYTES) OF PRIMARY AND LOGGED-OVER FOREST IN DANUM VALLEY AND ULU SEGAMA FOREST COMPLEX**

IJAZAH : **MASTER OF SCIENCE (ECOLOGICAL PROCESSES)**

Saya **Florina Anthony**, Sesi **2014-2017,** mengaku membenarkan tesis Sarjana ini disimpan di Perpustakaan Universiti Malaysia Sabah dengan syarat-syarat kegunaan seperti berikut:-

1. Tesis ini adalah hak milik Universiti Malaysia Sabah.
2. Perpustakaan Universiti Malaysia Sabah dibenarkan membuat salinan untuk tujuan pengajian sahaja.
3. Perpustakaan dibenarkan membuat salinan tesis ini sebagai bahan pertukaran antara institusi pengajian tinggi.
4. Sila tandakan ( / ):

| | | |
|---|---|---|
| ☐ | SULIT | (Mengandungi maklumat yang berdarjah keselamatan atau kepentingan Malaysia seperti yang termaktub di dalam AKTA RAHSIA 1972) |
| ☐ | TERHAD | (Mengandungi maklumat TERHAD yang telah ditentukan oleh organisasi/badan di mana penyelidikan dijalankan. |
| ☐ | TIDAK TERHAD | |

Disahkan Oleh,

**FLORINA ANTHONY**
**MX1411005T**

(Tandatangan Pustakawan)

Tarikh: 06 September 2017

(Prof. Madya. Dr. Monica Suleiman)
Penyelia

# CHAPTER 1

# INTRODUCTION

## 1.1 Background of the study

Tropical rainforest cover around 1571 million ha of the earth's surface and represent one of the world's largest terrestrial biomes (Van Gardingen *et al.*, 2003). However the expansion of deforestation and forest degradation especially in Southeast Asia have led to major concern regarding the fate of tropical biodiversity (Gardner *et al.*, 2009; Frumhoff, 1995). As a result of deforestation, secondary and logged-over forests cover more than 600 million ha of the land area in the tropics (Brown and Lugo, 1990). Selective logging has been the principal cause of disturbance in Southeast Asia (Collins *et al.*, 1991). Today, large areas of primary forests in Southeast Asia have been converted into secondary forests and logged-over forest. High rates of deforestation have led to an unprecedented loss of biodiversity in the humid tropics.

Southeast Asia comprises of 11 countries with a combined population of over 565 million. It is one of the most biologically diverse regions of the planet. Even though the region occupies just 3 percent of the world's surface, it accommodates about 20 percent of all plants, animals and marine species (Braimoh *et al.*, 2010). Southeast Asia includes three major diverse countries (Indonesia, Malaysia and the Philippines) and contains four of the world's 25 biodiversity hotspots as designated by Conservation International (CI). However, vast areas of primary rainforests in Southeast Asia today are rapidly dwindling through exploitation, large scale fires and conversion into agriculture. As a result, degraded vegetation types and secondary forests and logged-over forest are replacing patches of species-rich lowland rainforests.

In Malaysia alone, 4.6 million hectares of forest have been lost, accounting for a 20 percent reduction of forested land. By 2010 all forest outside of conservation areas in Borneo is likely to have been logged at least once, leading to severe pressure for conversion to oil palm plantations or other uses (Laurance 2007; Wilcove and Koh, 2010; Berry *et al.,* 2010). Cultivation and logging have been practiced for centuries, but increased dramatically in the twentieth century (Hamer *et al*., 2003; Collins *et al.,* 1991). The area of primary forest in Southeast Asia has decreased from 66.3 million ha in 2000 to 64.0 million in 2010, a loss of 2.3 million ha (Koh *et al*., 2012). For secondary forest, it was reported that it comprised of 67.5 percent of the total native forest area (Koh *et al*., 2012) and secondary forest may represent a key habitat for biodiversity conservation in Southeast Asia because of their great extent and their potential to regenerate into old-growth forests (Koh, 2007). As the extent of primary forest is shrinking throughout the tropics, a growing body of work has quantified the biodiversity values of degraded tropical forests. So, secondary forest may become important to biodiversity conservation efforts in the future.

Ferns can be specific to certain areas, thus the presence or absence of ferns is a good indicator to define primary and secondary forests. Previous study suggests that species richness of ferns group could be utilized as forest habitat quality indicator. This group contains enough differing species thriving in distinct habitats that respond to environmental factor changes brought on by disturbances caused by human activities (Beukema and Noordwijk, 2007). In addition, species information of ferns is able to show the compositional patterns of other vascular plant species (Duque *et al*., 2005; Ruokolainen *et al*., 1997). Thus, ferns can be used to detect and forecast changes in the forest composition of an area. However, only a few studies on the impacts of forest disturbances on fern communities have been conducted, and most of the studies were focused on edge effects or forest fragmentation towards fern communities. Several studies on ferns have been conducted in Sabah but they focused mainly on distribution patterns and species inventories.

In this study different gradient of logging activities (selectively logged and highly degraded) was selected to quantify the residual value of logged forests in terms of their capacity to support ferns diversity. Through this study, the consequences of selective logging towards fern communities are investigated in thorough depth and will provide new information on the status of ferns in Danum Valley Conservation Area, Sabah. This is important in terms of management of the conservation area. This research will not only provide information that can be applied in the management plan for the conservation area, but also can indirectly protect the other organisms especially herbivore insects. This is because most of ferns communities act as keystone species and habitat of insect and other small organisms. In addition, this study can investigate to what degree the secondary forest can maintain the characteristics and function of primary forests by using fern communities as indicators.

## 1.2 Objectives of the study

There are three main objectives of this study:

1. To determine the ferns diversity between primary and secondary forests of high and low biomass in Danum Valley and Ulu Segama Complex.
2. To compare the species composition of ferns between the primary and secondary forests of high and low biomass.
3. To determine the ecological variables affecting the diversity of ferns between different forest types.

# CHAPTER 2

# LITERATURE REVIEW

## 2.1 Ecology of Ferns

Ferns are the most conspicuous spore-bearing land plant. They have evolved remarkable adaptations to extreme environments from tropical to cold temperate regions, from lowland to alpine zones and from xeric to aquatic conditions. These groups make up an important component of tropical and temperate floras and serve important functions in ecosystem processes in both canopy and the forest floor habitats (Hill and Silander, 2001).

The influence of the abiotic environment on fern distribution patterns is seen at all life stages. The majority of studies on fern ecology examined the sporophytes phase of the fern. However, the gametophyte is also sensitive to its environment, most notably of moisture conditions. Moisture is required to enable flagellated sperm to swim to the egg contained in the archegonium. At the same time, moisture also has predominant influences on species distribution and community structure.

Based on previous studies, fern communities are richest in diversity in wet tropical regions, especially in tropical rainforest where some 65 percent of extant fern species are found. Fern richness is highest in regions with high potential evapotranspiration, high number of rainy days and in mountainous areas (cloud forest) (Dzwonko and Kornas, 1994; Kessler, 2000; Pausas and Sáez, 2000).

Temperature, soil moisture and pH have also been cited as the dominant factors influencing fern distribution (Lellinger, 1985). Climatic differences are salient at a regional scale (Marquez *et al.,* 1997; Dzwonko and Kornas, 1994). However, at a local scale, fern diversity is influenced by soil fertility and the distance from putative Pleistocene forest refugee, for example in Uganda (Lwanga *et al.,* 1998). Within the canopy habitat, study has demonstrated that water use efficiency and drought tolerance of sporophytes have affected the distribution of epiphytic ferns (Andrade and Nobel, 1997; Hietz and Briones, 1998). Other factors that influenced the distribution of epiphytic ferns are morphological photoprotective mechanism (Watkins *et al.,* 2007), microclimate (Cardelus, 2002), tree characteristics and the individual plant adaptations. Substrate preference also seems to play a significant role in structuring some species distributions (Moran *et al.,* 2003). Kluge and Kessler (2011) reported that forest types might have no influence on species ferns elevational gradient.

In the humid tropics, the highest fern diversity is often found in cloud forests at mid-elevation, which is basically a result of water availability. For example, fern diversity is maximum at about 1800 m in both Costa Rica (Kluge, *et al.,* 2006), and on Mount Kinabalu, Borneo, 2000 m in Bolivia, (Kluge and Kessler, 2006), and at 2400 m on Mount Kilimanjaro, Tanzania (Kessler, 2001; Hemp, 2002). Elevational gradient also influenced the composition of fern communities (Mehltreter, 1996). Except for *Asplenium* and Dryopteridaceae that were found in each elevation, other fern species show distinct elevation preferences (Mehltreter, 1996). For example within the Hymenophyllaceae, the trichmanoid ferns typically occur at low and middle elevations, whereas the Hymenophylloid ferns are mainly found at mid to high elevations (Dubuisson *et al.,* 2003). Studies on elevational gradients have found two main patterns of species richness. First a monotonic decrease in richness (with increasing elevation or relatively constant richness at lower elevation and with decreasing richness at higher elevations) and secondly a humped distribution (with species richness high near the middle of the gradient) (Rahbek, 1995).

Differences in topography and soil condition, especially in the climatically fairly homogeneous region, also influence changes in species composition of communities of fern and other plant groups. Individual species of fern have distinct 'preferences' for specific soil conditions. The specificity of fern assemblages to soil conditions renders them suitable as indicators for forest types (Ruokolainen *et al.,* 1997; Salovaara *et al.*, 2004) and for distribution patterns of other plant groups (Vormisto *et al.,* 2000).

However, ferns can tolerate a wide range of environmental conditions (Hemp, 2002) and some have the capability to colonize such highly disturbed habitats as lava flows, dunes, landslides or floodplains as well as areas of forests that have been damaged by fire, ice storms, hurricanes or logging (Russell *et al.,* 1998; Arens and Baracaldo, 2000). One of the reasons why some ferns can readily colonize recently disturbed areas is that they have widely dispersed spores that reach even the most remote island. Furthermore, ferns also can inhabit disturbed areas vegetatively from adjacent areas. However, disturbance-adapted ferns must have characteristics that allow them to establish and grow under such condition.

The most crucial gradients separating fern species are the amount of soil moisture and soil nitrate concentration. Contrary to other findings, pH had little influence in controlling fern distribution. Greer *et al*. (1997) claimed that light has a relatively minor gradient in determining the structure of ferns communities in some temperate forests. They also found that neither diversity nor total abundance correlated with canopy cover. Moreover, species inhabiting the understory did not segregate along a canopy cover gradient.

### 2.1.1 Ferns response on different microclimates

Microclimate is the set of climatic conditions measured in localized areas near the earth's surface (Chen *et al.,* 1999). Ferns are regarded as environmental indicators because of their sensitivity to microclimatic variations in the forest's understory (Karst *et al.,* 2005). Ferns are also used as an indicator group in forest classification because they are easily identified in the field and their relation to edaphic conditions is relatively understood (Tuomisto and Ruokolainen, 1994). These plants have adaptations to low levels of light incidence, low evaporative potentials, a unique water dependent reproductive biology and high sensibility to abiotic features such as substrate. Therefore, fern community structure is expected to be very subjective to environmental changes related to solar irradiance and atmospheric moisture level (Barrington, 1993).

Epiphytes and terrestrials fern distribution are affected by the microclimates conditions. Based on previous studies, microclimates are varied in the understory and canopy of the lowland forest. Watkins and Cardelús (2009) stated that microclimatic variation and the extremes were differently distributed over the trunk. They pointed out that the buttress zone was consistently darker and exhibited significantly wetter air and less variation than the mid-trunk or bifurcation zone. They found that variation increased along the trunk with the most extreme and variable microclimate occurring in the bifurcation zone. Buttress zone is homogeneously dark and wet whereas the mid-and upper-trunk are brighter and drier and exhibit greater environmental heterogeneity. They suggested that microenvironmental heterogeneity, rather than absolute values, is particularly imperative for epiphytic ferns.

Leaf morphology of ferns reacts to different microclimatic conditions. For example, the majority of epiphytic species has simple leaves, whereas terrestrial and hemiepiphytic species have compound morphologies. Watkins and Cardelus (2009) suggested that these traits are adaptive and are under direct selective pressure. This is because canopy habitats tend to be hotter, drier and experienced more wind, in contrast to terrestrial habitats in most tropical forest. Epiphytic species from the bifurcation zone also had significant increased specific leaf weight when compared to terrestrial and buttress epiphytes.

Pacencia and Prado (2005) stated that a decrease in the number of fern species is related to the degree of habitat disturbance. Microclimate change at the edge of the forest may determine the substitution of forest species by exotic species to the forest, thus reducing the number of the original species. Fern species reacts differently with microclimatic changes. On forest fragmentation's edges, in which the environment tend to present a concentrated biotic and non-biotic dynamic, only forest species of fern capable of standing high luminosity and drought conditions, are abundant and well established, for example *Lomagramma guianensis*.

Watkins *et al*. (2007) also found that increased terrestrial gametophyte density and richness were related to both increased of light and disturbance. Nonetheless, increased of light and disturbance have no influence or had negatively affected epiphytes density. Disturbance and light seem to have negligible influence on the canopy gametophytes density. When measured against the typical rainforest understory habitat, the canopy is less buffered and has larger temperature and humidity extremes. It is plausible that microclimate and water availability play larger roles in these habitats relative to terrestrial sites.

Humidity gradient is the main factor determining the differentiation of the flora, vegetation and species richness of fern in Rwanda (Dzwonko and Kornas, 1994) and rainfall is the best predictor of species richness (Kornas, 1993). In Rwanda, the local distribution of ferns along a gradient of decreasing rainfall and increasing length of the dry season in Zambia has been extensively studied. It was found that the number of taxa in the squares relied on the level of rainfall, whereas the number of vegetation types was of minor importance.

Kluge and Kessler (2007) discovered that terrestrial species are higher along ravines than in the adjacent zonal forest. They showed that the constant high local air and soil humidity in the proximity of the stream and the availability of special habitat such as exposed earth banks influence the growth of terrestrial ferns. According to Kessler *et al.* (2007), ferns leaf length appears to decrease with increasing aridity while locations with greater moisture normally have greater diversity and abundance. Climatic data (rainfall, relative humidity and relative air temperature) is one of the variables that influence fern species richness in a given environment and the determining factor for the occurrence of the ferns.

## 2.2 Ferns Classification

Fern classification has undergone several changes over a long period, resulting in various name changes. Its classification has often being regarded as instable during the second half of the 20th century (Christenhusz *et al.,* 2011). Researchers with collecting experience in the Asian tropics (Copeland, 1947 ; Holttum, 1947) incorporated major changes into fern classification in the mid of the 20th century. A phylogenetic and molecular phylogenetic study has rapidly improved our understanding of fern relationships through phylogenetic analyses of DNA sequence data. The results of the studies were summarized in a ground-breaking classification published by Smith *et al.* (2006).

Fern classification is based upon sporophytic part of its life cycle. In the new classification, some plants once regarded as fern allies, are now viewed as true ferns, while others are relatively unrelated to ferns and therefore no longer considered as 'allies'. Similarly, the term 'pteridophyte' is no longer used because it includes ferns and the formerly recognized 'fern allies'.

An alternative term for ferns is 'monilophyte', an expression that recently has come into the literature for use with the ferns and implying that this group is monophyletic. Several factors have resulted in these changed concepts, one concerning molecular data, particularly DNA. Another has to do with the way in which characters are analyzed, especially with the help of computer techniques, where it is possible to take large numbers of traits into consideration, in determining the classification.

A third advance method involves data from fossil records, which although still woefully inadequate, has been substantially augmented by new fossil discoveries and recent interpretations of earlier records. To conclude, specified derivation in lieu of primitive characters should be used in interpreting evolutionary relationships. The new concepts have resulted in what now seem to constitute ten principal groups (or orders): the Ophioglossales, Marattiales, Equisetales, Osmundales, Hymenophyllales, Gleicheniales, Schizaeales, Salviniales, Cyatheales, and Polypodiales (Beaman and Edwards, 2007).

Christenhusz *et al.* (2011) have provided a linear classification to the exact lycophytes and ferns based on current phylogenetic knowledge. This provides a standardized guide for organization of fern collections into a more natural sequence. In this classification new families have been introduced, for example Diplaziopsidaceae and Rhachidosoraceae. Some genera also were merged with others, three families with five genera were accepted for lycophytes, and 45 families with about 280 genera were classified under ferns. There was a weakness in the collection by Smith *et al.* (2006). Their collection was not design as linear classification for implementation in collection, flora and textbooks and they excluded a classification of lycophytes, which are generally included with ferns in herbaria and many books.

The classification of Smith *et al.* (2006) and Christenhusz *et al.* (2011) has mostly reduced the number of genera, resulting in an expansion of several families (e.g *Asplenium*, *Blechnum*, *Cyathea* and *Hymenuphyllum*) but also resulted in the acceptance of narrower generic concepts in other groups (e.g . Hymenophyllaceae, Polypodiaceae and Pteridaceae). Again in 2014, fern classification underwent revisions and the new classification was presented based on current understanding of relationships of fern and lycopod clades.

### 2.2.1 Ferns Diversity and Composition

Malesia is a major hotspot of global biodiversity and harbors one of the greatest numbers of species of ferns and ferns allies in the world (Bellefroid *et al.,* 2007). The Philippines, Indonesia and Malaysia are among the most biologically diverse countries in Malesian region and recognized as four of the world's 25 biodiversity hotspots. The studies of fern in this region begun in 1824, originating from the Philippines (Underwood, 1906). A preliminary survey of ferns in Mindoro by John Smith listed 197 species of true ferns, of which 100 were presumed to be new (Merrill, 1946). They estimated about 925 valid species of true ferns and ferns allies found in the entire archipelago of the Philippines. The last study on ferns in the Philippines was carried out in 2003 by Julie F. Barcelona (Barcelona *et al.,* 2006). The study was based on collections made from those of Cuming in 1841 and Barcelona *et al.* (2006) own surveys from 2003 to 2004. They reported a total of 169 species of ferns from Bohol Island, of which 91 species were new additions to the Philippines flora.

The first collection of ferns in Malaysia was carried out by G. Finlayson in 1822 and his specimens were kept at the Kew Herbarium (Parris *et al.,* 1992). At the end of the 19th century, H.N Ridley, the director of the Botanic Garden of Singapore, had conducted an extensive collection of ferns in Peninsular Malaysia. He described five species of *Selaginella*, which was formerly known as fern allies. Subsequently, R.E. Holttum collected 2, 000 specimens of ferns from Mount Kinabalu from 1922 to 1954. He was well-known for his montane ferns collection. Another pteridologist in Peninsular Malaysia was A.G. Piggot, who described new species of ferns *Coryphoteris badia* (Thelyteridaceae) and *Ctenopterella khaoluangensis* (Grammatidaceae) (Parris, 1992). There are several local researchers who continued the studies on ferns in Peninsular Malaysia, such as A. Aziz, Haja Maideen and Razali Jaman from Universiti Kebangsaan Malaysia (UKM).

In Sabah, prior researchers on ferns had focused on Mount Kinabalu. This mountain has high plant diversity, thus had attracted many foreign botanists to conduct research and studies on its plants. The first collector of fern specimens from Mount Kinabalu was F.W. Burbidge; he collected 29 specimens between the periods of 1877 to 1878. This was followed by L.S. Gibbs, who described 27 species in February, 1910. In 1934, Christensen and Holttum reported a total of 417 species of ferns from Mount Kinabalu. They published a checklist of ferns of the mountain based on previous collection and publications.

The collection of fern specimen by Joseph and Mary Strong Clemens was the highest among all the collections of previous researchers, with probably over 9000 numbers including other kinds of plants (Parris *et al.*, 1992). They spent almost two years on the mountain from 1931 to 1933, with the sole purpose to collect specimens. Due to better facilities on Mount Kinabalu, later researchers like Parris took only four months to collect fern specimens, in equal numbers to the amount collected by Joseph and Mary Strong Clemens (Parris *et al.,* 1992). Parris *et al.* (1992) reported 608 species in 30 families and 138 genera of ferns from Mount Kinabalu. They noted that about half of the species were collected in lower montane forests, about 1500 m a.s.l. In addition, Razali Jaman and Abdul Latiff reported 106 species and six varieties of ferns in 59 genera and 24 families during the Sayap-Kinabalu expedition in 1998 (Jaman and Latiff, 1998).

Apart from Mount Kinabalu, there are other reports of fern collections in other parts of Sabah. Bidin and Jaman (1999) reported 83 species in 55 genera and 24 families from Tawau Hills Park and its adjacent areas. Parris (1997a) also reported 157 taxa of ferns from the Danum Valley, Lahad Datu. In general, the lowland mixed dipterocarp forest of the Danum Valley was poor in ferns at ground level, but many species are epiphytes high up in the crowns of trees (Parris, 1997a). She noted that the lowland forest of Danum Valley contains a more diverse pteridophyte flora that the lowland forest of Mount Kinabalu (Parris, 1997a). However, a substantial number of taxa known from the lowland forest of Danum Valley were also recorded from hill forest and lower montane forests of Mount Kinabalu, suggesting that these taxa are also present on the lowland forest of the mountain (Parris, 1997a).

A checklist of ferns and ferns allies was also reported by Akin (1998) from Maliau Basin Conservation Area collected during the Maliau Basin Scientific Expedition in 1996. Although the expedition lasted for only 14 days but they managed to collect 34 species in 26 genera and 14 families of ferns from the expedition site. In 2012, Parris described a new species of fern which is *Oreogrammitis translucens* (Parris *et al.,* 2012) from Maliau Basin. Parris (1997b) also recorded 174 taxa of ferns from the Crocker Range Park. However, Rimi *et al.* (2004) only managed to collect 10 families of ferns and ferns allies belonging to 13 genera and 14 species during the Crocker Range Scientific Expedition in 2002. This could be due to the small area explored, which was based at Ulu Kimanis, during the expedition.

Contrarily to Sabah, there is no extensive collection of ferns from Sarawak in recent years except for Mount Mulu. Mount Mulu itself is estimated to harbor about 446 species of ferns (Bellefroid *et al.,* 2007; Parris *et al.,* 1984). The first report of ferns collections from various localities within Sarawak was overseen by Mr. and Mrs. Joseph Clemens in 1929 with a total of 290 specimens (Copeland, 1931).

Similar to Sarawak, information on ferns of Kalimantan is very limited. The only significant collection of ferns from Kalimantan was from Bukit Baka-Bukit Raya National Park, in Central Kalimantan. Belleford *et al.* (2007) reported 59 new records of ferns for the National Park and Central Kalimantan. Among their collection, nine species were new records to Kalimantan. In addition, about 10 percent of the ferns found in the area are endemic to Borneo. The local species composition of ferns in Bukit Baka shows significant resemblance to those of Gunung Mulu and Mount Kinabalu.

### 2.3 Emergence of logged-over forest due to deforestation

Deforestation is one of the main factors of the extinction of our biodiversity. This is a global issue that should be prevented by all nations. Southeast Asia has the highest deforestation rate as compared to the other tropical region. The rate was 1.6 percent between 1990 and 1991, compared with 0.9 percent in the rest of the tropics. This rate also further increased between the periods of 1990 to 2000 and 2000 to 2005. Based on the latest Global Forest Resources Assessment (FAO, 2010), Southeast Asia has lost 33.2 million ha of forest in the past two decades.

The growth of human population and its greater demand for resources are the main cause of deforestation. Human population density as a whole in Southeast Asia has been increasing steadily over the past decade at a rate of 1.5 percent per annum, indicating that the remaining forest will continue to face high risk of degradation both locally and global level. The lowland forest of Southeast Asia is among the most diverse biomes, and globally a significant terrestrial carbon sinks (Saatchi *et al.,* 2011). However, based on current deforestation rates of 0.71 percent per year (Achard *et al.,* 2002)  74 percent of Southeast Asia's forest could be cleared by 2100, which is predicted to cause the extinction of between 13 percent and 42 percent of all species in the region by the end of this century (Brook *et al.,* 2003).

Reductions of the primary forest area due to deforestation have caught the attention of ecologists and foresters. A few studies have been carried out to investigate the potential of the secondary forests and logged-over forest to take over the role of primary forest to conserve the remaining biodiversity. Selectively logged forest retain substantial biodiversity, carbon and timber stocks, this 'middle way' between deforestation and total protection deserves more attention from researchers, conservation organization, and policy-makers (Putz *et al.,* 2012).

Logged-over forests are resulted from the logging activities consists of both selectively logged and the highly degraded forest. Throughout the tropics, forest continue to be commercially logged, resulting in considerable ecosystem degradation and fragmentation, to the point that logged-over and the degraded habitat now comprise more than 50% of the tropical estate (Blaser *et al.,* 2011; Struebing *et al.,* 2013).

Most logged-over forest are located in tropical Asia (47 percent), followed by tropical America (32 percent) and tropical Africa (21 percent) (Brown and Lugo, 1990). Logged-over forests will take about 40 to 100 years to begin to resemble the original old-growth forest. Numerous researches comparing primary and logged-over forest in the tropics had been carried out and they showed that logged-over forests generally had lower biodiversity values. However, they are more valuable than most other types of degraded forests.

Logged-over forests play a role in conserving the biodiversity. These forests have many other biotic characteristics, which in turn makes these forests as conservation's resources. Logged-over forests can foster within their understory species that will form forest and produce mature ecosystems. On the other hand, logged-over forests also provide conditions that help improve soil and water quality, as well as conserve genetic material, nutrients, moisture and soil organic matter (Brown and Lugo, 1990). Logged-over forests are behaving as a species refugee, in which the higher the initial number of species that it can sustain, the greater its role as a foster ecosystem. All the functions of logged-over forests depend on their ecological characteristics. Proper management of these characteristics will depend on how well they are understood.

### 2.3.1 Impacts of logging on animals and plant diversity

Logging of a tropical rainforest not only changes the composition of its flora and fauna, but it changes the original forest structure. Logging with high machinery can cause extensive damages to soils and residual trees. The logging activities trigger degradation of soil, through compaction and such degradation will render the regeneration poor. This can significantly slow down the rate of forest structural recovery and can have long lasting effects on species composition. After logging, the intensity of hunting increase primarily because of easier access to remote forests by new roads and of the greater human presence in the area, and this impact is an important component of the effect of logging (Bennett and Robinson, 2000; Putz *et al.,* 2012).

Most logging done in tropical rainforests is selective (Mertens *et al.,* 2001) in which the activity is performed by removing just a few trees per hectare rather than clearing the forest. In Malaysia, 3.5 percent of production forests are selectively logged annually. An increasing proportion of these forests has been subjected to a first cycle of selective logging and is in the phase of regeneration. However, forest disturbance due to selective logging can cause major changes in vegetation structure and the forest environment (Sayer and Whitmore, 1991). But some of the studies about the effects of selective logging on other flora and fauna groups show merely insignificant impacts.

### 2.3.2 Impacts on animal group

Studies on the effect of selective logging on South-east Asia vertebrates have commonly shown that a gap and forest-edge specialists may benefit, but species restricted to climax forest decline in abundance or become locally extinct (Johns, 1997; Heydon and Bulloh, 1997). Based on Bernard (2004), there was no consistent trend observed in small mammals microhabitat use patterns in primary compared to logged forest. In general, small mammals appeared to be able to utilize different sets of microhabitats in dissimilar habitat types. He suggested that habitat utilization by small mammals is closely linked to food or foraging areas and shelter or refuge sites, with possibly a strong influence for predator avoidance. Peters *et al.* (2006) found that the abundance of nectarivorousbn bm and frugivorous taxa of bat communities were higher in logged-over forest where canopy openness and understory foliage density were greatest. However, insectivorous and omnivorous species of bat were more abundant in primary forest where canopy foliage density and variability in the understory stratum were greatest. This study concluded that low intensity logging is important to some taxa, leading to changes in bat community and trophic structure.

Another study on the effect of selective logging in Borneo did not show evidence that logging could be change in the composition of butterfly assemblages from species with a local distribution to more widespread species (Willott *et al.,* 2000). This is may be because logged forest site in close proximity to primary forest or low intensities of logging do not necessarily reduce the species richness or abundance of butterflies, although assemblage composition is changing. Within the study Willott *et al.* (2000) found that there was no significant difference in the number of species between primary and logged forest. However, there was a significant difference in the relative abundance of species, but this was due largely to the abundance of one or two species. Result from this study support other studies in the same area showing little effects of selective logging on butterflies (Hamer *et al.,* 2003).

Hill *et al.* (2003) reported that butterfly abundance and distribution did not differ between selectively logged and unlogged areas in the pattern of post-drought recovery. This result indicated that the abundance of *Ragadia makuta* was significantly reduced both after high rainfall and during severe drought, but these impacts were short-lived and were not affected by habitat disturbance. Hamer *et al.* (2003) restated that there was little difference in butterfly diversity between primary forest and the selectively logged forest of 10–12 years previously.

In contrast, the composition of the butterfly assemblages in the two habitats differs with each other. The differences are being strongly associated with species' gap preferences and geographical association. For example, those species with higher shade preferences and narrower geographical distributions were most adversely affected by logging, whereas cosmopolitan species with high light preferences benefited from logging. Changes in vegetation structure following selective logging which resulted in much lower habitat heterogeneity with less dense shade and fewer open gaps in logged forest influence the changes in butterfly assemblages. Area of dense shade, which is more common in unlogged forest supported species with shade preferences and restricted geographical distribution (Satyrinae and Morphinae).

Eggleton *et al.* (1999) reported that selective logging appeared to have relatively little effect on termites' assemblages, although soil-feeding termites may be moderately affected by this level of disturbance. They also found that species composition changed, but to a small extent when considered against the background level of compositional differences within the Sunda region. They suggested no major shift in termite assemblage structure in sites that have been logged, as long as a substantial tree canopy remained. Selective logging followed by a long fallow period may not adversely affect termite population.

Studies on bird showed that changes of forest structure by selective logging did not reduce the overall richness and diversity of the bird's community although it altered the vegetation structure in logged forest. In fact, richness and diversity in the bird's community in logged forest had increased with the changes in vegetation structure with the production of large area of the secondary habitat (Aleixo, 1999). Similarly Bernard *et al.* (2014) reported that no evidence of differential habitat disturbance effects on the primate community and no evidence supporting differential habitat disturbance effect on the primate based on animal body size or feeding habit. Surprisingly, they found the presence of primates within the heavily logged forest sampling sites, which included endemic species and species of high conservation concern. This study exhibited that highly disturbed forests are still valuable for primate conservation. However Brodie *et al.* (2015) reported that newly logged sites had 11 percent lower species richness of mammals than unlogged sites, but sites logged for more than 10 years previously had richness levels similar to those in old-growth forests.

In contrast, a study by Parry *et al.* (2007) on large-vertebrate assemblages of primary and logged-over forests in the Brazilian Amazon had discovered that large prehensile-tailed primates were absent or at very low abundance in the logged-over forest. The abundance of large frugivorous or granivorous birds was also low in logged-forests compared with primary forests. Nevertheless, they reported that logged-over forests held a similar abundance of ungulates browsers and smaller-bodied primates compared with primary forests. Therefore, they concluded that although logged-over forests were crucial habitats for large mammals and birds, they were limited in terms of faunal richness, particularly disperser of large-seeded plants.

Schleuning *et al.* (2011) stated that ecosystem processes responded in distinct ways to forest fragments and selective logging in Kakagema forests (Kenya). Their findings strongly implied that forest ecosystem processes mediated by animal communities with highly mobile species, may respond primarily to changes in the local forest structure and resource distributions, for instance caused by selective logging. They alluded that selective logging influences the processes directly by modifying local environmental conditions and resource distributions. The positive to neutral effects of selective logging on ecosystem processes showed that the functionality of tropical forest can be maintained in moderately disturbed forest fragments.

### 2.3.3 Impacts on plants

Impacts of selective logging on tree diversity are varying, showing little effect or negative effect. In Kalimantan, a study on the effects of selective logging on vegetation found that harvesting removed 62 percent of dipterocarp basal area (Cannon *et al.,* 1994). However, a subsequent study from the same area showed that there was an increase in tree diversity eight years after selective logging. Similarly, other studies comparing plots 18−20 years after logging also presented that tree diversity was not significantly affected, although species composition had been altered and varied among the logged plots (Berry *et al.,* 2008; Verburg and van Eijk-Bos, 2003). Conversely, in a study in Peninsular Malaysia (Pasoh Forest

Reserve), the mean canopy height (24.8 versus 27.4), mean canopy surface area (12.0 versus 17.4 $m^2$), mean crown size (42.9 versus 94.5$m^2$) and diversity of trees were significantly lower in a forest 41 years later after logging than a neighboring primary forest (Okuda *et al.,* 2003). These results illustrated that tree structure and diversity in a regenerating forest may match poorly with those in the primary forest, even after more than four decades.

Nonetheless, a study by Imai *et al.* (2012) found that tree diversity pattern following occurrence of disturbance usually varies with spatial scale of sampling. Based on the study’s findings, species richness was higher in the RIL (Reduced Impact Logging) than in the primary forest at the 2-ha scale, while it was similar between the two forests at 0.2-ha scale. Community of the RIL forest resembled greatly with the primary forest than that of the CL (Conventional Logging) forest, regardless of plot size. In this study, they examined impacts of RIL on species richness and community composition of tree species of different spatial scales, and the scale (plot size) dependency of the two metrics, species richness versus community similarity.

Plant diversity, especially ferns, in different forest types is poorly understood. The early work by Hietz (1997) suggested that the performance, survival and distribution of epiphytes in disturbed forest or secondary vegetation are influenced by stand density and microclimate, distance from seed source, tree size and sometimes tree species, types and history of disturbance, population dynamic of epiphytes and trees, and epiphyte physiology.

Hietz e*t al.* (2006) compared the abundance and distribution of mature individuals of three orchid and five bromeliad species between a relatively undisturbed and an adjacent disturbed plot in a Mexican humid montane forest. Most species attached at a lower position on trees in the disturbed plot, and in a few cases showed differences in preferred branch diameter or inclination. They concluded that generally higher individual species numbers and mostly higher epiphytes biomass in the disturbed plot.

Majit *et al.* (2011) studied the diversity and abundance of orchids in Crocker Range Park and found that old secondary forest had the most diverse orchid species compared to young secondary forests and primary forests. They alluded that the distribution patterns of every single species of orchid have influenced the diversity of orchid in old secondary forest. According to the study, the orchid in old secondary forest had high number of even distribution than in the primary forest. In this study, old secondary forest has higher orchid diversity because of its high evenness. They also concluded that humidity, temperature and canopy did not influence the species composition. Other variables that characterized primary and secondary forest may influence the diversity of orchid between secondary forest and primary forest. For example, a study carried out by Gobilik (2002) has stated that the abundance of gingers was influence by the tree DBH, soil PH and soil nutrient. He was discovered that diversity of gingers was higher in logged forest and significantly different compared to primary forest. In general, species preferred to primary forest habitat had negative correlations with the species that preferred logged forest and with the variables that characterized it.

The only study on the relationship between ferns diversity and different forest types was in Jambi Sumatra (Indonesia) by Beukema and van Noordwijk, (2004). Their study focused on the terrestrial fern species composition between primary forest, jungle rubber and rubber plantation based on comparison for plot level diversity (average number of species per plot) and landscape level diversity (species-area curve). According to the study, terrestrial fern species were grouped accordingly to their ecological requirement in order to assess forest habitat quality in rubber production system as compared to primary forest. They determined that the average number of terrestrial ferns species per plot in the current study was indeed higher in jungle rubber than in primary forest. However, the difference found was not statistically significant. They observed that higher species richness in jungle rubber and in rubber plantations was largely due to an increase in species that acquired optima in environments other than the shady forest understory.

On the other hand, Ariyanti *et al.* (2008) investigated bryophytes assemblages on trunk bases in natural forest, selectively logged forest and cacao agroforests that are shaded by remnants of natural forest in Central Sulawesi. They came to the conclusion that the bryophytes assemblages of selectively logged forests and cacao agroforests were as rich as in natural forest, but species turn-over was particularly high towards cacao agroforests, probably due to microclimatic changes. The study found a marked difference between mosses and liverworts in their response to forest types. Mosses were affected in terms of species richness, whereas liverworts were affected only in terms of cover. They related the pronounced differences between mosses and liverworts in their response to habitat changes with their adaptations to environmental drought.

Paciencia and Prado (2005) had compared the species richness and diversity among areas of large forest fragments (> 900 ha), small forest fragments (<100ha), and landscape matrix. Their study indicated that fragmentation has negative impacts on species richness of the matrix and the edges of forest remnants, but small forest fragments have ferns species richness and diversity rate similar to large forest fragments. Through their study, a reduction of richness was noticed in the edge areas, indicating that loss of forest species might be occurring parallel to the ongoing fragmentation process and the increase of forest edge. One of the factors responsible for the species richness reduction could be the habitat structure which, in the case of forests, reflects the degree of anthropogenic alteration in them. The study's results suggested that the regional deforestation process, which happened in the past, initiates the decreased in ferns species number. This is due to the fact that such process is the main determining factor of the existing structure in forest fragments and matrix.

Additional work by Murakami *et al*. (2005) investigated the relationship between species diversity of ferns and the environmental variables of 39 fragmented forest patches in the Kyoto city, Japan. They concluded that fern species diversity depends on patch size and isolation distance from mountain forest. The number of ferns species increased with the patch area and ferns with high disperability were affected by isolation distance in the city area. At the same time temperature may influence the diversity of fern. For example, their study noted the higher average temperature in the city environments was the products of human population alteration of the landscape which causes dryness of the forest floor environment. The prothallus (gametophytes) of ferns is susceptible to dehydration. Therefore, isolation may be an indirect impact that reduces the ferns species diversity by drying forest floors.

Silva *et al*. (2011) had investigated how edge effects influence the fern community of Jaguarao Forest located in the distinct of Rio Formoso, Pernambuco, Brazil. A comparative analysis was made of the interior and edge of the fragment of forest, regarding the richness, abundance and diversity of ferns in the two areas. It was concluded that the edge effect causes a decrease in richness and abundance of the fern species found in Jaguarao Forest, where the highly sensitive species are being replaced by species with greater tolerance to the disturbance caused by the creation of an edge.

Impacts of logging vary depending on extraction methods used, spatial scale, size and the amount of timber harvested and local condition. It must be noted that the observed high tree diversity in logged forest does not equate to high conservation value, as disturbance can facilitate the growth of fast growing early successional species, while old growth forests may have less diversity and yet contain higher abundance of ranges-restricted species (Sheil *et al.*, 1999; Slik *et al.*, 2008).

## 2.4 Microclimate of primary and logged-over forest in lowland tropical rainforest

Primary rainforest are characterized as usually dark and humid, with stable temperatures, little wind and almost continuous canopy cover (Laurance *et al.,* 2002; Lovejoy *et al.,* 1986). Forest canopy cover, also known as canopy coverage or crown cover, is the proportion of the forest floor covered by the vertical projections of the tree crowns (Jennings *et al.,* 1999). Primary rainforest has uniform, unbroken canopy with little radiation penetrates through the many foliage layers to ground level, except where gaps occur.

In general, lowland rainforests show an almost uninterrupted canopy with small light gaps. This conditions differs with the microclimate of secondary lowland rainforest, as logged-over forest have elevated temperature, reduced humidity and increased sunlight to the forest floor. Silva *et al.* (2011) also reported that the interior and edge sites of the forest showed differences in relation to air temperature and relative air humidity. The forest interior was observed to have lower mean temperature (27°C) and higher mean relative humidity (79 %) than the edge (31°C and 69 %). Overall rainforest vegetation produces a complex three-dimensional mosaic of microclimates, which vary vertically from canopy top to forest floor and horizontally from point to point beneath the canopy.

On a larger scale, microclimates fluctuate between canopy gap of different sizes, between regenerating and mature forest phases and between different rain-forest types. For example, ravines have higher humidity compared to ridges. The climatic gradients depend on the structure of the forest, particularly height and spacing of the trees. Beneath the canopy of lowland rainforest along the height gradient of single trees, humidity decreased from 50 cm above ground to inner crown (Kluge and Kessler, 2007).

Brown (1993) measured relative humidity in primary forest at Danum Valley at 1.2 m above the forest floor over a one year period (April 1987 March 1988).

Mean maximum and minimum relative humidity rarely fell below 90 percent and never below 85 percent. Maximum temperature, its means and ranges also significantly decrease below tree canopy, but minimum temperatures were very similar at all levels of the forest. The same study by Brown (1993) found that the mean maximum and minimum temperature was 28°C and 21.2°C in primary forest, respectively. He concluded that canopy gaps play an essential role to influence the reading of temperature measurement.

Any changes in forest canopy structure will result in a change in the transmission of radiation. The radiation regime in a canopy gap is strongly dependent on the size of the gap. The larger the gap, the greater the amount of solar radiation that may reach the forest floor (Barton *et al.,* 1989). Based on the measurement by Brown (1993), between center of each gap and two control plot beneath closed canopy forest, there are strong relationships between gap size and microclimate at the center of each gap. Maximum temperatures averaged 38.2°C in the largest gap, compared with 28.4°C in closed-canopy forest.

Selective logging as one of the principal cause of disturbance in Southeast Asia produces canopy gaps on much larger scale, which involves the loss of 25 percent or more of the canopy cover and causes some extremely large gaps where logs are gathered at log landings, as well as linear gaps along all logging roads and trails. Nussbaum (1995) who recorded minimum and maximum air temperatures at representatives' location in logged forest of the Ulu Segama region, found that maximum temperature had increased substantially after logging disturbance. At a log-landing site the maximum temperature recorded on a sunny day was 38.5°C, on an open skid trail 36.8˚C and on a shaded skid trail 30.7°C.

# CHAPTER 3

# METHODOLOGY

## 3.1 Study site

Danum Valley Conservation Area (DVCA) located at the Southeast of Sabah (Figure 3.1) and classified as Class I (Protection) Forest Reserve, covering approximately 428 $km^2$ of undisturbed forest. It is situated approximately 80 km west of Lahad Datu, at 5’01 ‘N, 117’E. The conservation area is characterized as a lowland primary dipterocarp forest and classified as tropical moist forest. It occupies rugged terrain at moderate elevation in a complex and actively eroding landscape (Marsh and Greer, 1992). DVCA is surrounded by a mosaic of forest blocks that have been selectively logged at different times and practices during the last 20 years. Some logged areas were recently assisted by enrichment planting but most of it was left regenerating naturally. The term Danum Valley and Ulu Segama Forest Complex was used in this study to refer to the sampling site within the adjacent conservation area and a forest reserve (Figure 3.1).

The study was carried out in three different forest areas, primary forest and logged-over forest of selectively logged and logged-over forest of highly degraded. DVCA consisted of undisturbed (primary forest) lowland tropical rainforest, whereas Ulu Segama consisted of degraded logged forest (selectively logged and highly degraded) which was logged in 1980s. Characteristics of the three forest types are almost similar in terms of topographic features, with altitudes ranging from about 150 m to 230 m a.s.l. The logging technique used was “conventional selective”, where timber trees above 0.6 m DBH were logged in a single operation (Poore *et al.,* 1989).

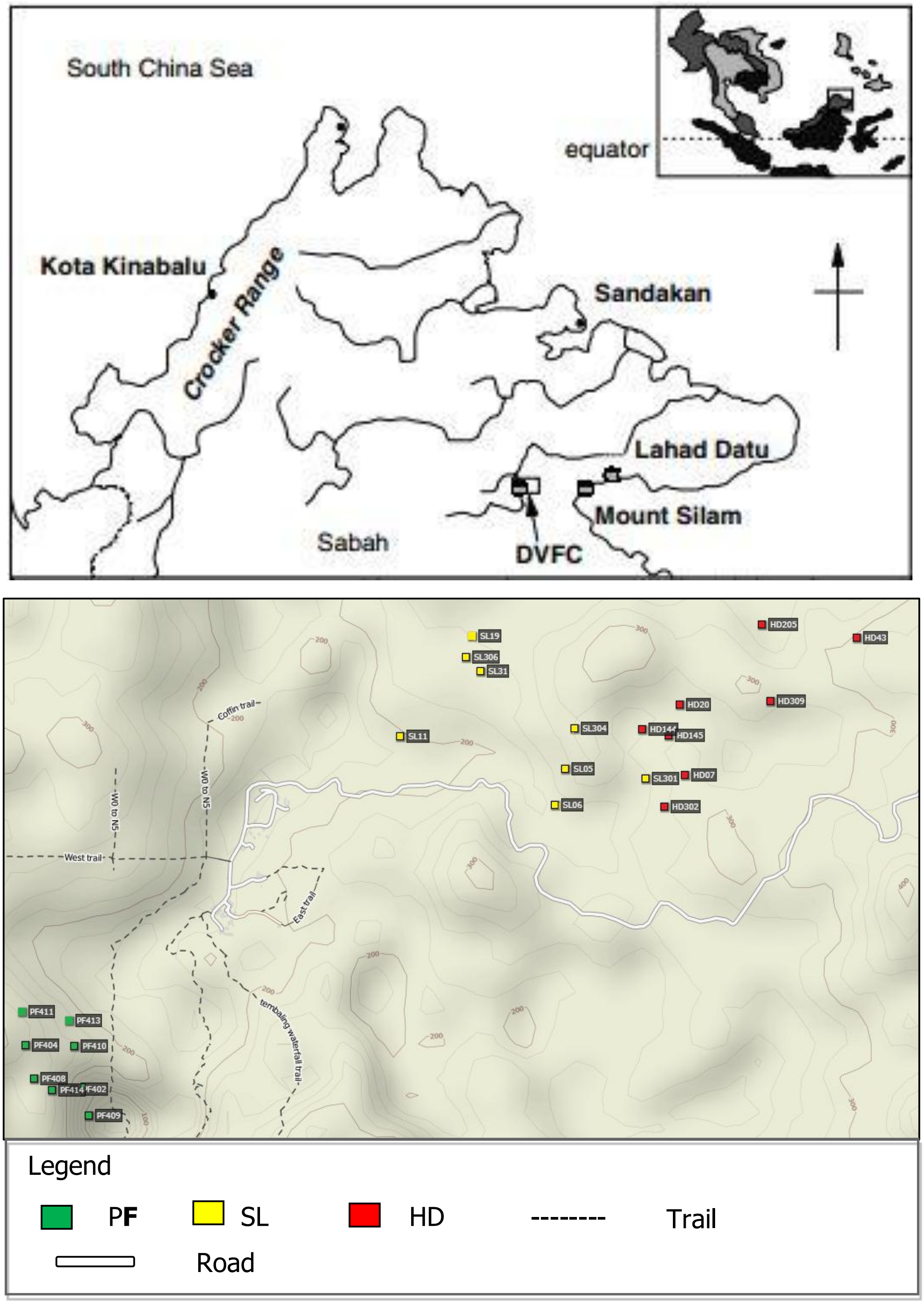


**Figure 3.1: Location of the sampling plots in Danum Valley and Ulu Segama Forest Complex. PF-Primary forest, SL- Selectively logged (logged-over forest), HD-Highly degraded (logged-over forest**

Although only about 7–10 percent of the trees were taken per hectare, the resultant damage was severe (Lambert, 1992) as heavy machinery utilized in the operation. The bulldozers and tractors made use of skid trails (network of bulldozer paths) to conduct the log extraction and at the log landings (log yarding areas); the logs were debarked, stored and loaded onto trucks for removal. The study areas receive averages mean annual rainfall of 2822 mm with a mean annual temperature of 26.7 °C (Marsh and Greer 1992; Walsh, 1996).

### 3.1.2 Primary forest

The primary forest in the lowland dipterocarp forest of the Danum valley lies on a moderately undulating terrain of small rounded ridges and valleys (Marsh and Greer, 1992). Sampling area is located at an elevation in a range of 200 to 300 m a.s.l along Rhino Trail and within the 50 ha permanent plots. The forest locally has been classified as the *Parashorea malaanonan* type (Newberry *et al.,* 1992) and *P.malaanonan* is prominent among the large dipterocarp in Danum plots. The understory is densely dominated by saplings and creepers in area of treefall gaps, belonging to Annonaceae, Rubiaceae and Sapindaceae.

### 3.1.3 Logged-over forest

The logged-over forests that selected in this study are located in Sapat Kalisun area. The Sapat Kalisun lies within the Brassey Range of hills in Eastern Sabah, approximately 50 km west of the eastern coast of Sabah, Malaysia Borneo. The summit of Atur hills (Bukit Atur) was the highest point and the altitudinal within the Sapat Kalisun itself is 132 to 435 m. The area was selectively logged in 1988 and 1989, with annual production volume of between 96 – 100 $m^3$ $ha^{-1}$ under the Yayasan Sabah Concession (Forestry Upstream Division). Timbers were selectively logged rather than clear felling system in order to produce sustainable harvesting coupes every 30 years (Chappell *et al.*, 1999). Logging was done through a combination of tractor yarding and high-lead logging leaving the complex structure of regenerating forest patches, areas of protection forest and areas of highly damaged forest (Putz, 1994). The area has notable skid trails and log landing areas with a relatively mixture of undisturbed forest area. The logged forest contained remnant primary forest trees and colonizing species. The Dipterocarp forest of

Sapat Kalisun is dominated by species from Dipterocarpaceae and Euphorbiaceae. Open areas created during logging were dominated by climbers.

### 3.1.4 Climate

The mean annual temperature recorded at the field centre was 26.7°C during the typical of a wet equatorial climate. The mean maximum temperature was 30.9°C and the mean minimum was 22.5 °C (Walsh, 1990). The highest recorded temperature of the typical of a rainforest environment during the absent of the extremes was 36.0°C and the lowest 18.2 °C. In term of humidity the annual mean was 94.5 % and falling to as low as 57 % in the early afternoon (14h00) during the dry month of May in 1986 (Walsh 1990). However, these results are not representative of conditions experienced beneath the forest canopy. This is because the DVFC's meteorology station is in a large clearing on a bank of a board river.

Brown (1990) measured temperature and relative humidity in undisturbed forest at DVCA, at 1.2 m above the forest floor over a one-year period (April 1987-March 1988). Mean maximum and minimum temperatures were 28.4°C and 21.2°C respectively, while the relative humidity rarely below 90% and never below 85%. Canopy gap are an important feature of both the natural forest and selectively logged forest, where they occur on a much larger scale. Canopy gap microclimates vary considerably with size of gap and the microclimates beneath a closed canopy. Maximum temperature average 38.2°C in the largest gap, compared with 28.4°C in closed-canopy forest.

## 3.2 Sampling design

There were three types of forest used in this study: Two forest types of logged-over forest (selectively logged and highly degraded forest) and primary forest. Selectively logged forests are referring to the high biomass logged-over forest and the highly degraded forests are referring to the low biomass logged-over forest. The AGB selection are based on the AGB value calculated on the 30 square plots that were randomly established within the study site in Sapat Kalisun area and follow the study by Phua *et al.* (2016). AGB values lower than 150 AGB t/ha are consider being low and over 250 AGB t/ha are consider being higher. The biomass selection are based on the eight plots with the size of 0.04 ha (20 x 20 m) were established in each of the forest types, totaling to 24 plots.

The number of individual ferns found on all substrates in every plot was counted, including epiphytic ferns on tree trunks up to 2 m above the ground. Vining species and species with interwoven rhizomes were treated as one individual on a single tree or branch. Duplicates of botanical vouchers were collected for every species to be deposited at BORNEENSIS Herbarium of the Institute for Tropical Biology and Conservation (ITBC), Universiti Malaysia Sabah (BORH) and Sandakan Herbarium of Sabah Forestry Department (SAN). As much as possible, only fertile plants were collected for voucher specimens.

Duplicates of specimens that could not be recognized in the field were also collected. Each specimen was tagged with a serial number and placed in a plastic bag. Duplicate specimens from the same plant were given the same serial number. The specimens were pressed in the field by placing them in between newspapers before putting into a leak proof plastic bag to preserve the specimens in the field. Seventy percent ethanol was poured onto the pressed specimens and the plastic bag was secured properly. The specimens were dried later in an oven, set at 70°C, for three to four days at ITBC.

### 3.3 Species identification

Ferns specimen was identified based on literature on Malaysian species (Beaman and Edwards, 2007; Piggot and Piggot, 1988; Kramer, 1971). To confirm the identification the collected specimen was compared with the ferns specimen in BORH and SAN herbarium. The ferns identification was verified by a ferns expert in BORH and SAN researcher.

## 3.4 Sampling of Environmental Variables

### 3.4.1 Relative Humidity and Air Temperature

Temperature and relative humidity were measured at several locations within the primary and secondary forests from March 2015 to March 2016 using Hobo Pro v2 loggers. Data loggers were established in each plot at two meters above the ground and the measurement interval were one hours. Data loggers were placed in the middle of each six plots in each of the forest areas. The two plots that were set up latter were excluded from the relative humidity and temperature analysis due to inadequate data loggers.

### 3.4.2 Canopy Cover

Canopy coverage was measured using a spherical densitometer (Lemmon, 1957) at every subplot during the data collection. The densitometer was held at elbow height in front of the recorder and canopy coverage was measured as the number of squares marked on the convex mirror of the densitometer which did not reflect any canopy. Four readings were taken at each subplot of the plot station facing north, south, east and west. All four readings were average and multiplied by 1.04 to obtain the percent of overhead area not occupied by canopy. The difference between this percentage and 100% is the estimated overstory density in percent.

## 3.5 Statistical analysis

Statistical tests and data manipulation were carried out using PAST (Paleontological Statistics) version 2.17 and the computer program "Species Richness and Diversity" (Henderson & Seaby, 1998). To detect differences between the canopy cover, relative humidity and relative air temperature at each of the forest areas, One-way ANOVA, and Kruskal Wallis test was run in the PAST (Paleontological Statistics) version 2.17. One-way ANOVA was also used to compare the diversity of ferns species at different sites (Primary forest and logged-over forest).

To test the difference between mean in ferns diversity in each forest area, the Shannon diversity of ferns species in each plots were calculated. All of the Shannon diversity was tested using normality test to fulfill the assumption of a One-way Anova and if the p-value greater than 0.05, the data is normal. Therefore, One-way Anova used to test the differences between mean of the Shannon diversity in each forest types. The $H_0$ are rejected and the $H_1$ accepted, if the p-value less than 0.05.

### 3.5.1 Biodiversity Index

The Shannon-Wiener Index was used to estimate species diversity which incorporates richness and evenness into a single measure. To estimate the species richness, margalef S Index was used and evenness index was used to measure the equitability or evenness in which can be derived from Shannon-Wiener Index.

#### a. Shannon-Wiener index

The Shannon-Wiener index (H') is an estimate of species diversity which incorporates richness and evenness into a single measure. The index assumes those individuals are randomly sampled from an "indefinitely large' population (Magguran, 1998).

$H' = \text{In } p,$

Where, *p* is the proportion of individuals found in the *ith* species.

#### b. Simpson's Index of Diversity 1-D

The value of this index ranges between 0 and 1 and the greater the value, the greater the sample diversity. In this case, the probability that two individuals randomly selected from a sample will belong to different species.

#### c. Evenness Index (E)

The measure of equitability or evenness can be derived from Shannon-Wiener Index. This measure also assumes that all species in the community are accounted for the sample (Magguran, 1998). The index of equitability or evenness was calculated as follow;

$E = H'/In\ S$

Where S is the total number of species recorded.

**d. Dominance-D Index**

Ranges from 0 (all taxa are equally present) to 1 (one taxa dominants the community completely).

**e. Similarity Index**

Similarity index was calculated to determine the similarity between any two sites. The Similarity Index that used in this study was Morisita Index using equation below;

$$C_D = \frac{2\sum_{i=1}^{S} x_i y_i}{(D_x + D_y)XY}$$

$x_i$ is the number of times species $i$ is represented in the total $X$ from one sample

$y_i$ is the number of times species $i$ is represented in the total $Y$ from another sample. $D_x$ and $D_y$ are the Simpson's Index values for the $x$ and $y$ samples respectively. $S$ is the number of unique species. Further analysis of similarities, cluster analysis, was carried out using PAST (Paleontological Statistics) version 2.17.

### 3.5.2 Species Accumulation Curve and Species Richness Estimation

The traditional method of plotting a species-accumulation curve starts by calculating and plotting the mean number of species of the smallest sample size. Then all combination of the next samples size was randomized and the mean cumulative number of species is calculated. For the randomized sample data, once a curve has been obtained it can be used to estimate species richness (Ugland *et al.,* 2003).

The Jacknife estimator and Chao's estimators are generally realible when H=10 but can grossly overestimate true richness when H=25 (Poulin, 1998). Jacknife and CHAO's estimator are among the best available estimator of species richness for many types of communities and tend to give reliable and generally unbiased estimates (Palmer, 1990; Colwell and Coddington, 1994).

### a. Jacknife (1*Order)

The first-order Jacknife estimators are based on the idea of changing the samples, seeing the effect on the number of species ($S_{obs}$) and estimating how many species would be recorded with more plots. The first-order Jacknife estimator drops one of the plots and calculates the number of species ($S_{obs}$) from the remaining plot.

### b. CHAO 1

CHAO 1 is an estimator for the number of species in the population based on the number of singletons and doubletons.

### c. ACE

The Abundance-based Coverage Estimator (ACE) is based on estimating sample coverage and uses the number of species with 10 or fewer individuals.

### 3.5.3 Cluster Analysis

In this study, cluster analysis was used to classify the similarities in species composition from 24 plots in three study sites. Cluster analyses were performed as a tree diagram using Jaccard distance and group average method in the PAST (Paleontological Statistics) version 2.17.

### 3.5.4 Linear regression

Linear regression was used to correlate or to investigate the effect of environmental variables to the diversity of fern in each of the forest areas.

# CHAPTER 4

# RESULTS

## 4.1 Species Diversity and Composition

In total, the 24 plots in primary and logged-over forests of Danum Valley and Ulu Segama forest complex has 26 species in 20 genera and 11 families of ferns (Table 4.1). This was represented by 476 individuals of ferns. Among the 26 species, 10 species are terrestrials and 16 are epiphytes. The primary forest has 15 species (10 are epiphytes) whereas logged-over forest of selectively logged has 18 species (12 are epiphytes) and logged-over forest of highly degraded has 11 species (six are epiphytes) (Figure 4.1).

The number of individuals per species was highly unevenly distributed. The Tectariaceae fern *Tectaria graffithii* with 103 individual had the highest numbers followed by the Pteridaceae fern *Antrophyum callifolium* (85 individuals) and *Leptochilus* cf. *decurrens* with 62 individuals (Figure 4.2). Based on family classification, Tectariaceae has the highest number of individuals count (103) followed by Pteridaceae (95) and Polypodiaceae (75) (Figure 4.3). Polypodiaceae has the most species rich-family with seven species, followed by Thelypteridaceae with four species and Pteridaceae with three species (Figure 4.4).

**Table 4.1: Checklist and abundance of ferns recorded in all forest types at Danum Valley and Ulu Segama Forest Complex**

| Species | Forest types | | | Total | Percentage (%) |
|---|---|---|---|---|---|
| | PF | SL | HD | | |
| **Tectariaceae** | | | | | **21.64** |
| *Tectaria griffithii* | 79 | 11 | 13 | 103 | 21.64 |
| **Pteridaceae** | | | | | **19.96** |
| *Antrophyum callifolium* | 61 | 17 | 7 | 85 | 17.86 |
| *Adiantum latifolium* | 0 | 8 | 0 | 8 | 1.68 |
| *Haplopteris elongata* | 0 | 0 | 2 | 2 | 0.42 |
| **Polypodiaceae** | | | | | **15.76** |
| *Leptochillus cf. decurrens* | 55 | 6 | 1 | 62 | 13.03 |
| *Christella quadrangularis* | 0 | 0 | 5 | 5 | 1.05 |
| *Pyrrosia christii* | 0 | 3 | 0 | 3 | 0.63 |
| *Phymatosorus membranifolius* | 0 | 2 | 0 | 2 | 0.42 |
| *Lepisorus mucronatus* | 1 | 0 | 0 | 1 | 0.21 |
| *Pyrrosia lanceolata* | 1 | 0 | 0 | 1 | 0.21 |
| *Lepisorus longifolius* | 1 | 0 | 0 | 1 | 0.21 |
| **Aspleniaceae** | | | | | **12.39** |
| *Asplenium phyllitidis* | 19 | 19 | 21 | 59 | 12.39 |
| **Dryopteridaceae** | | | | | **11.13** |
| *Teratophyllum aculeatum* | 41 | 8 | 4 | 53 | 11.13 |
| **Thelypteridaceae** | | | | | **7.35** |
| *Sphaerostephanos penninger* | 3 | 6 | 19 | 18 | 3.78 |
| *Mesophlebion dulitense* | 12 | 1 | 0 | 13 | 2.73 |
| *Sphaerostephanos heterocarpus* | 0 | 3 | 0 | 3 | 0.63 |
| *Pronephrium menisciicarpon* | 0 | 1 | 0 | 1 | 0.21 |
| **Davalliaceae** | | | | | **3.15** |
| *Davallia denticulata* | 8 | 4 | 0 | 12 | 2.52 |
| *Davallia triphylla* | 0 | 3 | 0 | 3 | 0.63 |
| **Athyriaceae** | | | | | **3.15** |
| *Diplazium cordifolium* | 9 | 0 | 4 | 13 | 2.73 |
| *Diplazium crenatoserratum* | 0 | 2 | 0 | 2 | 0.42 |
| **Hymenophyllaceae** | | | | | **2.94** |
| *Crepidomanes bipunctatum* | 8 | 1 | 0 | 9 | 1.89 |
| *Didimoglossum mindoroense* | 4 | 1 | 0 | 5 | 1.05 |
| **Lygodiaceae** | | | | | **1.89** |
| *Lygodium longifolium* | 0 | 0 | 8 | 8 | 1.68 |
| *Lygodium circinnatum* | 1 | 0 | 0 | 1 | 0.21 |
| **Nephrolepidaceae** | | | | | **0.63** |
| *Nephrolepis falcata* | 0 | 2 | 1 | 3 | 0.63 |
| **Total of individuals** | **303** | **98** | **75** | **476** | **100** |

Notes: PF - Primary forest, SL – Selectively logged HD – Highly degraded

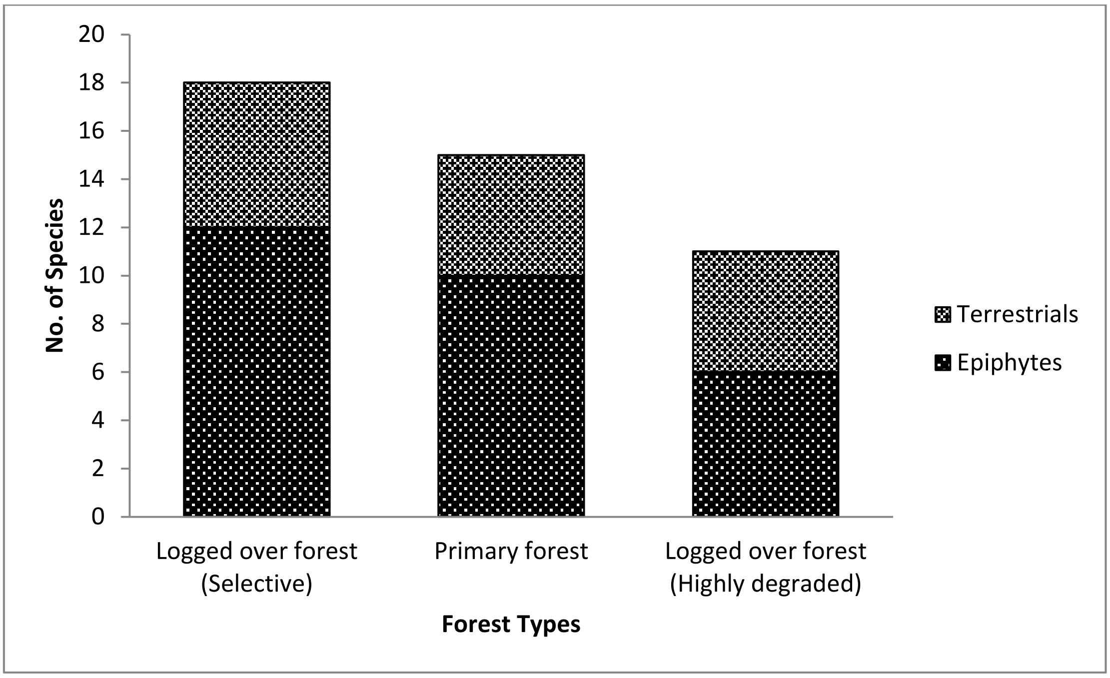


**Figure 4.1: Number of species in each forest types in primary forest, logged-over forest of selectively logged and logged-over forest of highly degraded**

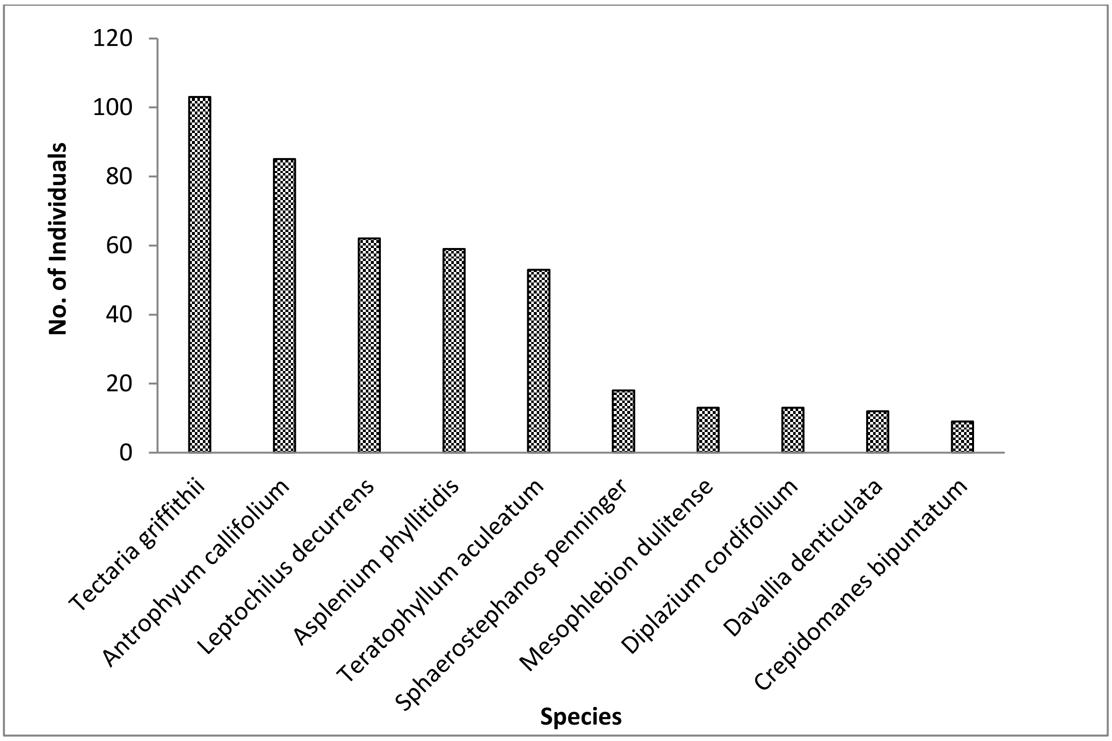


**Figure 4.2: Top ten dominant fern species in term of abundance**

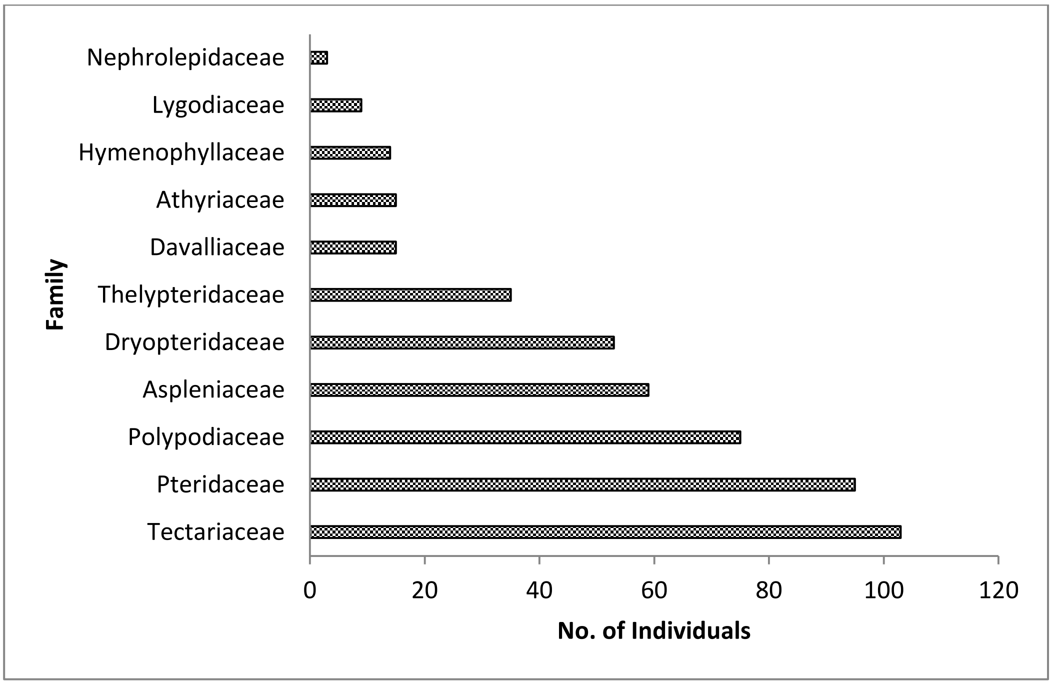


**Figure 4.3: Abundance of ferns by family**

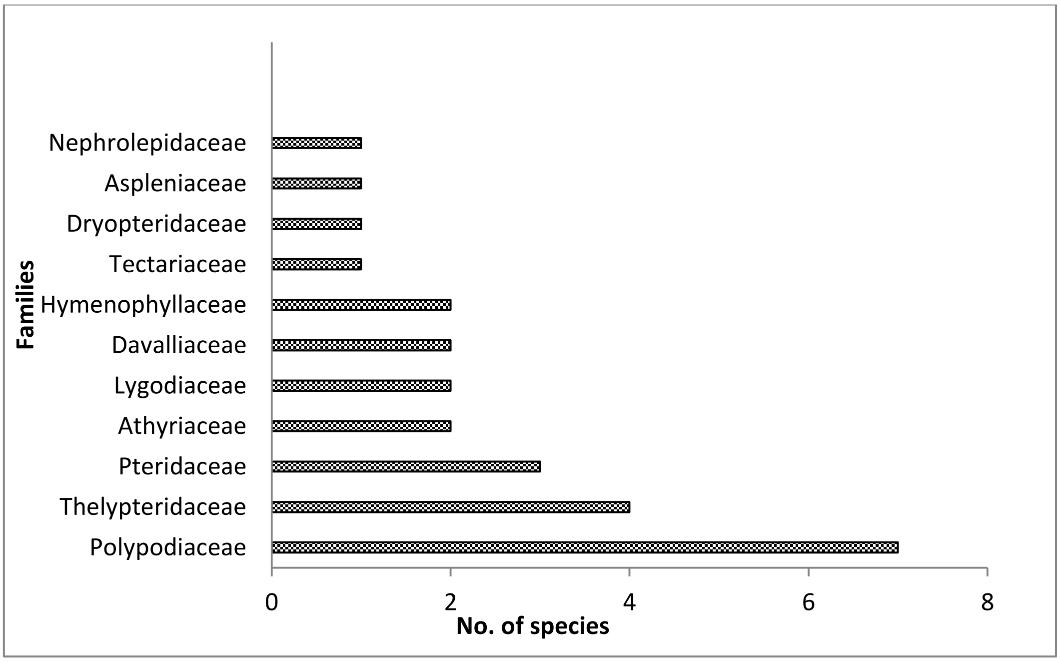


**Figure 4.4: Species richness of fern by family**

## 4.2 Ferns species diversity in primary and logged-over forest

### 4.2.1 Species richness

Ferns were more diverse in primary forest than in logged-over forest (Table 4.2). The Shannon-Wiener diversity Indices show that primary forest is more diverse (H= 1.43) followed by logged-over forest of selectively logged (H=1.17) and logged over forest of highly degraded (H=0.96). Diversity of primary and both categories of logged over forests is not significantly different about F = 2.54 ($P>0.05$). The Diversity of the epiphytes ferns in the primary forest was higher as compared to the both categories of logged over forests and the differences was significant (Table 4.2). Similarly, terrestrials’ ferns also showed higher diversity in the primary forest however the differences were not significant. The Simpson_1-D Index showed the same trend in which primary forest was more diverse compared to logged-over forest of selectively logged and logged-over forest of highly degraded (Table 4.3). The greater the value of Simpson_1-D index, the greater the sample diversity. Most of the other diversity indices used supported the foregoing trend.

**Table 4.2: Comparison of diversity of epiphytes and terrestrials ferns in Primary Forest (PF), Logged-over forest of selectively logged (SL) and Logged-over forest of highly degraded (HD)**

| | **PF** | **SL** | **HD** | **Significant test** |
|---|---|---|---|---|
| **Mean of H’<br>(All species/Overall)** | 1.425 | 1.165 | 0.960 | One-way ANOVA<br>F(2.54)<br>$P>0.05$ (0.1028)<br>**NS** |
| **Mean of H’ (Epiphytes)** | 1.079 | 0.832 | 0.429 | One-way ANOVA<br>F(4.918)<br>df=2<br>$p<0.05$ (0.01771)<br>**SF** |
| **Mean of H’ (Terrestrials)** | 0.463 | 0.287 | 0.456 | One-way ANOVA<br>F(0.521)<br>df=2<br>$P>0.05$ (0.601)<br>**NS** |

**Table 4.3: Comparison of diversity and evenness of ferns in Primary Forest (PF), Logged-over forest of selectively logged (SL) and Logged-over forest of highly degraded (HD)**

| | PF | (SL) | (HD) |
|---|---|---|---|
| **Taxa_S** | 15 | 18 | 11 |
| **Individuals** | 303 | 98 | 75 |
| **Dominance_D** | 0.3096 | 0.3789 | 0.4442 |
| **Simpson_1-D** | 0.6903 | 0.6210 | 0.5558 |
| **Shannon_H** | 1.4253 | 1.1653 | 0.9600 |
| **Evenness_e^H/S** | 0.7323 | 0.8525 | 0.8703 |

### 4.2.2 Species Richness Estimation

The estimated numbers of species between forest types by a same estimator were apart from each other and the difference was significant. The Jacknife (1st Order) and ACE estimate showed that logged-over forest of selectively logged was slightly richer than the primary forest, whereas CHAO 1 showed the opposite (Table 4.4). Species richness estimation at most sites did not reach an asymptote (figure 4.5) based on Chao 1, Jacknife (1st order) and ACE. The completeness of all samples was assessed by comparing the number of species observed with the estimated total species richness. In general, only logged-over forest (selectively logged and highly degraded) has reached the range of sample completeness above 90% with the sampling, ranging from 67.30 percent−95.00 percent (Table 4.4). Meanwhile, the primary forest ranged below 90% of sampling completeness of species richness (71.42 percent–79.23 percent).

**Table 4.4: Species richness estimates and completeness for each site according to three species richness estimators (Chao 1, Jacknife- 1st Order, and ACE)**

| | PF | SL | HD |
|---|---|---|---|
| **Number of species observed** | 15.00 | 18.00 | 11.00 |
| **Estimated species richness** | | | |
| **Chao 1** | 21.00 | 19.50 | 11.50 |
| **Jacknife (1st Order)** | 19.38 | 26.75 | 16.25 |
| **ACE** | 18.93 | 20.70 | 11.98 |
| **Sample completeness (%)** | | | |
| **Chao 1** | 71.42 | 92.30 | 95.00 |
| **Jacknife (1st Order)** | 77.39 | 67.30 | 67.69 |
| **ACE** | 79.23 | 86.96 | 91.80 |

The accumulation curve for primary forest (PF) and logged-over forest of selectively logged (SL) and logged-over forest of highly degraded (HD) showed the greatest number of species diversity in terms of abundance in primary forest. The accumulation curve of primary forest has leveled higher compared to the logged-over forest (Figure 4.5).

### 4.3.2 Species Abundance

Primary forest has the highest abundance totaling to 303 (63.66%) individuals, followed by the logged-over forest of selectively logged by 98 (20.59%) individuals and the logged-over forest of highly degraded with 75 (15.76%) individuals (Figure 4.6) (Appendix 1). This was supported by the rank abundance curve (Figure 4.7). Logged-over forest of selectively logged exhibited the longest tail of the curve, which indicated that it was more diverse than primary forest and logged-over of highly degraded in terms of species richness. The shortest tail was observed in logged-over forest of highly degraded suggesting low species diversity as compared to the other two forest types. Primary forest has steeper abundance curves compared to both categories of logged-over forest.

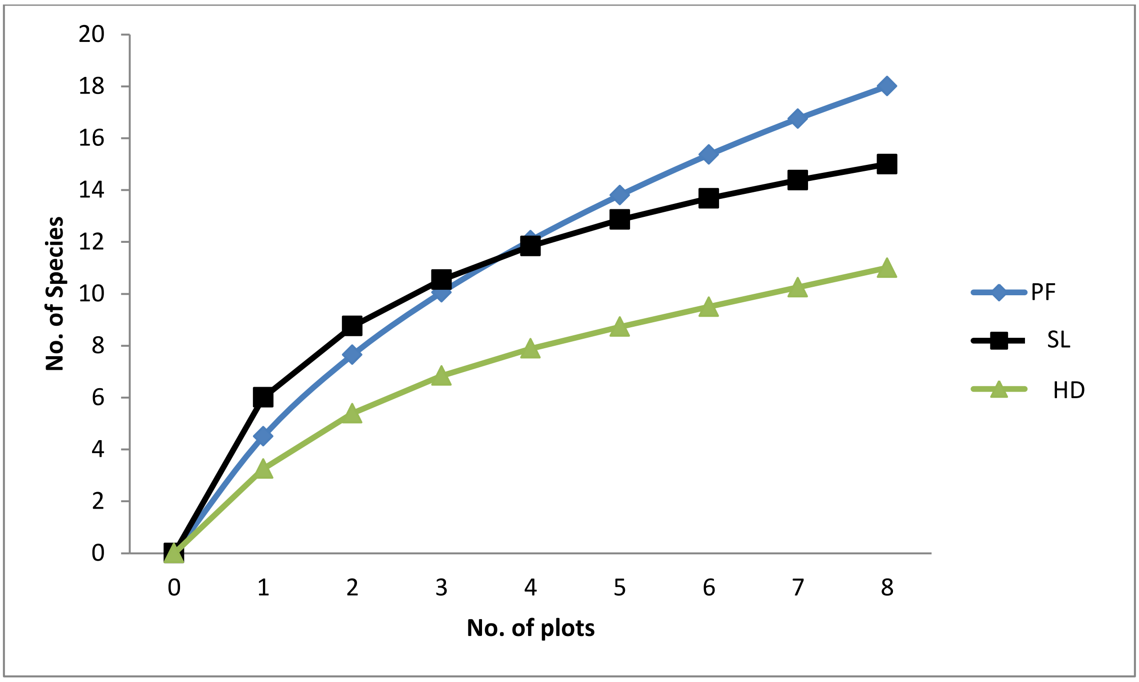


**Figure 4.5: Species accumulation curve for three forest types. PF – Primary forest, SL – Selectively logged, HD – Highly degraded**

This indicated that primary forest was dominated by several species with high abundance. The dominant species in the primary forest were *Tectaria grafithii* with 72 individuals, while it was *Asplenium phyllitidis* with 39 individuals in the both categories of logged-over forest (Table 4.1). In terms of evenness, the logged-over forest of highly degraded has more evenness (0.870) followed by the logged-over of selectively logged forest (0.853) and primary forest (0.732) (Table 4.3). This indicated that the total number of individuals in the highly degraded forest is quite evenly distributed between the 11 species.

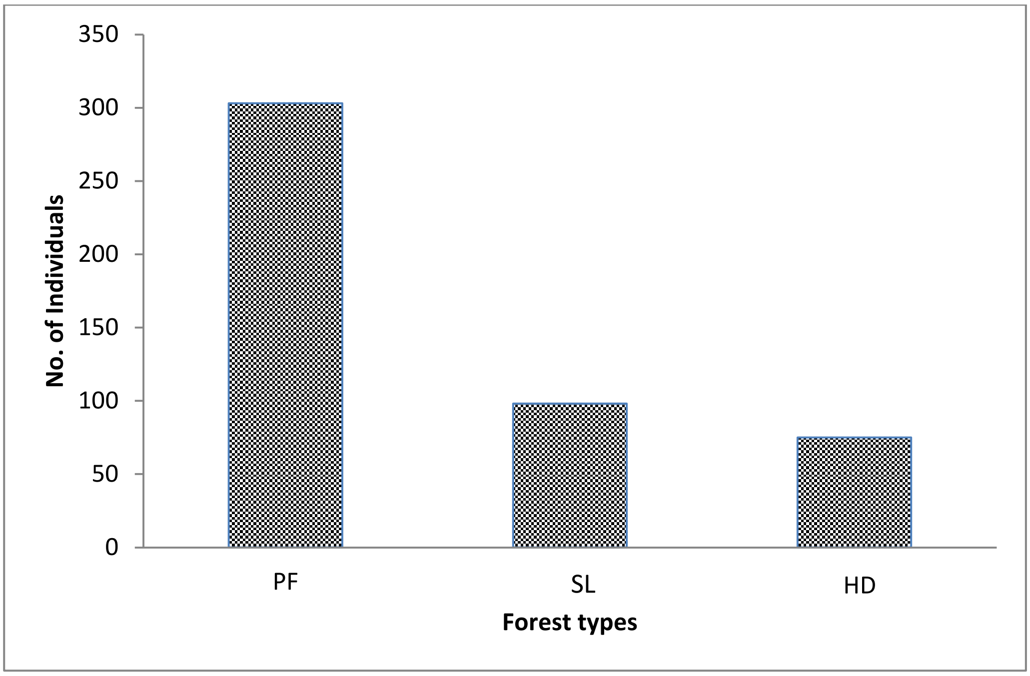


**Figure 4.6: Total abundance of ferns in different forest types. PF – Primary forest, SL – Selectively logged, HD – Highly degraded**

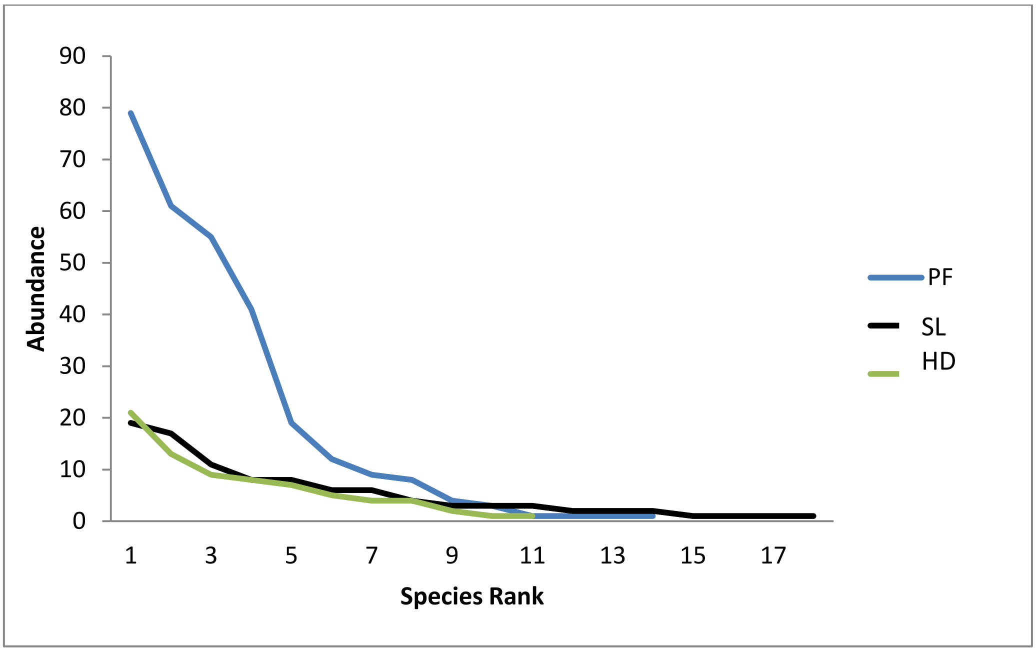


**Figure 4.7: Rank abundance distribution of ferns in three forest types. The species were ranked in order from the most to the least abundant. PF – Primary forest, SL – Selectively logged forest, HD – Highly degraded**

## 4.4 Species composition in primary and logged-over forest

Similarity indices showed that the similarity between primary and logged-over forests of selective logging was higher (0.734) than the highly degraded forest (0.584). On the other hand, the similarity between the logged-over forest of selectively logged and highly degraded was higher (0.786) compared to the primary forest (0.734). However the similarities difference does not significance. Cluster analysis was used to classify the similarities between species types and the 24 plots in Danum Valley and Ulu Segama forest complex (Figure 4.8). The dendrogram showed 24 chains with four large groups. The first group consisted mainly of plots from selectively logged and plots from highly degraded forest. The second group included plots from primary forest and selectively logged forest whilst, the third and fourth group was formed mainly by primary forest plots.

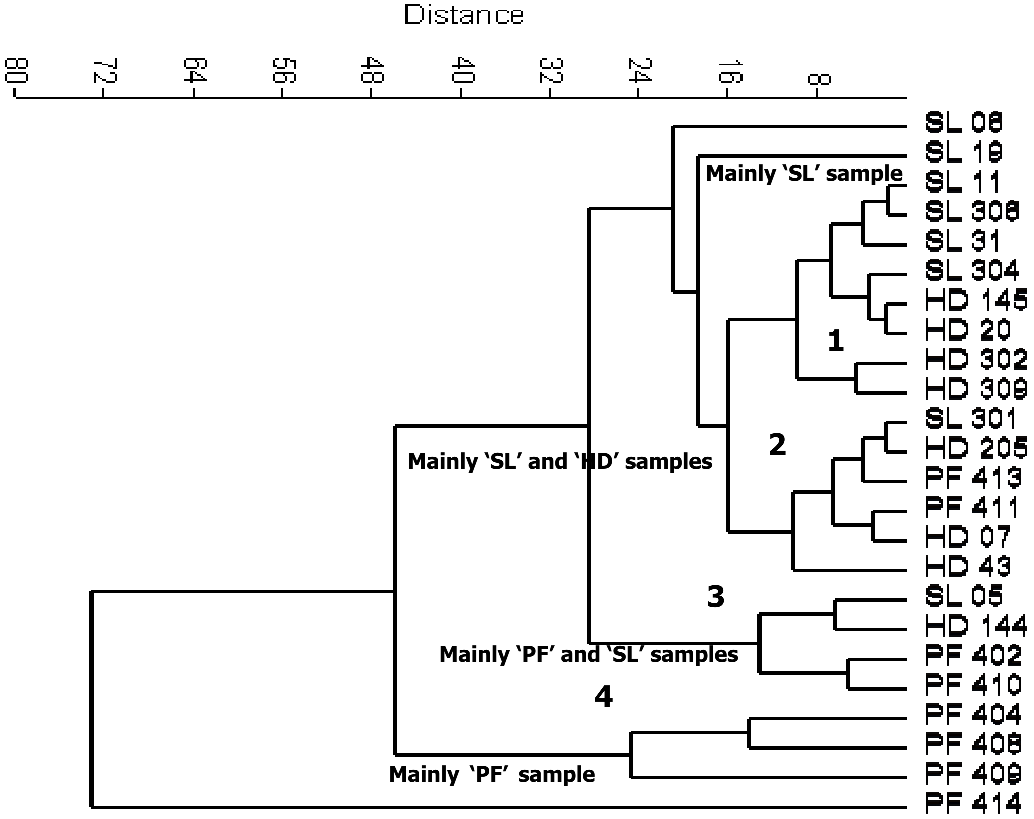


**Figure 4.8: A dendrogram from the cluster analysis using species parameters data. PF – Primary forest, SL – Selectively logged, HD– Highly degraded**

### 4.5 Effects of Environmental Variables on the Diversity of Ferns

In each of the forest types canopy covers, relative humidity and temperature has been recorded. Primary forest had the highest mean of canopy cover that was 90.3% (Appendix II). Again, the mean reading of temperature in primary forest was higher as compared to the logged over forest of selectively logged and highly degraded forest (Appendix III). For the relative humidity, primary forest had higher mean of relative humidity than both of the logged over forest categories (Table 4.5) (Appendix IV). The difference of the mean of canopy cover and temperature in each forest area are not significant (p> 0.05). However, the mean of relative humidity was significantly different in each of the forest types (p<0.05) (Table 4.5).

**Table 4.5: The mean of canopy coverage, temperature and relative humidity in each of the forest areas**

| | **PF** | **SL** | **HD** | **Significant test** |
|---|---|---|---|---|
| **Mean of Canopy coverage (%)** | 90.3 | 87.6 | 81.48 | One-way ANOVA<br>F(2.911)<br>P>0.05 (0.08545)<br>**NS** |
| **Mean of Temperature (℃)** | 25 | 24.9 | 24.7 | One-way ANOVA<br>F(0.3609)<br>df=2<br>p>0.05 (0.703)<br>**NS** |
| **Mean of Relative humidity (%)** | 95.8 | 92.6 | 88.63 | Kruskal-Wallis Test<br>Chi square:<br>6.924<br>df=2<br>**p<0.05** (0.0311) |

#### 4.5.1 Forest Types

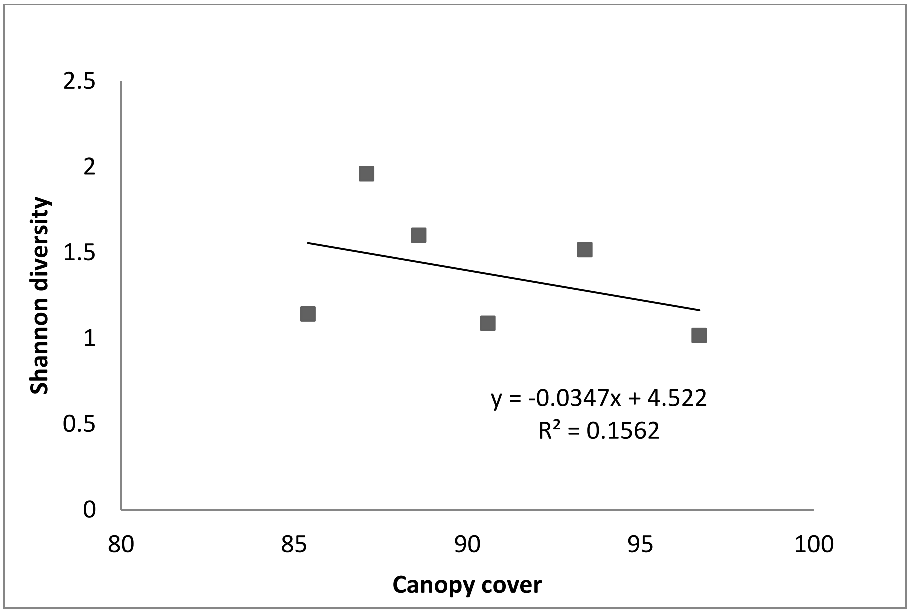
Across the study site, canopy cover showed a weak negative correlation with the species diversity of ferns (Figure 4.9) (Table 4.6). This indicates that increasing the percentage of the canopy cover will decrease the species diversity of ferns. In term of temperature, there was a weak negative correlation between the temperature and species diversity in primary forest and logged-over of highly degraded forest (HD) (Figure 4.10). However logged-over of selectively logged forest showed almost no correlation between species diversity and temperature when the R value is almost zero. This mean that increase in temperature does not really affect the species diversity of ferns in each of the forest types. Only in primary forest showed a weak positive correlation between relative humidity and species diversity of ferns. However, both logged-over forest of selectively logged and highly degraded forest's relative humidity showed a weak negative correlation with the species diversity of ferns (Figure 4.11)

The R square is a statistical measure of how close the data are, to the fitted regression line. It is also known as the percentage of the response variable variation that is explained by a linear model. Based on the linear regression analysis, only relative humidity in selectively logged forest has a higher R square value. This indicated that only relative humidity in selectively logged have variance that closer to fall to the fitted regression line (Table 4.6). Overall the R square value of each environmental variable, are lower (lower than 30%) except for relative humidity in logged-over forest of selectively logged (80%) and all are not significant. Therefore, the model explains only a little of the variability of the environment variables around its mean. In the other words, only a few of the data points would fall on the fitted regression line.

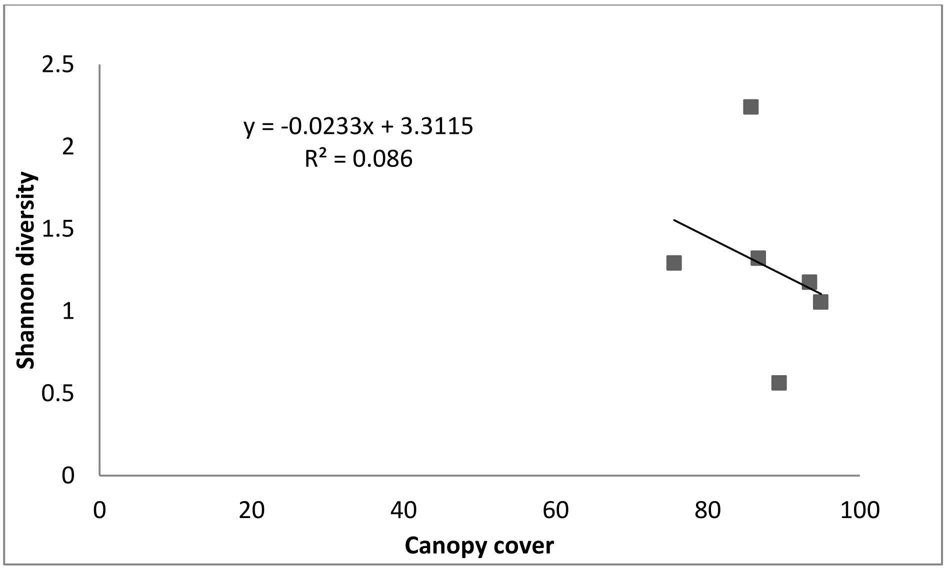
**Table 4.6 Linear regression of each environment variable with the diversity of ferns**

| Forest types | Ecological factors | p | R-value | R^2 |
|---|---|---|---|---|
| PF | **Constant** | NS | 0 | 0 |
| | **Canopy coverage** | NS | -0.39 | 0.15621 |
| | **Temperature** | NS | -0.49 | 0.24616 |
| | **Relative humidity** | NS | 0.24 | 0.05836 |
| SF (HB) | **Constant** | NS | 0 | 0 |
| | **Canopy coverage** | NS | -0.29 | 0.08599 |
| | **Temperature** | NS | 0.02 | 0.00060 |
| | **Relative humidity** | NS | -0.91 | 0.82375 |
| SF (LB) | **Constant** | NS | 0 | 0 |
| | **Canopy coverage** | NS | -0.30 | 0.092095 |
| | **Temperature** | NS | -0.06 | 0.004386 |
| | **Relative humidity** | NS | -0.02 | 0.000521 |

a).


2.5
2
1.5
1
0.5
0
Shannon diversity
y = -0.0347x + 4.522
R² = 0.1562
80
85
90
95
100
Canopy cover

b)


2.5
2
1.5
1
0.5
0
Shannon diversity
y = -0.0233x + 3.3115
R² = 0.086
0
20
40
60
80
100
Canopy cover

c)

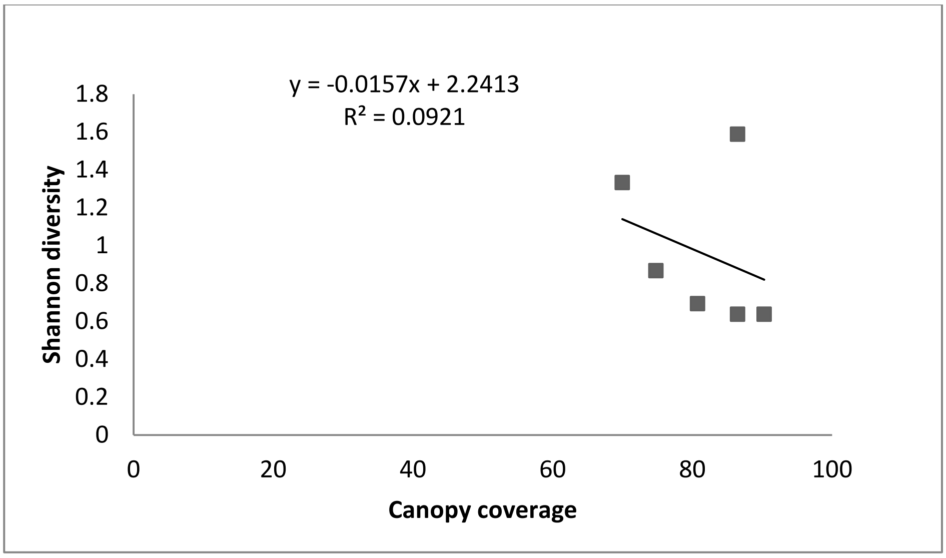
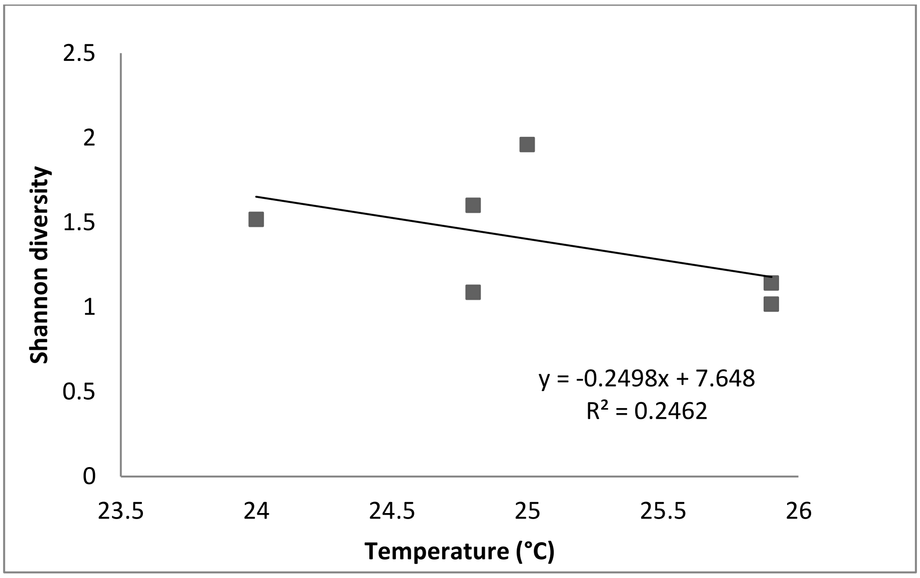
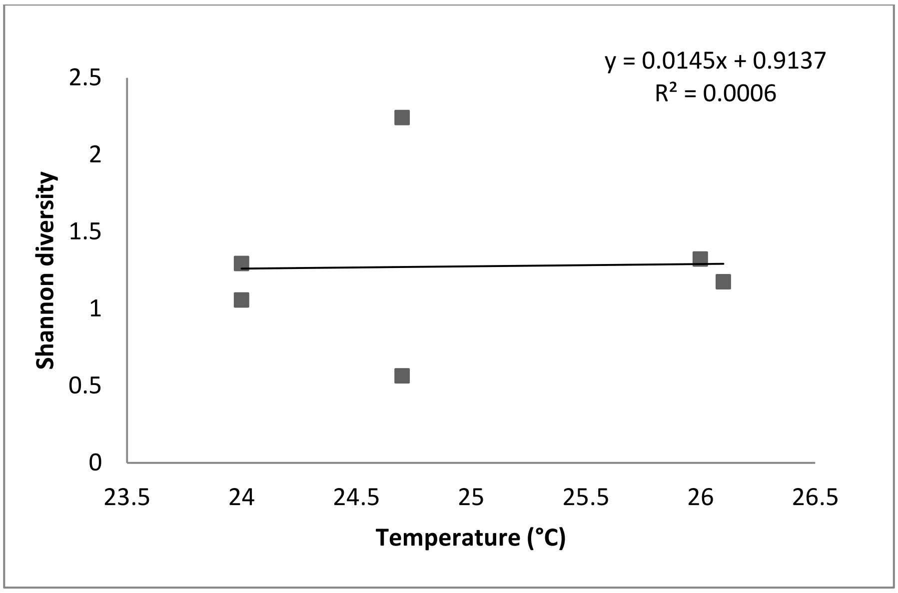


**Figure 4.9: Linear regressions of species diversity (H’) and canopy coverage. a) Primary forest, b) Selectively logged, and c) Highly degraded**

a).



b).



c).

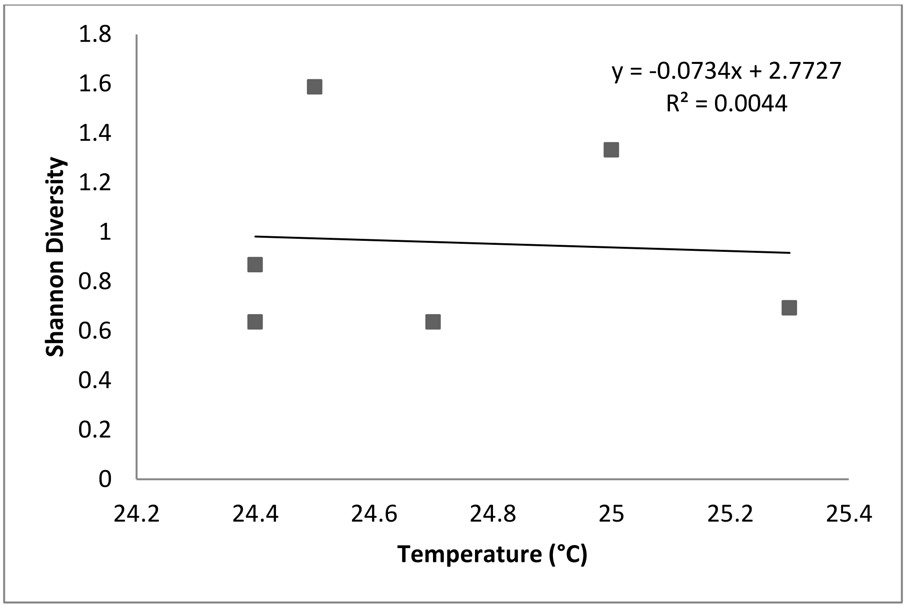
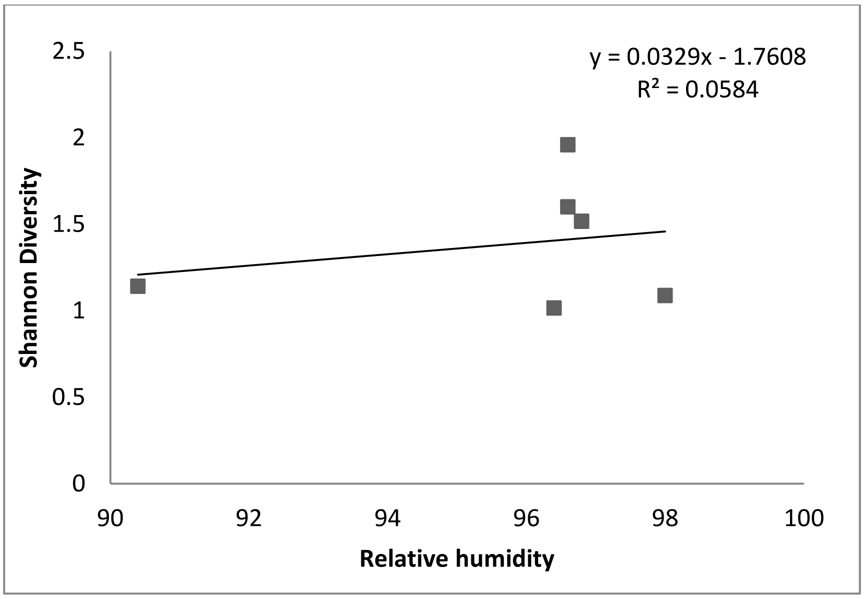
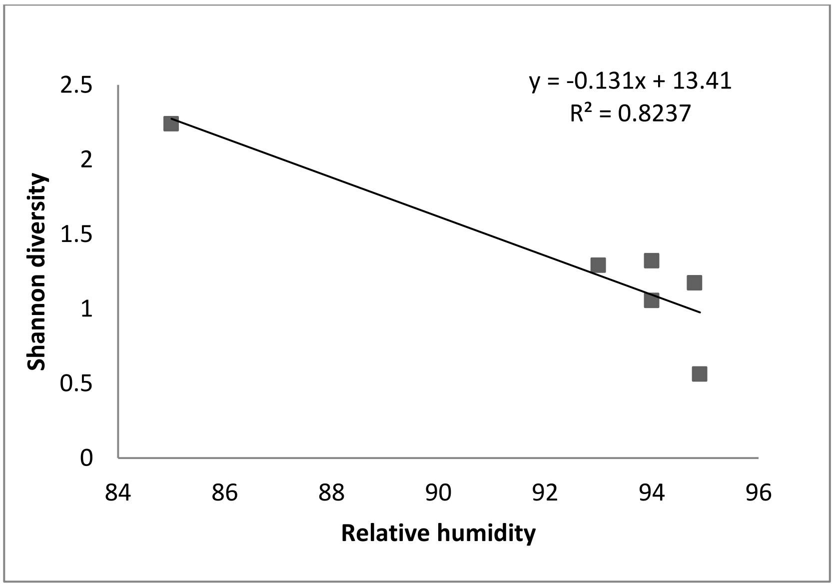


**Figure 4.10: Linear regressions of species diversity (H') and temperature a) Primary forest b) Selectively logged, and c) Highly degraded**

a).


y = 0.0329x - 1.7608
R² = 0.0584
Shannon Diversity
2.5
2
1.5
1
0.5
0
90
92
94
96
98
100
Relative humidity

b)


y = -0.131x + 13.41
R² = 0.8237
Shannon diversity
2.5
2
1.5
1
0.5
0
84
86
88
90
92
94
96
Relative humidity

c).

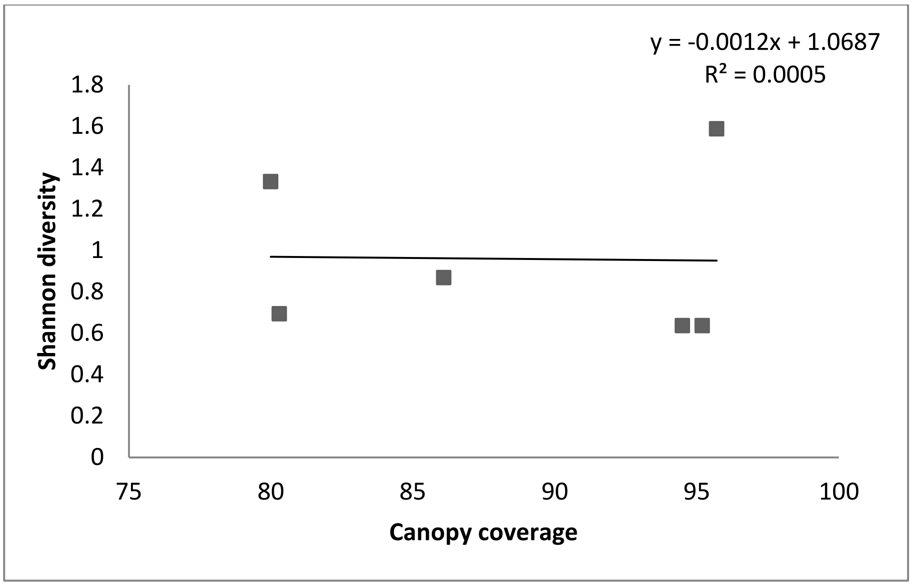


**Figure 4.11: Linear regression of species diversity (H’) and relative humidity a) Primary forest b) Selectively logged, and c) Highly degraded**

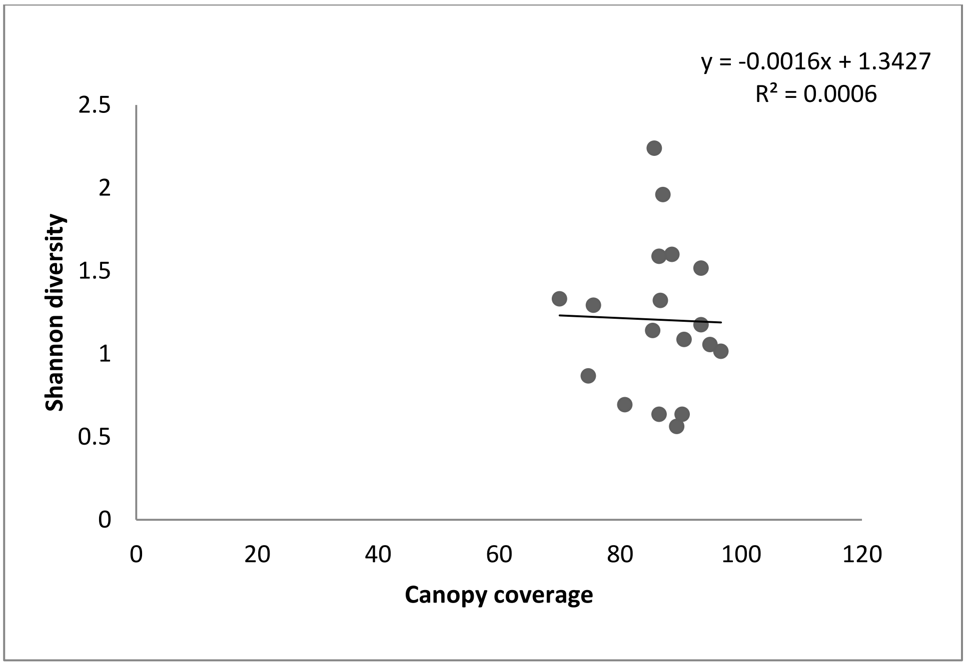
### 4.5.2 Forest Complex level

All of the variables have positive correlation with the species diversity of ferns. However, the correlation were not significant (P>0.05) and only temperature variables showed positive correlation with the higher R square value compared to the other environmental variables (Table 4.7). None of the R square value is more than 10 % and the model cannot explain well the variability of the environmental variables around its mean.

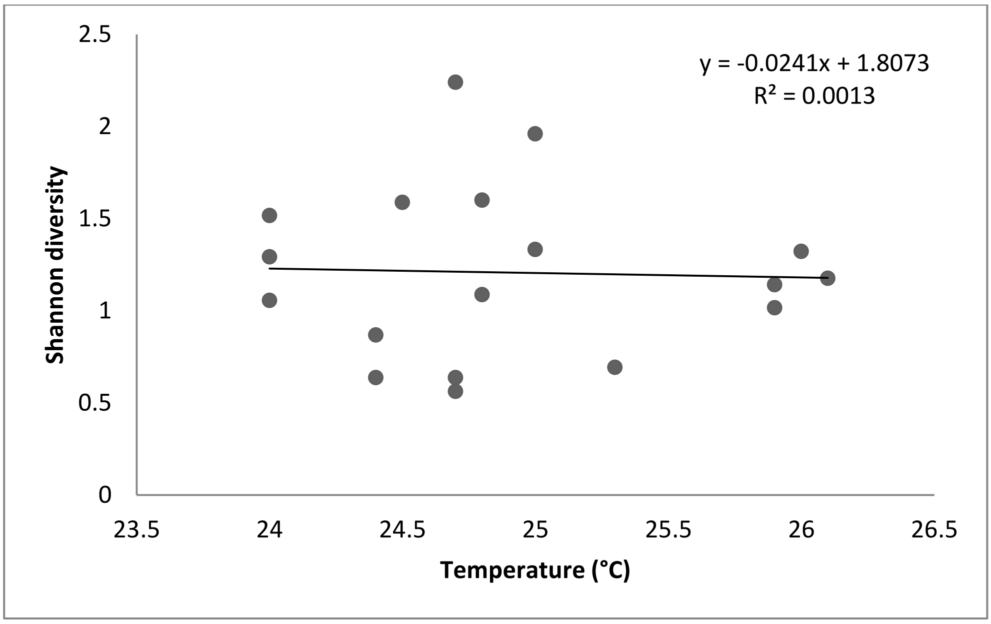
**Table 4.7 Linear regression of each environment variable with the diversity of ferns in forest complex level**

| | Coeff. | Std.err. | T | p | R | R^2 |
|---|---|---|---|---|---|---|
| **Constant** | 2.4529 | 3.9301 | 0.62413 | 0.5425 | 0 | 0 |
| **Canopy cover** | -0.00382 | 0.0250 | -0.15285 | 0.8807 | 0.024 | 0.0006 |
| **Temperature** | -0.050379 | 0.1237 | -0.40706 | 0.6901 | 0.115 | 0.0134 |
| **Relative humidity** | 0.003600 | 0.0322 | 0.11166 | 0.9126 | 0.0179 | 0.0003 |

a)


y = -0.0016x + 1.3427
R² = 0.0006
Shannon diversity
2.5
2
1.5
1
0.5
0
0
20
40
60
80
100
120
Canopy coverage

b)


y = -0.0241x + 1.8073
R² = 0.0013
Shannon diversity
2.5
2
1.5
1
0.5
0
23.5
24
24.5
25
25.5
26
26.5
Temperature (°C)

c)

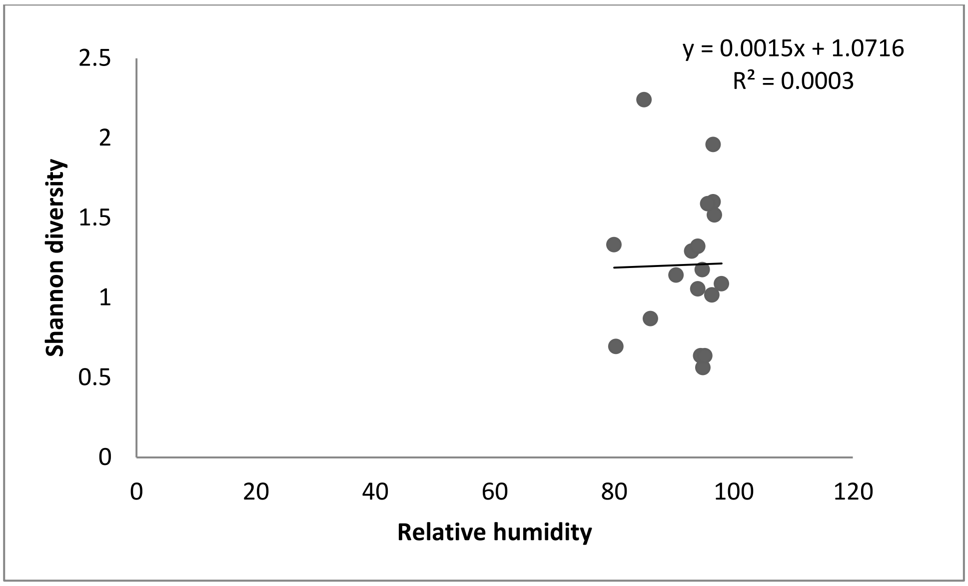


**Figure 4.12: Linear regression of species diversity (H') and environmental variables of ferns. a) Linear regression species diversity vs. Temperature, b) Linear regression species diversity vs. Relative humidity, and c) Linear regression species diversity vs. Canopy coverage**

# CHAPTER 5

# DISCUSSION

## 5.1 Ferns Diversity

### 5.1.1 Ensemble species richness of ferns

The diversity of ferns in lowland forests of Danum Valley and Ulu Segama forest complex was low as compared to the ferns diversity in other areas in Sabah. Based on the results of this study, there were only 26 species of ferns in the study sites of the primary and logged-over forests. This number is far lower than the number of taxa reported by Parris (1985) from Danum Valley, which were 157 taxa. However, she also collected epiphytic ferns from tree trunks above 2 m height and in the canopy. Parris (1997) has remarked that the Danum Valley appears to be poor in ferns at ground level but many species are epiphytes high up in the crowns of trees. Her finding is in agreement with the results of this present study. In addition, only terrestrial ferns and epiphytes ferns below 2 m above ground level were collected in the present study.

The fern richness of the lowland forests of Danum Valley and Ulu Segama forest complex is in no way comparable to other areas in Sabah. For example, Mount Kinabalu has 608 species of ferns  Parris *et al.* (1992).  Tawau Hills Park had recorded 83 species of ferns (Bidin and Jaman, 1999). While Gunung Kuli, in Imbak Canyon Conservation Area (ICCA) has 104 taxa in 24 families of ferns (Shim *et al.,* 2010). They noted that waterfall area and lower montane forests of Mount Kuli are areas of the highest fern diversity in ICCA. According to Kessler (2010), fern richness is highest in regions with high potential evapotranspiration, a high number of rainy days and marked topographical relief. He added that tropical mountain is no doubt the hot spots of fern diversity. This is supported by other studies on mountainous areas (Dzwonko and Kornas, 1994; Michael Kessler, 2000;

Pausas and Sáez, 2000). Basically, the composition and the richness of ferns are influenced by elevational gradient and water availability especially at mid elevation on mountains (Mehltreter, 1996). Fern richness in lowland forests is lower than montane areas due to the low moisture level in lowland forests.

Other lowland tropical forests in other parts of the world also showed poor fern richness compared to montane forests. The lowland forest of Jaguarao Forest Pernambuco in Brazil, a Neotropical area, has only 25 species of ferns belonging to 12 families (Silva *et al.,* 2011). This is quite similar with the species richness of the current study. Similarly, Beukema and van Noordwijk (2004) had recorded only 65 species of terrestrial ferns from sampling plot of 8.16 ha in the lowland forest of Jambi, Sumatra (Indonesia). Pacencia and Prado (2005) also recorded only 60 species in 17 families of ferns within a 4.44 ha sampling plot in the lowland ombrophilous forests of Bahia, Brazil. Nonetheless, ferns richness in tropical lowland forests appear to be exceptionally high compared to temperate forests. For example, Richard *et al.* (2000) only found 19 species of ferns in a one hectare plot of old growth forest at Mont St. Hilaire, Quebec in North America.

The lower species richness in the study site was constricted by the sampling effort and sampling technique. Most of the fern collections in the previous studies in Sabah were extensive and usually covering larger areas. These studies were focused on collecting randomly as many species as possible. The larger the sampling area covered the higher chances to record more species of ferns. In contrast, the collection of ferns in the present study was restricted by the fixed plot size in order to compare the species richness among the forest types.

### 5.1.2 Species Diversity

Primary forest had shown higher species diversity as compared to both categories of logged-over forest. This trend is similar to the findings by Kluge and Kessler (2006) and Pacencia and Prado (2005). They stated that the lower number of ferns species are related to the degree of habitat disturbance. The diversity of ferns in the logged-over forest should therefore be lower than in a primary forest. Barthlolt *et al.* (2001) reported that a secondary forest in the Venezualan Andes harboured only 81 species of vascular epiphytes as compared to 178 species in a primary forest. They suggested that the low species numbers and abundance as well as the differences in species composition are mainly due to the less diverse phorophyte structure and less differentiated microclimate in the disturbed and secondary vegetation as compared to the primary forest. These result also consistent with the study by Okuda *et al.* (2003) and Brearly *et al*. (2004) on tree diversity in logged-over forest. They found that the selectively logged forest had lower diversity. However, poison-girdling of non-commercial species and shifting cultivation may have influenced the results of the study in both cases (Berry *et al.,* 2008).

The present study however, inconsistent to previous study on plants in secondary forest due to logging activities. Cannon *et al.* (1994) reported that there was an increase in tree diversity eight years after selective logging in West Kalimantan, Indonesia. Similarly, Roberts and Gilliam (1995) found that species diversity of herbs in forest floor layer increased after disturbance on mesic sites, but shade-tolerant tree species decreased. They concluded that increase in resource availability, particularly light, in disturbed stand have influenced the species composition in disturb areas.

Majit *et al.* (2011) reported that there were more diverse orchids in old secondary growth than those in primary forest and young secondary forest in Crocker Range Park, Sabah. They suggested that illegal harvesting of wild orchids in the primary forest by the local people have caused the diversity in primary forest to be lower than the old secondary forest. They also stated that the evenness of orchid distribution in old secondary forest had influenced the diversity in the old secondary forest. The higher species richness found in the logged-over forest of high biomass in the present study does not agree with the findings of Gobilik (2002) who studied ginger diversity in primary and logged forests. He observed that diversity of gingers was higher in logged forest compared to primary forest and stated that the abundance of gingers was influenced by the tree DBH, soil PH and soil nutrient. Similarly, Berry *et al.* (2010) had outlined that floral species richness was typically higher in logged forest than in primary forest. In most studies on quantifying the impacts of logging on biodiversity within lowland dipterocarp forests of Sabah, more than 90% of species recorded in primary forests were also present in logged forests, including species of conservation concern (Berry *et al.,* 2010).

Ariyanti *et al.* (2008) also reported that bryophytes assemblages of selectively logged forests and cacao agroforests were as rich as in the natural forest. Similarly, Verburg and van Eijk-Bos (2003) showed that logging had no significant effect on tree diversity nor did diversity significantly change over time after logging. They found that Fisher' alpha did not differ significantly among forest types (virgin forest and two types of selectively logged forest). Costa and Magnusson (2002), who examined the selective logging effects on abundance, diversity and composition of tropical understory herbs, had demonstrated that the ground herb community was not severely affected by selective logging at the intensities used in their study. Berry *et al.* (2008) who studied the tree diversity at Danum Valley Conservation Area found no difference between logged and unlogged forests in a localized alpha-diversity of trees 18 years after logging. However, they reported that there was a significantly higher diversity of small trees in larger spatial scale. There is no published data on ferns diversity in primary and logged-over forests in Malaysia for comparison with the present findings. The closest study was by Beukema and van Noordwijk (2004) in Jambi, Sumatra (Indonesia) whereby they found that ferns diversity was higher in jungle rubber than in primary forests.

The higher diversity of ferns in the primary forest compared to the logged-over forest could be influenced by the distribution of the epiphytic species of ferns (Table 4.2). In each of the forest types, the number of epiphytes fern was higher than the terrestrial fern (Figure 4.1). Primary forest has higher abundance of epiphytic ferns as compared to the logged-over forest. However in term of species richness, more species of epiphytes ferns present in selectively logged forest. Epiphytes abundance, species numbers and community composition were shown to be significantly lower in disturbed forest and in secondary forest than in primary forest (Turner *et al.*, 1994). Disturbed forests are generally drier and more sun exposed the general adaptation of epiphytes to temporarily water stress (Zotz, 1995). Therefore, in the present study most species that is partially drought tolerant for examples *Crepidomanes bipunctatum* and *Didimoglossum mindoroense* (Hymenophyllaceae) presence abundantly in primary forest. In the other hand epiphytes species present in logged over forest are the drought tolerant species. Nevertheless, some moist and shady habitats have been preserved in the disturbed forest and in the clearing to which shade tolerant species are adapted. For examples in the study site both of the partially tolerant species (*Crepidomanes bipunctatum* and *Didimoglossum mindoroense*) presence in selectively logged forest.

Other factor that influences the high species diversity in primary forest is related to the degree of disturbance. This was supported by Pacencia and Prado (2005) in who stated that a decrease in the number of fern species is related to the degree of habitat disturbance. The degrees of disturbance influence the environmental condition within the study site. Relative humidity, rainfall and relative air temperature are the variables that influence ferns diversity (Kessler et al., 2007). According to Kessler et al. (2007) location with greater moisture normally has greater diversity and abundance. In the present study the primary forest has higher canopy cover, relative humidity and temperature compared to the logged-over forest (Table 4.4). This condition enables the original ferns species to establish abundantly. In contrast, the logged-over forest may have higher species composition because of the habitat heterogeneity but the presences of the individual's ferns are not abundant as compared to the primary forest.

In term of species richness, ferns species richness was high in selectively logged forest than the primary forest. Habitat heterogeneity can explain this condition. Different taxa of ferns react in different ways to disturbance. Greater resource heterogeneity in the selectively logged forest enables the coexistence of various ferns taxa that have different ecological preferences (Bazzaz, 1975; Cannon *et al.,* 1994). Berry *et al.* (2008) and Hill and Hamer (2004) showed that species richness of flora in logged forest is higher than in unlogged forest, probably reflecting increased landscape scale heterogeneity resulting from spatial variation in the intensity of disturbance due to logging. The high species richness in the selectively logged forest was largely due to an increase in species of fern that have their most favourable conditions for growth and reproduction in logged-over forest environments compared to shady forest understory (Beukema and van Noordwijk, 2004). Similarly, Watkins (2007) has reported that fern gametophytes had varying levels of disturbance tolerance. The nature of disturbed habitats can create positive feedback for species that prefer disturbed sites.

### 5.1.3 Species Abundance

In the present study, the primary forest has the highest abundance of fern communities as compared to the other types of forest. Community diversity of ferns is influenced by habitat quality, for examples, the presence of high moisture, substrate pH and available calcium (Greer *et al.,* 1997). On the other hand, abundance is associated only with diversity. Changes in community composition could be associated with environmental factors through the effects on relative abundance.

The abundance of the fern communities is influenced by water availability (Barrington, 1993). Marquez *et al.* (1997) reported that water availability was an important factor underlying fern distribution on the Iberian Peninsula in Spain, and a humidity gradient was the main factor influencing differentiation of ferns species in Rwanda (Dzwonko and Kornas, 1994). In the present study, most of the plots in the primary forest were located near water sources. Thus, a few fern species that prefers high moisture had the highest abundance. This is not surprising due to the fact that fern gametophytes need high moisture level to complete fertilization. Therefore, ferns distribution and abundance are strongly associated with water regime. In addition, the primary forest at the study site had the highest relative humidity when compared to the other forest types and was significantly different. This factor could have attributed to the high ferns abundance in the primary forest.

The high abundance of ferns in the primary forest was also due to the number of individuals of *Tectaria griffithii*, *Antrophyum callifolium* and *Leptochilus* cf. *deccurens*. These three species were present in all of the forest types, but were found to be the most abundant in the primary forest. Thus, the lower species diversity in the primary forest could be due to these dominant species. Higher ferns abundance in the primary forest is also associated with the fact that these plants prefer humid, shaded environments with good water availability, a condition commonly found in primary forests.

The alteration of the canopy structure in the logged-over forests may stimulate the growth of different fern species that is light demanding but not the abundance of fern communities. Shade tolerant species are capable of germinating and establishing in the shade, and as a result population tends to skew size and age structure, with many small individuals and few large individuals. As a result, both of the logged-over forests in the study site are low in species abundance.

## 5.2 Species Composition in Primary and Logged-over Forests

Most of ferns that are regarded as forest species (species that can be found in shade or deep shade, light shade and grow in the forest) were found in both categories of logged-over forests. However, their abundance was lesser in the logged-over forest. Examples of these species are *Crepidomanes bipuntatum* and *Didymoglossum mindorense* that belong to Hymenopyllaceae group. Hymenophyllaceae group has been known to tolerate some form of disturbance mediated condition such as low level of sunlight exposure and have been classified as forest species (Beukema and van Noordwijk, 2004). Selectively logged forest may have preserve moist and shady habitats to which shade tolerant species like *Crepidomanes bipuntatum* and *Didymoglossum mindorense* are adapted to. Other forest species are *Tectaria griffithii*, *Antrophyum callifolium*, *Leptochilus* cf. *deccurens*, *Mesophlebion dulitense* and *Lygodium circinnatum*. All of these species have a high abundance in primary forest and among those species only *Lygodium circinnatum* is restricted to the primary forest in this study.

A number of forest species of ferns such as *Sphaerostephanos heterocarpus* and *Diplazium crenatosseratum* were recorded in the selectively logged forest but not in the primary forest. This is due to the light condition in the forest. *Sphaerostephanos heterocarpus* and *Diplazium crenatosseratum* need a light shade to grow and this condition is available in the logged-over forest of high biomass. Most of the fern species present in the logged-over forest of highly degraded are non-forest species (species of open and open/light shade condition and prefer habitat other than forest) and a few are forest species. The grouping of ferns as forest and non-forest species does not imply that 'non-forest' does not grow in the forest. A number of them occur in the forest, especially in gaps, but they are more abundant in open condition.

Selectively logged forest has higher similarity to the highly degraded forest compared to the primary forest. Both categories of logged-over forests share similar environmental condition. Most of the fern species in the logged-over forests are light demanding species. Logging had changed the canopy structure and altered the composition of the stand, reducing the number of shade tolerant species and stimulating light demanding species (Silva *et al.,* 1995). Examples of the light demanding species are the *Nephrolepis falcata* and *Lygodium longifolium* . The shade tolerant species are the *Lygodium circinnatum, Sphaerostephanos heterocarpus* and the Hymenophyllaceae group (*Didimoglossum mindoroense*) (Beukema and Noordwjick, 2004). However, selectively logged forest harbours higher species richness than the highly degraded forest. This is related to the age of the logged-over forest and the logging technique used.

Classification of the plots based on species showed that the plots of the primary forest are clustered separately from plots of both categories of the logged-over forest. This implies that most plots in primary forest have similar species of ferns. Nevertheless, a few plots from the primary forest and selectively logged forest were grouped together, which indicated that these plots have similar species of ferns. Several plots in the selectively logged have similar environmental condition with the primary forest in terms of canopy coverage and this has resulted in the grouping of these plots. Therefore, micro-level site differences pertaining to environmental conditions greatly influenced the species diversity of regenerating forests whereby certain species prefer specific forest types whilst the more robust and adaptive fern species thrive in all forest types.

## 5.3 Relationship between fern diversity and ecological variables

### 5.3.1 Comparison of ecological variables between forest areas

Based on the result of the study, the primary forest indeed has the highest mean reading of canopy cover, temperature and relative humidity. However, the differences are not significant except for the relative humidity. Bernard (2004) remarked that differences in environmental factors other than those related to the effects of logging are believed to be minimal if the primary and secondary forests were located close to each other. This is similar with the description of primary forest by Laurance *et al.* (2002), Brown (1993) and Lovejoy *et al.* (1986). They stated that lowland rainforest has stable, unbroken canopy with little radiation penetrates through the many foliage layers to ground level, except where gaps occur. This trend also follow that of Silva *et al.* (2011) which reported that the interior and edge sites of the forest differ in relation to air temperature and relative humidity.

However higher temperature in the primary forest as compared to the logged-over forest are in contrast with a theory proposed by Brown (1993) whereby the larger the gap the greater the amount of solar radiation that may reach the forest floor. In the case of the present study, the percentage of canopy cover in primary forest was higher than the logged-over forest. So the air temperature of the primary forest should be lower than the logged-over forest.

### 5.3.2 Effect of Environmental Variables to Ferns Diversity

In plot level and forest level, none of the environmental variables had significantly affected the diversity of ferns. In forest area level, all of the ecological variables showed weak positive correlation with the fern diversity. The regression coefficient ($R^2$) values for all of the ecological variables are less than 0.1. This means that the ecological variables cannot be determined as a good regulator of the phenomenon in lowland rainforest. There was no significant correlation between canopy coverage, relative humidity and the air temperature with the diversity of ferns in all of the forest types.

The results are in contrast with the study by Dzwonko and Kornas (1994) which reported that humidity gradient was the main factor influencing the differentiation of flora and species richness of ferns in Rwanda. In addition, the result of the current study also disagree with the study by Silva *et. al.* (2011). They found that high fern diversity in interior forest was associated with high mean of relative humidity and low mean temperature.

Canopy cover was also not significantly correlated with fern diversity. This trend adhered to previous study by Greer *et. al* (1997) where it was reported that neither diversity nor total abundance was correlated with canopy cover. They suggested that understory light environment may be too complex and fluctuating for segregation among species of long-lived, low structure plants such as ferns. In the present study, canopy covers for all of the forest areas are fairly homogenous and not distinctly different from each other. Thus, canopy cover only has minor influence on fern diversity. Similarly, a study of species richness and density of ground herbs within a plot of lowland rainforest showed that there was no significant difference in canopy openness between upper and lower subplots (Poulsen, 1996). However, his study indicated that it is possible that a gap may have a major effect on the ground herb abundance and cover which will still be noticeable even long after the canopy was closed.

In the present study, humidity and air temperature did not affect much on the fern diversity. The communities of ferns in lowland rainforests are different from those found in montane areas that are usually affected by relative humidity and air temperature. All of the forest areas in the study site are climatically fairly homogeneous.

Soil conditions and topography are two environmental variables that influence the distribution of ferns. However, these two variables were not tested in this study. Karst *et al.* (2005) have demonstrated that ferns have predictable distribution along a soil moisture gradient and reported that at a mesoscale, soil moisture have a dominant role in determining ferns distributions. Similarly Greer *et al.* (1997) reported a separation among fern species along a soil moisture gradient in Southeastern Ohio. In addition, Tuomisto and Poulsen (2000) showed that the difference in ferns communities' was attributed to variation in soil drainage in the Amazon rainforest. Kessler and Lenhert (2009) reported that ridges had fewer species of ferns than slope in Ecuador. They found that ridges had significantly higher abundance of terrestrial, but not epiphytic, species of ferns than slopes. Similarly, Costa *et al.* (2005) also found that slopes affected the diversity of fern in mesoscale. However, this is only applicable for terrestrial fern and not the epiphytic ferns.

Watkins *et al.* (2007) reported that increased in terrestrial gametophyte density and richness of ferns were related to both increased in light and disturbance. In contrast, they stated that increased in light and disturbance had no influence or negatively affected epiphytes density. In this study, more epiphytic fern species were recorded than terrestrial fern species. Thus, the abundance and the richness of epiphytic fern species could have influenced the correlation between fern abundance and the environmental variables.

### 5.4 Implication of the study to the management of conservation area

The findings of the current study clearly support previous studies on the importance of primary forest and logged-over forest as habitats for various flora species. It is obvious that primary forest has higher species diversity of ferns compared to the logged-over forest. However the logged-over forests of selectively logged harbour both forest dependent and the light demanding species. Selectively logged forest can provide forest-like habitat, but priority for conservation and protection should be given to the remaining primary forests. All categories of the forest play important roles to support various herbs species, especially ferns and providing resources to other organisms.

The present research has demonstrated that the logged-over forests, especially selectively logged forest, are important landscape that provides habitat and dispersal pathway and help to conserve local species assemblages of ferns. The presence of forest species of ferns in the logged-over forest showed that both categories of logged-over forest provide suitable habitat for forest-dependent species.

Low species diversity in the highly degraded forest suggested that the regional deforestation process, which happened in earlier times, contributed primarily to the decrease in fern species with regards to the community as a whole. High diversity of ferns in the primary forest does not mean that all species assemblages present in primary forests are rare and endemic. It does not mean that logged-over forests are less important in conserving biodiversity than primary forests.

# CHAPTER 6

# CONCLUSION

The primary forest had the highest diversity of ferns compared to the selectively logged forest and highly degraded forest in Danum Valley and Ulu Segama forest complex. Epiphytes fern were more diverse than terrestrial fern species in primary forests, which contributed to the diversification of fern species in the primary forest. Abundance of individual ferns was high in the primary forest compared to both categories of logged-over forest. Dominant species such as *Tectaria graffithii*, *Antrophyum callifolium* and *Leptochillus deccurens* contributed to the high abundance of ferns in the primary forest.

Some species of ferns that have been classified as forest species or forest dependent species were recorded in both categories of forest. Selectively logged forests are able to preserve some moist and shady habitats to which shade tolerant species are adapted. Both categories of logged-over forest in the study site have higher similarities of species composition than the primary forest. Plots in the primary forest are clustered separately, indicating that it has similar species of ferns as compared to the plots in the logged-over forests. The similar environmental condition shared by the logged-over forest of selectively logged and highly degraded enable the growth of similar species in such forest conditions. Micro-level site difference pertaining to environmental conditions has influenced the species diversity in each forest type. The Primary forest recorded the highest mean reading of canopy cover, relative humidity, and the air temperature among all of the forest types. However, all of the mean readings of the environmental variables were not significant except for the relative humidity. These mean that all of the forest types have fairly homogenous microclimate conditions.

Canopy coverage, relative humidity and air temperature have showed weak positive correlation with ferns diversity and cannot be determined as good regulator of the phenomenon in study sites. All of the linear regression relationships are not significant and only have minor influence on ferns diversity. Other environmental variables such as topography and soil condition may affect the changes in species composition of ferns communities rather than air temperature, relative humidity and canopy cover. These variables should be taken into consideration for future study. Both categories of logged-over forest are important landscape that provide habitat for forest-dependent species. However, the role of primary forest for conserving local species should be the priority in the management of conservation areas.

## REFERENCES

Achard, F., Eva, H. D., Stibig, H. J., Mayaux, P., Gallego, J., Richards, T. & Malingreau, J. P. (2002). Determination of Deforestation Rates of the World's Humid Tropical Forests. *Science*, **297** (5583): 999–1002.

Aleixo, A. (1999). Effects of Selective Logging on a Bird Community in the Brazilian Atlantic Forest. *The Condor*, **101**(3): 537–548.

Andrade, J. L. & Nobel, P. S. (1997). Microhabitats and Water Relations of Epiphytic Cacti and Ferns in a Lowland Neotropical Forest1. *Biotropica*, **29**(3): 261–270.

Arens, N. C. & Baracaldo, P. S. (2000). Variation in Tree Fern Stipe Length with Canopy Height: Tracking Preferred Habitat through Morphological Change. *American Fern Journal*, **90**(1): 1–15.

Ariyanti, N. S., Bos, M. M., Kartawinata, K., Tjitrosoedirdjo, S. S., Guhardja, E. & Gradstein, R.S. (2008). Bryophytes on tree trunks in natural forests, selectively logged forests and cacao agroforests in Central Sulawesi, Indonesia. *Biological Conservation*, **141**(10): 2516–2527.

Barcelona, J. F., Dolotina, N. E., Madroñero, G. S., Granert, W. G. & Sopot, D. D. (2006). The Ferns and Fern Allies of the Karst Forests of Bohol Island, Philippines. *American Fern Journal*, **96**(1): 1–20.

Barrington, D. S. (1993). Ecological and Historical Factors in Fern Biogeography. *Journal of Biogeography*, **20**(3): 275–279.

Barthlott, W., Schmit-Neuerburg, V., Nieder, J. & Engwald, S. (2001). Diversity and abundance of vascular epiphytes: a comparison of secondary vegetation and primary montane rain forest in the Venezuelan Andes. *Plant Ecology*, **152**(2): 145–156.

Barton, A. M., Fetcher, N., & Redhead, S. (1989). The Relationship between Treefall Gap Size and Light Flux in a Neotropical Rain Forest in Costa Rica. *Journal of Tropical Ecology*, **5**(4): 437–439.

Bazzaz, F. A. (1975). Plant Species Diversity in Old-Field Successional Ecosystems in Southern Illinois. *Ecology,* **56**(2): 485–488.

Beaman J. H., Edwards P. J. 2007. *Ferns of Kinabalu: an introduction.* Kota Kinabalu: Natural History Publications (Borneo).

Bellefroid, E., Chaerle, P., Leroux, O., & Viane, R. L. L. (2007). Additions to the Pteridophyte Flora of Kalimantan, Indonesian Borneo. *American Fern*

*Journal*, **97**(1): 1–18.

Bennett, E. L., & Robinson, J. G. (2000). *Hunting of Wildlife in Tropical Forests: Implications for Biodiversity and Forest Peoples*. Washington: The World Bank.

Bernard, H. (2004). Effects of selective logging on microhabitat-use patterns of non-volant small mammals in a Bornean tropical lowland mixed-dipterocarp forest. *Nature and Human Activities*, **8**: 1–11.

Bernard, H., Bili, R., Wearn, O. R., Hanya, G., & Ahmad, A. H. (n.d.). The distribution and persistence of primate species in disturbed and converted forest landscapes in Sabah, Malaysia: Preliminary results.

Berry, N. J., Phillips, O. L., Lewis, S. L., Hill, J. K., Edwards, D. P., Tawatao, N. B., & Hamer, K. C. (2010). The high value of logged tropical forests: lessons from northern Borneo. *Biodiversity and Conservation*, **19**(4:, 985–997.

Berry, N. J., Phillips, O. L., Ong, R. C., & Hamer, K. C. (2008). Impacts of selective logging on tree diversity across a rainforest landscape: the importance of spatial scale. *Landscape Ecology*, **23**(8): 915–929.

Beukema, H., & van Noordwijk, M. (2004). Terrestrial pteridophytes as indicators of a forest-like environment in rubber production systems in the lowlands of Jambi, Sumatra. *Agriculture, Ecosystems and Environment*, **104**(1): 63–73.

Blaser, J., Sarre, A., Poore, D., & Johnson, S. (2011). Status of tropical forest management. ITTO Techinal Series 38. *International Tropical Timber Organization, Yokohama, Japan*.

Bidin, A.A. & Jamal, R. (1999). The Pteridophytes Of Tawau Hills Park, Sabah. *ASEAN Review of Biodiversity and Environment Conservation (ARBEC)*, **2**, 1-11.

Braimoh, A. K., Subramanian, S. M., Elliott, W. S., & Gasparatos, A. (2010). Climate and human-related drivers of biodiversity decline in Southeast Asia. *UNU-IAS Policy Report*.

Brearley, F. Q., Prajadinata, S., Kidd, P. S., Proctor, J., & Suriantata. (2004). Structure and floristics of an old secondary rain forest in Central Kalimantan, Indonesia, and a comparison with adjacent primary forest. *Forest Ecology and Management,* **195**(3), 385–397.

Brodie, J. F., Giordano, A. J., Zipkin, E. F., Bernard, H., Mohd-Azlan, J., & Ambu, L. (2015). Correlation and persistence of hunting and logging impacts on tropical rainforest mammals. *Conservation Biology*, **29**(1): 110–121.

Brook, B. W., Sodhi, N. S., & Ng, P. K. L. (2003). Catastrophic extinctions follow deforestation in Singapore. *Nature*, **424**(6947): 420–426.

Brown, N. (1993). The Implications of Climate and Gap Microclimate for Seedling Growth Conditions in a Bornean Lowland Rain Forest. *Journal of Tropical Ecology*, **9**(2): 153–168.

Brown, S., & Lugo, A. E. (1990). Tropical secondary forests. *Journal of Tropical Ecology*, **6**(1): 1–32.

Cannon, C. H., Peart, D. R., Leighton, M., & Kartawinata, K. (1994). The structure of lowland rainforest after selective logging in West Kalimantan, Indonesia. *Forest Ecology and Management*, **67**(1): 49–68.

Cardelus, C. L. (2002). Distribution and abundance of vascular epiphytes in tropical wet forests: A multi-scale approach.

Chen, J., Saunders, S. C., Crow, T. R., Naiman, R. J., Brosofske, K. D., Mroz, G. D. & Franklin, J. F. (1999). Microclimate in Forest Ecosystem and Landscape Ecology Variations in local climate can be used to monitor and compare the effects of different management regimes. *BioScience,* **49**(4): 288–297.

Clarke, M. A., Walsh, R. P. D., Sinun, W., Larenus, J., & Hanapi, J. (2002). Soil erosion and surface roughness dynamics in relation to slope angle in primary and logged tropical rainforest, eastern Sabah. (pp. 1471–1484). GEOFORMA Edicions, S.L.

Collins, N. M., Sayer, J. A., & Whitmore, T. C. (1991). Western Pacific Islands. In *The Conservation Atlas of Tropical Forest Asia and the Pacific* (pp. 240-249). Palgrave Macmillan UK.

Copeland, E. B. (1931). Sarawak Ferns Collected by J. and M. S. Clemens. *Brittonia*, **1**(2): 71–78.

Copeland, H. F. (1947). Progress Report on Basic Classification. The American *Naturalist*, **81**(800): 340–361.

Costa, F., & Magnusson, W. (2002). Selective Logging Effects on Abundance, Diversity, and Composition of Tropical Understory Herbs. *Ecological Applications*, **12**(3): 807–819.

Costa, F. R. C., Magnusson, W. E., & Luizao, R. C. (2005). Mesoscale distribution patterns of Amazonian understorey herbs in relation to topography, soil and watersheds. *Journal of Ecology,* **93**(5): 863–878.

Christenhusz, M., Zhang, X. C., & Schneider, H. (2011). A linear sequence of extant families and genera of lycophytes and ferns. *Phytotaxa.*

Dubuisson, J. Y., Hennequin, S., Rakotondrainibe, F., & Schneider, H. (2003). Ecological diversity and adaptive tendencies in the tropical fern Trichomanes L. (Hymenophyllaceae) with special reference to climbing and epiphytic habits. *Botanical Journal of the Linnean Society*, **142**(1): 41–63.

Duque, A. J., Duivenvoorden, J. F., Cavelier, J., Sánchez, M., Polanía, C., & León, A. (2005). Ferns and Melastomataceae as indicators of vascular plant composition in rain forests of Colombian Amazonia. *Plant Ecology*, **178**(1): 1–13.

Dzwonko, Z., & Kornas, J. (1994). Patterns of Species Richness and Distribution of Pteridophytes in Rwanda (Central Africa): A Numerical Approach. *Journal of Biogeography*, **21**(5): 491–501.

Eggleton, P., Homathevi, R., Jones, D. T., MacDonald, J. A., Jeeva, D., Bignell, D. E., & Maryati, M. (1999). Termite assemblages, forest disturbance and greenhouse gas fluxes in Sabah, East Malaysia. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **354**(1391): 1791–1802.

Frumhoff, P. C. (1995). Conserving wildlife in tropical forest managed for timber. *BioScience*, 45 (7): 456-464.

Gardner, T. A., Barlow, J., Chazdon, R., Ewers, R. M., Harvey, C. A., & Sodhi, N. S. (2009). Prospects for tropical forest biodiversity in a human-modified world. *Ecology letters,* 12 (6): 561-582.

Gibson, L., Lee, T. M., Koh, L. P., Brook, B. W., Gardner, T. A., Barlow, J., & Sodhi, N. S. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, **478**(7369): 378–381.

Gobilik, J. 2002. Some aspects of abundance and distribution of Zingiberaceae and Costaceae in Tabin Wildlife Reserve, Sabah. Master Thesis, Universiti Malaysia Sabah (unpublished).

Greer, G. K., Lloyd, R. M., & McCarthy, B. C. (1997). Factors Influencing the Distribution of Pteridophytes in a Southeastern Ohio Hardwood Forest. *The Journal of the Torrey Botanical Society*, **124**(1): 11–21.

Hazebroek, H. P., Adlin, T. Z & Sinun, W. 2004. Maliau Basin: *Sabah's Lost World*. Natural History Publication (Borneo) Kota Kinabalu.

Hamer, K. C., Hill, J. K., Benedick, S., Mustaffa, N., Sherratt, T. N., Maryati, M., & K., C. V. (2003). Ecology of butterflies in natural and selectively logged forests of northern Borneo: the importance of habitat heterogeneity. *Journal of Applied Ecology*, **40**(1): 150–162.

Hemp, A. (2002). Ecology of the pteridophytes on the southern slopes of Mt. Kilimanjaro – I. Altitudinal distribution. *Plant Ecology*, **159**(2): 211–239.

Henderson, P. A., & Seaby, R. M. H. (1998). Species diversity and richness. *PISCES Conservation Limited*.

Heydon, M. J., & Bulloh, P. (1997). Mousedeer Densities in a Tropical Rainforest: The Impact of Selective Logging. *Journal of Applied Ecology*, **34**(2): 484–496.

Hietz, P. (1997). Population Dynamics of Epiphytes in a Mexican Humid Montane Forest. *Journal of Ecology*, **85**(6): 767–775.

Hietz, P., & Briones, O. (1998). Correlation between water relations and within-canopy distribution of epiphytic ferns in a Mexican cloud forest. *Oecologia*, **114**(3): 305–316.

Hietz, P., Buchberger, G., & Winkler, M. (2006). Effect of forest disturbance on abundance and distribution of epiphytic bromeliads and orchids. *Ecotropica*, **12**(2): 103–112.

Hietz-Seifert, U., Hietz, P., & Guevara, S. (1996). Epiphyte vegetation and diversity on remnant trees after forest clearance in southern Veracruz, Mexico. *Biological Conservation*, **75**(2): 103–111.

Hill, J. D., & Silander, J. A. (2001). Distribution and dynamics of two ferns: Dennstaedtia punctilobula (Dennstaedtiaceae) and Thelypteris noveboracensis (Thelypteridaceae) in a Northeast mixed hardwoods–hemlock forest. *American Journal of Botany*, **88**(5): 894–902.

Hill, J. K., Hamer, K. C., Dawood, M. M., Tangah, J., & Chey, V. K. (2003). Rainfall but not selective logging affect changes in abundance of a tropical forest butterfly in Sabah, Borneo. Journal of Tropical Ecology, **19** (1): 35–42.

Hill, J. K., & Hamer, K. C. (2004). Determining impacts of habitat modification on diversity of tropical forest fauna: the importance of spatial scale. Journal of Applied Ecology, **41**(4): 744–754.

Holttum, R. E. (1947). A revised classification of Leptosporangiate ferns. *Journal of the Linnean Society of London, Botany*, **53**(350): 123–158.

Imai, N., Seino, T., Aiba, S., Takyu, M., Titin, J., & Kitayama, K. (2012). Effects of selective logging on tree species diversity and composition of Bornean tropical rain forests at different spatial scales. *Plant Ecology*, **213**(9): 1413–1424.

Jaman, R. & Latiff, A. 1998. On some pteridophytes of Sayap-Kinabalu Park, Sabah. *ASEAN Review of Biodiversity and Environmental Conservation (ARBEC),* **4**: 1-35.

Jennings, S. B., Brown, N. D., & Sheil, D. (1999). Assessing forest canopies and understorey illumination: canopy closure, canopy cover and other measures. *Forestry*, **72**(1): 59–74.

Johns, A. G. (1997). *Timber Production and Biodiversity Conservation in Tropical Rain Forests*. Cambridge University Press.

Karst, J., Gilbert, B., & Lechowicz, M. J. (2005). Fern Community Assembly: The Roles of Chance and the Environment at Local and Intermediate Scales. *Ecology*, **86**(9): 2473–2486.

Kessler, M. (2000). Elevational gradients in species richness and endemism of selected plant groups in the central Bolivian Andes. *Plant Ecology*, **149**(2): 181–193.

Kessler, M. (2001). Pteridophyte species richness in Andean forests in Bolivia. *Biodiversity & Conservation*, **10**(9): 1473–1495.

Kessler, M. (2010). Biogeography of ferns. *Fern Ecology*, 22–60.

Kessler, M., Siorak, Y., Wunderlich, M. & Wegner, C. (2007). Patterns of morphological leaf traits among pteridophytes along humidity and temperature gradients in the Bolivian Andes. *Functional Plant Biology*, **34**(11): 963–971.

Kessler, M., & Lehnert, M. (2009). Are Ridge Habitats Special Sites for Endemic Plants in Tropical Montane Rain Forests? A Case Study of Pteridophytes in Ecuador. *Folia Geobotanica*, 44(4): 387–398.

Kluge, J., & Kessler, M. (2006). Fern endemism and its correlates: contribution from an elevational transect in Costa Rica. *Diversity and Distributions*, **12**(5): 535–545.

Kluge, J., & Kessler, M. (2011). Phylogenetic diversity, trait diversity and niches: species assembly of ferns along a tropical elevational gradient. *Journal of Biogeography*, **38**(2): 394–405.

Kluge, J., Kessler, M., & Dunn, R. R. (2006). What drives elevational patterns of diversity? A test of geometric constraints, climate and species pool effects for pteridophytes on an elevational gradient in Costa Rica. *Global Ecology and Biogeography*, **15**(4): 358–371.

Koh, L. P. (2007). Impending disaster or sliver of hope for Southeast Asian forests? The devil may lie in the details. *Biodiversity and Conservation*, **16**(13): 3935–3938.

Koh, L. P., Gibbs, H. K., Potapov, P. V., & Hansen, M. C. (2012). REDDcalculator.com: a web-based decision-support tool for implementing Indonesia's forest moratorium. *Methods in Ecology and Evolution*, **3**(2): 310–316.

Kornas, J. (1993). The Significance of Historical Factors and Ecological Preference in the Distribution of African Pteridophytes. *Journal of Biogeography*, **20**(3): 281–286.

Lambert, F. R. (1992). The Consequences of Selective Logging for Bornean Lowland Forest Birds. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, **335**(1275): 443–457.

Laurance, W. F., Lovejoy, T. E., Vasconcelos, H. L., Bruna, E. M., Didham, R. K., Stouffer, P. C., & Sampaio, E. (2002). Ecosystem Decay of Amazonian Forest Fragments: a 22-Year Investigation. *Conservation Biology*, **16**(3): 605–618.

Laurance, W. F. (2007). Forest destruction in tropical Asia. *Current Science*, 1544-1550.

Lellinger, D. B. (1985). The distribution of Panama's pteridophytes. *Monographs in Systematic Botany from the Missouri Botanical Garden (USA).*

Lemmon, P. E. 1957. A new instrument for measuring forest overstory density. *Journal of forestry*, **55**(9): 667-668.

Lovejoy, T. E., Bierregaard, R. O. J., Rylands, A. B., Malcolm, J. R., Quintela, C. E., Harper, L. H., & Hays, M. B. (1986). Edge and other effects of isolation on Amazon forest fragments. *Nature,* **478** (7369): 378-381.

Lwanga, J. S., Balmford, A., & Badaza, R. (1998). Assessing fern diversity: relative species richness and its environmental correlates in Uganda. *Biodiversity & Conservation*, **7** (11): 1387–1398.

Magguran, A. E. 1988. *Ecological diversity and its measurement*. Princeton University Press, New Jersey. 192 pp.

Majit, H. F., Suleiman, M. & Rimi, R. (2011). Diversity and abundance of orchids in Crocker Range Park, Sabah, Malaysia. *Journal of Tropical Biology and Conservation*, **8**: 73–81.

Marquez, A. L., Real, R., Vargas, J. M., & Salvo, A. E. (1997). On Identifying Common Distribution Patterns and Their Causal Factors: A Probabilistic Method Applied to Pteridophytes in the Iberian Peninsula. *Journal of Biogeography*, **24** (5): 613–631.

Marsh, C. W., & Greer, A. G. (1992). Forest Land-Use in Sabah, Malaysia: An Introduction to Danum Valley. Philosophical Transactions: *Biological Sciences,* **335**(1275): 331–339.

Mehltreter, K. (1996). Species richness and geographical distribution of montane pteridophytes of Costa Rica, Central America. *Feddes Repertorium*, **106** (58), 563–584.

Merrill, E. D. (1946). A Mindoro Fern Adventure. *American Fern Journal*, **36** (2): 33–47.

Mertens, B., Forni, E., & Lambin, E. F. (2001). Prediction of the impact of logging activities on forest cover : A case-study in the East province of Cameroon. *Journal of Environmental Management*, **62**(1): 21–36.

Moran, R. C., Klimas, S., & Carlsen, M. (2003). Low-Trunk Epiphytic Ferns on Tree Ferns Versus Angiosperms in Costa Rica. *Biotropica*, **35** (1): 48–56.

Moura-Costa, P., Karolus, A. (1992). Timber extraction volumes (1970–1991) of Ulu Segama Forest Reserve. Innoprise Corporation Sdn. Bhd. Unpublished.

Murakami, K., Maenaka, H., & Morimoto, Y. (2005). Factors influencing species diversity of ferns and fern allies in fragmented forest patches in the Kyoto city area. *Landscape and Urban Planning*, **70** (3–4): 221–229.

Okuda, T., Suzuki, M., Adachi, N., Quah, E. S., Hussein, N. A., & Manokaran, N. (2003). Effect of selective logging on canopy and stand structure and tree species composition in a lowland dipterocarp forest in peninsular Malaysia. *Forest Ecology and Management*, **175**(1–3): 297–320.

Paciencia, M. L. B., & Prado, J. (2005). Effects of Forest Fragmentation on Pteridophyte Diversity in a Tropical Rain Forest in Brazil. *Plant Ecology*, **180**(1): 87–104.

Parris, B. S. (1984). Ecological aspects of distribution and speciation in Old World tropical ferns. *Proceedings of the Royal Society of Edinburgh Section B: Biological Sciences*, **86**, 341–346.

Parris, B. S., Beaman, R. S & Beaman, J. H. (1992).The plants of Mount Kinabalu.1. Ferns and ferns allies. Royal Botanical Gardens, Kew.

Parris, B.S. (1997a). A provisional checklist of pteridophytes from the Danum Valley, Sabah. *Sandakania,* **9**: 77-85.

Parris, B. S. (1997b). A provisional checklist of pteridophytes from the Crocker Range National Park, Sabah. *Sandakania,* **9**: 103-111.

Parris, B. S., Jaman, R. & Latiff, A. (2012).RESEARCH NOTES Oreogrammitis translucens, a New Species of Fern (Polypodiopsida: Grammatidaceae) from Maliau Basin Conservation Area, Sabah, Malaysia. *Sains Malaysiana*, **41**(3): 299–301.

Parry, L., Barlow, J., & Peres, C. A. (2007). Large-vertebrate assemblages of primary and secondary forests in the Brazilian Amazon. *Journal of Tropical Ecology*, **23**(6): 653–662.

Pausas, J. G., & Sáez, L. (2000). Pteridophyte richness in the NE Iberian Peninsula: biogeographic patterns. *Plant Ecology*, **148** (2): 195–205.

Peters, S. L., Malcolm, J. R., & Zimmerman, B. L. (2006). Effects of Selective Logging on Bat Communities in the Southeastern Amazon. *Conservation Biology*, **20**(5): 1410–1421.

Phua, M.H., Hue, S.W., Ioki, K., Hashim, M., Bidin,K., Musta, B., & Maycock, C.R. (2016). Estimating logged-over lowland rainforest aboveground biomass in Sabah, Malaysia using airborne LiDAR data. *Terrestrial, Atmospheric & Oceanic Sciences*, **27** (4): 481-489.

Piggot, A.G., Piggot, C.J., 1988. *Ferns of Malaysia in Colour*. Tropical Press Sdn. Bhd., Kuala Lumpur.

Poore, D., Burgess, P., Palmer, J., Rietbergen, S., Synnott, T., Kula, E., & Kumar, S.

K. (1989). *No timber without trees: sustainability in the tropical forest (No. GTZ-22)*. ITTO, London (RU).

Poulsen, A. D. (1996). Species Richness and Density of Ground Herbs within a Plot of Lowland Rainforest in North-West Borneo. *Journal of Tropical Ecology*, **12**(2): 177–190.

Putz, F. E., Zuidema, P. A., Synnott, T., Peña-Claros, M., Pinard, M. A., Sheil, D. & Zagt, R. (2012). Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. *Conservation Letters*, ***5*** (4): 296–303.

Rahbek, C. (1995). The elevational gradient of species richness: a uniform pattern? *Ecography*, **18** (2): 200–205.

Richard, M., Bernhardt, T., & Bell, G. (2000). Environmental heterogeneity and the spatial structure of fern species diversity in one hectare of old-growth forest. Ecography, **23**(2): 231–245.

Ripin, R., Dolois, S., Rahimah, P., Yabainus, J. & Gusilli, J. (2004). Additions to the checklist of plants of Crocker Range Park. *In*: Maryati, M., Zulhazaman, H., Takuji, T. & Jamili, N. (eds). *Crocker Range Scientific Expedition 2002*. Kota Kinabalu: Universiti Malaysia Sabah.

Roberts, M. R., & Gilliam, F. S. (1995). Disturbance effects on herbaceous layer vegetation and soil nutrients in Populus forests of northern lower Michigan. *Journal of Vegetation Science*, **6** (6): 903–912.

Ruokolainen, K., Linna, A., & Tuomisto, H. (1997). Use of Melastomataceae and pteridophytes for revealing phytogeographical patterns in Amazonian rain forests. *Journal of Tropical Ecology*, **13**(2): 243–256.

Russell, A. E., Raich, J. W., & Vitousek, P. M. (1998). The ecology of the climbing fern Dicranopteris linearis on windward Mauna Loa, Hawaii. *Journal of Ecology*, **86**(5): 765–779.

Saatchi, S. S., Harris, N. L., Brown, S., Lefsky, M., Mitchard, E. T. A., Salas, W., & Morel, A. (2011). Benchmark map of forest carbon stocks in tropical regions across three continents. *Proceedings of the National Academy of Sciences*, **108**(24): 9899–9904.

Salovaara, K. J., Cárdenas, G. G., & Tuomisto, H. (2004). Forest classification in an Amazonian rainforest landscape using pteridophytes as indicator species. *Ecography*, **27**(6): 689–700.

Sayer, J. A., & Whitmore, T. C. (1991). Tropical moist forests: Destruction and species extinction. *Biological Conservation*, **55**(2): 199–213.

Schleuning, M., Farwig, N., Peters, M. K., Bergsdorf, T., Bleher, B., Brandl, R., Böhning-Gaese, K. (2011). Forest Fragmentation and Selective Logging Have Inconsistent Effects on Multiple Animal-Mediated Ecosystem Processes in a Tropical Forest. *PLOS ONE*, **6**(11): 27-85.

Sheil, D., Sayer, J. A., & O'Brien, T. (1999). Tree Species Diversity in Logged Rainforests. *Science*, **284**(5420): 1587–1587.

Shim, P.S., Jaman, R. & Thani N. E. A. (2012). Preliminary survey of ferns in the Imbak Canyon Conservation Area. *In:* Mohamad A.L & Sinun, W. (eds). *Imbak Canyon Conservation Area, Sabah*. Kota Kinabalu: Akademi Sains Malaysia and Yayasan Sabah.

Silva, J. N. M., de Carvalho, J. O. P., Lopes, J. do C. A., de Almeida, B. F., Costa, D. H. M., de Oliveira, L. C. & Skovsgaard, J. P. (1995). Growth and yield of a tropical rain forest in the Brazilian Amazon 13 years after logging. *Forest Ecology and Management*, **71**(3): 267–274.

Silva, I. a. A., Pereira, A. F. N., & Barros, I. C. L. (2011). Edge effects on fern community in an Atlantic Forest remnant of Rio Formoso, PE, Brazil. *Brazilian Journal of Biology*, **71**(2): 421–430.

Slik, J. W. F., Bernard, C. S., Breman, F. C., Van Beek, M., Salim, A., & Sheil, D. (2008). Wood Density as a Conservation Tool: Quantification of Disturbance and Identification of Conservation-Priority Areas in Tropical Forests. *Conservation Biology*, **22**(5): 1299–1308.

Smith, A. R., Pryer, K. M., Schuettpelz, E., Korall, P., Schneider, H., & Wolf, P. G. (2006). A Classification for Extant Ferns. *Taxon*, **55**(3): 705–731.

Struebing, M. J., Turner, A., Giles, E., Lasmana, F., Tollington, S.,Bernard, H., & Bell, D. (2013). Quantifying the biodiversity value of repeatedly logged rainforest gradient and comparative approaches from Borneo. *Advances in ecological research,* **48**: 183-224.

Sukup Akin. (1998). Checklist of ferns and ferns allies in the Maliau Basin Conservation Area, Sabah. *In*: Maryati, M., Sinun, W., Anton, A., Dalimin, M. N. & Abdul, H. A. (eds). *Maliau Basin Scientific Expedition*. Kota Kinabalu: Universiti Malaysia Sabah

Tuomisto, H., & Ruokolainen, K. (1994). Distribution of Pteridophyta and Melastomataceae along an edaphic gradient in an Amazonian rain forest. *Journal of Vegetation Science*, **5**(1): 25–34.

Tuomisto, H., & Poulsen, A. D. (2000). Pteridophyte Diversity and Species Composition in Four Amazonian Rain Forests. *Journal of Vegetation Science*, **11**(3): 383–396.

Ugland, K. I., Gray, J. S., & Ellingsen, K. E. (2003). The species-accumulation curve and estimation of species richness. *Journal of Animal Ecology*, **72** (5): 888-897.

Underwood, L. M. (1906). American Ferns-VI. Species Added to the Flora of the United States from 1900 to 1905. *Bulletin of the Torrey Botanical Club*, **33**(3): 189–205.

Van Gardingen, P. R., McLeish, M. J., Philips, P. D., Fadillah, D., Tyrie, G., & Yasman, I. (2003). Financial and ecological analysis of management options for logged-over Dipterocarp forests in Indonesia Borneo. *Forest Ecology and Management*, **183** (1): 1-29

Verburg, R., & van Eijk-Bos, C. (2003). Effects of selective logging on tree diversity, composition and plant functional type patterns in a Bornean rain forest. *Journal of Vegetation Science*, **14**(1): 99–110.

Vormisto, J., Phillips, O. L., Ruokolainen, K., Tuomisto, H., & Vásquez, R. (2000). A comparison of fine-scale distribution patterns of four plant groups in an Amazonian rainforest. Ecography, **23**(3): 349–359.

Walsh, R. P. D. (1996). Drought frequency changes in Sabah and adjacent parts of northern Borneo since the late nineteenth century and possible implications for tropical rain forest dynamics. Journal of Tropical Ecology, **12**(3): 385–407.

Walsh, R. P., & Newbery, D. M. (1999). The ecoclimatology of Danum, Sabah, in the context of the world's rainforest regions, with particular reference to dry periods and their impact. *Philosophical Transactions of the Royal Society B: Biological Sciences,* **354**(1391): 1869–1883.

Watkins, J. E., Mack, M. K., & Mulkey, S. S. (2007). Gametophyte ecology and demography of epiphytic and terrestrial tropical ferns. American Journal of Botany, **94**(4): 701–708.

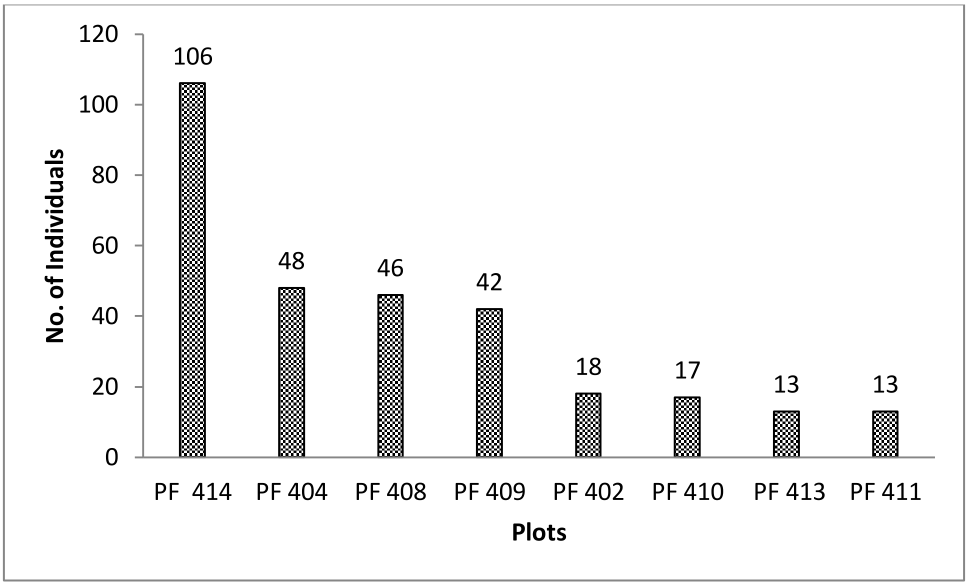
Watkins, J. E., & Cardelús, C. (2009). Habitat Differentiation of Ferns in a Lowland Tropical Rain Forest. *American Fern Journal*, **99**(3): 162–175.

Wilcove, D., S., & Koh, L.P. (2010). Addressing the threats to biodiversity from oil-palm agriculture. *Biodiversity and Conservation*, **19**(4): 999-1007.

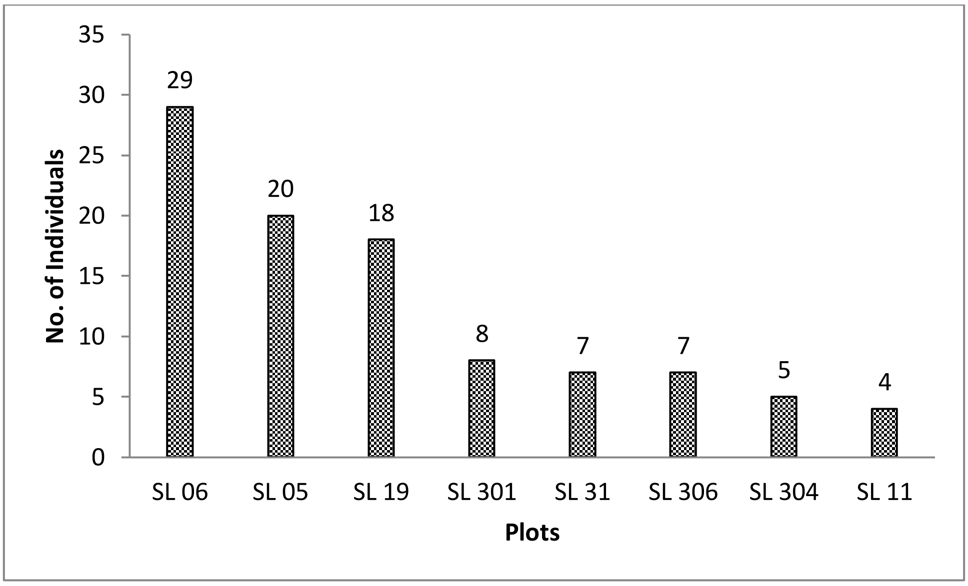
Willott, S. J., Lim, D. C., Compton, S. G., & Sutton, S. L. (2000). Effects of Selective Logging on the Butterflies of a Bornean Rainforest. *Conservation Biology*, **14**(4): 1055–1065.

## APPENDIX I

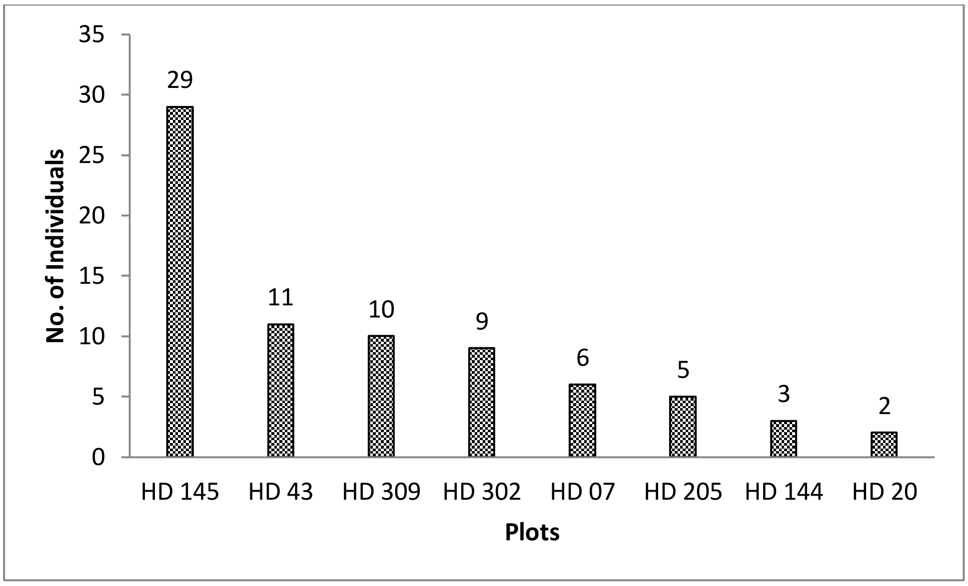
a) Plot with descending number of ferns individuals in Primary forest



b) Plot with descending number of ferns individuals in logged-over forest of selectively logged forest



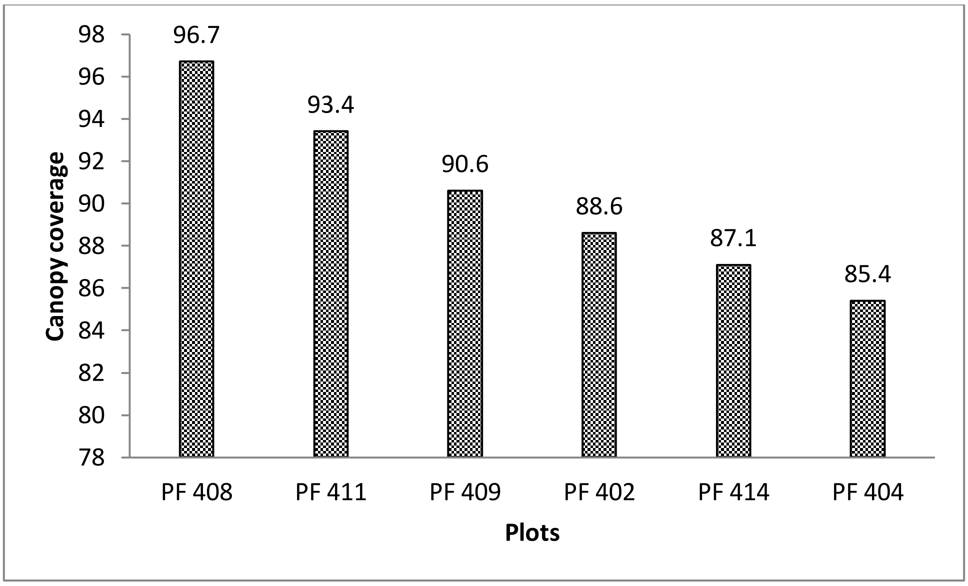
c) Plot with the descending number of ferns individuals in logged-over forest of highly degraded



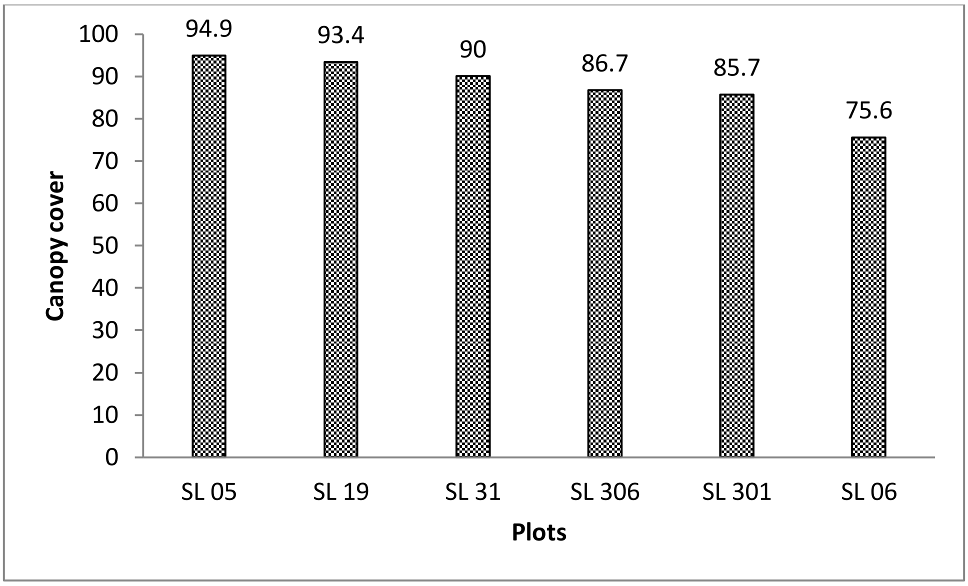
**APPENDIX II**

## Canopy coverage reading in each plot of different forest types

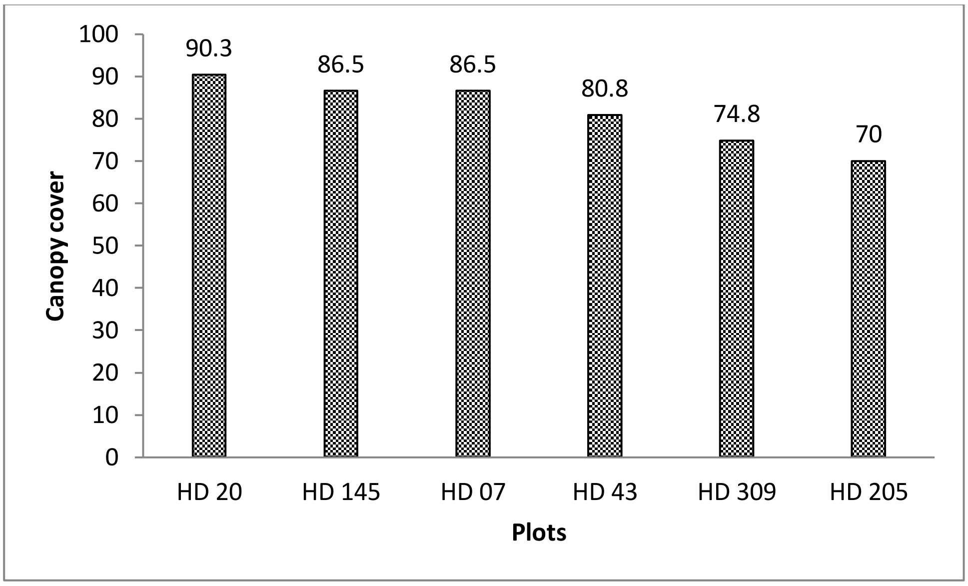
a) Canopy coverage reading in each plot of the primary forest



b) Canopy coverage reading in each plot of the selectively logged forest

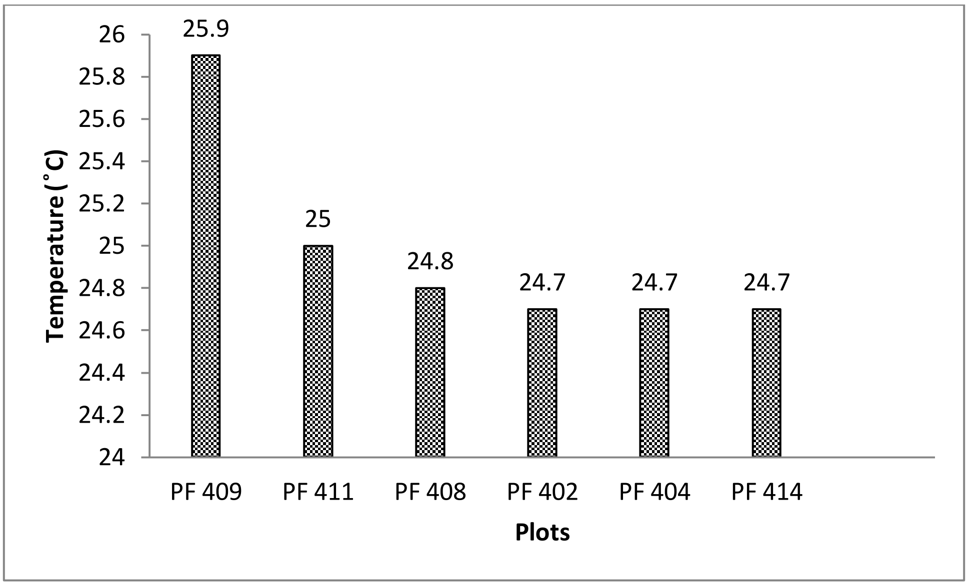


c) Canopy coverage in each plot of highly degraded forest

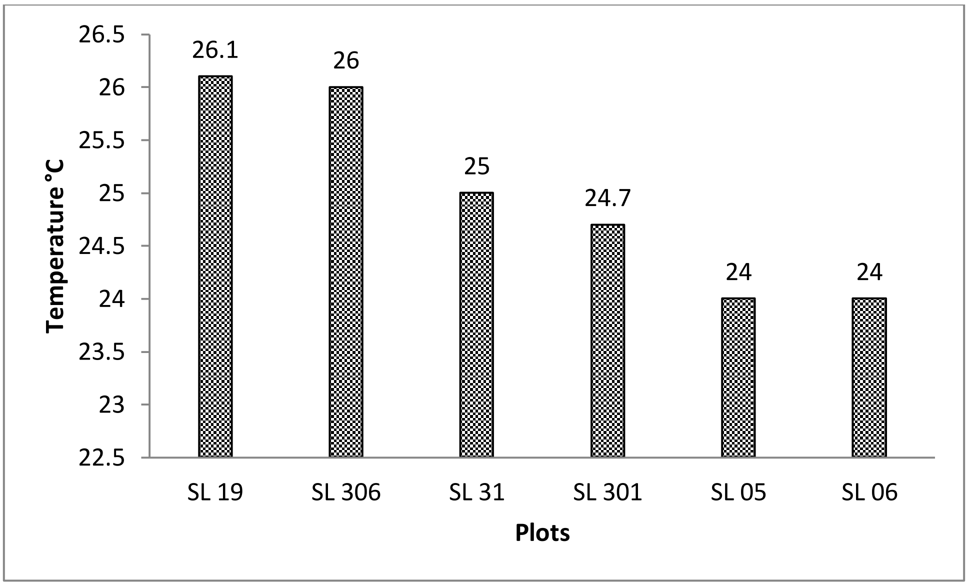


## Temperature recorded in each plots of the different forest types

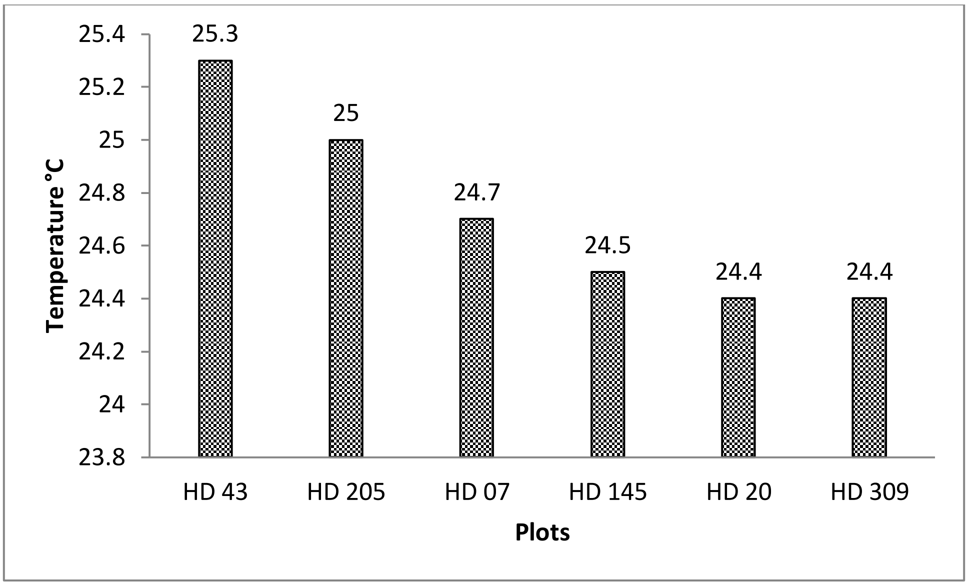
a) Temperature recorded in each plots of primary forest



b) Temperature recorded in each plots of selectively logged forest



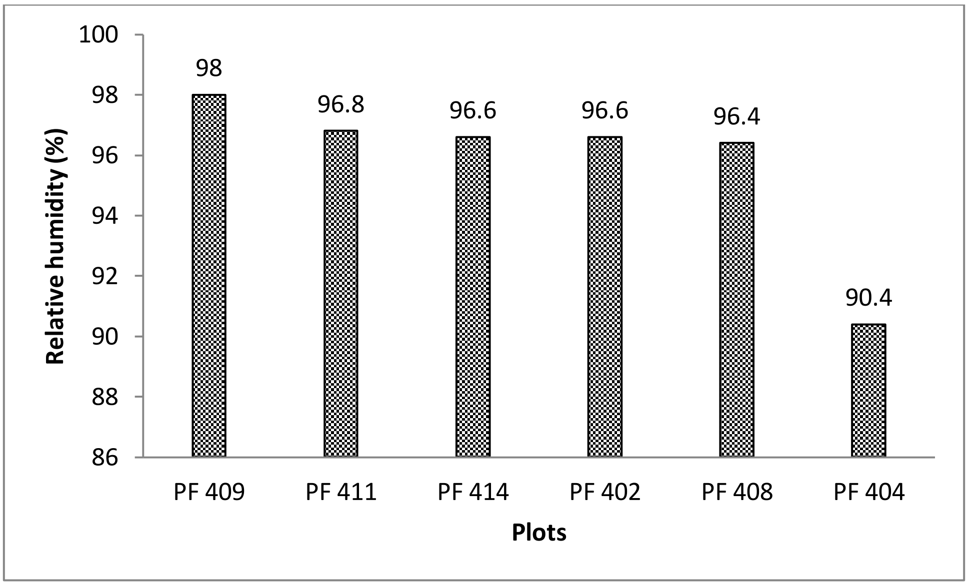
c) Temperature recorded in each plots of highly degraded forest



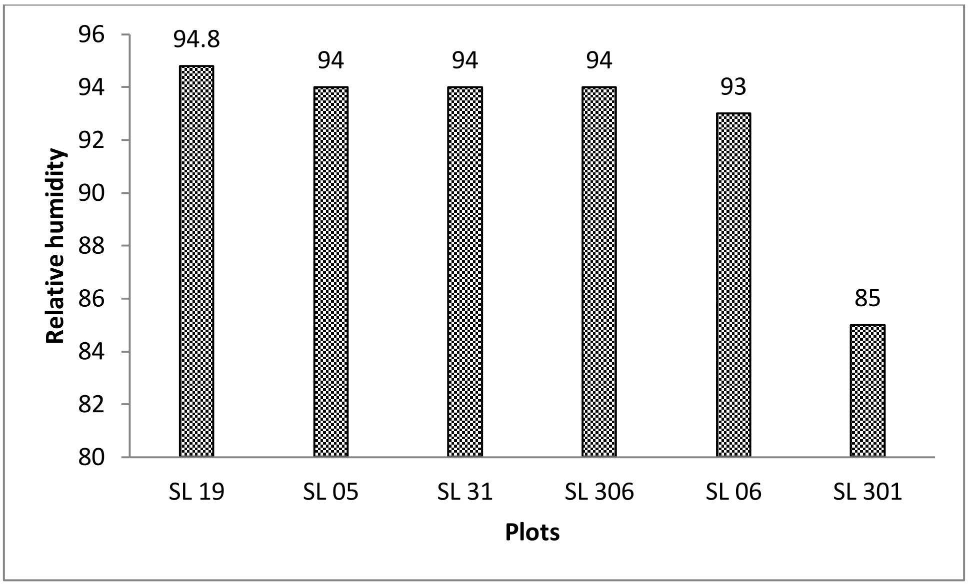
**APPENDIX IV**

**Relative humidity in each plot of different forest types**

a) Relative humidity in each plot of primary forest



b) Relative humidity in each plot of selectively logged forest



c) Relative humidity in each plot of Highly degraded forest